US ARMY CORPS
OF ENGINEERS
NEW YORK DISTRICT

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
REGION 2

February 9, 2000

MEMO FOR THE RECORD

SUBJECT: Review of Compliance with the Testing Requirements of 40 CFR 227.6 and 227.27, and Site Designation Provisions of 40 CFR 228.15 for the Federal Navigation Project No. 36, Buttermilk Channel, New York Bay, New York.

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I. SUMMARY

This memorandum provides comprehensive review and analysis of the Federal Navigation Project No. 36, Buttermilk Channel (Buttermilk Channel) sediment test results. This memorandum addresses compliance with the regulatory testing criteria of 40 CFR Sections 227.6 and 227.27, and the requirements of the rule establishing the Historic Area Remediation Site (HARS) set out in Section 228.15(d)(6). These requirements hereinafter are referred to as the "Regulations."

As discussed in the HARS rulemaking preamble (See 62 Fed. Reg. 46142 (August 29, 1997) and 62 Fed. Reg. 26267 (May 13, 1997)) and its accompanying documentation, the need to remediate the Historic Area Remediation Site is amply supported by the presence in the HARS of toxic effects, dioxin bioaccumulation exceeding Category I levels in worm tissue, as well as TCDD/PCB contamination in area lobster stocks. Individual elements of the aforementioned data do not prove that sediments within the Study Area are imminent hazards to the New York Bight Apex ecosystem, living resources, or

human health. However, the collective evidence presents cause for concern, justifies that a need for remediation exists, that the site is Impact Category I (see, 40 CFR 228.10), and that the site should be managed to reduce impacts to acceptable levels (see, 40 CFR 228.11(c)). Further information on the conditions in the Study Area and the surveys performed may be found in the Supplemental Environmental Impact Statement (EPA, 1997c).

This evaluation confirms that: 1) all tests required under the Regulations were conducted; 2) this project meets the criteria at 40 CFR Section 227.6 for trace contaminants and Section 227.27 for Limiting Permissible Concentration (LPC); and 3) the dredged material is Category I under U.S. Environmental Protection Agency (USEPA) Region 2/Corps of Engineers, N.Y. District (CENAN) guidance and is suitable for placement at the HARS as Remediation Material.

II. PROJECT DESCRIPTION

The proposal is to dredge and place approximately 112,000 cubic yards (yd) of dredged material at the HARS. The project encompassed one reach; sediment core samples were taken from seven locations to characterize the sediment (see sampling plan (CENAN, 1999)).

III. REGULATORY REQUIREMENTS

In order for dredged material to be suitable for use as Remediation Material to be placed at the HARS, it must conform to the Regulations. The Marine Protection, Research, and Sanctuaries Act (MPRSA) or "The Act" prohibits dumping of materials into the ocean except as authorized by USEPA or, in the case of dredged materials, by the U.S. Army Corps of Engineers (USACE). Section 102 of the Act directs the USEPA to establish and apply criteria for reviewing and evaluating permit applications (33 U.S.C. Section 1412). The USEPA has adopted such criteria in the Regulations. 40 CFR Section 227.6(a) lists constituents that are prohibited from being placed in the ocean unless only present as trace contaminants in material otherwise suitable for dumping (hereinafter referred to as "listed constituents"). Section 227.27 addresses compliance with the LPC. See also, Section 227.13(c).

Section 227.6(b) states that constituents are considered to be present as trace contaminants only when they are present in such forms and amounts that the "dumping of the materials will not cause significant undesirable effects, including the possibility of danger associated with their bioaccumulation in marine organisms." The regulations set forth criteria for determining the potential for significant undesirable effects in Section 227.6(c). In order to be found environmentally acceptable for ocean placement, it must be found that the liquid phase does not contain any of the listed constituents in concentrations that would exceed applicable marine water quality criteria after allowance for initial mixing (Section 227.6(c)(1)). For the suspended particulate phase (Section 227.6(c)(2)) and the solid phase (Section 227.6(c)(3)), bioassay results must not indicate occurrence of significant mortality or significant adverse sublethal effects due to the ocean placement of wastes containing the listed constituents.

Section 227.27 of the regulations addresses the LPC. For the liquid phase, Section 227.27(a) provides that the LPC is that concentration which does not exceed applicable marine water quality criteria after initial mixing, or when there are no applicable marine water criteria, that concentration of material that, after initial mixing, would not exceed 0.01 of a concentration shown to be acutely toxic to appropriate sensitive marine organisms in a bioassay carried out in accordance with procedures approved by USEPA and USACE. For the suspended particulate phase and the solid phase, Section 227.27(b) provides that the LPC is that concentration of material which will not cause unreasonable acute or chronic toxicity or other sublethal adverse effects based on results of bioassays using appropriate sensitive organisms and conducted according to procedures that have been approved by USEPA and USACE, and which will not cause accumulation of toxic materials in the human food chain.

The HARS encompasses an area which includes the Mud Dump Site (MDS), and which has exhibited the potential for adverse ecological impacts and has been identified for remediation. The site will be remediated with uncontaminated dredged material (i.e., dredged material that meets current Category I standards and will not cause significant undesirable effects including through bioaccumulation) (hereinafter referred to as "Remediation Material" or "Material for Remediation"). Under 40 CFR 228.15 (d)(6)(v)(A) the site will be managed to reduce impacts within the Primary Remediation Area (PRA) to acceptable levels in accordance with 40 CFR Section 228.11(c). Use of the site is restricted to dredged material suitable for use as Material for Remediation. This material shall be selected so as to ensure it will not cause significant undesirable effects including through bioaccumulation or unacceptable toxicity in accordance with 40 CFR 227.6.

Section 228.15(d)(6) of the Regulations describes the locations where material from NY/NJ Harbor and surrounding areas may be placed in the HARS, provided that it is suitable as Remediation Material. The HARS consists of a PRA, a Buffer Zone, and a No Discharge Zone. Under 228.15(d)(6) placement of Remediation Material is limited to the PRA.

IV. GUIDANCE FOR TESTING AND EVALUATION OF DREDGED MATERIAL

The discussion below describes how the material proposed for placement at the HARS as Remediation Material, resulting from the maintenance dredging of Buttermilk Channel was evaluated for compliance with the requirements of 40 CFR 227.6, 227.27, and 228.15(d)(6). Testing of the material was conducted following procedures approved by USEPA and USACE, and contained in the joint USEPA/USACE national guidance "Evaluation of Dredged Material Proposed for Ocean Dumping - Testing Manual" (February, 1991) (the "Green Book") (USEPA/USACE, 1991), and the regional implementation manual developed by the USEPA Region 2 and CENAN (USEPA/CENAN, 1992).

These test results were analyzed in accordance with the Regulations to ensure that the proposed placement meets the criteria of Part 227 and the requirements of 228.15(d)(6). As explained in the preamble to the HARS Rule, Remediation Material is uncontaminated dredged material (i.e., dredged

material that meets current Category I standards and will not cause significant undesirable effects including through bioaccumulation). The determination of whether materials meeting the Part 227 criteria are assigned to Category I is based upon technical and scientific judgment as set out below.

Applying the USEPA Region 2/CENAN guidance to this project, the material would be Category I if it meets the Part 227 criteria (including the requirements regarding acute toxicity) and:

< bioaccumulation test results are below the regional Matrix levels for cadmium, mercury, total PCBS, and total DDT, and are below the regional Category I values for dioxin; and
< bioaccumulation test results for the other bioaccumulative chemicals of concern identified in USEPA/CENAN (1992) do not indicate a potential for undesirable effects using conservative assessment techniques.

Sediments that meet this Category I definition are suitable for placement at the HARS as Remediation Material as they will improve sediment conditions at the HARS to reduce impacts to acceptable levels in accordance with 40 CFR Section 228.11(c). Sediments that do not meet this definition are not suitable for placement at the HARS.

V. RESULTS OF EVALUATION OF THE MATERIAL

A. Evaluation of the liquid phase

The liquid phase of the material was evaluated for compliance with Sections 227.6(c)(1) and 227.27(a). There are applicable marine water quality criteria for constituents in the material, including listed constituents, and the applicable marine water quality criteria would not be exceeded after initial mixing. In addition, liquid phase bioassays run as part of the suspended particulate phase on three appropriate sensitive marine organisms, show that after initial mixing (as determined under 40 CFR 227.29(a)(2)), the liquid phase of the material would not exceed a toxicity threshold of 0.01 of a concentration shown to be acutely toxic to appropriate sensitive marine organisms. Accordingly, it is concluded that the liquid phase of the material would be in compliance with 40 CFR 227.6(c)(1) and 227.27(a). The specific test results and technical analysis of the data underlying this conclusion are described and evaluated in CENAN (2000).

B. Evaluation of the suspended particulate phase

The suspended particulate phase of the material was evaluated for compliance with Sections 227.6(c)(2) and 227.27(b). Bioassay testing of the suspended particulate phase of the material has been conducted using three appropriate sensitive marine organisms: inland silversides (*Menidia beryllina*), mysid shrimp (*Mysidopsis bahia*), and blue mussel (*Mytilus galloprovincialis*). That information shows that when placed in the HARS and after initial mixing (as determined under 40 CFR 227.29(a)(2)), the suspended particulate phase of this material would not exceed a toxicity threshold of

0.01 of a concentration shown to be acutely toxic in the laboratory bioassays, and thus would not result in significant mortality. The material may be discharged on either incoming or outgoing tides. However discharge may occur only at distances of at least 2,605 feet from the northern and southern boundaries and at least 2,463 feet from eastern and western boundaries of the HARS, independant of the direction of tidal transport at the time of placement. The specific test results and technical analysis of the data underlying this conclusion are described in CENAN (2000). The factor of 0.01 was applied to ensure that there would be no significant adverse sublethal effects. Moreover, the fact that after placement, the suspended particulate phase would only exist in the environment for a short time, means the suspended particulate phase would not cause significant undesirable effects, including the possibility of danger associated with bioaccumulation, since these impacts require long exposure durations (see USEPA, 1994). Accordingly, it is concluded that the suspended phase of the material would be in compliance with 40 CFR 227.6(c)(2) and 227.27(b).

C. Evaluation of the solid phase

The solid phase of the material was evaluated for compliance with Sections 227.6(c)(3) and 227.27(b). This evaluation was made using the results of two specific types of evaluations on the solid phase of the material, one focusing on the acute (10-day) toxicity of the material, and the other focusing on the potential for the material to cause significant adverse effects due to bioaccumulation. Both types of tests used appropriate sensitive benthic marine organisms according to procedures approved by USEPA and the USACE. The following sections address the results of those tests and further analyze compliance with the regulatory criteria of Sections 227.6(c)(3), 227.27(b), and 228.15 and with EPA Region 2/CENAN guidance.

1. Solid phase toxicity evaluation

Ten-day toxicity tests were conducted on project materials using mysids (*M. bahia*) and amphipods (*Ampelisca abdita*), which are appropriate sensitive benthic marine organisms. These organisms are good predictors of adverse effects to benthic marine communities (see, USEPA, 1996a). The mortality in project sediments did not exceed mortality in the reference sediment by 10% for mysid shrimp or 20% for amphipods and was not statistically greater than reference for either mysids or amphipods. These results show that the solid phase of the material would not cause significant mortality and meets the solid phase toxicity criteria of Sections 227.6 and 227.27.

