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United States Department of the Interior

FISH AND WILDLIFE SERVICE

Boqueron Field Office
P.O. Box 491
Boqueron, Puerto Rico 00622



MAY 23 2006

Mr. Christopher T. Penney
Eastern Vieques Project Coordinator
Naval Facilities Engineering Command
6506 Hampton Blvd
Norfolk, VA 23508-1278

Re: RI/FS for SWMU-6, Mangrove Site, Vieques

Dear Mr. Penney:

Enclosed are the Service's comments regarding the draft Remedial Investigation/Feasibility Study (RI/FS) for Solid Waste Management Unit 6 (SWMU-6), also known as the mangrove site, on the western side of the Vieques National Wildlife Refuge. SWMU-6 consists of various trash piles scattered throughout a mangrove estuarine area. This site was first reported by the Navy and its consultants in the late 1970's as part of the initial studies for the Navy's 1979 Vieques Environmental Impact Statement. It was subsequently reported in various Navy sponsored follow up mangrove surveys in the 1980's. The various Land Use Management Plans for the Naval Facility, Vieques, recommended the removal of these trash piles as a top priority. We understand that the existing trash piles will be removed under an ongoing Engineering Evaluation/Cost Analysis (EE/CA) for this site.

SPECIFIC COMMENTS ON VIEQUES REMEDIAL INVESTIGATION REPORT

Section 2 - Site History, Previous Investigations:

P. 2-2:

The report states: "No vegetation stresses were observed in the plant communities." The Service suggests that the lack of visible signs vegetation stress be eliminated as a criterion from site evaluations, because the lack of stressed vegetation on a 26 year old site does not provide any information regarding possible impacts.

P. 2.4:

The report states: "There was no evidence that the site has had an impact on wildlife or habitat." The Service suggests that data needs to be included to substantiate this conclusion.

P. 2.5, Section 2.6.1:

The results of the confirmation study conducted in 1988 (ESE 1988) reportedly included soil concentrations of chromium (Cr) and lead (Pb) of from 18.5 to 48.2 mg/kg and 10.2 to 345 mg/kg, respectively. Chromium and lead concentrations in sediment ranged from 5.28 to 88.4 mg/kg and 2.82 to 312 mg/kg, respectively. In the next sentence, the report states that no elevated levels of any chemicals of concern (COCs) were detected in soil, surface water, or sediment samples. While it appears as though the Remedial Investigation (RI) for SWMU-6 is only reporting the conclusions of the 1988 study, it is unclear what criteria were used to make the assessment that COCs were not elevated. We recommend that this be clarified since several currently accepted benchmarks are below the maximum concentrations reported. For example, Efroymsen et al. (1997a,b) report lower soil Cr benchmarks for plants (1 mg/kg), earthworms (0.4 mg/kg), and soil microorganisms (10 mg/kg), and a lower Pb benchmark for earthworms (50 mg/kg). Similarly, Ecological Soil Screen Levels (EcoSSLs) for Pb in soil are 110 (plants), 16 (birds), and 59 (mammals) mg/kg (U.S. EPA 2003). In marine sediments, the threshold effect level (TEL) for chromium reported by Long et al. (1995) is 81 mg/kg; the TEL listed in MacDonald (1994) is 52.3 mg/kg. The corresponding thresholds for lead are 46.7 and 30.2 mg/kg, respectively. Additionally, while the confirmation study did not recommend additional investigation of the site, it does not appear to have evaluated any inorganic compounds besides Cr and Pb. We recommend that those compounds from the 1988 study whose maximum concentrations exceed the currently accepted benchmarks be reevaluated and the conclusion of the 1988 study regarding additional investigations be modified in necessary.

P. 2.6:

The report states that upgradient and downgradient concentrations of detected compounds were similar. Although not stated specifically in the report, it appears as though average downstream concentrations were compared to maximum background concentrations. It should be noted that at some downgradient sampling locations, maximum detected concentrations of chemicals of potential concern (COPCs) were higher than the maximum background (upgradient) concentration. We recommend that this conclusion be reexamined for those downgradient chemicals that were higher than maximum background concentrations.

The report states that the relative similarity in metal concentrations among different surface soil samples could be an indication that contaminants are present as a result of background conditions, rather than being site-related. We recommend that this statement be deleted as it is conjecture.

P. 2-8 (Table 2-1):

Under the "Findings" column for the confirmation study in 1988, it states, "Two metals were detected". This implies that other metals were screened for, but not detected. However, in the discussion on p. 2-5 (section 2.6.1) it sounds as though samples were only screened for

chromium and lead (see comment above). The Service requests clarification as to whether other metals, besides Cr and Pb, were evaluated in the confirmation study.

Section 3 - Summary of Field Investigation

P. 3-2:

The discussion of surface soil sampling locations is somewhat confusing. In the third paragraph of this section (3.2.2), the document states that 15 surface soil samples (NDW06SS09 through NDW06SS23) were collected at the site. However, the corresponding figure (3-1) shows an additional eight surface soil locations that are not mentioned in this paragraph. In the fourth paragraph of section 3.2.2, it is explained that an additional eight samples were taken at previous soil boring locations, but only figure 3-2 is referred to as indicating the location where these samples were taken. We suggest that it would be less confusing if figure 3-2 was deleted entirely and a statement to the effect that "Previous soil boring locations sampled during the Expanded Preliminary Assessment/Site Inspection (PA/SI) include NDW06SS01 through NDW06SS08" was added to the third paragraph of section 3.2.2.

