NOTICE OF DATA AVAILABILITY (NODA)

RESPONSE TO COMMENT DOCUMENT

PART I

June 1998

U.S. Environmental Protection Agency
Office of Solid Waste
401 M Street, SW
Washington, DC 20460

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General Comments

Comment 1: The National Petroleum Refiners Association (NPRA) supports EPA's decision which is based on both the initial and new data analyses not to list eleven of the fourteen residuals identified in this rulemaking. However, NPRA does not believe that the new data analyses demonstrate that the risks from the CSO sediment, hydrotreating catalysts, and hydrorefining catalysts warrant their listing as RCRA hazardous wastes. EPA overstated the potential risks because of errors in the methodology used to determine the risks as described below (NPRA, 00004, pg 1):

a)   EPA's Monte Carlo analysis conducted with the new data supports a no-listing decision for Hydrotreating and Hydrorefining Catalyst . The Monte Carlo analysis is recognized by EPA as a superior technical approach to the deterministic approach which EPA used to make the listing determination. EPA should use results of the Monte Carlo risk analysis as basis for the listing decision. The deterministic analysis, on the other hand, does not allow EPA to determine what percentile of the risk distribution is represented by the high-end analysis. This information is necessary to make a defensible listing decision.

b)   EPA failed to follow its own policy decision by using the OSW database to determine the distance between landfills and groundwater wells. EPA previously stated that existing data such as the OSW database was not acceptable because it was too general and was not specific enough for the refining sector which is the focus of this rulemaking. As EPA stated at the start of this rulemaking, the OSW database is not refinery specific. On the other hand, the Section 3007 survey is specific to this rulemaking.

Response: (a) The Agency agrees with the commenter that the Monte Carlo approach allows the EPA to determine the percentile of the risk distribution represented by the two high-end analysis. However, both the two high-end parameter and the Monte Carlo analyses show risk above the discretionary range for waste streams proposed to be listed based on the groundwater pathway analysis (see Table 5.7 in the Additional Groundwater Pathway Risk Analysis, Supplemental Background Document, USEPA, 1998; risks for benzene and arsenic in hydrotreating catalyst and hydrorefining catalyst and for benzene in crude oil tank sediment are all above 1E-05).
(b) For responses to comments in paragraph (b), see discussion in Section I.A.6.a.

Comment 2: Comments on Notice of Data Availability

NPRA comments on March 14, 1996 supported EPA's original proposal not to list eleven residuals: sludge from sulfuric acid alkylation; catalyst from sulfuric acid alkylation; Claus and Scot-like catalyst; catalyst from catalytic reforming; catalyst and fines from catalytic cracking; crude oil storage tank sludge; sludge from sulfur complex and H_2S removal facilities; spent caustic from liquid treating; unleaded gasoline storage tank sediment; off-specification product and fines from thermal processes; and sludge from HF alkylation. EPA's original proposal was supported by sound data and a conservative risk assessment.

The new analysis conducted by EPA and provided in the NODA does not demonstrate any reason to change EPA's original decision. In fact, the results of the Monte Carlo risk analysis strengthen EPA's decision not to list these eleven residuals even considering co-disposal. Further, the new analysis on additive risks did not identify any conditions where individuals would be simultaneously exposed to residuals being managed in a landfill or land treatment unit. (NPRA, 0004)

Response: The Agency appreciates the commenter's supportive comments. The Agency has conducted additional analyses that are presented in the Additional Groundwater Pathway Supplemental Background Document[1]. The final decisions are based on those results as discussed in the preamble to the final rule.

For waste-specific comment responses, see Section I.A.5. For co-disposal responses, see Section I.A.3. For additive risk responses, see Section I.C.2 of this Response to Comment Document and Section III.R of the Proposed Rule Response to Comment Document.

Comment 3: EPA Has Ample Support for its Proposal That Eleven of the Fourteen Refinery Residuals Considered for Listing Do Not Warrant Listing

As part of the NODA, EPA conducted several new analyses, including evaluation of co-disposal and additive risk scenarios. EPA's proposal not to list sludge from sulfuric acid alkylation, catalyst from sulfuric acid alkylation, crude oil storage tank sludge, Claus catalyst and SCOT-like catalyst, catalyst from catalytic reforming, catalyst and fines from catalytic cracking, sludge from sulfur complex and H_2S removal facilities, spent caustic from liquid treating, unleaded gasoline storage tank sediment, off-specification product and fines from thermal processes, and sludge from HF alkylation is adequately supported by these new analyses as well as the original listing proposal. As discussed in API's comments of March 21, 1996, EPA's proposal not to list these eleven residuals is supported by sound data and a conservative, iterative risk assessment process. Additionally, as discussed in API's March 1996 comments, some of these residuals are often used in such a manner that they are not solid wastes and, therefore, are not appropriate candidates for listing.

The new analyses clearly support EPA's no list decision for the eleven residuals listed above. In the case of additive risks, EPA could find no circumstance which would suggest that individuals would be simultaneously exposed to residuals managed in an on-site landfill or land treatment unit. Similarly, for co-disposal, EPA's analysis emphasizes the fact that these eleven residuals do not warrant listing, even with great conservatism in the risk assessment. EPA's Monte Carlo analysis strengthens these results and further highlights the conservatism in EPA's risk assessment for both the high-end analysis and the co-disposal scenario. In fact, the Monte Carlo analysis

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[1]Additional Groundwater Pathway Risk Analysis, Supplemental Background Document, USEPA, 1998.

illustrates that the risks predicted by the high-end analysis clearly fall at levels greater than the 95th percentile.

In summary, EPA's analyses do not demonstrate that these eleven residuals pose a substantial hazard to human health and environment as required for listing by the 261.11(a)(3) listing criteria. (API, 00009)

Response: The Agency acknowledges the commenter's supportive comments. However, EPA has decided to list crude oil storage tank sediment as discussed in the preamble to the final rule.

For waste-specific comment responses, see Section I.A.5. For co-disposal responses, see Section I.A.3.

Comment 4: EPA Should Not List Any of the Refining Residuals Proposed for Listing

As detailed in API's comments on the flaws and errors contained in the revised NODA risk assessments, and in Mobil's and API's comments on the original November 20, 1995 proposal, EPA has failed to properly acknowledge and weigh the existing regulatory protections provided by the Toxicity Characteristic rule and other federal transportation regulations, and continues to rely on flawed risk assessments that overstate risk. As a result, the record in this rulemaking does not adequately support listing any of the three residuals proposed for listing. (Mobil, 00002, pg 2)

Response: EPA has refined its risk assessment analyses for the petroleum refining waste listing based on the latest approach and data. The two high-end parameter analysis has been confirmed using a full Monte Carlo approach, and the analysis continues to indicate that the waste streams proposed for listing show risk above the discretionary range. See other sections of this document for discussion of specific comments raised, and see also the preamble discussion in the final rule.

Comment 5: As discussed at length in these comments, while several positive changes in the modeling were made (i.e., EPA included non-ingestion risks associated with the use of contaminated groundwater, and attempted to evaluate a scenario involving the co-disposal of refinery wastes), the bulk of the modeling shortcomings remain. In most cases, EPA either ignored the previous criticisms or attempted to justify the errors with supplementary information. As a result, the NODA risk assessments are also fundamentally flawed. Moreover, in many cases, the NODA materials actually demonstrate the weaknesses in EPA's methodologies and rationales.

For example, the NODA materials prove the ineffectiveness of the TCLP on oily and tarry wastes, document the potential for free-phased flow of contaminants in refinery landfills, demonstrate EPA's landfill active life assumptions are extremely and arbitrarily short (thus understating associated waste volumes substantially), and reveal co-disposal occurs to a much greater extent than EPA has considered.

In addition, EPA did not conduct risk evaluations for waste management practices such as storage in piles, road spreading, and surface impoundment management that are plausible mismanagement practices associated with many wastes in this rulemaking and refinery practices generally. Consequently, EPA's proposed no-list decisions continue to be based upon incomplete risk analyses.