2. Solid phase bioaccumulation evaluation

USEPA/USACE (1991) describes an approved process of evaluating bioaccumulation potential using comparative analysis of project sediment bioaccumulation to reference sediment bioaccumulation, FDA Action levels and evaluation of eight additional factors for assessing the significance of bioaccumulation. These factors are:

- number of species in which bioaccumulation from the dredged material is statistically greater than bioaccumulation from the reference material

- number of contaminants for which bioaccumulation from the dredged material is statistically greater than bioaccumulation from the reference material

- magnitude by which bioaccumulation from the dredged material exceeds bioaccumulation from the reference material

- toxicological importance of the contaminants whose bioaccumulation from the dredged material exceeds that from the reference material

- phylogenetic diversity of the species in which bioaccumulation from the dredged material statistically exceeds that from the reference material

- propensity for the contaminants with statistically significant bioaccumulation to biomagnify within aquatic food webs

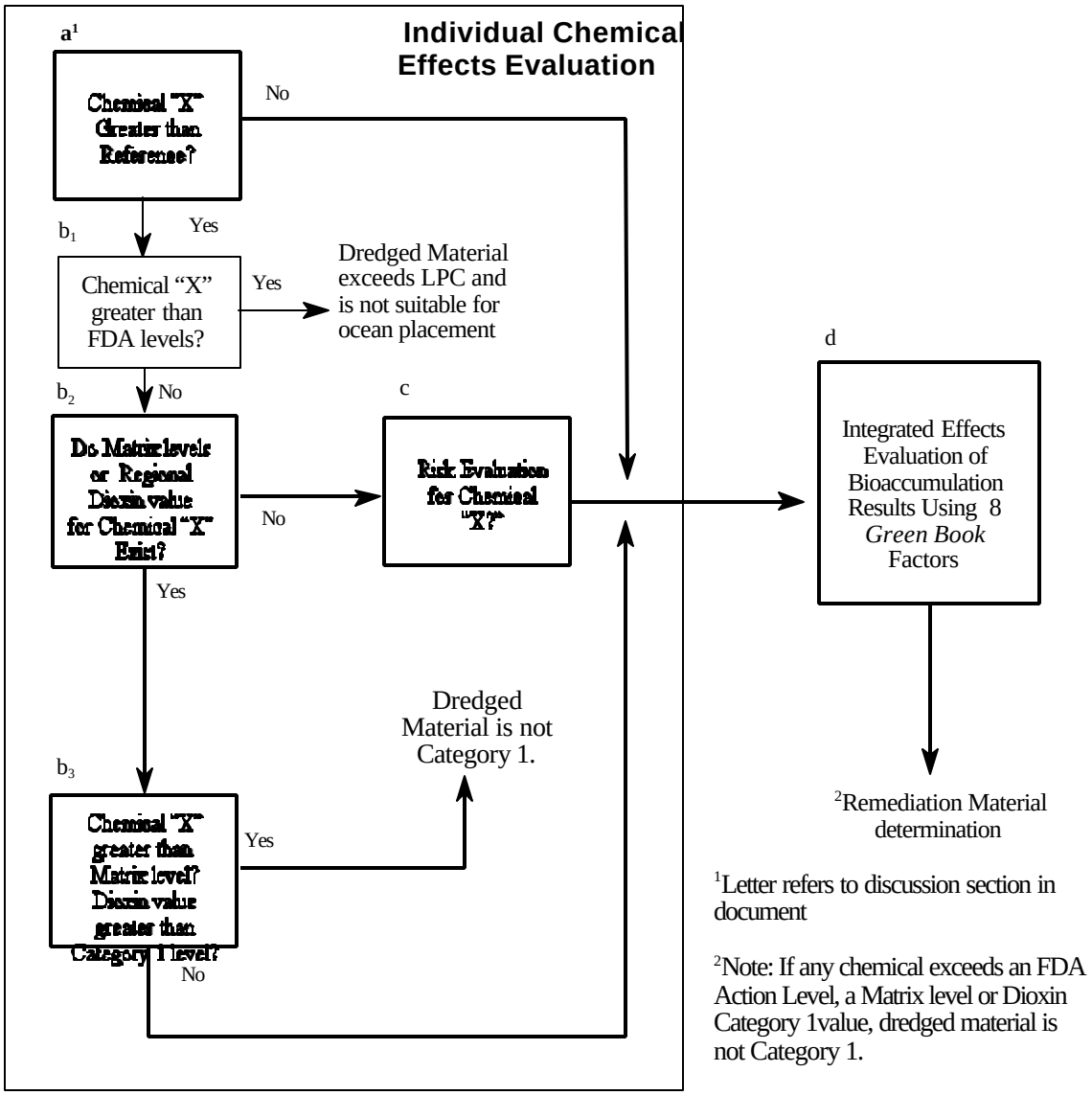
- magnitude of toxicity and number and phylogenetic diversity of species exhibiting greater mortality in the dredged material than in the reference material

- magnitude by which contaminants whose bioaccumulation from the dredged material exceeds that from the reference material also exceed the concentrations found in comparable species living in the vicinity of the proposed site

In following this guidance, USEPA Region 2 and CENAN used a framework (described in Figure 1) for evaluating project sediment bioaccumulation results. As shown in Figure 1, this process involves four consecutive evaluations. In the first three evaluations, the project sediment bioaccumulation test results for each compound of concern are sequentially compared to:
a) reference test results; b_1) FDA Action levels; b_2) Regional Matrix levels; b_3) regional dioxin values; and, c) general risk-based evaluations (including comparison to background tissue concentrations). If these evaluations show that the project sediment does not exceed the reference test results in step (a), the FDA levels in step (b_1), and the Regional Matrix levels/dioxin Category I values in steps (b_2 to b_3) for a particular compound, this indicates that the placement of the material would not result in adverse effects due to that chemical, and there is no need to further evaluate that individual chemical in the next step. Markings in columns 5 or 7 of Table 1 indicate where project test results were statistically greater than the reference levels for the clam or the worm. If any species are marked for a particular compound, the evaluation will proceed to the next step. General risk-based evaluations are conducted in step (c) for compounds not resolved in steps (a) or (b_1 to b_3). The fourth evaluation (d) uses all the information and results of the individual chemical evaluations (particularly as these results relate to the

eight Green Book factors listed above), to evaluate the solid phase of the dredged material as a whole. These evaluations for this project are discussed below in the order described in Figure 1.

Figure 3. **EPA Region 2/CENAN Framework for Evaluating Bioaccumulation Test Results for the Federal Navigation Project No. 36, Buttermilk Channel.**



Bioaccumulation tests were conducted on the solid phase of the project material for contaminants of concern identified in USEPA/CENAN (1992) and the project sampling plan (CENAN, 1999) using two appropriate sensitive benthic marine organisms, sand worm (*Nereis virens*) and bent-nosed clam (*Macoma nasuta*). These species are considered to be good representatives of the phylogenetically diverse base of the marine food chain. Contaminants of concern were identified for the regional testing manual from the NY/NJ Harbor Estuary Program Toxics Characterization report (Squibb, *et al.*, 1991). That report was prepared as part of development of the Harbor Estuary Program in order to identify and characterize contaminants in Harbor sediments. Those compounds with the potential to

bioaccumulate (K_{ow} of approximately 4 or greater) are included on the testing list and evaluated by use of bioaccumulation tests when expected to be present in project sediments based upon the location of contaminant inputs and results of previous sediment sampling. The bioaccumulation test results were used in evaluating the potential impacts of the material. The determination is that the combined results of the toxicity and bioaccumulation tests indicated that the material meets the criteria of Sections 227.6(c)(3) and 227.27(b) and
228.15(d)(6)(v)(A) of the Regulations, and that the material is suitable for placement at the HARS as Remediation Material.

a. *Comparison of Bioaccumulation Test Results to Reference Sediment Test Results*

Concentrations of contaminants in tissues of organisms exposed for 28 days to project sediments were compared to concentrations in tissues of organisms exposed for 28 days to reference sediment. Reference sediment serves as a point of comparison to identify potential effects of contaminants in the dredged material (USEPA/USACE, 1991). In essence, exposing test organisms to this sediment allows for the prediction of contaminant levels that would result in the test organisms were they "in the wild" at the area from which the reference sediment was taken. The tissue concentrations in two species of appropriate sensitive benthic marine organisms resulting from 28-day exposure to project sediments is compared to the tissue concentrations in the same species of organisms resulting from 28-day exposure to reference sediment. In order to make a statistically valid determination that the project sediment does/does not cause greater bioaccumulation than the reference sediment, several sub-samples of the dredged material and reference are run; these separate sub-samples are called replicates. A mean can then be calculated with a standard deviation for each sediment (i.e., Buttermilk Channel sediments, and Reference sediment). The means and standard deviations are compared using a standard statistical approach, and a determination is made, with 95 percent confidence, that there is or is not a true difference between the test and reference sediments. A statistical analysis is merely a quantification of the variability between the test and reference data, and a measure of the probability of the difference being real. Throughout this memorandum, statements regarding project sediment having "greater" or "less" bioaccumulation are referring to calculated differences which are statistically significant at the 95% confidence level. To be environmentally conservative, test values which were below detection levels were estimated at very conservative levels for purposes of statistical comparisons (USEPA/CENAN, 1997)

The reference sediment used for this project was collected at the Reference Site, in an area of clean, sandy sediments located in the New York Bight near the HARS, where the sediments are unaffected by prior dredged material disposal (see reference values in Table 1, Columns 2 and 3). When bioaccumulation in organisms exposed to project sediments is not greater than bioaccumulation in organisms exposed to appropriate reference sediments, this means that placement of the material would not result in bioaccumulation above that found to occur in the "clean" reference sediment. Accordingly, such material would not result in bioaccumulation that would cause unreasonable degradation of the environment or human health, or significant adverse effects. In cases where bioaccumulation levels are statistically greater (at the 95% confidence limit) than in the reference, further evaluation for potential

effects is warranted. A statistically significant difference between test and reference bioaccumulation is not itself a quantitative prediction that an impact would occur in the field, nor is it related to any cause and effect. A key to understanding bioaccumulation and potential adverse impacts is that bioaccumulation is a phenomenon and does not necessarily result in an effect. In addition, depending upon the exposure (concentration and duration), bioaccumulation may cause no harm. On the other hand, as exposure and subsequent bioaccumulation increases, the potential for adverse effects increases.

The following text summarizes the test results comparing bioaccumulation from the project sediments to that in the reference sediments. (Contaminants for which bioaccumulation from the dredged material was statistically greater than the reference in the clam and/or the worm are indicated by a mark in columns 5 and/or 7 for that compound in Table 1.)

Metals

- Of the nine metals tested, six in the clam and four in the worm were bioaccumulated greater in the project sediment than the reference. Cadmium and mercury are the only metals that are listed constituents in Section 227.6(a). Cadmium bioaccumulated greater in project sediment than reference only in worm tissue. Mercury bioaccumulated greater in the project sediment than the reference only in the clam tissue.

Pesticides

- Of the 15 pesticides (including DDT congeners) tested, seven in the clam and five in the worm were bioaccumulated greater in the project sediment than the reference.

Industrial Chemicals

- Total PCBs and 1,4-dichlorobenzene bioaccumulated greater than reference in both the clam and the worm.

Dioxins

- Of dioxin congeners (including 2,3,7,8-TCDD and 16 other dioxin congeners), seven in the clam and four in the worm were bioaccumulated greater in the project sediment than the reference.

PAHs

- Of the 16 PAHs tested, 15 in the clam and 15 in the worm were bioaccumulated greater in the project sediment than the reference.