In addition on this page, while it is stated that additional surface soil samples were taken to further characterize the horizontal extent of contamination, many of the samples appear to be fairly bunched while other areas appear to be overlooked (for example, close to the water near the western edge of the SWMU). Also, relatively few samples (perhaps 4 of the 23) appear to have been located close to or at the debris piles, where the highest concentrations of COPCs are likely to occur.

Figure 3-1:

Although the text describing the soil sampling locations discusses their placement relative to borings and/or wells, the location of the borings and wells described are not noted on Figure 3-1, making it difficult to determine which soil sample locations are being referring to. For example, the text states that four soil samples were taken to the north, southeast, and west of well NDW06MW04, but since the location of that well is not indicated on Figure 3-1, it cannot be determined which soil samples comprise those four. We recommend that the locations of borings and wells be included in this figure.

Page 3-3:

Both of the monitoring wells installed to assess background conditions are locations very close to the road. We wonder if it is possible for contaminants from the road surface to enter the wells through ground water seepage, thereby artificially increasing background concentrations.

P. 3-6 and Figure 3.4:

The description of samples used for the RI does not include any mention of the locations from the 2000 PA/SI that were not re-sampled. It is confusing that additional sampling locations (NDW06SW01, NDW06SW04) not mentioned in the text are evident in the corresponding figure (Figure 3.4), and it makes it hard to reconcile the text with the figures when the locations and number of samples don't appear to match. We suggest that relevant information about all sampling locations presented in the figures should be included in the text, or the figure should be modified to indicate which samples are from the 2000 vs. the 2003 investigation.

P. 3-7:

We noticed that co-located sediment samples were not taken with the newly collected surface water samples (NDW06SW08 and NDW06SW09). We also noticed that co-located surface water samples were not taken with the newly collected sediment samples (e.g., NDW06SD09, NDW06SD10, NDW06SD12, and NDW06SD14). We recommend that surface water samples or sediments always have a co-located sediment or surface water sample.

P. 3-8:

The discussion of the background locations for surface water and sediment samples is confusing. The first paragraph in section 3.5.3 states that two background surface water and sediment samples were collected from nearby Arenas Lagoon. The next sentence states that two surface water and sediment samples were previously collected from this water body and included in the background sampling report. In the second paragraph of this section, it states that two new surface water and sediment sample were collected from south Kiani Lagoon. These are not located in the site boundary, and so would seem to be background samples (particularly because they are described in the section entitled, "Background Surface Water and Sediment Sampling"). However, in the last paragraph of this section, it states that two background surface water and sediment samples were collected from Arenas Lagoon. Are those the same two referred to in the first paragraph? If so we suggest that, redundancy (and confusion) would be reduced by putting the sample numbers in the first paragraph and deleting the last paragraph entirely. Only two are shown in Table 4.2 (water: locations NDW06SW10 and NDW06SW11) and Table 4.3 (sediment: locations NDW06SD15 and NDW06SD16). Thus, according to these tables, the samples from Kiani Lagoon (locations NDW06SW/SD06B and NDW06SW/SD07B) do not seem to be background samples. We recommend that the Navy clarify or correct this information about the background samples.

P. 3-10:

We suggest that results from the geophysical investigation would be more meaningful if close-ups of the transect lines and interpreted fill boundary were overlain with the outline of the access restriction boundary and sampling locations.

Section 4 - Nature and Extent of Contamination

Tables 4-1 to 4-4 and 4-12 to 4-14:

There are some inconsistencies between the tables listing background concentrations (Tables 4-1 to 4-4) and tables comparing screening values to background concentrations (Tables 4-12 to 4-14). For example, the groundwater summary table (4-12) does not contain background values for the following elements listed in table 4-1 with the qualifier 'J': barium (Ba), beryllium (Be), cobalt (Co), manganese (Mn), mercury (Hg) (there appear to be tap water preliminary remediation goals (PRGs) for all of these compounds), but it does list background values for other elements with the qualifier 'J' (ex. chromium, selenium (Se)). Similarly, the sediment summary table (4-14) does not contain background values for the following elements listed in table 4-3 with the qualifier 'J': beryllium, cobalt, chromium, iron (Fe), manganese, selenium, and vanadium (V). Although most of these compounds are not listed in the sources for the ecological screening values, chromium is listed in both documents cited as sources of the sediment ecological screening values (Long et al. 1995; U.S. EPA 2000). Background values for barium and manganese should be added to Table 4-13, with the screening value listed as 'NA', since no screening values are available from the listed sources. Additionally, the PRGs listed for chromium and lead in Table 4-12 don't match those listed in U.S. EPA (2002). Finally, what does the shading in Tables 4-12 through 4-14 indicate? We recommend you correct or clarify these points.

P. 4-8:

We recommend that all soil background values (from CH2M Hill 2002) used for this RI should be presented in table form; the accuracy of statements comparing concentrations of detected compounds in soil to background levels cannot be adequately assessed without this information.

Table 4-4 (P. 4-43):

We recommend that you identify the source of the daily soil intake rates.

Soils

P. 4-11 and 4-44 (Table 4-5):

Several compounds with detected values exceeding ecological criteria are not listed in this table (Aluminum (Al), Cadmium (Cd), Cr, Mn, V, Thallium (Tl)). For example, the plant benchmark for vanadium is 2 mg/kg (Efroymson et al. 1997b), and this value was exceeded in soil samples from all locations. It is possible that the compounds are not listed because the concentrations do not exceed background levels, but we cannot determine whether this is the case, since background concentrations are not presented. For this entire section as well as the corresponding tables, we feel it would be useful to provide a table that provides a complete list of analytes and the corresponding background concentrations and screening criteria, and indicates which detected compounds do not have screening criteria available.