Accordingly, the fact remains that a proper risk evaluation of the refinery wastes in the instant rulemaking would result in the listing of at least eight wastes as hazardous. EPA's groundwater modeling methodology remains inappropriate for the instant rulemaking because it fails to address the potential for the free-phased flow of contaminants within the landfill and upon leaving the landfill, and does not take into account existing contamination when evaluating potential risks. Moreover, even using EPA's inappropriate methodology, revising several incorrect modeling parameters would substantially change EPA's modeling results and warrant the additional hazardous waste listings. These conclusions are now supported by additional groundwater risk analyses, and data from groundwater experts on the potential for free-phased flow of contaminants at refinery sites, as introduced in the next section of these comments and discussed in greater detail throughout this document.

Finally, these comments address the propriety of several exemptions previously proposed by the Agency. In 1995, EPA proposed an exemption for oil-bearing residuals inserted into the coking process, but as discussed below, neither the NODA materials nor the 1995 proposal adequately address the well established legitimacy recycling criteria necessary to warrant such a broad, poorly defined exclusion, particularly when the residuals are used for quenching purposes. In addition, EPA proposed an exemption for the placement of certain listed sludges into refinery wastewater treatment systems (the so-called Headworks Exemption). While the NODA preamble includes an EPA acknowledgment the proposal as drafted was improperly broad, the exact intent of the Agency remains unclear. Specific approaches to appropriately limit the exemption are provided below. (EDF, 0006)

Response: EDF's individual comments (*e.g.* on Active Life, codisposal, and waste management, and use of TCLP issues) are addressed elsewhere in this document. The Agency's analysis of the petroleum residuals indicates that no free oil was present in any of the samples (Comment 7a in Section I.C.1); therefore, EPA's modeling methodology assuming aqueous phase transport of contaminants was appropriate. See Comment 8f in Section I.C.1 concerning findings in the Waterloo report. Please also see comment 1 in section I.A. 6g. for response to the issue of existing contamination.

Comment 6: The additional analyses included in the NODA are not adequately responsive to the criticisms of the risk assessment modeling EPA performed in 1995 and relied upon for its proposed negative listing determinations. They contain many of the same flaws, and additional shortcomings, that result in substantial understatements of the risks posed by the wastes at issue.

Correction of only some of these flaws produces risk results that justifying listing many of the refinery wastes as hazardous, as demonstrated by other groundwater modeling referenced in these comments.

In addition, even with the NODA materials regarding the proposed recycling exemption, EPA has yet to address many of the requisite findings needed to reach a valid legitimacy determination, including whether the alleged recycling activities are economically motivated largely by avoided disposal costs, whether the specifications for the residual are selected to meet a recycling purpose, whether the hazardous constituents in the waste contributes properties of value to or are actually necessary for the product, and whether adequate records are kept. The failure to consider these legitimacy criteria is particularly problematic in the case of the use of residuals in the coker for quenching purposes. (EDF, 00006, pg. 77)

Response: See Section I.A response to Comment 2 for detailed discussion of comments on additional groundwater pathway analyses performed for NODA. See Section II.B for discussion of the proposed recycling exemption.

Comment 7: In RCRA §3001(e), Congress directed EPA to "make a determination of whether or not to list" certain wastes, including petroleum refining wastes. EPA lists a waste as hazardous if "the waste is capable of posing a substantial present or potential hazard to human health or the environment when improperly . . . managed," after considering certain factors such as the "potential of [toxic] constituents . . . to migrate from the waste into the environment under the types of improper management" that are considered. 40 CFR §261.11.

In the Proposed Petroleum Refinery Waste Listing Rule, EPA failed to consider plausible mismanagement scenarios and to reasonably analyze the potential for toxic constituents in petroleum wastes to cause groundwater contamination from land disposal. As discussed below, the extensive re-evaluation that EPA conducted in response to public comments described in the NODA continues to violate the legal standard for making the listing determination required by RCRA § 3004(e). (ETC, 0005)

Response: The agency believes it has considered plausible management scenarios. Individual issues raised by ETC are addressed elsewhere in this document and the preamble for the final rule.

Comment 8: Phillips has participated in the development of comments by API. Phillips endorses and incorporates by reference those comments. There are several specific aspects of the NODA that Phillips wishes to particularly emphasize by submitting comments in addition to the comprehensive comments prepared by API. The Phillip's comment relate to:

•   EPA's proposal not to list eleven residuals is support by the NODA analyses

•   Groundwater pathway risk analysis oversights

- Need for listing tailored for only demonstrated high-risk management practices (conditional/listings/contingent management)

- Appropriateness of the proposed headworks exemption

Phillips urges EPA to reevaluate its risk calculations taking into account the technical comments made by API. Proper application of TC cap values, distances/dispersion from onsite landfills to receptor wells, the influence of biodegradation, and consideration of the time lag (>1500 years) for arsenic to encounter a receptor well (long after any risk from benzene would have occurred) are just a few of the changes that will likely result in anticipated risks that do not justify a hazardous waste listing for the three residuals proposed for listing in addition to those eleven residuals not proposed for listing. If after the risk assessment is complete, there still remains specific pathways that do pose a risk to human health or the environment, the listing should be tailored to regulate only those pathways or management methods of concern. Other methods of management (i.e., those without demonstrable risk) including under the toxicity characteristic. Phillips applauds EPA's approach in the recently-issued military munitions waste rule (62 FR 6622, 6655-56 (Feb. 12, 1997)) for distinguishing and allowing a contingent management regulatory approach. (Phillips, 00014)

Response: The Agency appreciates the commenter's views regarding the contingent management approach. The individual issues raised by API such as TC Capped analysis, well distances, dispersion of contaminants, biodegradation and retardation of arsenic are addressed later in this document.

I.   Additional Information

Comment 1: Revised Risk Assessments

EPA's proposal to list clarified slurry oil sediment and spent hydrotreating/hydrorefining catalysts is based on a flawed risk assessment. The following section identifies numerous flaws in the groundwater and Non-groundwater pathways of the risk assessment which should be corrected prior to a listing determination. API is confident that once these flaws are corrected and the risk assessment models are rerun, EPA will agree that the three refinery residuals currently proposed for listing should not be listed as hazardous wastes. (API, 00009, pg 10)

Response: EPA does not agree with the commenters claim, and responses to specific criticisms of EPA's modeling are presented in the appropriate sections of this response document. EPA notes, however, that the decision to list CSO sediment was not based primarily on groundwater pathway risk (see proposed rule 60 FR 57764, November 20 ,1995, and the NODA 62 FR 16747, April 8, 1997). Furthermore, EPA's revised analysis for individual risk using both a two high-end

parameter analysis and a full Monte Carlo approach are presented in detail in background documents.[2,3]

I.A.   Supplemental Background Document: Groundwater Pathway Risk Analysis: Petroleum Refining Process Waste Listing Determination

Comment 1: Relying on EPA data gathered by the Superfund program and other sources, University of Waterloo scientists document the contamination of groundwater with nonaqueous-phase liquids (NAPLs) resulting from waste management practices, including practices associated with refinery wastes and refinery sites.[4] As explained further below, this report is relevant to the instant rulemaking (and is thus provided for the record) because EPA's modeling assumes neither NAPL formation nor NAPL migration of contaminants once the contaminants reach the groundwater will occur, when in fact, NAPL contamination is an important part of an actual and potential plausible mismanagement scenario for the refinery industry.

Throughout these comments, groundwater modeling performed by King Groundwater Science, Inc. (KGS) is referenced. This groundwater modeling was performed for the Environmental Technology Council, and will be submitted to the docket by that organization.[5] KGS performed a series of modeling runs reflecting modified input parameters that would correct some of the deficiencies in EPA's methodology. The KGS modeling results indicate that the refinery wastes previously identified by EDF as appropriate for listing do pose substantial human health risks via the groundwater exposure pathway. (EDF, 00006, pg 4)

Response: EPA does not agree that consideration of NAPL formation and migration are important in this rulemaking. See Section I.A.6.g dealing with comments related to existing contamination and the Waterloo report. EPA also does not agree with many of the assumptions made in the KGS modeling analysis, as described in Section I.A.