For all metals, the magnitude of exceedance is less than 10 times the reference. For the remaining contaminants that bioaccumulated from project sediment to greater concentrations than the reference, 11 PAHs, four pesticides, total PCBs, and 2,3,7,8-TCDF exceeded the reference greater than ten times in the clam and/or the worm. Exceedance of the reference values is common when reference values are very low or “non-detect,” as here. In such cases the potential for the actual tissue

concentration to be related to an effect on the organism or the food chain (including human health) is further evaluated.

b. ***Comparison to FDA Action levels, Regional Matrix Levels and Dioxin Values***

i.) Comparison to FDA Action levels (b_1)

There are FDA Action levels for seven compounds (aldrin, dieldrin, a-chlordane, heptachlor, heptachlor epoxide, PCBs, and mercury). None of the contaminants for which there are FDA Action levels exceed such thresholds in the tissues of organisms exposed to project sediments for 28 days (see also Table 1). The source of FDA Action levels is described in USEPA/USACE (1991). Table 1, Column 18, identifies the relevant FDA Action levels.

Exceedance of an FDA Action level results in a conclusion that the placement of the dredged material would result in significant adverse effects. No contaminants for which there are FDA Action level exceeded any such level in either the clam or the worm tissues.

ii) Comparison to Regional Matrix Levels (b_2)

There are regional Matrix levels for four compounds (cadmium, mercury, PCBs and total DDT). The source of regional Matrix levels is described in USACE (1981). Table 1, Column 20, identifies the relevant regional Matrix levels. Bioaccumulation results that exceed the regional Matrix level indicate that the sediment is not Category I under USEPA Region 2/CENAN guidance. Total DDT, total PCB, cadmium, and mercury did not bioaccumulate in either clam or worm tissue at concentrations exceeding the Matrix level.

iii) Comparison to Regional Dioxin Values (b_3)

Regional dioxin values are intended for comparison to the results of 28-day bioaccumulation test results, and the source and use of the values are described in the USEPA (1997a). Table 1, Sheet B, Column 20, identifies the relevant regional dioxin values. Twenty-eight day bioaccumulation test results that equal or exceed the regional Category I dioxin value of 1 pptr for 2,3,7,8-TCDD (or 4.5 pptr using the TEQ approach described in USEPA (1997a) for the non 2,3,7,8-TCDD congeners) indicate that the sediment is not Remediation Material under USEPA Region 2/CENAN guidance. Bioaccumulation test results were below the regional Category I dioxin values for both the worm and the clam.

iv) Steady State Considerations for Matrix Compounds

When the end point to which the test data is compared potentially represents a steady-state level, rather than a 28-day level, consideration may need to be given to whether the 28-day test results are representative of bioaccumulation levels that could be expected to occur in the field after placement.

The literature was reviewed to determine the degree to which the test results reached steady state, as appropriate. The relevance of adjusting project data to steady state for comparison to regional Matrix levels is discussed below.

PCBs

To assess the rate of bioaccumulation of PCBs and other compounds, Rubinstein, *et al*. (1990) and Pruell, *et al*. (1993) exposed three species of organisms, the grass shrimp *Palaemonetes pugio*; the sandworm *Nereis virens*; and the clam *Macoma nasuta*, to sediments collected from the Passaic River, N.J. Sub-samples of the exposed organisms were removed on various days into the study including days 0, 10, 28, 42, 84, and 180. For the clam tissue, the variance in the concentrations on day 28 and day 84 (by which point the maximum concentration had been reached) overlap, thus indicating that the two are not statistically different and the bioaccumulation on day 28 is at or very close to steady-state. Thus, the clam bioaccumulation for the project sediments using 28-day exposures is acceptable for use as steady-state tissue levels, and was below the Matrix level for total PCB. For the worm tissue, variances for days 28 and 180 do not overlap, thus indicating that steady-state was probably not reached in 28 days, although the variance in the data makes it difficult to quantify a real difference. However, if the means for days 28 and 180 from Rubinstein *et al*. (1990) are compared (approximately 1,750 ng/g (nanograms per gram or parts per billion, ppb) for 28 days, and 3,000 ng/g for 180 days) this indicates approximately 58% of steady-state would have been reached in 28 days. If on this basis the worm project data are conservatively adjusted upward by even a factor of two to calculate a steady-state tissue concentration, the dredged material tissue concentration is still below the Matrix level for PCBs in the worm.

Total DDT

With regard to DDT and its metabolites, the degree to which these compounds reached steady-state was also evaluated. Table 1 contains the project test results for the total DDT, which is the sum of the results for DDT and its metabolites (i.e., DDE and DDD). This level is compared to the Matrix level for total DDT. To assess the rate of bioaccumulation of the DDTs and their metabolites, Lee, *et al.*, (1994) exposed the clam *Macoma nasuta*, to sediments collected from the vicinity of the United Heckathorn Superfund site in Richmond California. The study measured tissue residues and uptake kinetics from exposure to pesticide-contaminated sediments. Results of the study indicate that one parent compound, 4,4-DDT, bioaccumulates much more slowly than 2,4-DDT and the DDT metabolites. The results range from approximately 9 percent of steady state after 28 days for 4,4-DDT, to 55 percent of steady state after 28 days for 2,4-DDT. (Lee, et al., 1994) In the Buttermilk Channel project, the parent compounds (4,4-DDT and 2,4-DDT) did not accumulate to concentrations statistically greater than reference. Metabolites were detected and were statistically greater than the reference in the bioaccumulation test results for both the clam and the worm tissue. In order to calculate a steady-state tissue concentration, based on the above study a factor of 11 was applied to the project data for 4,4-DDT, a factor of three to the project data for 4,4-DDD, and a factor of two for 2,4-DDT and the remaining DDT metabolites, assuming the detection limit represents the amount

present when "not detected." Using these conservative assumptions, the dredged material tissue concentration is below the Matrix level for total DDT in both the worm and the clam.

Cadmium and Mercury

Cadmium and mercury are not regulated in marine organisms as are essential metals, and, thus, no adjustment for steady state is applicable. The Matrix levels for cadmium and mercury, therefore, do not represent "steady state." Bioaccumulation of these metals is affected by many complex factors, and is essentially linear (Dethlefsen, 1978; Giesy, *et al.*, 1980; V-Balogh and Salanka, 1984). Therefore, there are no adjustments that can be made to reproduce "steady state," and so 28-day test results are used to compare to the Matrix levels.

c. Risk-based evaluations

The potential for impacts due to compounds that produced greater bioaccumulation from project sediments than the reference sediments and for which Matrix levels or Regional dioxin values did not exist, was determined using risk-based evaluations. As noted in Table 1 and the previous discussions, for this project PAHs, chlordane, dieldrin, heptachlor epoxide, 1,4-dichlorobenzene, chromium, copper, lead, nickel, and silver fall into that group for the worm and/or clam.

The toxicological significance of this bioaccumulation was evaluated by: i) consideration of steady-state bioaccumulation and food-chain transfer; ii) comparison to background tissue concentrations; iii) consideration of potential ecological effects; and, iv) consideration of potential carcinogenic and non-carcinogenic effects on human health.

i) *Consideration of Steady-State Bioaccumulation and Food-Chain Transfer*

Bioaccumulation tests were conducted using 28-day exposure of appropriate sensitive benthic marine organisms to sediment. As previously discussed, for bioaccumulation evaluations involving comparisons with "steady-state" tissue concentrations (as opposed to evaluations using other 28-day tissue concentrations such as the comparison to reference sediment), it may be necessary to understand the extent to which the organism tissue concentration has reached steady-state. Steady-state may be defined operationally as the lack of any significant difference (ANOVA, alpha = 0.05) among tissue residues taken at three consecutive sampling intervals (Lee, *et al.*, 1989). The 28-day test exposure period was selected as appropriate because most chemicals of concern will reach at least 80% of steady-state in benthic marine organisms within that time frame (Boese and Lee, 1992). For the few chemicals that may not meet steady-state tissue concentrations in 28 days, a factor may be used to adjust the data to steady-state when necessary. In order to better use the tissue concentration results of 28-day bioaccumulation exposure tests to assess the risks posed to the environment from the chemicals requiring further evaluation (see discussion above for the identification of such chemicals), consideration was given to the steady-state concentration of these compounds that could occur in the HARS after

extended periods of time. In addition, the potential movement of these compounds through the food chain was considered and appropriate trophic transfer factors applied to adjust the data accordingly, as described below.

Metals

In general, metals bioaccumulate more rapidly than organics and 28-day tests are sufficient to evaluate potential effects (see USEPA/USACE, 1991), for example, arsenic (Naqvi, *et al*., 1990; Riedel, *et al.*, 1987; Oladimeji, *et al.*, 1984).

Trophic transfer of most metals is not sufficient to qualify as biomagnification (Brown and Neff, 1993). The lack of observed biomagnification for such metals as chromium, copper, lead, nickel, and silver is the result of incomplete absorption of metals across the gut, rapid excretion, and dilution in muscle, which represents a large part of the total body weight of most marine animals (Fowler, 1982; Suedel *et al.*, 1994). For purposes of conducting the human health and ecological evaluations below, a conservative trophic transfer coefficient equal to one will be used for these non-biomagnifying metals (Suedel *et al*., 1994 and references cited therein).

Pesticides and Industrial Chemicals

Uptake of non-polar organic contaminants from food is highly dependent on its hydrophobicity, a property measured by the octanol/water partition coefficient, K_{ow}. The higher the value of K_{ow}, the longer it takes to reach steady-state in benthic marine organisms. For the organochlorine compounds aldrin, dieldrin, heptachlor, trans nonachlor, and alpha-chlordane that have log $K_{ow} > 6$, it is possible that steady-state was not reached within 28 days. Information contained in Boese and Lee (1992) indicates that 28-day bioaccumulation tests for these chemicals achieve at least 50% of steady-state for aldrin, dieldrin, and a-chlordane. The remaining compounds, heptachlor and trans nonachlor, were not addressed in Boese and Lee (1992), however estimates of the fraction of steady state achieved after 28-days can be calculated using the equations contained in McFarland (1995). Results of such calculations indicate heptachlor and trans nonachlor reach approximately 50% of steady state after 28 days. Comparison of the project data for these compounds with the effects data discussed below after using an appropriate factor (a factor of two) to adjust the project data to an appropriate steady-state concentration, indicates that project sediment results would be below conservative ecological and human effects levels. Calculating the fraction of steady state achieved for the remaining pesticides with log $K_{ow}<6$ (i.e., heptachlor epoxide) and for 1,4-dichlorobenzene ($K_{ow}=3.44$) indicates that these compounds reach steady state within 28 days, so no adjustment is necessary.

The potential for these chemicals to biomagnify was also evaluated. Although organic contaminants with values of log $K_{ow} > 4$ tend to biomagnify in the marine food chain, studies (USACE, 1995) have shown that this is not true for higher molecular weight compounds such as the most highly chlorinated PCBs or for easily metabolized compounds such as PAHs. Those organic compounds which are not

efficiently excreted, such as certain pesticides (including dieldrin, a-chlordane, and heptachlor epoxide), can biomagnify in the food chain. For the organic constituents with a potential to biomagnify in the marine food chain, trophic transfer factors were calculated, using the approach described by Gobas (1993). The values are summarized in Attachment B. These factors, which ranged from 1.4 to 2.3, were taken into account in assessing potential human health and ecological risk effects of these compounds, as discussed below.