The list of compounds that exceeded leachability criteria in soil (Soil Screening Levels (SSLs) in Table 4-5) should also include chromium, which has an SSL of 19 mg/kg (using a Dilution/Attenuation Factor (DAF) of 10, as is indicated in the RI/FS). Also, we question why is a DAF of 10 used, instead of the default of 20 as is in the source document (U.S. EPA 2002). (A DAF of 10 is more conservative than a DAF of 20).

Also on P. 4-11, the first paragraph states that “eleven inorganic analytes were detected above screening criteria in at least 1 surface soil sample.” This number (eleven) appears to be correct. The second paragraph on this page states, “seven metals...exceeded their respective human health or ecological screening criteria in at least one surface soil sample.” In fact, ten metals exceeded ecological screening criteria (Al, Antimony (Sb), Cr, Cu, Fe, Mn, Pb, Tl, V, Zinc (Zn)) and eight metals exceeded their human health screening criteria (Al, Sb, As, Fe, Mn, Pb, Tl, V). Only two metals exceeded the leachability criteria (Sb and Cr). We recommend that this error be corrected.

P. 4-11 to 4-12:

The fourth paragraph states, “Each of the [inorganic] chemicals that exceed available screening criteria is discussed below.” This is followed by a list of seven analytes. However, a total of 11 inorganic compounds were detected above screening levels (see above); Al, Cr, Mn, and V are missing from the list (see Table I-2 in Appendix I). These compounds are also not listed on the corresponding figure (Figure 4-1). We suggest presenting the background data from CH2M Hill (2002), so that we could determine whether any samples contained concentrations of these compounds that exceeded background.

We also suggest presenting the background data for organics from CH2MHill (2002), because without a presentation of background data, the accuracy of statements comparing sample concentrations of organics in soil to background levels cannot be assessed (with the exception of background values for COPCs in Table 4-10). It should be noted that several discrepancies were found between which metals actually exceeded benchmarks and which compounds are stated to have exceeded benchmarks (see discussion of P. 4-11). There may be similar discrepancies for organic compounds as well.

P. 4-12:

Two additional semivolatile organics (benzo(k)fluoranthene and dibenzofuran) appear to have been detected above ecological screening values and should be mentioned in the discussion on pages 4-12 to 4-13. These include the following exceedences:

Compound	Location	Value (ug/kg)	PRG (ug/kg)	Eco (ug/kg)	SSL (ug/kg)
benzo(k)fluoranthene	W6-SB05	1,230	6,200	100 ¹	25,500
	NDW06SS12	205			
	NDW06SS13	509			
	NDW06SS18	173			
Dibenzofuran	W6-SB05	784	29,000	4 ¹	NA

¹ Value is from (CCME 1991 and CCME 2002). This comparison assumes the benchmark values for benzo(k)fluoranthene and dibenzofuran are the same in CCME (1991, 2002) as those from CCME (1996), which was not available.

Surface Water

The total metal concentrations should be presented in Appendix I (only dissolved concentrations are included), so the information presented regarding inorganic analytes in unfiltered samples can be verified.

P. 4-18:

Di-n-octyl phthalate was detected in two samples, not one, as is stated in the report. We suggest that this be corrected.

Sediment

P. 4-20:

In the final paragraph on this page, the report says detected organic chemicals were not above the available screening criteria. We noticed that is not the case for bis (2-ethylhexyl phthalate), which was detected above the ecological criterion in two samples (see P. 4-19) or for

dichlorodiphenylethane (DDD), dichlorodiphenylethylene (DDE), and dichlorodiphenyltrichloroethane (DDT), which were detected at concentrations above the screening criteria in two, three, and one sample, respectively (see P. 4-20). We recommend this be corrected.

Appendix I:

Pages I-188 to I-192 are missing.

Table I-2 – We recommend that methylene chloride be in the table of detected chemicals in surface soils (it was detected at locations NDW06SS14, NDW06SS16, and NDW06SS23), although it was measured at concentrations lower than the eco screening value.

Section 6 Human Health Risk Assessment (HHRA)

P. 6-6:

The document states, “Frequency of detection, concentration, site-related use or release, and the structural activity relationships available in the EPA guidance were taken into account in deciding whether to retain each chemical without a PRG for further consideration.” It appears from Tables 6-1 through 6-4 that only chemicals exceeding PRGs were retained as COPCs; no compounds without PRGs (or SSLs for subsurface soils) were retained. How were the factors listed taken into account? We suggest that specific reasons why each chemical without a PRG or SSL was not retained be provided.

P. 6-14:

The document states that the screening level for the conservative health protection-based lead concentration protective of a residential child (PRG) is 400 mg/kg. As was pointed out on P. 6-5, the non-carcinogenic PRGs were reduced by a factor of 10 to account for the possibility that multiple chemicals may act synergistically. The screening value in Table 6-11 (p. 6-33) appears to reflect this, since a value of 40 mg/kg is listed. However, by comparing the mean lead concentration (78.5 mg/kg) to the un-adjusted PRG of 400 mg/kg (P. 6-14, second paragraph) the conclusion was reached that there is no unacceptable risk of lead (P. 6-18, first paragraph). Comparing the mean lead concentration to the adjusted screening PRG listed in Table 6-11 would yield a Hazard Quotient (HQ) greater than the target of 1.0. In addition, the average lead concentration is also much higher than the maximum background concentration. We suggest that you reevaluate your conclusion about there being no unacceptable risk from lead.