Comment 2: KGS conducted a series of composite runs to determine the potential risks using the EPACMTP model using the adjustments discussed above. These composite runs more closely

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[2]USEPA. Additional Groundwater Pathway Risk Analysis, Supplemental Background Document, 1998

[3]USEPA. Additional Groundwater Pathway Risk Analysis, Supplemental Background Document, 1998

[4] Hubbard & Panders, Release of Dense Nonaqueous-Phase Liquids to Groundwater from Waste Disposal Areas: Part 1 (Case Studies), March 1997 (hereafter "Waterloo NAPL Report").

[5]Analyses Using EPACMTP to Estimate Groundwater Pathway Risks from Disposal of Petroleum Refinery Wastes, King Groundwater Science, Inc., July 7, 1997.

reflect the groundwater pathway risks posed by landfilling of refinery wastes compared to EPA's NODA efforts. In sum, KGS made the following groundwater model input revisions:

- Receptor well placed in the center of the plume line consistent with EPA listing determination policy and statutory mandate;

- TCLP values reflect the average benzene capture efficiency of 53% for all wastes;

- CSO landfill characteristics used as standard size for onsite landfills (except for the co-disposal scenario where the EPA modeled landfill was used);

- Median and high-end annual offsite waste volumes derived from facility-received volume data in the NODA were used where those volumes were higher than corresponding values EPA computed on a refinery-generated basis;

- In the co-disposal runs, annual median waste volumes were derived by including the estimated contribution of refinery wastes not covered by the listing determinations;

- A TCLP volume-weighted value was computed in the co-disposal cases by assigning a benzene value of 0.25 mg/l (half the TC characteristic threshold) to the codisposed wastes not covered by the listing determinations; and

- Total waste volume was computed based upon a landfill active life of 40 years.

KGS also conducted limited sensitivity analyses to determine whether EPA's high-end parameters were the most sensitive parameters given these input adjustments and the absence of EPA sensitivity analyses for unleaded gasoline storage tank sludge and the co-disposal scenarios. Since EPA did not conduct any analyses in the NODA testing the relative importance of a 30 year exposure period, this high-end value was included in the sensitivity analysis.

The risk results for the KGS composite runs demonstrate than many of the refinery wastes covered by this rulemaking warrant a listing determination. The following table compares the EPA and KGS composite modeling results for benzene:

Refinery Waste	EPA GW Risk	KGS GW Risk
Crude Oil Tank Sludge	2.7×10^{-5}	1.6×10^{-4}
HF Alkylation Sludge	4.3×10^{-6}	2.3×10^{-5}
Unleaded Gas Storage Tank Sludge	1.7×10^{-6}	8.8×10^{-5}
Co-disposal Onsite Landfill	9.3×10^{-6}	3.5×10^{-5}
Co-disposal Offsite Landfill	1.2×10^{-5}	3.9×10^{-5}

KGS Report at 17, Table 9.

In addition, a composite run for PAHs for Off-Spec Product and Fines disposed in onsite and offsite landfills resulted in risks >1×10^{-5}. KGS Report at 18, Table 10.

KGS also conducted several Monte Carlo analyses of crude oil tank sludge using the modified input parameters. One analysis, based upon 2,000 groundwater chemical transport simulations, yielded a 90% risk of 5.4×10^{-5}, a 95% risk of 1.2×10^{-4}, and a 99% risk of 3.6×10^{-4}, consistent with the deterministic results for this waste. The second analysis, based upon 10,000 realizations, yielded a 90% risk of 4.8×10^{-5}, a 95% risk of 1×10^{-4}, and a 99% risk of 3.7×10^{-4}. Accordingly, the Monte Carlo analyses essentially confirm the high risks associated with the deterministic modeling.

In the case of off-spec products and fines, and CSO sludge, hazardous waste listings are warranted based upon the PAHs found in these wastes. For the off-spec product and fines onsite scenario, KGS conducted high-end deterministic modeling for benz(a)anthracene using modified input values reflecting a standard onsite landfill (which in this case represented a reduction in size from EPA's modeled landfill area), increased volume (calculated for a 40 year active landfill life), and placing the well in the plume centerline. Setting high-end values for volume and distance to receptor well (as EPA did in its modeling), KGS found the resulting risk more than doubled from EPA's 2.1×10^{-5} to the revised 5×10^{-5}.

Moreover, as discussed above in Section II.A of the comments, KGS also computed the leaching values that would produce a 10^{-4} risk level using modified input parameters. The resulting leaching values were extremely small fractions of the waste concentrations, indicating such leaching values were more than plausible for these oily wastes.

Conclusion

KGS concluded: "This study indicates that for alternative disposal scenarios for five waste streams constructed suing realistic assumptions and decisions, higher modeled receptor well concentrations and associated risks were obtained compared to those reported by EPA in their groundwater pathways risk analysis." KGS Report at 21.

The groundwater pathway risks associated with land disposal scenarios are greater than 1×10^{-5} and clearly require the listing of crude oil tank sludge, HF alkylation sludge, unleaded gasoline storage tank sludge, and off-spec product and fines as hazardous wastes under RCRA § 3001(b) and (e). (ETC, 00005, pg 3 and 16, and EDF)

Response: The Agency completed some additional analyses to fully respond to comments based on the results of King Groundwater's analyses. While EPA revised some input parameters based on these comments, EPA did not agree with many of the assumptions in the KGS analysis. Responses to the issues of individual parameter revisions can be found elsewhere in this document as listed below:

*a)   Receptor well placed in the center of the plume line consistent with EPA listing
        determination policy and statutory mandate*

See Response to Comment 1 in Section I.A.6.e.

b)   TCLP values reflect the average benzene capture efficiency of 53% for all wastes.

See Response to Comment 6.a in Section I.C.1.

*c)   CSO landfill characteristics [were] used as standard size for onsite landfills (except for
        the co-disposal scenario where the EPA modeled landfill was used).*

See Response to Comments 1 and 2 in Section I.A.6.c.

*d)   Median and high-end annual offsite waste volumes derived from facility-received volume
        data in the NODA were used where those volumes were higher than corresponding values
        EPA computed on a refinery-generated basis.*

See Response to Comment 1 in Section I.A.3.

*e)   In co-disposal runs, annual median waste volumes were derived by including the
        estimated contribution of refinery wastes not covered by the listing determinations.*

See Response to Comment 1 in Section I.A.3.

*f)   A TCLP volume-weighted value was computed in the co-disposal cases by assigning
        benzene a value of 0.25 (½ its TC rule value).*

See response to Comment 5 in Section I.A.3.

g)   Total waste was computed based on a landfill active life of 40 years.

See response to Comment 1 in Section I.A.6.b for a discussion on the active live adjustments for
off-site landfills.

Responses to other issues raised can also be found elsewhere in this document as noted below..

(1) In response to this comment a high-end sensitivity analysis was performed for Unleaded
Gasoline Tank Sediment. The analysis is discussed in the revised risk analysis presented in
*USEPA, Additional Groundwater Pathway Risk Analysis, Supplemental Background Document,
1998* in the docket (results are summarized in Tables A.1 and A.2).

(2) While EPA did not perform a full sensitivity analysis for the high-end assessment for the projected co-disposal scenario, the Agency does not believe this is important. EPA performed a Monte Carlo analysis for the codisposal scenario, which indicates that co-disposal does not appear to cause significant risk. See Section I.A.3 for further discussion of codisposal.

(3) Exposure Duration: In the sensitivity to exposure duration presented in KGS's groundwater modeling report, 30-year exposure risks appear to be erroneously calculated by simply multiplying the 9-year exposure groundwater risk by a factor of 3.33, corresponding to the ratio of 30/9. The 9-year exposure risks presented in the NODA and in the proposal were calculated from the modeled maximum 9-year average receptor well concentrations and the 9-year exposure duration health-based numbers (HBNs). The more accurate approach to model 30-year exposure would be to calculate maximum 30-year average groundwater concentrations from the modeling results, and then calculate the risk based on 30-year health-based numbers. Maximum 30-year average well concentrations may be smaller than 9-year average well concentrations depending on the peak concentration period. By simply scaling from risks based on 9-year exposure, the extrapolated risk for a 30-year exposure is likely to be overestimated. EPA conducted its own sensitivity analysis for select representative waste streams including exposure duration as a varied parameter (see Appendix A B to Additional Groundwater Pathway Risk Analysis: Supplemental Background Document, U.S. EPA, 1998). Exposure duration was not a sensitive parameter in any of the sensitivity analyses, and the resulting two high-end parameter combination was not changed with inclusion of exposure duration as a sensitive parameter. Therefore, EPA does not believe this factor greatly affects the risk calculations.