PAHs

The time required for a given PAH to attain a steady-state concentration following exposure to bedded sediments (t_{ss}) is determined primarily by the log K_{ow} of the compound in question (McFarland, 1995; Meador, *et al.*, 1995). Meador, *et al.*, (1995) reviewed nine studies that investigated the attainment of steady-state tissue concentrations of PAHs by various marine invertebrates. In each case, tissue concentrations approached steady-state within several days to two weeks after initiating exposure to both low molecular weight PAHs and high molecular weight PAHs. McFarland (1995) estimated the time to steady-state (t_{ss}) for 15 PAHs based on their hydrophobicity. The t_{ss} values ranged from 3.5 to 326 days. The estimated steady-state concentration of the sum total of the 15 PAHs analyzed by McFarland for sediments collected from typical harbor areas revealed that the mean concentration attained after 28-day bioaccumulation tests was approximately 86% of steady-state. McFarland (1995) concluded that 28-day tests are likely to reflect steady-state. However, even using the conservative approach of adjusting the data to calculate steady-state for the individual PAHs in the project based on McFarland (1995) (using a factor of one, two, or three, as indicated) and summing the results, the project data would still fall below the effects levels as discussed below.

With regard to the potential for biomagnification of PAHs, feeding studies show that assimilation rates from ingested food are extremely low, e.g., more than 98% of the target contaminant remained in an undigested form in fish gut 48 hours after feeding squid containing radio-labeled benzo[a]pyrene to young cod (Corner, *et al.*, 1976) and juvenile Atlantic herring (Whittle, *et al.*, 1977) . PAH metabolites are also transferred through the marine food chain; however, they are absorbed even less efficiently than their parent compounds (McElroy and Sisson, 1989; McElroy, *et al.*, 1991). Up to 99% of the PAH compounds taken up by fish are metabolized and excreted into bile, the usual elimination mode, within 24 hours of uptake (Varanasi, *et al.*, 1989). Similar results are described in Brown and Neff (1993) who evaluated various studies describing trophic transfer. The studies cited in Brown and Neff (1993) indicate a trophic transfer rate for BaP from invertebrates to fish of between 0.02 and 0.23 times the concentration in the ingested invertebrates (Corner, *et al*., 1976, O'Connor, *et al*., 1988, McElroy, *et al*., 1991). This was taken into account when assessing the ecological and human health effects of the project material as discussed below.

ii) *Comparison of Test Results to Background Tissue Concentrations*

Where data regarding tissue levels of organisms living in the general area of the HARS are available ("background levels"), it is useful to compare those levels with the test levels as part of the risk

evaluation (Figure 1, Box c). However, this comparison is not, by itself, definitive. When bioaccumulation in organisms exposed to project sediments is not greater than tissue concentrations in organisms from the vicinity of the remediation site (the background levels), this means that placement of the material would not result in bioaccumulation above existing ambient levels in the general area and thus does not have a potential to cause undesirable effects. When bioaccumulation in organisms exposed to project sediments is greater than these levels, it may or may not be predictive of adverse effects (e.g., it may reflect extremely low "background" levels). Depending on the exposure (concentration and duration), bioaccumulation may cause no harm. However, as exposure increases, the potential for adverse effects increases.

Organisms collected from a broad area of the sea floor in the vicinity of (but not inside) the HARS have been collected and analyzed for tissue concentration for bioaccumulative contaminants of concern (Charles and Muramoto, 1990; USACE, 1994; USEPA, 1996f; USEPA, 1997b). These field-generated bioaccumulation results provide a measure of the tissue residues for organisms living outside the HARS. Table 1, Columns 16 and 17 summarize the most recent background data. For clam background, data were collected only for the following constituents: all PAHs, aldrin, two DDT compounds, PCBs, and seven of the nine metals analyzed. Where background values exist, the following constituents exceeding background are identified: nine PAHs exceeded background levels in the clam, six PAHs exceed worm background values. Total PAHs exceeded background levels for both clam and worm tissue. Dieldrin, a-chlordane, total residual chlordane/heptachlor, 4,4-DDD, 2,4-DDE, total DDTs, and total PCBs exceeded background levels for the worm, and 4,4-DDE exceeded background levels for the clam.

iii) *Consideration of Potential Ecological Effects*

A review of scientific information was also done to further evaluate the test results with respect to potential ecological impacts for the chemicals requiring further evaluation (above reference and for which there is no Matrix level or dioxin value).

Metals, Pesticides, and Industrial Chemicals

The potential for ecological effects from the bioaccumulation over reference of dieldrin, total chlordane, 1,4-dichlorobenzene, cadmium, chromium, copper, lead, nickel, and silver was evaluated by comparing to a Water Quality Criterion Tissue Level (WQCTL). The WQCTL is calculated by multiplying the Clean Water Act Section 304(a)(1) Federal water quality criterion chronic value (CV) for the chemical by the empirically determined bioconcentration factor (BCF) for the chemical for a representative marine organism (Lee, *et al.*, 1989). A BCF is the ratio of the concentration of a contaminant in an organism to the concentration of the contaminant in water. Thus, the WQCTL represents the tissue concentration that would be expected in an organism exposed to water containing the chemical at the CV concentration. This level is set to protect 95% of all tested organisms included in the water quality criterion database, thus representing a conservative level of protection (USEPA, 1985b). Table 1 lists the calculated WQCTLs. Sources of CVs and BCFs are USEPA ambient water quality criteria

documents (USEPA 1980b, 1980c, 1980d, 1980e, 1980f, 1984a, 1984b, 1985a, 1985c, 1986, 1987b and 1992a) and Calabrese (1984)(for silver). Calculations are shown in attachment A. None of the WQCTLs were exceeded. Therefore, these bioaccumulation test results do not indicate a potential for undesirable ecological effects.

PAHs

For PAHs, a more definitive method is available for evaluating the potential ecological effects. This method makes use of a direct comparison of total PAH tissue residues and the Critical Body Residue (CBR). This approach is supported by a review of the scientific literature. The CBR approach described by McCarty (1991) was used to evaluate the potential impacts of total PAHs accumulated in the dredged material bioaccumulation test organisms. CBRs are concentrations of chemical residues in organisms which elicit a deleterious biological response associated with narcosis, which is the primary non-cancer effect of PAHs. Narcotic responses measured can be acute (e.g., immobilization or death) or chronic endpoints (e.g., reduced reproduction, fecundity or growth). CBRs are represented as the ratio of the mass of toxicant to the mass of the organism, such as millimoles or micrograms of toxicant per kilogram (mmole or ug/kg) of organism. For the narcosis endpoint, each molecule of individual PAH congeners is generally equipotent, thus the total PAH concentration is compared to the CBR. For example, a 400 ppb dose of naphthalene would elicit a similar toxicity response as 400 ppb of fluorene; if both chemicals are present together at these concentrations, then the dose would equal 800 ppb (see Appendix for Table 1).

As shown in Table 1, total PAH levels in tissues from the dredged material bioaccumulation test were below levels at which chronic adverse effects might be expected from a narcotic mode of action in sensitive aquatic organisms (i.e., fish) as estimated by the CBR.

Effects of Mutagenic, Carcinogenic and Teratogenic PAHs. Applying the uncertainty factor (UF) of 10 and a trophic transfer factor of 0.1 described in the Appendix for Table 1, to the no-effects level for BaP calculated from Hannah, *et al.* (1982), as discussed in the Appendix for Table 1 (8,021 ppb) results in a no-effect level for BaP of approximately 8,000 ppb in benthic tissue, which is considerably greater than the highest tissue concentration of BaP found in the project bioaccumulation test results (approx. 29.8 ppb). Even when applying the more conservative steady-state factors for BaP and the other carcinogenic PAHs derived from McFarland (1995), as identified above, the calculated concentrations (59.63 ppb for BaP only and 78.44 ppb for total BaP equivalents) are still below the no-effects level; the project tissue concentrations would still be below this no-effect level if the higher trophic transfer factor (0.23) reported by McElroy, *et al*. (1991) was used. Therefore, the most relevant aquatic effects information reviewed indicates that the highest tissue levels accumulated in the dredged material bioaccumulation tests are below the no-effect level.

Another study that was reviewed considered the carcinogenicity of BaP in rainbow trout resulting from embryo microinjection (Black, *et al*., 1988). A statistically significant number of liver neoplasms was found at a concentration of approximately 200,000 ppb, with non-significant effects at up to one half that concentration. Therefore, using the above across-species UF of 10 and trophic transfer factor of 0.1 results in an aquatic no-effect level of 100,000 ppb. Since this is several orders of magnitude above the highest tissue concentration of BaP for this project, as described above (and even the highest BaP-equivalent levels for human health, as discussed above), this provides additional support for a finding that the test results do not indicate a potential for undesirable effects to the marine environment due to mutagenic, carcinogenic or teratogenic contaminants.

Hall and Oris (1991) reported on experiments that exposed fathead minnows to anthracene during long-term exposures and observed adverse effects on reproduction. The paper reported that a concentration of anthracene in the tissue of the egg in the range of 3,750 to 8,000 ppb resulted in no significant effects on egg hatching or survivorship. Using the same approach for accounting for species-to-species uncertainty and food chain transfer described above and in the Appendix for Table 1, yields a conservative benthic tissue level of 3,750 ppb. Anthracene tissue concentrations from the project bioaccumulation tests are well below this level.

iv) *Consideration of Potential Carcinogenic and Non-carcinogenic Effects on Human Health*

Human health effects screening levels were developed for those chemicals requiring further evaluation with risk-based methods using conservative estimates of exposure to assess whether these contaminants would accumulate to levels in fish and shellfish that could lead to significant adverse effects to humans. The approach assessed consumption of fish and shellfish to derive conservative estimates of contaminant concentrations in benthic tissue protective of human health using USEPA standard risk-assessment assumptions and the process described in the Appendix for Table 1. Table 1, Column 14 lists conservative human cancer protection levels in benthic organisms for the chemicals which are known or suspected carcinogens that would lead to a human cancer risk level of 10^{-4}. When the bioaccumulation test results for those chemicals are adjusted for steady-state (as previously described), the results are below the human cancer protection levels in Table 1.

Since the analysis used conservative methods, the result represents conservative estimates of risk, or what are in effect plausible upper-bound estimates. Thus, the true risk is highly unlikely to be greater than estimated and could be much lower. None of the human health cancer protection levels were exceeded in the bioaccumulation test results.

The potential for non-cancer impacts can be expressed as a hazard quotient (HQ), which is the ratio of the average daily intake divided by the toxicological reference dose for the chemical. If the HQ is less than unity (i.e., 1), an adverse noncarcinogenic effect is highly unlikely to occur. If the HQ exceeds unity, an adverse health impact may occur. The higher the HQ, the more likely that an adverse noncarcinogenic effect will occur as a result of exposure to the contaminant in the dredged material after placement. Table 1, Column 15 includes the noncancer protection levels in benthic organisms for the

chemicals requiring further analysis that are known to cause, or suspected of causing, non-carcinogenic effects, that would result in a human HQ equal to unity. Those numbers were derived using the conservative assumptions and source materials described in the Appendix for Table 1. The concentrations of the chemicals requiring further evaluation were below the non-cancer protection level.

d. Evaluation of Solid Phase Bioaccumulation Results for Dredged Material as a Whole

The evaluation of the testing results performed above indicates that the material does not have a potential to cause undesirable effects to aquatic marine biota due to chronic adverse effects including such effects due to mutagenic, carcinogenic, or teratogenic contaminants, or to human health due to cancer or non-cancer effects from the individual contaminants. That evaluation includes the information relevant to the eight factors identified in the Green Book for assessing bioaccumulation test results (USEPA/USACE, 1991). As a final and additional step in the evaluative process, however, it is appropriate to go beyond assessing the individual test results in order to look at the results as a whole so as to provide an opportunity for an integrated assessment of the individual test results (Figure 1, Box d). For example, if a number of the individual bioaccumulation test results were only marginally at or below the relevant levels of concern, it is appropriate to consider this and the other relevant factors to evaluate whether, taken as a whole, the material is unsuitable for placement at the HARS, even though no single individual test result would indicate that outcome.