Table 6-11 (P. 6-33):

It is unclear to us why are maximum and not mean background concentrations presented are and used for comparisons to detected concentrations? We recommend you provide clarity here.

P. 6-18:

The document implies that the elevated iron concentration found at location NDW06SS01 occurred because that location is adjacent to piles of rusted metal debris observed at location NDW06SS23, which also had an elevated iron concentration. However, the two sampling locations are not exactly “adjacent”; appearing approximately 80 feet apart (see Figure 6-2).

P. 6-19:

The document states that none of the wells at the site had dissolved antimony levels above the PRGs. However, according to Table 6-12, one location had a detected dissolved antimony concentration of 1.8 ug/L, which is above the PRG (1.46 ug/L, according to the table). We suggest that the statement be revised.

P. 6-19 to 6-20:

Despite the fact that Cd was not found in elevated concentrations in soils, all Cd concentrations detected in groundwater are above base-wide background levels and PRGs. Additionally, given the degree to which groundwater flows throughout the site (described in previous sections of the report), it seems possible that elevated groundwater concentrations could occur some distance from the source. Therefore we suggest that it might be precipitous to discount the importance of the potential hazard from Cd at the site.

P. 6-20:

The document states that because the soil sampling locations with elevated concentrations of iron do not correspond to the groundwater sampling locations with elevated concentrations of iron, groundwater iron levels do not appear to be related to waste disposal activities at the site. However, given the degree to which groundwater flows throughout the site (described in previous sections of the report), it seems possible that elevated groundwater concentrations could occur some distance from the source. We suggest it might be premature to discount the importance of elevated iron concentrations at the site.

P. 6-21:

The document states that the elevated concentration of Tl found in sediment from location NDW06SD02 may not be site related, “as these samples are collected from the canal that is

separated from the site, near the metal bridge.” This statement is confusing; since the site maps indicate that this sampling location is within the boundary of SWMU-6 (see Figure 6-5). Additionally, how is the canal separated from the site? We recommend clarification or revision of the statement.

Comments on Ecological Risk Assessment (ERA) for SWMU-6, Vieques, Puerto Rico

Overall Comments:

Generally, the ERA was performed following standard and accepted practices for analytical procedures and the selection of benchmark values, food chain model parameters, and toxicity reference values. However, we note that there are a few areas which lend uncertainty as to whether the overall conclusions and recommendations (that conditions at SWMU 6 do not pose an unacceptable risk to ecological receptors) are credible. The primary reasons for this uncertainty are:

1) Selection of receptor species:

The receptor species chosen were not generally those that would provide the most conservative estimates of risk. First, the Service suggests that the use of species with lower body weights would have provided higher mass-specific food ingestion rates. Second, we suggest that the use of receptors with more restricted diets (e.g., herbivores or invertivores) in addition to omnivorous species would have provided a more complete assessment of risk, since the risk to receptors could be diluted by the inclusion of multiple food sources, if only some food sources contain high quantities of contaminants. In addition, the food web models for the spotted sandpiper did not use the most conservative soil ingestion estimate.

2) Use of mean HQs in Baseline Ecological Risk Assessment (BERA)

Instead of looking at the hazard quotient obtained for each location separately, the average HQ was calculated across all sample locations. If the average was < 1.0 , it was assumed that risks are very unlikely. However, it is possible (and did indeed occur) that COPC concentrations indicate that risk is present at some locations and not others. The authors argue that since they are concerned with population-level effects, and since a conservative area use factor (AUF) of 1.0 was used, average HQs are more realistic than location-specific HQs. We suggest that it seems possible; however, that exposure to high concentrations at a particular location could present risk to even wide-ranging assessment endpoints, depending upon the distribution of resources relative to COPC concentrations.

3) Use of Lowest Observable Adverse Effects Levels (LOAELs) only in determining HQs in BERA

When hazard quotients were calculated, risk was assumed to exist only if LOAEL values were exceeded; No Observable Adverse Effects Levels (NOAELs) were only considered in the Screening Level Ecological Risk Assessment (SLERA). However, the LOAEL is not necessarily the lowest possible adverse effect level, only the lowest reported in the literature used as the source of toxicity reference values. In reality, the lowest adverse effect level is likely somewhere between the LOAEL and the NOAEL. Therefore we recommend, to be protective of sensitive species, HQs calculated using the NOAEL that are > 1.0 should be considered in estimating potential risk.

4) Overlooking the significance of calculated HQs and/or comparisons to background concentrations.

In several cases, HQs > 1.0 were discounted as significant (e.g., when the average concentration was close to the background concentration or where HQs were “close to” 1.0), or, when HQs could not be calculated because screening values were not available, the occurrence of elevated sample concentrations relative to background was discounted as significant (e.g., where elevated concentrations were not “widespread”), leading to the conclusion that there was not an unacceptable level of risk at the site. However, the whole point of using HQs and background concentrations as tools to evaluate risk is to provide a quantitative, unobjective assessment. Therefore we suggest that discounting evidence of risk based on these quantitative approaches seems unjustified.

5) The results of back-calculating minimum contaminant concentrations in crabs that would pose an unacceptable risk to birds foraging at the site from food web models cannot be replicated.

The ERA states that calculations indicated no unacceptable risk exists to avian receptors eating crabs containing the measured concentrations of contaminants. However, the toxicity reference values (NOAELs) used to perform those calculations were developed for other bird species, and were not the most conservative NOAELs listed in the ERA. We suggest that if more conservative NOAELs are used, calculations indicate that risk from DDE and Pb may exist to avian receptors at the site via consumption of crabs.