The impacts of assumptions made in the KGS modeling on the listing decisions are discussed elsewhere in this document and in the individual waste sections (Section I.A.5). Other issues related to codisposal analysis are given in Section I.A.3.

Comment 3: Impact of Selected Groundwater Modeling Revisions

The KGS modeling results still substantially understate the risks posed by landfilling these refinery wastes. First, the modeling revisions still do not reflect the true leachability of the wastes since the modeled TCLP value still assumes no primary leachate generation and no free-phased contaminant migration, and thus understates actual benzene availability.

Second, the high-end waste volumes do not reflect a high-end landfill active life duration, since all volumes were derived using an average landfill life assumption. Third, the offsite median and high-end landfill size is very small when compared to comparable values in EPA's municipal landfill survey, and landfill characteristics employed in the carbamates listing determination.[6]

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[6] KGS frequently mentions the relatively small area of the standard offsite landfill as an important factor in its groundwater modeling results. Therefore, the arbitrarily small standard offsite median and high-end landfills assumed by EPA may be one of the most important deficiencies in EPA's modeling.

Fourth, the risk results do not take into account the contribution of pre-existing contamination at refinery sites, and therefore does not reflect actual contaminant exposures under plausible mismanagement conditions. Fifth, only a limited number of sensitivity analyses were conducted, particularly for the co-disposal scenario. High-end values for codisposed volume were not tested, a potentially significant omission considering the average annual onsite co-disposal increment attributable to other wastes not covered by the listing determination was seven times the median value, based upon EPA's NODA data as discussed above. Sixth and finally, the above risk estimates are for benzene contamination only, and do not reflect the risks posed by other contaminants in the wastes. (EDF, 00006, pg 46)

Response: The Agency does not consider the modified input parameters used by KGS in the modeling to be appropriate (see Comment 2 above); therefore, the increased risks shown in KGS's modeling are not appropriate. As for the modeling analyses of benz(a)anthracene, they were performed by KGS with inappropriately modified input parameters as discussed above. Individual responses to items 1 - 6 above are listed below:

1.   *The true leachability of the waste because the TCLP test does not assume primary leachate generation and free-phased contaminant migration;*

See responses to comments 7.a, 8.c, 7.f and 7.g in Section I.C.1.

2.   *High-end waste volumes that incorporate high-end landfill active life duration, since all volumes were derived using only an average landfill life assumption;*

The selection of a high-end active landfill life would make the analyses a three high-end parameter analyses and as such would have been overly conservative. As noted in the response to comments on estimating active landfill life (see Section I.A.6.B), the data for active life are limited, and EPA does not believe that adequate data for off-site landfills exist to allow the assumption of an appropriate high-end for this parameter. Further, EPA would typically use the median value if evaluating the sensitivity of the results to active life, but the median value also was not available. In the absence of adequate data, therefore, EPA decided to use the average estimate for landfill life.

3.   *Appropriate offsite median and high-end landfill areas;*

See response to the comment given in Section I.A.6.c.

4.   *The contribution of pre-existing contamination at refinery sites to receptor well contamination;*

See response to Comment 2 in Section I.B.

5.   *Risks posed by contaminants other than benzene in the waste; and*

The Agency does not know what other contaminants to which the commenter is referring. EPA evaluated risks from all chemicals of concern in the waste, either in bounding analysis or high-end/Monte Carlo analyses.[7] In addition, EPA notes that risks for individual contaminants practically are not necessarily additive, since peak concentration arrival times vary as a function of chemical transport properties. Therefore, consideration of only benzene is not overly conservative.

6.   Risks posed by contaminants other than benzene in the waste; and

EPA has performed full sensitivity analyses for all wastes of potential concern in the revised modeling results.[8] Concerning codisposal, EPA believes that the Monte Carlo analysis, which considered all volumes (not just the median values) is an appropriate way to evaluate this scenario; see Section I.A.3 for further discussion.

I.A.1. Revised High End Analysis

Comment 1: EPA's Determination of High-end Parameters Fails To Consider Correlated Variables

EPA's determination of high-end parameters in the groundwater model fails to consider the potential correlation between waste volume and surface area. As a result, EPA determined risks using a 90th percentile value for waste volume and a 50th percentile value for surface area. EPA's groundwater model is thus forced to generate unrealistic values of waste depth. For example, in the case of off-site hydrorefining catalyst analysis for arsenic, the inappropriate use of 90th and 50th percentile values for waste volume and surface area results in a depth of waste of 4.1 m. To remedy this error, API recommends that EPA reevaluate the high-end parameters selected for the groundwater model considering the correlation between waste volume and surface area, thereby, keeping both parameters at their 90th or 50th percentile values. (API, 00009, pg 16)

Response: The Agency believes that there may not necessarily be a strong correlation between surface area and waste volume, particularly when evaluating single waste stream disposal. Furthermore, because the waste quantity and waste concentration were constant values in this analysis and were not derived parameters, the total mass of contaminant, a key element of valid risk analysis, was conserved in the analysis and did not vary as a function of landfill area and depth.

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[7]U.S. EPA. *Additional Groundwater Pathway Analyses,* 1998; *Supplemental Background Document: Groundwater Pathway Risk Analysis,* 1997; and *Petroleum Refining Waste Listing Determination: Background Document for Groundwater Pathway Analysis.* August 1995.

[8]U.S. EPA. *Additional Groundwater Pathway Analyses,* 1998.

I.A.2. Monte Carlo Analysis

Comment 1: <u>The Monte Carlo Risk Analysis Should Be the Basis for the Listing Decision.</u>

The Monte Carlo based risk assessment is well recognized both at EPA and in the statistical modeling community as a superior technical approach to conducting a risk assessment. Use of the Monte Carlo risk assessment would be consistent with the Agency policy to use the best and most accurate modeling techniques in developing a regulation. The deterministic risk assessment, on the other hand, does not allow EPA to determine what percentile of the risk distribution is represented by the "high-end" analysis. This information is necessary to make a defensible listing determination. (NPRA, 00004, pg 2)

Response: The Agency's "Policy for Use of Probabilistic Analysis in Risk Assessment" states that "...such probabilistic analysis techniques as Monte Carlo analysis, given adequate supporting data and credible assumptions, can be viable statistical tools for analyzing variability and uncertainty in risk assessments." The policy also states that "It is not the intent of this policy to recommend that probabilistic analysis be conducted for all risk assessments supporting risk management decisions." In addition, as one of the conditions for using Monte Carlo analysis, the policy states that "Calculations of exposures and risks using deterministic (e.g. point estimate) methods are to be reported if possible." Thus the Agency policy is a far cry from the commenter's contention that information from Monte Carlo analysis is necessary to make a defensible listing determination. The Agency believes that, for this listing determination, Monte Carlo analysis can be a helpful tool for supplementing the deterministic analysis by providing additional information on variability and uncertainty.

The Agency agrees with the commenter that the Monte Carlo approach allows the EPA to determine the percentile of the risk distribution represented by the two high-end analysis. However, as is evidenced in the revised risk results, both the two high-end parameter and the Monte Carlo analyses show risk above the discretionary range for waste streams being listed.[9]

Comment 2: Proper Consideration of Potential Individual Risks Supports Not Listing Spent Hydrotreating Catalysts

The results of EPA's risk assessment for landfilling of spent hydrotreating catalyst clearly indicate that this residual should not be listed as hazardous waste. As reported by EPA, potential TC-capped risks associated with both on-site and off-site disposal range from $2x10^{-5}$ to $4x10^{-6}$ from exposure to benzene (62 FR 16752). Clearly, these high-end risk levels are within EPA's discretionary range for not listing (<u>see</u> 59 FR 66073) and are likely overestimates of the actual risk level (EPA appropriately reports the risks from arsenic as not applicable because spent catalysts containing arsenic at levels sufficient to pose such risks are already covered by the TC

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[9]U.S. EPA. Additional Groundwater Pathway Risk Analysis, Supplemental Background Document, 1998.