As indicated above, the following chemicals of concern were bioaccumulated above reference for the clam and/or the worm: PAHS, chlordane, dieldrin, DDTs, total PCB, 1,4-dichlorobenzene, copper, lead, nickel, 2,3,7,8-TCDD, 1,2,3,4,6,7,8-HpCDD, OCDD, and 2,3,7,8-TCDF bioaccumulated in both the clam and worm; heptachlor epoxide, chromium, mercury, silver, 1,2,3,6,7,8-HxCDD, 1,2,3,4,6,7,8-HpCDF, and OCDF bioaccumulated only in the clam; and cadmium bioaccumulated only in the worm. In the case of those contaminants with test results exceeding reference, and which have regional Matrix levels or dioxin values, none exceeded the relevant Category I value. For the non-Matrix or non-dioxin contaminants with test results that exceeded reference levels, except for PAHs, only dieldrin, and chlordane bioaccumulated in the project sediments greater than background levels. Total PAHs bioaccumulated to concentrations slightly exceeding background levels in both clam and worm tissue. Dieldrin bioaccumulated to greater than 10 times background in the worm, all others were no more than three times background in the worm. Although some of the contaminants that bioaccumulated in the tests can be toxicologically important, in no case did they accumulate to toxicologically important concentrations, even when conservative assumptions were used to evaluate the test results exceeding reference, as described above. Dieldrin, chlordane, heptachlor epoxide, PAHS, 1,4-dichlorobenzene, cadmium, chromium, copper, lead, nickel, and silver exhibited bioaccumulation test results above reference which were all below the acceptable human health risk range and acceptable aquatic effects range using conservative approaches and analyses as described above to evaluate those test results. Thus, an evaluation of the solid phase bioaccumulation test results

for the dredged material as a whole considering the factors in the Green Book (Figure 1, Box d) would not indicate a different outcome than that shown by the individual test results themselves; i.e., that the material does not have the potential to cause undesirable effects due to bioaccumulation.

Taking into account all of the above information, it is determined that this material will not cause undesirable effects due to bioaccumulation as a result of the presence of individual chemicals or of the solid phase of the dredged material as a whole. Therefore, it is concluded that the solid phase of the material proposed for HARS placement is classified as Category I under USEPA Region 2/CENAN general guidance and meets the requirements of 40 CFR §227.6(c)(3), 227.27(b), and 228.15(d)(6).

VI. OVERALL CONCLUSION ON THE PROPOSED PROJECT

Based upon this review of the results of testing of the sediments proposed for dredging and ocean placement from Buttermilk Channel, the material meets the criteria for acceptability for ocean placement as described in Sections 227.6, and 227.27 of the Regulations, is Category I under USEPA Region 2/CENAN guidance, and is suitable for placement at the HARS under Section 228.15(d)(6) as Remediation Material.

Placement of this material at the HARS will serve to reduce impacts to acceptable levels and improve benthic conditions. Sediments in the HARS have been found to be acutely toxic to sensitive benthic marine organism in laboratory tests, whereas project sediments used in laboratory acute toxicity tests with the same species were determined not to be toxic. Placement of project material over existing toxic sediments would serve to remediate those areas for toxicity. In addition, by covering the existing sediments in the site with this project material, surface dwelling organisms will be exposed to sediments exhibiting Category I qualities (e.g., 2,3,7,8-TCDD bioaccumulation less than 1 pptr) whereas the existing sediments exceed these Category I levels.

Thus, this material meets the requirements for placement at the HARS as Remediation Material as described in 40 CFR Section 228.15 (d)(6).

TABLE 1

Template Version: 4/7/98

Bioaccumulation Table for NY/NJ Harbor Projects, ALL VALUES ARE IN WET WEIGHT

Project Name: Buttermilk Channel FP No. 36

	PROJECT DATA												COMPARISON DATA						
Col. 1	**Col. 2 [11]**	**Col. 3 [11]**	**Col. 4 [11]**	**5**	**Col. 6 [11]**	**7**	**Col. 8**	**Col. 9**	**Col. 10**	**Col. 11**	**Col. 12**	**Col. 13**	**Col. 14**	**Col. 15**	**Col. 16**	**Col. 17**	**Col. 18**	**Col. 19**	**Col. 20**
Sample I.D.	**Reference**	**Reference**	**Test Sed.**	**[1]**	**Test Sed.**	**[1]**	**Conv.Fac.**	**Test Sed.**	**Test Sed.**		**Test Sed.**	**Test Sed.**	**Human Health**	**Human Health**				**Ecological**	
							clam/worm	**SS**	**SS**	**Carcinogenic**	**BaP Tox. Equiv.**	**BaP Tox. Equiv.**	**Cancer (10E-4)**	**Non-Cancer**	**Background**	**Background**	**FDA**	**Non-Specific**	**Regional**
	(clam)	**(worm)**	**(clam)**		**(worm)**		**[2]**	**(clam)**	**(worm)**	**TEF [3]**	**Conc.(clam)[4]**	**Conc.(worm)[4]**	**Level[5]**	**Level (HQ=1)**	**(clam)**	**(worm)**	**Limits [7]**	**Effects Level**	**Matrix [12]**
Compound	**(ug/Kg)**	**(ug/Kg)**	**(ug/Kg)**		**(ug/Kg)**			**(ug/Kg)**	**(ug/Kg)**		**(ug/Kg)**	**(ug/Kg)**	**(ug/Kg)**	**(ug/Kg)[10]**	**(ug/Kg)**	**(ug/Kg)**	**(ug/Kg)**	**(ug/Kg)**	**(ug/Kg)**
PAHs																			
Acenaphthene	**0.34 U**	0.28	2.14	X	1.64	X	1/1	2.14	1.64					8,775,000	**8.1**	**0.5**		[15]	
Acenaphthylene	**0.40 U**	**0.65 U**	2.17	X	0.51	X	1/1	2.17	0.51					[6]	**4.6**	**1.3**		[15]	
Anthracene	0.26	**0.50 U**	11.4	X	0.60	X	1/1	11.45	0.60					43,605,000	**10.0**	**1.6**		3,750	
Fluorene	**0.44 U**	0.17	2.10	X	0.51	X	1/1	2.10	0.51					5,805,000	**7.4**	**0.3**		[15]	
Naphthalene	1.81	1.43	1.92		1.27		1/1	1.92	1.27					[6]	**26.4**	**4.5**		[15]	
Phenanthrene	0.68	0.27	16.0	X	1.33	X	1/1	16.01	1.33					43,605,000	**32.7**	**4.7**		[15]	
Benzo(a)anthracene*	0.35	0.59	40.0	X	2.03	X	2/2	80.06	4.06	0.1	8.006	0.406		[6]	**25.1**	**4.8**		[15]	
Benzo(a)pyrene*	0.46	**0.73 U**	29.8	X	1.89	X	2/2	59.63	3.79	1	59.629	3.787		[6]	**24.2**	**7.6**		8,000 [8]	
Benzo(g,h,i)perylene	**0.33 U**	**1.02 U**	8.69	X	1.05	X	3/3	26.08	3.16					[6]	**22.2**	**7.6**		[15]	
Benzo(b)fluoranthene*	0.67	**0.93 U**	24.1	X	1.87	X	2/2	48.19	3.73	0.1	4.819	0.373		[6]	**40.6**	**16.5**		[15]	
Benzo(k)fluoranthene*	0.41	**1.00 U**	25.8	X	2.12	X	2/2	51.57	4.25	0.01	0.516	0.042		[6]	**81.6**	**5.6**		[15]	
Chrysene*	1.05	0.48	51.9	X	10.8	X	2/2	103.82	21.61	0.001	0.104	0.022		[6]	**29.4**	**6.7**		[15]	
Dibenzo(a,h)anthracene*	**0.37 U**	**0.98 U**	1.78	X	1.74	X	2/2	3.56	3.48	1	3.565	3.479		[6]	**8.5**	**1.1**		[15]	
Fluoranthene	1.84	0.68	71.1	X	23.5	X	1/1	71.11	23.51					5,805,000	**43.8**	**10.6**		[15]	
Indeno(1,2,3-cd)pyrene*	**0.52 U**	**1.05 U**	6.01	X	0.41	X	3/3	18.02	1.23	0.1	1.802	0.123		[6]	**16.6**	**4.4**		[15]	
Pyrene	1.13	0.69	103	X	40.6	X	1/1	102.82	40.64					4,387,000	**51.3**	**26.6**		[15]	
TOTAL PAHs								600.66	115.31		78.440	8.233	2,000 (BaP eqv.)		**432.7**	**104.60**		40,000 [9]	
PESTICIDES																			
Aldrin	**0.00U**	**0.04 U**	**0.18 U**		**0.06 U**		2/2	0.36	0.12				33	167	**0.9**	**0.1**	300	299 [10]	
Dieldrin	0.20	0.75	0.60	X	1.20	X	2/2	1.20	2.40				65	518		**0.1**	300	4.37 [10]	
a-Chlordane	0.05	0.53	0.73	X	0.81	X	2/2	1.46	1.62							**0.7**	300		
Trans nonachlor	**0.02 U**	1.50	0.21		1.59		2/2	0.43	3.19							**0.5**			
Heptachlor	0.08	0.02	0.09		0.04		2/2	0.17	0.08							**0.05**	300		
Heptachlor epoxide	0.21	0.11	0.51	X	0.21		1/1	0.51	0.21							**0.2**	300		
Total Residual Chlordane/ Heptachlor								2.57	5.09				114	135		**1.7**		64# [10]	
Endosulfan I	0.02	**0.07 U**	**0.05 U**		**0.10 U**		1/1	0.05	0.10				[6]			**0.2**			
Endosulfan II	0.06	**0.04 U**	**0.04 U**		**0.05 U**		1/1	0.04	0.05				[6]			**0.1**			
Endosulfan sulfate	0.09	0.13	0.17		0.15		1/1	0.17	0.15				[6]			0.1			
Total Endosulfans								0.26	0.30					87,000		**0.4**		2.86# [10]	
4,4-DDT	0.05	0.16	0.08		0.21		11/11	0.85	2.35						**0.6**	**1.0**			
2,4-DDT	**0.09 U**	**0.07 U**	**0.06 U**		**0.10 U**		2/2	0.12	0.20							**0.4**			
4,4-DDD	0.15	1.14	1.78	X	2.24	X	3/3	5.35	6.72							**3.9**			
2,4-DDD	0.08	0.29	0.46	X	0.50		2/2	0.92	1.01							**1.4**			
4,4-DDE	0.20	0.38	3.45	X	1.75	X	2/2	6.90	3.49						**3.5**	**4.3**			
2,4-DDE	0.03	0.32	1.33	X	0.78	X	2/2	2.66	1.56							**0.1**			
Total DDT								16.81	15.32							**11.1**			**40**
TOTAL PCBs	5.38	47.9	77.5	X	87.7	X	1/2	77.50	175.40						**106.6**	**88.1**	2,000		**100 (clam) 400 (worm)**
1,4-Dichlorobenzene	**0.16 U**	0.16	0.35	X	**0.82 U**	X	1/1	0.35	0.82				60,000	[6]				11,820 [10]	
METALS	**(mg/Kg)**	**(mg/Kg)**	**(mg/Kg)**		**(mg/Kg)**			**(mg/Kg)**	**(mg/Kg)**				**(mg/Kg)**	**(mg/Kg)**	**(mg/Kg)**	**(mg/Kg)**	**(mg/Kg)**	**(mg/Kg)**	**(mg/Kg)**
Arsenic	3.12	2.64	3.08		2.49		1/1	3.08	2.49				[17]	[17]		**4.89**		12.6 [10]	
Cadmium	0.04	0.03	0.04		0.03	X	[16]	0.04	0.03						**1.21**	**0.11**			**0.3**
Chromium (total)	0.22	0.14	0.59	X	0.14		1/1	0.59	0.14				[6]	73	**1.28**	**1.31**		11.8 [10]	
Copper	2.45	0.90	3.26	X	1.21	X	1/1	3.26	1.21				[6]	540	**5.58**	**2.78**		9.6 [10]	
Lead	0.19	0.12	0.96	X	0.16	X	1/1	0.96	0.16				[6]	1.3	**1.41**	**1.64**		11.9 [10]	
Mercury	0.01	0.01	0.02	X	0.01		[16]	0.02	0.01						**0.04**	**0.03**	1		**0.2**
Nickel	0.33	0.09	0.50	X	0.18	X	1/1	0.50	0.18				[6]	290	**1.10**	**0.77**		3.8 [10]	
Silver	0.05	0.01	0.08	X	0.01		1/1	0.08	0.01				[6]	73		**0.15**		1.4 [10]	
Zinc	16.9	8.22	17.0		7.14		1/1	17.03	7.14				[6]	4,400	**11.50**	**20.61**		1,517 [10]	