6) Several errors or discrepancies were found in the report, either between literature sources and data values presented in the ERA or within the document itself. In some cases, for example, additional COPCs should have been retained in the screening level risk assessments. We suggest that the ERA process be completed for all the contaminants with noted discrepancies and then reexamining the new list of COPCs.

SPECIFIC COMMENTS ON VIEQUES SWMU-6 ECOLOGICAL RISK ASSESSMENT:

P. 7-5:

The report states that the species composition and structure of the site and the background location were similar, that no stresses were observed in the plant communities, and that there was no visible evidence that the SWMU 6 site has had an impact on wildlife or wildlife habitat. However, no data are presented to support these statements. We recommend that visible evidence of adverse impacts on a 26 year old site should not be a criterion.

P. 7-9:

The report states that SWMU 6 lacks significant habitat for amphibians and reptiles; however, according to Table 7-1 (P. 7-27), *anolis* lizards were observed at the site. We suggest that since the site is an estuarine mangrove it provides more habitat for amphibians and reptiles than stated in the report and we request that the statement regarding lack of habitat for amphibians and reptiles be reevaluated.

Table 7-4 (P. 7-29):

Selection of receptors: Why was the red-tailed hawk selected as the terrestrial avian carnivore receptor? We suggest a more conservative (i.e., more in line with the first and fourth stated criteria) choice would have been the peregrine falcon, since the peregrine falcon is potentially present on site, and the peregrine falcon, being considerably smaller than the red-tailed hawk, would be expected to have higher mass-specific ingestion rates. Another raptor species which are resident and found in the area would be the American kestrel a permanent resident in Puerto Rico and Vieques. We suggest that any species of omnivorous mouse would likely represent a more conservative model than the Norway rat or Indian mongoose. Depending on the contaminant, a more limited diet may result in higher risk estimates, depending upon the bioaccumulation factors of contaminants in various dietary items.

Table 7-6 (P. 7-33):

The food chain models used in the RI/FS for SWMU-6 used only the biological accumulation factors (BAFs) for omnivorous small mammals (see Sample et al. 1998b) to estimate dietary ingestion of contaminants in carnivorous birds. We note this may not be the most conservative assumption. Depending upon what food item carnivorous birds are assumed to consume, food chain models could end up with very different results, due to the differences in bioconcentration factors in the different types of small mammals. For example, Sample et al. (1998b) list an uptake factor for Cd for soil invertebrate-feeding small mammals that is much higher than that listed for omnivorous small mammals; if shrews were assumed to be consumed by carnivorous birds, higher estimates of risk may have resulted. We recommend it may make more sense to use either the general uptake factors (UFs) (since carnivorous birds are likely to consume whatever small mammals they can get), or, to be conservative, use the highest soil-small mammal UF listed for each contaminant in Sample et al. (1998b).

P. 7-12 & Table 7.7 (P. 7-35):

The sediment to invertebrate BAF values taken from (Bechtel Jacobs Co. 1998a) for As and Hg were not based on depurated worms, but were instead based on both depurated and non-depurated worms. We suggest that values for depurated worms (which happen to be higher than values for all worms combined for As and Hg), should be used for all analytes. In addition, we recommend that maximum surface water contaminant concentrations should be compared to benchmarks (in addition to sediment chemical concentrations) to assess risk to fish communities. The references that exist for water to fish BAFs (for Cd and nickel (Ni)) are much higher than the sediment to fish BAFs used in this ERA.

Table 7-8 (P. 7-37 to 7-38):

Note that several of the references for the values cited (e.g., Dunning 1993; U.S. EPA 1993a,b) are not the original sources. We suggest checking the original references to verify that cited values are correct.

Additionally, it is not clear how the values were arrived using allometric equations. The values do not appear to check out. This is evident even without calculating the ingestion rates, by comparing just the intake rates (e.g., the conservative and central tendency water ingestion rates for the green heron are 0.0227 and 0.0209 L/d; if these were calculated using the corresponding conservative and average body weights [0.158 and 0.212 kg], the second water ingestion calculation would have to yield a larger value, as a result of using the heavier body weight. This is true for food ingestion rates using allometric equations as well). If ingestion rates were expressed on a per unit of body weight basis, then you would expect the conservative value to be higher, but according to the table legends, that is not how ingestion rates are expressed. Perhaps the maximum body weight, rather than the minimum body weight was used, to calculate maximum ingestion rates. We suggest that these calculations be verified and if they are incorrect they be corrected.

Finally, on this page, a soil ingestion rate of 18 percent is listed for the spotted sandpiper, which is said to have come from Beyer et al. (1994). However, the source document does not list a soil ingestion rate for the spotted sandpiper. The value of 18 percent may be the average of the values for all sandpipers listed in Beyer et al. (1994). Since the object in the screen is to be as conservative as possible, we suggest it would make more sense to either model the spotted sandpiper using the most conservative soil ingestion rate for all sandpipers listed (30 percent for the semi-palmated), or to use the semi-palmated sandpiper (which, is an abundant visitor to Puerto Rico throughout the year) in the food chain models.

Table 7.9 and 7.10 (P. 7-39 to 7-46):

The values from Sample et al. (1996), the reference cited for some of the ingestion screening values (LOAELs and NOAELs) for mammals and birds (although, as for the food chain model values, this document is a compilation of results from various sources and so is not the primary literature citation) does not contain all the values used. While we did not check every compound, we couldn't find some of the values used - e.g., the NOAEL for As and the LOAEL for Cr. We suggest that ingestion screening values be verified and correct references added as necessary.