Rule, and regulated under RCRA as hazardous wastes; thus, these residuals cannot be disposed in Subtitle D landfills). When evaluating other residuals with potential risks in the same range, EPA appropriately proposed not to list these residuals. For example, TC-capped risks from on-site landfilling of crude oil storage tank sediment range from $2x10^{-5}$ to $3x10^{-6}$. Based upon these risk levels, EPA did not consider landfilling of crude oil storage tank sediment to warrant a listing.

More importantly, EPA cannot ignore the results of the Monte Carlo analysis for hydrotreating catalyst which report on-site and off-site risks of only $4x10^{-6}$ at the 95th percentile. According to recently released policy on Monte Carlo analysis, "...risk assessments using Monte Carlo analysis ... will be evaluated and utilized in a manner that is consistent with other risk assessments..."[10]. In addition, EPA's guidance on Monte Carlo analysis[11] states that Monte Carlo analysis is "...useful when screening calculations using conservative point estimates that fall above the levels of concern. Other situations [where it could be useful] could include when it is necessary to disclose the degree of bias associated with point estimates of exposure;...when the cost of regulatory or remedial action is high and the exposures are marginal; or when the consequences of simplistic exposure estimates are unacceptable." Based upon this guidance, it is clear that Monte Carlo analysis is the appropriate tool to use in evaluating petroleum refinery residuals.

Although the results of EPA's point estimate analysis do not fall above the level of concern, they do fall within EPA's discretionary zone. Due to subjective nature of the listing decision, it is important that EPA clearly understand the degree of bias associated with the high-end results. The Monte Carlo analysis is further necessary because the costs of the listing decision are relatively great compared to the marginal exposures reported from the population level risk assessment. As a result, the finding of a $4x10^{-6}$ risk is an appropriate basis for reaching a decision not to list spent hydrotreating catalyst.

A no list decision for hydrotreating catalyst is further supported by EPA's evaluation of potential population risks. The maximum population risk associated with this residual is an annual cancer incidence rate of 0.00003 or only three cases of cancer in 100,000 years. These population risks are clearly negligible and do not support EPA's proposed listing decision (as described further below). (API, 00009, pg 3)

Response: As indicated above, the Agency is using the Monte Carlo analysis as a source of additional information on the variability and uncertainty of the deterministic analysis. In conducting the Monte Carlo analysis for the Notice of Data Availability (NODA), the Agency made a key assumption concerning well location which was inconsistent with the assumption made for the deterministic analysis. In the deterministic analysis, the down gradient wells of concern were assumed to be those within the plume of contamination from the landfill. For the

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[10]U.S. EPA. Additional Groundwater Pathway Risk Analysis, Supplemental Background Document, 1998.

[11]U.S. EPA, 1997b. Background document for EPACMTP. User's Guide. p. 5.

Monte Carlo analysis, all potential wells on the downgradient side (within a 180 degree arc) from the landfill were included, thus including wells which would never be affected by contamination from the landfill. Since the Agency is interested in risk only at well locations which could be affected by the landfill, the Monte Carlo analysis was corrected to include only those well locations which were within the plume. The results of this correction, along with the other corrections to waste volume and landfill area estimates which were described earlier, show that risks are higher than previously reported for different percentiles on the Monte Carlo distribution as well as for the deterministic analysis. The results are presented in, Additional Groundwater Pathway Risk Analysis, Supplemental Background Document.[12]

Furthermore, it is important to note that the Agency policies do not indicate there is any particular point on a Monte Carlo distribution which should be the point at which the Agency regulates or does not regulate. The 1992 memorandum the then Deputy Administrator F. Henry Habicht "Guidance on Risk Characterization for Risk Managers and Risk Assessors" states that "The 'high end' of the risk distribution [generally the area of concern for risk managers] is, conceptually above the 90th percentile of the actual (either measured or estimated) distribution. This conceptual range is not meant to precisely define the limits of this descriptor, but should be used by the assessor as a target range for characterizing 'high end risk'." Therefore, a high end estimate which falls within the range (above the 90th percentile but still realistically on the distribution) is a reasonable basis for a decision.

A revised analysis conducted by EPA including a 30 year active life and a municipal landfill area distribution indicates that the TC-capped risk for exposure to benzene from hydrotreating catalyst is 2.6E-05.[16] Furthermore, no TC-capped risk for benzene from hydrotreating in any of the revised two high-end parameter analyses is below 2.0E-05. See also response to comment 1 in Section I.A.2.

EPA responds to the general comments on population risk in Section IV.B. EPA believes that the individual risks noted above are sufficient to warrant listing, despite relatively small population risks.

Comment 3: Proper Consideration of Individual Risk Supports Not Listing Spent Hydrorefining Catalysts

The results of EPA's risk assessment for landfilling of hydrorefining catalysts support not listing hydrorefining catalysts as a hazardous waste. According to EPA, potential TC-capped risks to an individual from benzene range from 2×10^{-5} to 6×10^{-6} (62 FR 16752). As discussed for hydrotreating catalysts, these risk levels are clearly within EPA's discretionary range for listing (59 FR 66073). The EPACMTP model predicts that these risks will occur in 13 to 50 years.

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[12]U.S. EPA. Additional Groundwater Pathway Risk Analysis, Supplemental Background Document, 1998.

EPA also reports that potential TC-capped risks from landfilling hydrorefining catalyst containing arsenic range from 4×10^{-4} to 1×10^{-4}. Although these high-end potential risks suggest that EPA could consider listing hydrorefining catalyst as a hazardous waste, it is clear that EPA did not adequately consider the results of the groundwater model before reporting these results. Unlike the risks reported for benzene, the EPACMTP model predicts that the arrival of peak arsenic concentrations at the receptor well will occur between 3400 and 8400 years after the source is released. Measurable concentrations would not begin to reach the well until after 1500 years. Obviously, these data suggest that there is no significant off-site migration of arsenic. Thus, EPA's entire risk assessment for arsenic is based upon the assumption that the landscape will not change over the next 3400 to 8400 years. This assumption is extremely implausible and renders the arsenic risk assessment unrealistic, at best.

In addition, the timing of peak concentrations at the receptor well prohibits EPA from adding together the risks from benzene and arsenic. API's analysis indicates that the risks from benzene and arsenic cannot occur simultaneously. In fact, there is a minimum of 3350 years between peak concentrations at the receptor wells for these constituents. As a result, EPA should base its listing determination for hydrorefining catalysts on the results of the more plausible benzene risk assessment. Any other decision would require costly protection against a risk that is extremely unlikely to occur.

When evaluating the risks from benzene, EPA cannot ignore the results of the Monte Carlo analysis. As stated above, the Monte Carlo analysis indicates that EPA's high-end analysis has conservatively predicted risks and should not be used when making a list/no list decision. In contrast to the findings of the high-end analysis, the Monte Carlo analysis reports a risk of 6×10^{-6} at the 95th percentile.