TABLE 1 (continued) ERR

Bioaccumulation Table for NY/NJ Harbor Projects, ALL VALUES ARE IN WET WEIGHT

Project Name: Buttermilk Channel FP No. 36

	PROJECT DATA												COMPARISON DATA						
Col. 1	**Col. 2**	**Col. 3**	**Col. 4 [11]**	**5**	**Col. 6 [11]**	**7**	**Col. 8**	**Col. 9**	**Col. 10**	**Col. 11**	**Col. 12**	**Col. 13**	**Col. 14**	**Col. 15**	**Col. 16**	**Col. 17**	**Col. 18**	**Col. 19**	**Col. 20**
Sample I.D.	**Reference**	**Reference**	**Test Sed.**	**[1]**	**Test Sed.**	**[1]**	**Conv.Fac.**	**Test Sed.**	**Test Sed.**		**Test Sed.**	**Test Sed.**	**Human Health**	**Human Health**				**Ecological**	**Regional**
							clam/worm	**SS**	**SS**		**TEQ**	**TEQ**	**Cancer (10E-4)**	**Non-Cancer**	**Background**	**Background**	**FDA**	**Non specific**	**Dioxin**
	(clam)	**(worm)**	**(clam)**		**(worm)**		**[2]**	**(clam)**	**(worm)**	**TEF [3]**	**(clam)**	**(worm)**	**Level**	**Level (HQ=1)**	**(clam)**	**(worm)**	**Limits [7]**	**Effects Level**	**Value**
Compound	**(pptr)**	**(pptr)**	**(pptr)**		**(pptr)**			**(pptr)**	**(pptr)**		**(pptr)**	**(pptr)**	**(pptr)**	**(pptr)[10]**	**TEQ (pptr)**	**TEQ (pptr)**	**(pptr)**	**(pptr)**	**(pptr)**
DIOXINS																			
2,3,7,8,-TCDD	**0.06 U**	**0.21 U**	0.42	X	0.40	X				1	0.42	0.401			**1.73**	2.50			**1 [13]**
1,2,3,7,8-PeCDD	**0.09 U**	0.15	**0.07 U**		0.18					0.5	0.035	0.0875				0.41			
1,2,3,4,7,8-HxCDD	**0.04 U**	**0.04 U**	**0.03 U**		**0.04 U**					0.1	0.003	0.004				0.10			
1,2,3,6,7,8-HxCDD	0.05	0.22	0.25	X	0.29					0.1	0.0252	0.0294				0.17			
1,2,3,7,8,9-HxCDD	**0.06 U**	0.08	0.12		**0.07 U**					0.1	0.0119	0.007				0.08			
1,2,3,4,6,7,8-HpCDD	0.38	1.18	2.26	X	1.61	X				0.01	0.02262	0.01608				0.13			
OCDD	1.92	5.58	14.8	X	8.13	X				0.001	0.014804	0.008134				0.11			
2,3,7,8-TCDF	0.10	1.51	1.07	X	2.23	X				0.1	0.107	0.2228				0.27			
1,2,3,7,8-PeCDF	**0.03 U**	0.15	0.21		0.28					0.05	0.0105	0.0138				0.05			
2,3,4,7,8-PeCDF	**0.10 U**	0.22	0.23		0.53					0.5	0.115	0.267				0.39			
1,2,3,4,7,8-HxCDF	0.07	0.16	0.12		0.23					0.1	0.012	0.0232				0.08			
1,2,3,6,7,8-HxCDF	**0.06 U**	**0.12 U**	0.07		**0.12 U**					0.1	0.007	0.012				0.09			
1,2,3,7,8,9-HxCDF	**0.05 U**	**0.10 U**	**0.03 U**		**0.11 U**					0.1	0.003	0.011				0.11			
2,3,4,6,7,8-HxCDF	**0.01 U**	**0.06 U**	**0.01 U**		**0.06 U**					0.1	0.001	0.006				0.08			
1,2,3,4,6,7,8-HpCDF	0.18	0.49	0.81	X	0.74					0.01	0.00806	0.00736				0.04			
1,2,3,4,7,8,9-HpCDF	**0.09 U**	**0.08 U**	**0.09 U**		**0.07 U**					0.01	0.0009	0.0007				0.02			
OCDF	0.18	0.39	0.84	X	0.52					0.001	0.00084	0.00052				0.01			
TEQs - non 2,3,7,8-TCDD	3.34	11.6	20.6		16.1						0.377824	0.716494				2.13			**4.5 [14]**
TEQs (all)	3.40	11.9	20.9		16.5						0.797824	1.117494			**1.73**	4.63			**10 [13]**

FOOTNOTES FOR TABLE 1:

*: Carcinogenic PAHs.

#: Levels represent the conservative level of protection for the sum of the related compounds and their metabolites.

na: Not Available

1. A "X" in this column indicates that the analyte concentration in the test sediment is statistically greater than that of the reference sediment. Means and statistical comparisons were determined using conservative estimates of concentrations for analytes that were below the detection limit (USEPA/CENAN, 1997).

2. Conversion factors from 28-day bioaccumulation results to steady state are obtained from the following sources: for PAH's: from McFarland, 1995; for Aldrin, Dieldrin, Chlordane, DDT, DDD, and DDE: from Lee and Lincroft, et al, 1994; for PCBs: from Pruell, *et al.*, 1993, and Rubinstein, *et al.*; for 1,4-Dichlorobenzene: from de Bruijn, *et al.*; for Endosulfan I, Endosulfan II, Endosulfan Sulfate, Heptachlor and trans nonachlor: from Syracuse Research Corporation, 1996, and McFarland, 1995; for Heptachlor Epoxide: from Veith, *et al.*, 1979.

3. PAH TEFs taken from: USEPA. 1993; Dioxin TEFs taken from: USEPA. 1989.

4. Toxic equivalence for the carcinogenic PAHs are from USEPA (1993).

5. This value represents the 10^{-4} cancer risk level for the carcinogenic PAHs. The total concentration of carcinogenic PAHs is expressed in BaP equivalents (see discussion in the text of the memo).

6. Cancer risk factor or reference dose are not assigned by USEPA in IRIS (USEPA, 1995).

7. FDA limits are from the USEPA/USACE, 1991.

8. This value represents the benthic level expected to result in a no-effect level for possible mutagenic and teratogenic effects in fish from exposure to BaP, which is the most toxic PAH.

9. This value represents the non-specific narcosis effects level (see discussion in Appendix). This value is compared to the sum of all PAHs measured.

10. Calculations are included in the appendix to Table 1.

11. Means of five tissue replicates calculated using conservative estimates where analytes were not detected (USEPA/CENAN, 1992); "U" indicates that all five replicates were not detected.

12. Chemicals for which the bioaccumulation from the dredged material was greater than the reference but less than the Matrix level are indicated by bolding the Matrix level in Column 20.

13. Levels are based on the Regional Dioxin Values.

14. Level is the sum of all dioxin congeners other than 2,3,7,8-TCDD.

15. For this PAH, the no-effect level for possible mutagenic and teratogenic effects in fish is estimated from exposure to BaP, which is the most toxic PAH.

16. Cadmium and mercury do not obey steady state kinetics, therefore, no adjustment is made (see discussion in the text of the memo).

17. Cancer and non-cancer protection levels, based on inorganic arsenic as contained in EPA's IRIS database, are not appropriate for evaluating the potential human health impacts of arsenic bioaccumulation from dredged material, and therefore, are not included in Table 1 (see discussion in Appendix to Table 1).

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Appendix for Table 1

I. CONSIDERATION OF ECOLOGICAL EFFECTS

A. Potential for ecological effects based on Water Quality Criteria (Column 19)

The potential for ecological impacts due to bioaccumulation of several compounds of concern was evaluated by calculating a Water Quality Criterion Tissue Level (WQCTL). The WQCTL is calculated by multiplying the Clean Water Act Section 304(a)(1) Federal water quality criterion chronic value (CV) for the chemical by the empirically determined bioconcentration factor (BCF) for the chemical for a representative marine organism (Lee, *et al.*, 1989). A BCF is the ratio of the concentration of a contaminant in an organism to the concentration of the contaminant in water. Thus, the WQCTL represents the tissue concentration that would be expected in an organism exposed to water containing the chemical at the CV concentration. This level is set to protect 95% of all tested organisms included in the water quality criterion database, thus representing a conservative level of protection (USEPA, 1985b). Table 1 lists the calculated WQCTLs. Sources of CVs and BCFs are from USEPA ambient water quality criteria documents (USEPA 1980b, 1980c, 1980d, 1980e, 1980f, 1984a, 1984b, 1985a, 1985c, 1986, 1987b and 1992a) and Calabrese (1984)(for silver). Calculations are shown in attachment A.

Several pesticides were evaluated based on the sum of their primary constituents and associated metabolites (e.g., total chlordane, total endosulfan, and total DDT). Alpha(trans)-chlordane, trans nonachlor, heptachlor and heptachlor epoxide represent the primary components of technical chlordane and its metabolites found in the tissue of aquatic organisms (Jarman, et. al., 1996; Verschueren, 1983; Sweeney, et. al., 1993). These constituents are summed as total chlordane as is consistent with current practice for chlordane (Jarman, et. al., 1996) and total DDT. The WQCTL for total chlordane was calculated using the WQC for chlordane as a conservative level of protection. While water quality criteria exist, and WQCTLs can be calculated, for heptachlor (133 ppb) and chlordane (64 ppb), the sum total chlordane is compared to the WQCTL for chlordane in order to be more environmentally conservative. The chlordane WQCTL provides a conservative level of protection as indicated by published residue effects levels (Sweeney, et. al., 1993; Bauman, et. al., 1987; Feroz, et. al., 1979; Parrish, et. al., 1976). Consistent with the above approach, the tissue concentration for endosulfan I, endosulfan II and endosulfan sulfate were also summed as total endosulfan and compared to the WQCTL for total endosulfan.

The WQCTLs were also calculated for all metals of concern which don't have Matrix values. For total chromium, the WQCTL was calculated based on chromium(VI), which is substantially more toxic than chromium (III) and elemental chromium in order to provide a conservative level of environmental protection.