Tables 7-11 to 7-14 (P. 7-47 to 7-68):

What does the shading indicate in these tables?

Table 7-14 (P. 7-67) is out of order, coming after Table 7-15 (P. 7-66).

P. 7-15:

In the Food Web Exposures section, the document states that, "Three pesticides and six Poly Chlorinated Byphenyls (PCBs) were retained as COPCs because the maximum reporting limits exceeded screening values." This text is confusing, since food web exposures are estimated by comparing an ingested dose to NOAELs and LOAELs, rather than using screening values directly. Perhaps the maximum reporting limits for detections below quantification limits as the input into food chain models, and compounds with resulting HQs greater than 1.0 (i.e., dose estimates were greater than toxicity reference values (TRVs)) were retained as COPCs. Therefore, we suggest the text should be clarified or be corrected to better reflect this. Additionally, it is stated that three pesticides were retained as COPCs for the reasons just stated; but hexachlorobenzene was not mentioned, which also should be retained based on HQs > 1.0.

P. 7-16 and Table 7-15 (P. 7-66):

No compounds are indicated in Table 7-15 as being retained for soil (compounds that should be indicated as being retained are listed in the text on P. 7-15). The compounds 4-bromophenyl-phenylether, 4-chlorophenyl-phenylether, hexachlorocyclopentadiene, hexachloroethane, and 1,1,2,2-tetrachloroethylene should not be indicated in this table as being retained, since undetected chemicals without screening values were not identified as COPCs (see last paragraph on P. 7-15).

We recommend summary tables of COPCs retained for sediment and water (similar to Table 7-15, for soil) be presented.

Table 7-16 (P. 7-78):

It is not clear where the values used for the soil to plants, soil to rat, soil to invertebrate, or sediment to invertebrate BAFs originated (cited as being taken from Bechtel Jacobs Co. 1998a,b and Sample et al. 1998a,b). The document states that “central tendency estimates (e.g., median or mean)” were used. Very few of the numbers match the means or the medians listed in the source documents. It is also not clear why the median was chosen for some compounds and the mean for others. We suggest using the values in the source documents to reevaluate the risk estimate unless some compelling reason is suggested for not doing so.

Table 7-18 (P. 7-71):

The food ingestion rate of the red-tailed hawk is listed as being lower than the food ingestion rate of the green heron. This might make sense if the food ingestion rate was listed on a kilogram per kilogram (kg) body weight per day basis, but it is not; it is listed on a kg/day basis. A 1.13 kg hawk would have to consume more per day than a 0.212 kg green heron. We suggest clarification or correction of these numbers.

P. 7-18:

An AET is by definition, the Apparent Effect Threshold; a concentration at which, “a particular adverse biological effect is always expected” (Barrick et al. 1988). Thus, compared to threshold effects levels (TELs) and effects range-low (ERLs), the AETs are fairly high, generally in the range of or higher than the probable effects levels (PELs) and effects range-medium (ERMs). We note AETs are not conservative screening levels and should not be presented as such.

The ERA states that literature screening values could not be found for the several detected volatiles of concern (VOCs); therefore the measured concentrations were compared to the lowest value among screening values for other VOCs. However, screening values for acetone, carbon disulfide, ethyle benzene, methyl ethyl ketone, methylene chloride, and toluene can be found in Jones et al. (1997). It is possible that the screening values for these compounds presented in Jones et al. (1997) were not used because they were derived for freshwater sediments, using equilibrium partitioning (EqP) methodology. However, U.S. EPA Regions 1 and 9 both apply them to marine sediments, and other benchmarks based in part upon freshwater sediment data (from Long and Morgan 1990) were used in this ERA. Using the Jones et al. (1997) screening value for acetone (0.0087 mg/kg) and carbon disulfide (0.85 mg/kg), calculated HQs would be 69 and 16 (acetone) and 26 and 11.4 (carbon disulfide), using the maximum and average concentrations, respectively. Therefore, we suggest that neither acetone nor carbon disulfide should be excluded from consideration as a CPOC.

P. 7-19 to 7-21:

Much of this section compares site COPC concentrations to mean background COPC concentrations; however, in many cases, the degree to which site COPC concentrations exceed mean background concentrations is not evident either from the text or from the corresponding table (7-24), in which only maximum upper threshold limits (UTL) background concentrations are presented. In addition, the background concentrations for Al are not presented, so statements comparing site specific concentrations with background for this COPC cannot be verified.

Evidence of risk from the remaining analytes (soil: Copper (Cu), Fe, Pb, Zn, DDD, DDE, DDT, and six Poly Aromatic Hydrocarbons (PAHs); surface water: Al, Arsenic (As), Ba, Co, Cu, Fe, Pb, Mn, Mn (dissolved), Hg, Ag, Tl, V; sediment: Al, As, Ba, Be, Co, Cu, Fe, Mn, Se, Tl, V, DDD, DDE) is explained away in every case. For example, the risk from some analytes is said to be low if there were only a few areas that exceeded background; if conservative benchmarks are exceeded they are replaced with less conservative values which are not exceeded; if dissolved metal concentrations do not exceed benchmarks, total metal concentrations that do exceed benchmarks are said to be ecologically unimportant; it is even argued that if the maximum values for some analytes are ignored, the re-calculated mean values would be lower than the HQs. We suggest several reasons these arguments are specious:

- In reality, the document is not using the HQs as thresholds to indicate if risk exists; the document implies that because the HQ's are "close to 1" risk is expected to be low. The very reason HQs are used is to quantitatively assess risk and avoid subjectivity.
- One reason for sampling different areas within a site is to determine if and where there are "hot spots". If there are localized areas where risk is shown to exist, remediation efforts can focus on those hotspots, but the risk should not be discounted. There obviously will be variation in COPC concentrations across a site; by sampling at several locations, the likelihood that an accurate depiction of risk at the entire site is obtained is maximized. If only areas that do not show risk above threshold levels are considered, the assessment of risk is biased.
- There are several reasons not to replace conservative benchmarks with less conservative ones after completing the analysis (e.g., replacing soil benchmarks for plants with soil benchmarks for invertebrates, as done in the document). First, conservative benchmarks protect sensitive species. If less conservative benchmarks are used, some species present at the site may not be protected. Second, it has not been conclusively demonstrated that the vegetation at the site was comparable to that at the reference location, or that the plant community has not been affected by COPCs at the site. Third, one reason benchmarks are selected *a priori* is to prevent bias. Benchmarks should not be replaced after analysis for "better" benchmarks which conveniently sweep the risk away. If the optimum benchmarks were not selected beforehand, then the study design was flawed.

- Total metal concentrations exceeding maximum benchmark concentrations may still present risk to ecological receptors. Elevated concentrations of total metals could result in elevated concentrations of dissolved metals under certain circumstances, depending upon water and soil quality parameters.
- Even if the premise that maximum COPC concentrations are not important at a population-wide and site-wide basis is accepted, several contaminants had mean concentrations that exceeded background and benchmark values. If the average HQ is greater than 1.0, it is an even stronger indication (compared to maximum HQs being greater than 1.0) that a population-level risk may exist. To state that risk is low because HQs are “close to” 1.0 when, in fact, the HQ is greater than the threshold of 1.0, negates the fundamental objectivity of using HQs to assess risk.
- For several analytes (soil: Cu, Pb, Zn, PAHs; surface water: Mn, Mn (dissolved), Hg; sediment: As, Ba, Cu, DDD, DDE), not only were **average** HQs greater than 1.0, but **average** (not maximum) concentrations **also** exceeded the **maximum** background concentrations. Some analytes also had a fairly high frequency of exceedence of background UTLs, with soil Pb and Zn showing 15 and 10 exceedences, respectively, out of 23 total samples (see Table 7-24) and Cu in sediment showing 10 exceedences (not counting 6 exceedences for which sample COPC concentrations were estimated) out of 19 total samples (see Table I-2 in Appendix I). To state that risk is negligible seems to be ignoring all evidence to the contrary.

P. 7-21:

The document states that concentrations of metals in sediment samples taken in 2003 at station NDW0SD02 were much lower than samples taken at that location in 2000, with 2003 sample concentrations of Sb, As, Ba, Be, Cd, Co, Cu, Pb, Hg, Ni, Se, Ag, Tl, and V being at or below background and screening values (where available). It goes on to say that, “Additional sediment samples were also collected in September 2003 in the immediate vicinity of NDW06SD02, including NDW06SD10, -11, -12, and -13, and analytical results were comparable among these stations”, implying that the concentrations of the metals listed above were also below background and screening values at locations NDW06SD10, -11, -12, and -13. However, this was not always the case: Cu concentrations were above background and screening values at locations NDW07SD10, NDW07SD11, and NDW07SD12. Additionally, concentrations of Al, Fe, and Mn were greater than background at NDW07SD10 and NDW07SD13, and concentrations of Fe and Mn were greater than background at NDW07SD12 (sediment concentrations of Al, Fe, and Mn could not be compared to screening values, since screening values were not available for these compounds). In fact, even when excluding NDW07SD02 from calculations, the mean concentrations of these compounds at the site exceed background concentrations; in the case of Cu, the mean concentration exceeds the screening value as well. It

is not unlikely that sediments have shifted slightly over the three year period; the apparent drop in contaminant concentrations at location NDW07SD02 from 2000 to 2003 may be due simply to the movement of contaminated sediments (or contaminants in sediment) rather than to erroneous readings having been obtained during the 2000 sampling event. We suggest that you do not drop the results from that event from the risk assessment.

The document states that potential toxicity of Be and Tl could not be quantitatively evaluated because screening values are not available for these compounds; this is true of Al, Fe, and Mn as well and needs to be noted in the document.

P. 7-22:

DDD, DDE, and DDT concentrations in sediment were found to be higher than screening values at some sampling locations. The authors then modified the screening values, replacing the procedure quantitation limits (PQLs) with the ERMs, and re-calculated the HQs. As mentioned above, it is not sound science to change the screening values *a posteriori*, as a means of reducing the HQ. Additionally, the ERM is the value at or above which effects would frequently occur (Long and Morgan 1990). Therefore, if the ERM is to be used in lieu of the PQL, an HQ equal to or greater than 1.0 would indicate effects *are likely*. To claim that HQs of 2.2 (for DDD) and 1.1 (for DDE) obtained using the ERMs are sufficiently close to 1.0 to warrant removing these compounds from further consideration is should be reconsidered.