Finally, a no list decision for hydrorefining catalyst is supported by EPA's evaluation of potential population risks. Maximum off-site population risks from landfilling of hydrorefining catalyst are limited to 0.2 cases of cancer in 10,000 years. As described in more detail below, on-site population risks are significantly smaller given that the entire population within one mile of the landfill is limited to a single individual. These population risks are clearly negligible and do not support EPA's proposed listing determination. (API, 00009)

Response: The Agency disagrees with the commenter concerning the listing of the two catalysts. The peak arrival time for benzene in the current revised analyses is between 30 and 40 years[13], while the peak arrival time for arsenic in the current revised analyses for both industrial and municipal landfill area distributions is between 3000 and 3500 years for the various analyses (industrial areas, municipal areas, 30 year landfill life, 20 year landfill life etc.). However, the modeling analysis has been conducted to protect the environment and human health for a period of 10,000 years. Therefore, if peak arsenic levels occur at greater than 3000 years, those peaks must still be considered. Also, if the landscape changes within those 10,000 years the

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[13]U.S. EPA. Additional Groundwater Pathway Risk Analysis, Supplemental Background Document, 1998.

contamination within the landfill will still be present. There is a very low chance of a geologic event occurring that will remove the landfill and its contamination from the landscape. It may be buried and/or excavated by natural processes, but that will not neutralize the contamination. Furthermore, the individual risks for benzene and arsenic are high enough to warrant listing without adding the risks and the decision to list hydrorefining catalyst waste was based on the independent risk results for both arsenic and benzene. Furthermore the current revised analyses show that the TC-Capped analysis receptor well risks are 1E-06 at about 1,700 years and they are 1E-05 at about 1,900 years, much earlier than the peak arrival time.

In response to comments on the NODA, the Agency further revised the input data for the groundwater pathway analysis as noted elsewhere. As a result of the revised risk analysis, the off-site landfill groundwater risks increased further. The revised off-site risks for hydrotreating catalyst are 1E-04 for benzene and 8E-05 for arsenic; the TC-capped results for this waste showed lower risk for benzene (3E-05), but arsenic was unchanged. Similarly, the revised off-site risks for hydrorefining catalyst are 7E-05 for benzene and 6E-04 for arsenic, and the TC-capped analysis for this wastes lowered the benzene risks (3E-05) but had no impact on arsenic risk. The revised Monte Carlo risks for hydrotreating catalyst (benzene 3E-5, arsenic 2E-5 at the 95th percentile) and hydrorefining catalyst (benzene 2E-5, arsenic 4E-4 at the 95th percentile) were somewhat lower, but still well above the listing benchmark of 1E-5. As in the NODA analysis, the high-end and Monte Carlo risks for arsenic were not lowered by the TC-capped analysis. While the TC-capped risks for both catalysts were somewhat lower in the high-end (both at 3E-5) and Monte Carlo analysis (9E-6 and 8E-6 for the hydrotreating and hydrorefining risks respectively), EPA believes that the overall results are strongly supportive of listing both spent catalysts. Even in the TC-capping results, both catalysts present risks in off-site landfills that exceed 1E-5. Specifically, for both hydrotreating and hydrorefining catalysts, the TC-capped arsenic risks exceed 1E-5 for the Monte Carlo and high-end evaluations, and the benzene risks exceed this benchmark in the high-end evaluation and approaches this level in the Monte Carlo analysis. In addition to the groundwater risks posed by these materials, the pyrophoric and self-heating nature of these catalysts also support EPA's conclusion that these materials present a substantial hazard.

Comment 4: EPA Should Base Its Listing Determinations for the Groundwater Pathways on the Monte Carlo Analysis Rather than the Deterministic Results

EPA presents both a revised high-end analysis and a Monte Carlo analysis for the groundwater pathways. API strongly recommends that the Monte Carlo analysis serve as the basis for a list/no-list decision, due to the superior quality of the Monte Carlo-based risk estimates in comparison to the deterministic risk estimates. In the NODA, EPA states that the Monte Carlo analysis is used only to "confirm" the risk findings (62 FR 16750-51). API disagrees with this approach and urges EPA to fully consider the outcome of the Monte Carlo results when finalizing the listing determination. The Monte Carlo results should be used as the primary determinant of individual risk.

EPA guidance[14,15] has recognized the superior quality of Monte Carlo-based risk estimates compared to high-end approaches. Specifically, EPA has recommended that, where actual dose distributions are not available, but information on the variability in concentrations, activity patterns, and other patterns are, that modeling such as Monte Carlo simulation is an appropriate substitute. EPA[16] also indicated that deterministic analyses aimed at estimating the high-end exposure (such as EPA's "high-end" analysis in the proposed rule) should be used "if only limited information on the distribution of the exposure or dose factors is available," which is not the case in this rulemaking. In the absence of the Monte Carlo analysis, it is impossible to determine what percentile of the risk distribution is represented by the "high-end" analysis. Using the risk estimate from the Monte Carlo analysis will ensure that EPA achieves "transparency, clarity, consistency, and reasonableness," as stated by EPA guidance (1995), in its risk characterization for this proposed rule. (API, 00009, pg 14)

Response: See Response to Comment 1 in Section 1.A.2.

Comment 5: Listing Determinations for the Groundwater Pathways on the Monte Carlo Analysis Rather than the Deterministic Results

EPA presents both a revised high-end analysis and a Monte Carlo analysis for the groundwater pathways. Phillips strongly recommends that the Monte Carlo analysis serve as the basis for a list/no-list decision, due to the superior quality of the Monte Carlo-based risk estimates in comparison to the deterministic risk estimates. In the Federal Register notice, EPA states that the Monte Carlo analysis is used only to "confirm" the risk findings. Phillips agrees with this statement and urges EPA to fully consider the outcome of the Monte Carlo results when finalizing the listing determination. (Phillips, 00014)

Response: See Response to Comment 1 in Section 1.A.2.

I.A.3. Co-disposal

Comment 1: In evaluating the risks posed by plausible mismanagement, EPA must consider the actual and potential co-disposal of refinery wastes in both on-site and off-site landfills. Such co-disposal can produce the free-phased flow of contaminants in a landfill, which makes EPA's reliance solely on the TCLP as a predictor of contaminant migration seriously flawed. The most likely scenarios are (1) co-disposal of different kinds of petroleum wastes covered by the

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[14]U.S. EPA, 1995. Listing Background Document, Petroleum Refining Listing Determination.

[15]U.S. EPA, 1997. Background Document for EPACMTP. User's Guide.

[16]U.S. EPA, 1995. Background Document for the Groundwater Pathway Analysis, Petroleum Refining Listing Determination. p. 15.

rulemaking in the same landfill; (2) co-disposal of the same waste from multiple refineries at one offsite landfill; and (3) co-disposal of covered wastes with other petroleum refinery wastes in the same on-site or off-site landfill.

Unfortunately, in the NODA the agency has addressed only the first co-disposal scenario listed above. By ignoring the other two co-disposal scenarios, EPA has substantially underestimated the risks posed by the refinery wastes covered by this rulemaking. To account for this deficiency, KGS considered the effect of other codisposed wastes. In composite run analyses, KGS adjusted the volume-weighted TCLP value for codisposed wastes to reflect the higher benzene values reasonably expected in soils contaminated with product and crude oil, other oily sludges and organic wastes, and other codisposed wastes not covered by the rulemaking. (ETC, 00005, pg 13)

Response: In the groundwater pathway analysis the Agency modeled both scenarios (1) and (2) described in the comment, contrary to the cementers assertion that the Agency considered only scenario (1) in this analysis. In both the single-wastestream and the co-disposal scenarios for offsite landfills the waste volumes were calculated on a facility received basis, i.e. a management unit loading method was used[17]. They were not based on the refinery of origin as the commenter claims. Scenario (3) was not considered because the intent of the 1994 consent decree was to analyze the incremental risk of the 29 residuals considered in this rulemaking, and the commenter's approach was highly speculative. EPA responds to the specific issues in more detail below. For this reason, residuals not included in the rulemaking were not included in the codisposal analysis. EPA responds to the specific TCLP concentration modeling assumptions used by the commenter in Comment 5, below.

Comment 2: In its comments on the 1995 proposal, EDF argued the Agency's risk assessments were flawed because they did not take into account both actual and potential co-disposal of refinery wastes. As discussed above, such co-disposal can produce the free-phased flow of contaminants in a landfill, rendering the TCLP ineffective as a predictor of contaminant migration from the landfill. In addition, properly taking into consideration co-disposal would increase the volumes of wastes that are landfilled, and affect other landfill modeling assumptions.

Co-disposal can be subdivided into three categories. First, the same type of waste (i.e., unleaded gasoline tank storage sludge) can be codisposed from multiple refineries at one offsite facility, a scenario that is both documented by EPA and probable given the geographic concentration of refineries in the United States. Second, a combination of wastes covered by this rulemaking (including the study wastes) can be codisposed in the same onsite or offsite facility. Third, the wastes covered by this rulemaking can be codisposed with other refinery wastes in the same onsite or offsite landfill.