B. Potential for ecological effects based on PAH toxicity (Column 19).

The Critical Body Residue (CBR) approach described by McCarty (1991) was used to derive values for use in evaluating the potential impacts of PAHs accumulated in the dredged material bioaccumulation test organisms. CBRs are concentrations of chemical residues in organisms which elicit a deleterious biological response associated with narcosis, which is the primary non-cancer effect of PAHs. Narcotic responses measured can be acute (e.g., immobilization or death) or chronic endpoints (e.g., reduced reproduction, fecundity or growth). CBRs are represented as the ratio of the mass of toxicant to the mass of the organism, such as millimoles or micrograms of toxicant per kilogram (mmole or ug/kg) of organism. For the narcosis endpoint, each molecule of individual PAH congeners are generally equipotent, thus the total PAH concentration is compared to the CBR. For example, a 400 ppb dose of naphthalene would elicit a similar toxicity response as 400 ppb of fluorene; if both chemicals are present together at these concentrations, then the dose would equal 800 ppb.

McCarty (1991) states that an average critical body residue of 400,000 - 1,200,000 ppb can be used as an estimate for acute effects for a narcosis-producing chemical (e.g., PAHs) on fish populations. (Note: McCarty reports the CBR in units of millimoles per kilogram; this value has been converted to ppb for PAHs using the average molecular weight of the PAHs analyzed in the bioaccumulation test). Chronic effect critical body residues can be estimated by applying an acute to chronic ratio of 10 to the acute CBR (McCarty, 1986; Call, *et al.*, 1985). Therefore, the chronic critical body residue for PAHs can be estimated at 40,000 - 120,000 ppb of PAHs in organism tissue, and Table 1 thus uses the 40,000 ppb level.

These CBRs were based on fish data. The use of CBRs based on fish toxicity represents a conservative estimate of potential toxicity due to exposure to dredged material because: (1) it is extremely unlikely that a fish would get its whole diet from the HARS; and (2) fish are generally more sensitive than the benthic organisms in direct contact with the dredged material placed at the HARS (e.g., Landrum, *et al*. (1988) estimated an acute CBR for crustaceans of 800,000 ppb - 42,000,000 ppb).

C. Potential ecological impacts of mutagenic, carcinogenic and teratogenic PAHs (Column 19)

USEPA and the USACE reviewed eleven scientific journal articles to obtain information about the potential for adverse effects to the marine environment due to the observed bioaccumulation of PAHs in the marine worm, *Nereis virens*, and the clam, *Macoma nasuta.* These articles reported the results of laboratory experiments that sought to relate the concentration of a contaminant(s) in water, as injected doses, or tissue concentrations, to mutagenic, carcinogenic, teratogenic and/or reproductive effects to fish. These studies all used fish species which are considered to be among the most sensitive organisms in the marine environment to exhibit the above effects (USEPA, 1996c). In addition, most of these studies focussed on the PAH most believed to cause such effects for which there is data, benzo(a)pyrene (BaP). One study (Breteler, 1984), discussed the possible sources and distribution of PAHs in the Hudson/Raritan estuary, and ranked the threat of PAHs to aquatic biota and humans. The main threat was believed to be carcinogenicity, with a greater threat ranking assigned to humans than biota. However, Breteler (1984) did not provide specific effects-based levels that could be used in the

following analysis. Two articles evaluated the effects of crude oil, and thus were not useful for evaluating the effects of specific PAHs measured in the bioaccumulation test (Rice, *et al.*, 1987; Lee, *et al.*, 1981). Three studies considered the effects of specific PAHs, but did not synoptically measure tissue concentrations in the organisms (Ward, *et al.*, 1981; Holcombe, *et al.*, 1983; Finger, *et al.*, 1985) and were not used, because the lack of tissue data for these studies makes their utility in evaluating the tissue concentration resulting from the dredged material bioaccumulation tests highly uncertain.

The remaining five papers reported measured tissue concentrations and observed reproductive effects in organisms exposed to PAH-spiked water. One article reported the tissue concentrations of adult fish and the observed effect on survival and health of the fish's offspring. Hose, *et al*. (1981) reported that adult English sole injected with benzo(a)pyrene (BaP) accumulated the chemical in the gonad and mature gametes. The amount of BaP taken up by the ovary ranged from 16,800 to 49,700 ppb. Two samples of ripe eggs contained 51,200 and 263,000 ppb of BaP and its metabolites. No adverse effects were reported for these concentrations. Hose, *et al*. (1981) also reported the results of injecting female flathead sole with BaP. Adverse effects to egg hatching success were reported for each female. The paper does not report tissue concentrations in either the parent fish or the egg of the flathead sole. Effects on reproductive success were reported but could only be correlated with the external dose injected into the parent. Therefore, since concentrations and effects were not synoptic in this report, it was not useful in the evaluation of the dredged material bioaccumulation results.

Three papers reported the results of experiments which measured fish egg or alevin concentrations of BaP and associated reproductive or carcinogenic effects (Hose, *et al.*, 1982; Hannah, *et al.*, 1982; Black, *et al.*, 1988). Hose, *et al*. (1982) exposed three species of sole, sand sole, English sole, and flathead sole, to BaP-spiked water. Tissue concentrations of 2,100 ppb were measured in sand sole on day 6 (24 hours after hatching) and were associated with reduced hatching success. However, we did not consider the results to be appropriate for use in setting effects levels because they may have been compromised by the methods of replication used in the experimental design.

Hannah, *et al*. (1982) estimated a concentration of BaP in tissue that caused abnormalities in development of rainbow trout eggs, using aqueous exposures and actual measured tissue concentrations in alevin tissues. An exposure to a 2.4 ppb mean aqueous BaP concentration accumulated an average of 12,340 ppb in alevins. This concentration was associated with an increase in percentage of abnormalities from approximately 6% at lower water concentrations (0.08, 0.21, 0.37 and 1.48 ppb) to approximately 13% at higher concentrations. From 0.08 to 1.48 ppb in the water, there were no increasing effects exhibited, therefore, the effects were apparently "real" (i.e., significantly greater than the threshold effect level of 6%) only at the aqueous 2.4 ppb concentration. The Hannah, *et al.*,(1982) study is considered the most reliable study for this evaluation since it used exposure series and measured tissue concentrations associated with observed effects, and therefore allows for the calculation of a no-effects level directly from the measured results.

In applying these studies to evaluations of dredged material, consideration must be given to uncertainties in converting these kinds of results to concentrations protective of other biota. Three uncertainties

needing to be considered are: (1) those associated with converting effect to no-effect concentrations, (2) across-species uncertainties, and (3) uncertainties in estimating the dose of contaminants to which the organism is exposed. These uncertainties are discussed below.

With respect to uncertainty when converting effect to no-effect concentrations, an uncertainty factor of one order of magnitude is often used when only an effect measure is reported. However, in Hannah, *et al.*(1982), the no-effect level can be estimated to be the next lowest concentration below the lowest-observed effect level, since the range of concentrations below this level did not exhibit significantly different responses. In this case, the no-effect level occurred at the water exposure concentration of 1.48 ppb. Although a tissue concentration was not measured at the 1.48 ppb water concentration, it can be calculated from the concentration measured at the effect level (i.e., the no-effect water concentration (1.48 ppb) is close to 65 percent of the observed effect concentration (2.4 ppb) so the no-effect tissue concentration should be about 65 percent of the lowest-observable effect tissue concentration (0.65 x 12,340 ppb = 8,021 ppb)). Thus, a factor to adjust these data from lowest observed effect tissue concentration to the calculated no-observed effect tissue concentration is obtained directly from the data.

There can also be uncertainty as to the proximity to the site of toxic action in the organism that a dose or concentration is measured, and with respect to species-to-species variability. Hannah (1982) reported dose concentrations in the tissue and, therefore, there is no need to account for variability associated with the large uncertainties encountered in typical water-only exposure studies where the actual concentration at the site of toxic action is unknown. When measured in the tissues, as was done for this project, concentrations of narcotic chemicals causing effects (i.e., critical body residues, CBRs) in aquatic organisms are reported to range only from 1.4 to 21 umoles/g wet weight (a factor of about one order of magnitude) for organisms as diverse as insects, crustaceans, and fish (McCarty, *et al*. 1992). Therefore, from a tissue concentration perspective, the species-to-species uncertainty factor appropriate for both total PAHs operating as narcotics and individual PAHs having teratogenic effects would be one order of magnitude, or a value of 10 (USEPA, 1996d).

In summary, a factor of 10 (representing species to species uncertainty) is an appropriate UF to use in these evaluations. Also, as described in memo SectionV, subsection C2(c)(i) above, Brown and Neff (1993) show that trophic transfer of PAHs up the food chain to fish decrease tissue levels by over an order of magnitude. Given this data and the fact that these studies included fish that spent 100% of their time feeding in the test sediment, whereas this would be highly unlikely to occur at an ocean site, a trophic transfer factor of 0.1 is used in this analysis. Applying this UF of 10 and a trophic transfer factor of 0.1 to the no-effects level for BaP calculated from Hannah, *et al*. (1982), as discussed above (8,021 ppb) results in a no-effect level for BaP of approximately 8,000 ppb in benthic tissue.

II. CONSIDERATION OF POTENTIAL EFFECTS ON HUMAN HEALTH (Columns 14 and 15)

Human effects screening levels were developed with risk-based methods using conservative estimates of exposure to assess whether these contaminants would accumulate to levels in fish and shellfish that could lead to significant adverse effects to humans. The approach assessed consumption of fish and shellfish to derive conservative estimates of contaminant concentrations in benthic tissue protective of human health using the following USEPA standard risk-assessment assumptions: a 70-kilogram adult eats 6.5 grams of fish and shellfish per day over a 70-year lifetime. This assessment considered potential for both cancer and non-cancer effects in humans. USEPA IRIS (USEPA, 1995) and effects information from USEPA's National Toxics Rule (USEPA, 1992b) were used in the human health assessment to calculate acceptable levels in fish and shellfish to protect human health. Trophic transfer factors, as discussed earlier, were then used to convert these fish and shellfish levels into benthic tissue concentrations.

For regulatory purposes, USEPA utilizes 10^{-4} to 10^{-6} (one in ten thousand to one in one million) as an acceptable incremental risk range for activities with potential for causing cancer in human beings (USEPA, 1980a; USEPA, 1988; USEPA, 1987a; Thomas, 1987; USEPA, 1991). USEPA considers a cancer risk within this range to be safe and protective of public health. This is supported by the World Health Organization Guidelines for Drinking Water Quality (WHO, 1993), where it selected a 10^{-5} guideline value, and then explained that the application could vary by a factor of ten (e.g., 10^{-4} to 10^{-6}). Since this analysis uses conservative methods, the results represent conservative estimates of risk, or what are in effect plausible upper-bound estimates. Thus, the true risk is highly unlikely to be greater than estimated and could be much lower.

Table 1, Column 14 lists human cancer protection levels in benthic organisms for chemicals which are known or suspected carcinogens that would lead to a human cancer risk level of 10^{-4}. For PAHs, this analysis used BaP-equivalents derived from the toxic equivalence factor for each carcinogenic PAH (from USEPA (1993); note: these factors are listed in Column 11 of Table 1 for each of the compounds).

The potential for non-cancer impacts can be expressed as a hazard quotient (HQ), which is the ratio of the average daily intake divided by the toxicological reference dose for the chemical. If the HQ is less than unity (e.g., 1), an adverse noncarcinogenic effect is highly unlikely to occur. If the HQ exceeds unity, an adverse health impact may occur. The higher the HQ, the more likely that an adverse noncarcinogenic effect will occur as a result of exposure to the contaminant in the dredged material after placement. Table 1, Column 15 lists noncancer protection levels in benthic organisms for the chemicals that are known to cause, or suspected of causing, non-carcinogenic effects, that would result in a human HQ equal to unity. Those numbers were derived using the conservative assumptions and source materials described in the introductory paragraph to this section.