The ERA states that six VOCs were identified as PCOCs because no screening values were available for these chemicals. However, according to Table 7-26, a total of eight VOCs for which screening values are not listed were detected: acetone, carbon disulfide, ethylbenzene, m,p-xylene, methyl ethyl ketone, methylene chloride, o-xylene, and toluene. In addition, the ERA states that since these VOCs are not related to solid waste materials discarded at the site, and since "many" are common lab contaminants, their presence may be an artifact of sample handling. However, it is unclear how it can be known with certainty that they are not related to solid waste discarded at the site, since there is little documentation of what or how materials were dumped there. Furthermore, if their presence is an artifact of handling, these VOCs should have been picked up in blank samples analyzed contemporaneously with site samples, which would have resulted in the qualification of field samples during the data quality evaluation. While this was the case for acetone, methylene chloride, m,p-xylene, and toluene, this was not the case for carbon disulfide, ethylbenzene, methyl ethyl ketone (2-butanone), or o-xylene.

Additionally on this page, the discussion of food web exposures only focuses on selenium, which the document states is identified as a PCOC because the NOAEL and LOAEL were both greater than 1.0 for the spotted sandpiper. However, the HQ calculated from NOAELs needs to be taken into account as well. The LOAEL is the lowest observed effect level, not necessarily the lowest effect level. The actual lowest effect level is expected to be somewhere between the LOAEL and the NOAEL; therefore, basing conclusions on LOAELs alone may underestimate risk to sensitive

species. Thus, as indicated in Table 7-23 (P. 7-82), Zn and Hg should be identified as PCOCs in the BERA.

Although a summary of HQ values obtained using food chain models are presented in Tables 7-14 (P. 7-67 to 7-68, step 2) and Table 7-23 (P. 7-82, step 3), the actual models, including input parameters, could not be found anywhere in the document or the appendices. While Table 7-10 (P. 7-43 to 7-46) presents the LOAELs and NOAELs obtained from the literature for use in the food chain models, there is no indication as to which specific sets were used in calculating HQs. For example, four sets of LOAELs and NOAELs are presented for DDD in birds; three of these were based on effects on reproduction and growth, which the text states were preferentially selected as endpoints. We recommend that the document indicate which of these three sets of LOAELs and NOAELs were used to calculate HQs so the conclusions drawn from HQs can be adequately evaluated.

P. 7-23:

The document states that they used the food web model to back-calculate the highest concentrations of COPCs (DDE, DDT, Cd, Pb, and V; identified in the DOI report) in crab tissues that would pose no adverse effects to three species of wading birds (great blue heron, yellow-crowned night heron, and green heron) consuming a diet solely of crabs. The value they obtained for DDE was 1.03 mg/kg (dry weight). For the green heron, this value appears to have been derived using the conservative body weight and food ingestion rate (from Table 7-8, P. 7-37) in conjunction with the NOAEL for the bald eagle (Table 7-10, P. 7-43) (note that body weights and food ingestion rates for the great blue heron and yellow-crowned night heron are not provided in the ERA). Although the NOAEL for bald eagles was likely chosen because this species is somewhat piscivorous, species-specific differences in susceptibility to contaminants, combined with the fact that bald eagles and wading birds are quite different physiologically and taxonomically, makes it not unlikely that the responses of wading birds and bald eagles to COPCs are substantially different. Therefore, we suggest that the most prudent and conservative approach to back-calculating the threshold above which adverse effects could occur would be to use the highest NOAEL measured among all birds, which in the case of DDE would be a value of 0.08, measured for the barn owl (according to the list presented in Table 7-10). If such a calculation is done, the resulting maximum threshold value for the concentration of DDE in crab tissue is 0.276 mg/kg, a value substantially lower than maximum concentrations measured. Similarly, the threshold value for Pb appears to have been calculated using the NOAEL for the kestrel. In addition to the significant taxonomic and physiologic differences that exist between the kestrel and wading birds, the diet is substantially different as well. Because it is impossible to know with certainty whether herons would be more or less susceptible to contaminants than the kestrel, we suggest it would be prudent to use the most conservative NOAEL available; that for the Japanese quail. Performing the back-calculations using the NOAEL for the Japanese quail results in a maximum dose threshold for Pb of 3.9 mg/kg, which again is substantially lower than the maximum concentration of Pb measured in crabs.

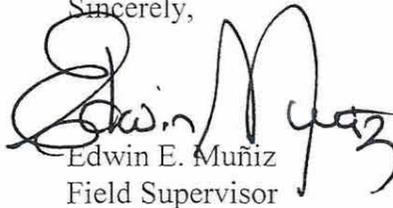
There are no toxicity reference values for vanadium listed in Table 7-10. They appear to have been inadvertently left out, since there are values presented in Sample et al. (1996), which, when used to back-calculate the threshold value for the concentration of V in crab tissue, yields a threshold value of 39.5 mg/kg as is indicated in the ERA. We suggest the correction of Table 7-10 to include the NOAELs and LOAELs for V.

Appendix L, Tables 2.1 to 2.5:

There are several compounds with COPC flags equal to "yes" where the rationale for deletion or selection is identified as "BSL" (below screening level). Other compounds have COPC flags of "no", with the rationale being identified as "ASL" (above screening level). We suggest clarification or correction of this text.

Thank you for the opportunity to comment on this action, if you have any questions please contact Felix Lopez at 787 851-7297 x 226 or Richard Henry US Fish and Wildlife Service, Environmental Response Team, at 732-906-6987 (office), or 973-204-5825 (cell).

Sincerely,



Edwin E. Muñiz
Field Supervisor

cc:

EPA, Vieques

FWS, Vieques

John Tomik, CH2MHILL ✓

Yarrisa Martinez, EQB, San Juan

Brett Doerr, CH2MHILL

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