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[17]U.S.EPA, 1995. Listing Background Document, Petroleum Refining Listing Determination. U.S. EPA, Office of Solid Waste, Washington, D.C., 20460.

In response to EDF's comments, EPA included in the NODA materials an onsite and offsite co-disposal scenario that attempts to reflect the second category of co-disposal only. By ignoring the other two co-disposal categories, and inadequately addressing the second category, the NODA groundwater risk assessment substantially understates the potential risks posed by the co-disposal of refinery wastes. (EDF, 00006)

Response: EPA used the first scenario described by the commenter in the proposed rule. The commenter correctly points out that EPA assessed the second scenario in the NODA, and did not model the third scenario. The commenter provides more detailed comments on each of these scenarios in comments 3, 4, and 5 in this section; EPA provides its response to these criticisms later in this section as well.

Comment 3: The Co-disposal of Individual Wastes at Offsite Landfills

As discussed above, the waste volume parameter for the groundwater modeling is derived by multiplying the annual waste quantity (based upon 1992 data) and the length of the unit active life. For the annual waste quantity, EPA computed median (50%) and high-end (90%) values based upon 1992 generation data provided from each refinery. Accordingly, EPA's computation reflects annual waste quantities on an individual refinery basis, not on a facility received basis. In the case of onsite landfills, this distinction does not matter, since onsite landfills would receive only wastes generated by that refinery. However, in the case of offsite facilities, the distinction matters because offsite facilities can and do receive the same refinery wastes from multiple sources. Therefore, median and high-end waste volumes received at offsite facilities may differ significantly from analogous values computed on an individual refinery basis.

In projecting plausible mismanagement scenarios, EPA must compute median and high-end annual waste volumes on both an individual refinery and receiving facility basis, and choose the higher values for its modeling. Volumes derived on a refinery basis reflect one plausible mismanagement scenario since refineries may adjust their disposal practices from year to year. Volumes derived on a facility received basis reflect a second plausible mismanagement scenario since they are based on actual co-disposal occurring in the surveyed year.[18] Since both scenarios are plausible, EPA's modeling should take into account the scenario posing the greatest potential risk for the individual waste stream.[19]

Table B.2 of the NODA Groundwater Risk Assessment provides the waste volumes for each waste type on an offsite facility received basis. From the data provided in this table, median and high-end volumes can be derived for each waste. For at least three wastes, the as received

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[18] EPA may also project co-disposal scenarios based upon geographic and offsite capacity considerations, as the industry and data warrant.

[19] Waste volumes computed on an as received basis also better reflect "regional" generation quantities, a relevant listing determination factor. See 40 CFR 261.11(a)(3)(viii).

volumes are or may be higher than EPA's refinery-based values. Those wastes are crude oil tank sludge, unleaded gas tank storage sludge, and off-spec thermal products and fines.[20] A comparison of the refinery-based and offsite facility-based median and high-end volume values for these wastes (based upon a 40 year active life for the reasons explained above) is as follows:

	EPA's Refinery-Based Volume	Facility-Based Volume
	(40 yrs, cubic m)	
Crude Oil Tank Sludge		
Median	782	1,678
High-End	16,632	21,424
Unleaded Gas Tank Sludge		
Median	193.8	154
High-End	Not Provided[21]	1,719
Off-Spec Products & Fines		
Median	2,880	10,978
High-End	20,920	112,562

Thus, for these wastes and perhaps others, EPA's volume parameters substantially understate the actual quantities of wastes codisposed, based upon the reported co-disposal offsite by multiple refineries contained in EPA's database. The offsite annual waste quantity values should be revised

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[20] Only five wastes at issue in this rulemaking were assessed in this manner. The two other wastes assessed were CSO sludge and HF alkylation sludge, and for these wastes, the volumes derived on a refinery basis were higher than those derived on a facility received basis. EPA should complete this assessment for all wastes covered by this rulemaking, and utilize the higher values in its modeling.

[21] In the case of unleaded gasoline tank sludge, it is not possible to precisely quantify the high-end volume using EPA's methodology since none was provided in the NODA, and the median offsite value was lowered in the NODA Groundwater Risk Assessment when compared to the 1995 modeling. See NODA Groundwater Risk Assessment at 3-3. However, the 40 year volume based upon the high-end volume value provided in 1995 is 2,237 cubic meters. See 1995 Listing Background Document, Table 3.1.9. Lacking a detailed explanation for the NODA median volume modification, and no data on whether such modification would also affect the high-end value had EPA presented one, the as-received high-end volume is presented as the minimum high-end value for this waste in the event EPA has reason to modify the 1995 high-end value.

to reflect the higher of the median/high-end values computed on a refinery and receiving facility basis.

KGS revised the annual offsite waste quantities to reflect the Table B.2 values where higher than EPA's offsite annual waste quantities, and incorporated the revised quantities into its composite modeling runs. Accordingly, the KGS composite modeling runs more closely reflect the groundwater risks posed by landfilling refinery wastes. (EDF, 00006)

Response: The commenter has misinterpreted EPA's waste quantity calculation methods. EPA did not calculate any landfill disposal quantities on an individual refinery basis, as the commenter claims, but rather calculated all quantities on a facility received basis. The commenter correctly points out that Table B.2 of the NODA and the 1995 Listing Background Document provide different results. However, EPA clarifies that Table B.2 of the NODA represents only offsite facilities, while the 1995 Listing Background Document represents both onsite and offsite landfills. EPA believes that each data set is uniquely appropriate for a given evaluation, and therefore disagrees with the commenter that EPA should use two different calculation methods and select whatever value is higher for its deterministic risk assessment.

In the deterministic risk assessment used for the proposed rule, EPA elected to assess onsite and offsite landfills using the same waste quantities. The basis for this was EPA's judgement that a refinery could likely dispose of its waste at either an onsite location or an offsite location. EPA combined the quantities received by both onsite and offsite refineries into the same data array so that a single set of values would be the basis for both assessments.

Conversely, the data presented in Table B.2 represents data used for the Monte Carlo assessment. EPA conducted separate Monte Carlo assessments for the onsite and offsite scenarios. For the offsite scenario, EPA selected specific landfills each with a corresponding waste volume, climatic region, etc. Onsite landfill quantities could not be used for the offsite Monte Carlo analysis, as they could for the deterministic analysis.

In response to the specific data values presented by the commenter, EPA wishes to correct several discrepancies. The first column, labeled "EPA's Refinery Based Volume," is mislabeled (as discussed above) because it actually represents onsite and offsite facility-based volumes. However, these quantities correctly correspond to those given in the 1995 Listing Background Document, taking density into account. EPA wished to note several discrepancies in the second column, labeled "Facility-Based Volume." For crude oil tank sludge, the quantities should match exactly because no onsite disposal occurred in 1992. However, slightly different densities were used in each case (which do not affect the modeling results). Two additional reasons account for the discrepancy: (1) the initial calculation (presented in the 1995 Listing Background Document) was based on a population of 12 offsite landfills while the 1997 Supplemental Background Document for Groundwater Pathway Risk Analysis, Table B.2 presents 11 offsite landfills. A landfill reporting zero quantity of waste received accounts for the difference. (2) The 50th percentile value in the 1995 LBD is an average of (the middle) two values. This was erroneously

calculated to be 29.75 MT. The correct value is 34.75 MT (however, all risk calculations were based on the 29.75 MT). The difference between these two 50th percentile values is less than 20 percent and is not expected to affect results.

For Unleaded Tank Sediment, the differences between the two sets of values are very small after eliminating the effect of using different values of density. Therefore, there is essentially no difference in the 50th and 90th percentile receptor well concentrations determined by each of the two methods.

For Off-spec Product and Fines, the high end value in the second column is based on an incorrect value given in Table B.2 of the NODA. This incorrect value was not used in the deterministic analysis used for the 1995 proposed rule but was inadvertently included in Table B.2. Substitution of this specific facility's incorrect value with the more appropriate, smaller value results in the same 90th percentile 40 year volume as presented in the first column.