For the following compounds, the following special considerations were used in evaluating the results in Table 1.

<u>Metals</u>

No reference dose has been established for lead. EPA has adopted a blood lead level of 10ug/dl as the level of concern and EPA policies are that regulatory programs should seek to minimize the number of children with blood lead levels above a target of 10 ug/dl (Final Rule for Lead and Copper NPDWR, 56FR26468, June 7, 1991), and this value was used to calculate the effects level in Table 1 (see USEPA 1996e).

When interpreting the importance of arsenic tissue concentrations for human health, consideration was given to the arsenic form present (i.e., inorganic vs. organic). Arsenic is found in marine organisms as an organic complex which includes such compounds as arsenobetaine and arsenocholine (Abel and Axiak, 1991). Organic arsenic in the tissues of aquatic organisms is not metabolized by predators or humans and is readily eliminated from the body through excretion (Hrudey *et al.*, 1995). As a result, the toxicity of organic arsenic ingested from seafood is low and appears to pose no significant hazard (Abernathy and Ohanian, 1992). For this reason, cancer and non-cancer protection levels, based on inorganic arsenic as contained in EPA's IRIS database, are not appropriate for evaluating the potential human health impacts of arsenic bioaccumulation from dredged material, and therefore, are not included in Table 1.

Pesticides

Alpha(trans)-chlordane, trans nonachlor, heptachlor and heptachlor epoxide represent the primary components of technical chlordane and its metabolites found in the tissue of aquatic organisms (Jarman, et. al., 1996; Verschueren, 1983; Sweeney, et. al., 1993). These constituents are summed as total chlordane as is consistent with current practice for chlordane (Jarman, et. al., 1996) and other pesticides (e.g., total DDT). Total chlordane is evaluated using the 10^{-4} cancer risk level and non-cancer level for heptachlor epoxide, which has the greatest potency of the chlordane constituents or metabolites. Similarly, endosulfan I, endosulfan II, and endosulfan sulfate are summed and the total is compared to the conservative non-cancer protection level for endosulfan.

PAHs

For PAHs, this analysis used BaP-equivalents derived from the toxic equivalence factor for each carcinogenic PAH (from USEPA (1993); note: these factors are listed in column 11 of Table 1 for each of the compounds).

Attachment A:
Tissue concentration is calculated using BCF * Water Quality Criteria (WQC) ambient aqueous concentration, and assuming the conversion factor of 1 kg=1L.

Compound	**Ambient Conc. (ug/L)[1]**	**BCF**	**Tissue Conc. (ug/Kg)**	**Remarks**
Aldrin	0.13	2,300	299	WQC was reduced by a factor of 10 to account for chronic effects; BCF estimate is based on Dieldrin since Aldrin rapidly transformed to Dieldrin in the environment; BCF is based on 1.1% lipid level for marine fish, Spot (*Leiostomus xanthurus*).
Dieldrin	0.0019	2,300	4.37	BCF is based on 1.1% lipid level for marine fish, Spot (*Leiostomus xanthurus*).
Total Chlordane	0.004	16,000	64	Total Chlordane includes alpha-chlordane, trans nonachlor, heptachlor, heptachlor epoxide; WQC for Chlordane is used for Total Chlordane; BCF is based on 3.6% lipid level for sheepshead minnow (*Cyprinodon variegatus*).
Total Endosulfan	0.0087	328	2.85	Total Endosulfan includes endosulfan I, endosulfan II and endosulfan sulfate; WQC for Endosulfan is used for Total Endosulfan; BCF is based on 3.6% lipid level for sheepshead minnow (*Cyprinodon variegatus*).
1,4-Dichlorobenzene	197	60	11820	Ambient conc. is based on lowest observed effect level (LOEL) for saltwater species from WQC, and reduced by a factor of 10 to account for chronic effects; BCF is based on the whole body for bluegill (*Lepomis macrochirus*).
Arsenic	36	350	12600	Ambient conc. is based on the saltwater criteria continuous conc. for arsenic (III); BCF is based on the Eastern Oyster (*Crassostrea virginica*).
Chromium	50	236	11800	Ambient conc. is based on Chromium (VI) since it is substantially more toxic than Chromium (III); BCF is based on polychaete worm.
Copper	2.9	3,300	9570	WQC is based on a hardness value of 100; BCF is based on soft shell clam.
Lead	8.5	1,400	11900	Ambient conc. is based on saltwater criteria continuous conc.; BCF is based on the Eastern Oyster (*Crassostrea virginica*).
Nickel	8.3	458	3802	Ambient conc. is based on saltwater criteria continuous conc.; BCF is based on the Eastern Oyster (*Crassostrea virginica*).
Silver	0.23	6,500	1495	Water Quality Criterion (WQC) was reduced by a factor of 10 to account for chronic effects; BCF is based on the Blue Mussel (*Mytilus edulis*).
Zinc	86	17,640	1517040	Ambient conc. is based on saltwater criteria continuous conc.; BCF is based on Eastern Oyster (*Crassostrea virginica*).

[1].The following documents were used to obtain the water quality criteria values.

Calabrese, A. 1984. "Effects of Long Term Exposure to Silver and Copper on Growth, Bioaccumulation and Histopathology in the Blue Mussel (Mytilus edulis)." *Mar. Envir. Res.* 1, 253-274.

USEPA. 1980b. Ambient Water Quality Criteria for Aldrin/Dieldrin; EPA 440/5-80-019; December 1980.

USEPA. 1980c. Ambient Water Quality Criteria for Chlordane; EPA 440/5-80-027; October 1980.

USEPA. 1980d. Ambient Water Quality Criteria for Heptachlor; EPA 440/5-80-052; October 1980.

USEPA. 1980e. Ambient Water Quality Criteria for Endosulfan; EPA 440/5-80-046; October 1980.

USEPA. 1980f. Ambient Water Quality Criteria for Dichlorobenzenes; EPA 440/5-80-039; October 1980.

USEPA. 1984a. Ambient Water Quality Criteria for Lead - 1984; EPA 440/5-84-027; January 1985.

USEPA. 1984b. Ambient Water Quality Criteria for Copper - 1984; EPA 440/5-84-031; January 1985.

USEPA. 1985a. Ambient Water Quality Criteria for Chromium - 1984; EPA 440/5-84-029; January 1985.

USEPA. 1985c. Ambient Water Quality Criteria for Arsenic - 1984; EPA 440/5-84-033; January 1985.

USEPA. 1986. Ambient Water Quality Criteria for Nickel - 1986; EPA 440/5-86-004; September 1986.

USEPA. 1987b. Ambient Water Quality Criteria for Zinc - 1987; EPA 440/5-87-003.

USEPA. 1992a. Draft Ambient Water Quality Criteria for Silver.

ATTACHMENT B

Benthic Cancer Protection Level Calculations for the Protection of Human Health from the Consumption of Fish Exposed to Dredged Material at the Historic Area Remediation Site

Basis[fn1]:

10^{-4} Benthic Tissue Level (ug/kg) = $\frac{[10^{-4} \text{ Conc. in Fish }] \times [\text{Whole Body/fillet Factor (1.35)}]}{\text{Trophic Transfer Factor}}$[fn5] [fn2]

10^{-4} Conc. in Fish (ug/kg) = $\frac{\text{Toxicological Dose (ug/day)}}{[\text{Seafood Consumption (6.5 g/day)}] \times [10^{-3} \text{kg/g}]}$[fn3]

Toxicological Dose (ug/day) = $\frac{[\text{Risk Level } (10^{-4})] \times [\text{Body Weight (70 kg)}] \times [10^{3} \text{ ug/mg}]}{\text{Potency Factor, } q_1^* \text{ (kg-day/mg)}}$[fn3] [fn4]

	Cancer Potency Factor q_1^* (kg-day/mg)	**Acceptable Concentration in Fish (ug/kg)**	**Trophic Transfer Factor**	**Benthic Protection Level (ug/kg)**
Pesticides				
Aldrin	17	63	2.6	33
Chlordane	1.3	828	2.3	486
Dieldrin	16	67	1.4	65
Heptachlor	4.5	239	2.7	120
Heptachlor epoxide	9.1	118	1.4	114
Industrial Organics				
1.4-Dichlorobenzene	0.024	44,872	1	60,577
PAHs				
Benzo(a)pyrene	7.3	147	0.1	2,000
METALS				
Arsenic	1.5	718	3	323

ATTACHMENT C

Benthic Non-Cancer Protection Level Calculations for the Protection of Human Health from the Consumption of Fish Exposed to Dredged Material at the Historic Area Remediation Site

Basis [fn1]:

$$\text{Benthic Tissue Level (ug/kg)} = \frac{[\text{Conc. in Fish}] \times [\text{Whole Body/fillet Factor (1.35)}]^{fn5}}{\text{Trophic Transfer Factor}^{fn2}}$$

$$\text{Conc. in Fish (ug/kg)} = \frac{\text{Toxicological Dose (ug/day)}}{[\text{Seafood Consumption (6.5 g/day)}^{fn3}] \times [10^{-3}\text{kg/g}]}$$

$$\text{Toxicological Dose (ug/day)} = [\text{Reference dose}^{fn4}] \times [\text{Body Weight (70 kg)}^{fn3}]$$

	Reference Dose RfD (ug/kg-day)	Acceptable Concentration in Seafood (ug/kg)	Trophic Transfer Factor	Benthic Protection Level (ug/kg)
Metals				
Arsenic	0.3	3,231	3	1,454
Chromium	5	54,000	1	73,000
Copper	37.1	400,000	1	540,000
Nickel	20	215,000	1	290,000
Silver	5	54,000	1	73,000
Zinc	300	3,230,769	1	4,361,538
Pesticides				
Aldrin	0.03	323	2.6	167
Chlordane	0.06	592	2.3	350
Dieldrin	0.05	538	1.4	518
Endosulfan	6	64,615	1	87,231
Heptachlor	0.5	5,385	2.7	2,692
Heptachlor epoxide	0.013	140	1.4	135
PAHs				
Acenaphthene	60	650,000	0.1	8,775,000
Fluorene	40	430,000	0.1	5,805,000
Phenanthrene	300	3,230,000	0.1	43,605,000
Anthracene	300	3,230,000	0.1	43,605,000
Fluoranthene	40	430,000	0.1	5,805,000

Pyrene	30	325,000	0.1	4,387,000

NOTES:

[fn1] Human health cancer and non-cancer assessments adapted from *Guidance for Assessing Chemical Contaminant Data for use in Fish Advisories: Volume II: Risk Assessment and Fish Consumption Limits*. U.S. Environmental Protection Agency, EPA823-B-94-004, Office of Science and Technology, Washington, DC, June 1994.

[fn2] Trophic transfer factors were calculated by Mr. Lawrence Burkhard, EPA Duluth, using the food chain transfer model developed by Gobas (1993).

[fn3] Default values were taken from EPA's national toxics rule for setting water quality criteria, USEPA (1992b).

[fn4] Cancer potency factors and non-cancer reference doses are taken from USEPA (1995).

[fn5] The acceptable concentration in seafood is defined on the basis of the fillet or edible portion for humans. Trophic transfer, however, was defined on the basis of whole body characteristics, including lipid concentrations. Experience in New York State indicates a whole body to fillet ration ranging from 1.2 to 1.5 is applicable to lipophilic substances. The mid range value of 1.35 is used in this analysis.