As for the issue raised in footnote 25 (Unleaded Gasoline high-end waste volumes), (1) the high-end Unleaded Gasoline Tank Sediment waste quantity was provided in the 1995 Listing Background as was the median value used in the modeling. The value is 2,077 cubic meters. Not 2,237 cubic meters. (2) An explanation for the difference in waste volumes used in the NODA as compared with the waste volumes used in the 1995 proposal was provided in the Supplemental Background Document for Groundwater Pathway Analyses[22] The waste fraction approach used in the proposal analysis resulted in derived waste volumes that were either too large or too small. Revised (reported) waste volumes were used in the NODA Analyses. This revision is the reason for the difference in the Unleaded Gasoline Tank Sediment volume. Furthermore, since high-end volumes were not used in the proposal analyses, they were not affected by the waste fraction issue and there were no revised waste volumes to report.

Comment 4: The Co-disposal of Wastes Covered by Listing Determination and Related Study

In the NODA materials, EPA assessed the groundwater risks posed by the co-disposal of multiple wastes covered by the instant rulemaking and a related study of refinery wastes. The onsite and offsite scenarios assessed deterministically assumed the wastes were codisposed at their median volumes, based upon a 20 year active life. Leachate values for the codisposed wastes were derived by computing a volume-weighted average value based upon the individual waste TCLP sampling results. High-end values were set for waste unit area and distance to well, without the benefit of a sensitivity analysis to determine whether these two parameters would produce the greatest risks at the receptor well.[23]

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[22]U.S. EPA. Additional Groundwater Pathway Risk Analysis, Supplemental Background Document, 1998.

[23] NODA Groundwater Risk Assessment at 3-5, 3-6.

In addition to the deficiencies discussed elsewhere in this section unique to the co-disposal scenario, EPA's co-disposal modeling suffered from the same shortcomings as the modeling performed on individual wastes. Specifically, the volume quantities are based on an incorrect 20 year active life, the TCLP values do not reflect the leachability of oily wastes, and the location of the receptor well was inappropriately located outside of the plume centerline (see next section of the comments).

Moreover, EPA's methodology assumes the volumes of listing determination and study wastes that are codisposed will remain forever constant, a projection for which EPA has not presented any factual basis. In contrast, API reports that 9,922 tons of "other oily sludges and organic wastes" were landfilled in 1992, the year on which EPA's volume data are based.[24] In 1993, API reports 180,221 tons of the same wastes were landfilled, a sum over 17 times greater than the corresponding 1992 volume.[25] Clearly, waste volumes can increase dramatically from the 1992 figures.

EPA's failure to conduct a sensitivity analysis for the co-disposal scenario further exacerbates the lack of emphasis on the potential for increased quantities of codisposed wastes. By modeling only median waste volumes, the annual onsite codisposed volume was only 2,170 cubic meters, and the corresponding offsite volume was only 2,727 cubic meters.[26] For comparison purposes, the annual high-end volume for CSO sludge onsite and offsite is 2,245 cubic meters, about 17 times the CSO median value and greater than EPA's codisposed onsite volume alone.[27] Therefore, EPA's modeling inputs fail to account for the co-disposal of high-end quantities of CSO sludge and other wastes. Similarly, the annual high-end volume quantity for HF alkylation sludge is 1,912.3 cubic meters for both onsite and offsite, therefore EPA's modeled co-disposal scenario assumes very little waste is also landfilled with that high-end volume.[28]

Significantly, the co-disposal of high-end volumes for various wastes is not merely a hypothetical scenario. In its comments on the 1995 proposal, EDF documented the co-disposal of high-end values of various wastes based upon the 1992 data alone.[29] The potential for such co-disposal to

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[24] API 1992-1993 Residual Report, p. B-18.

[25] Id., p. B-38.

[26] See NODA Groundwater Risk Assessment, Tables D.1, D.5.

[27] See NODA Groundwater Risk Assessment, Tables C.3, C.4. In addition, EPA's co-disposal CSO volume did not reflect the larger contingent CSO volumes based upon EPA listing CSO sludge as hazardous when land farmed only.

[28] See NODA Groundwater Risk Assessment, Tables C-30, C-31.

[29] See EDF March 1996 Comments at 16-17.

occur is thus both actual and potential, given the geographic concentration of refineries in some parts of the country.

Moreover, a review of the sensitivity analyses performed by EPA on the individual wastes indicates waste volume is an extremely important parameter in EPA's modeling. More often than not, waste volume proved to be one of the two high-end parameters that produced the highest risks at the receptor well.[30]

Response: EPA addresses the commenter's concerns regarding landfill active life, appropriateness of the TCLP to oily wastes, and plume centerline elsewhere in these comments. Specifically, greater discussion of active life is presented in Section I.A.6.b of this response to comments, application of the TCLP to oily wastes is discussed in Section I.C.1, and plume centerline is discussed in Section I.A.6.e.

As addressed elsewhere (for example, response to comment 3 of Section III.I of the NPRM), EPA believes its use of 1992 data for the purpose of selecting waste volumes is appropriate, because EPA has no reason to believe otherwise. In reference to the specific API data cited by the commenter, API's report acknowledges the increase in volume, explaining that it is due to a single refinery accounting for 62 percent of the total 1993 quantity (page 22 of API's "Generation and Management of Residual Materials, 1992-1993"). Analysis of wastes generating in years 1987 to 1991 shows relatively constant waste quantities (40,000 to 61,000 wet tons, from page 25 of that source) in comparison to the 1992 value of 38,000 tons. Further, the API category of "other oily sludges/organic residuals NOS" does not necessarily refer to the wastes of concern in this rulemaking and could be a wide variety of materials. Therefore, EPA disagrees with the commenter's conclusion that 1992 was not a representative year.

As for the issue of high-end waste volumes, the high-end parameter groundwater analysis does not assume that high-end values of parameters are hypothetical and exceed all possible values of a particular parameter. Rather, high-end values are based on actual statistical distributions of parameter values from the petroleum refining database. Therefore, the occurrence of a landfill possessing a property equal to the high-end value of that property is not unexpected and does not in any way invalidate the Agency's high-end analysis. For each landfill property, ten percent of all landfills should exceed the 90th percentile value for that property. However, in this case for the co-disposal scenario, waste volume was not one of the selected high-end parameters. Nevertheless, a comparison of the Monte Carlo co-disposal results with the two high-end parameter co-disposal results shows that the two high-end analysis produced risk values well above the 95th percentile. So, although a sensitivity analysis was not conducted for the co-disposal scenario, the two high-end parameters selected for the analysis based on a 1995 sensitivity analysis produced sufficiently conservative results.

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[30] See NODA Groundwater Risk Assessment, Table 3.3.

The two offsite landfills receiving large quantities of codisposed waste noted in EDF's comments to the 1995 proposal (EDF, 00036; pp. 16 and 17; March 21, 1996) did receive high-end quantities of waste. That waste, however, consisted largely of FCC Fines, Off-Spec Product and Fines (neither of which contain significant amounts of benzene) and hydrotreating and hydrorefining waste (which are excluded from the co-disposal analysis).

Comment 5: The Co-disposal of Wastes Not Covered by the Listing Determination and Related Study

EPA's co-disposal modeling does not take into account the large quantities of refinery wastes that are not covered by the listing determinations or the related study. Information gleaned from the NODA materials and other sources indicate large quantities of other refinery wastes are codisposed with the refinery wastes at issue in this rulemaking. EPA is obligated to consider the potential risks posed by such co-disposal, since such co-disposal is a plausible type of improper management. See 40 CFR 261.11(a)(3)(vii).

Attachment 1 to Chapter 6 of the NODA Background Document provides the onsite landfill capacity used in 1992 for 23 refinery waste landfills.[31] Importantly, the capacity utilization value is not restricted to the wastes covered by the instant rulemaking, therefore the collected values provide a useful measure of the extent of co-disposal that occurred in these landfills during 1992.

The median 1992 codisposed waste volume for these 23 landfills was 3,361 cubic meters, while the average value was 23,879 cubic meters. As noted above, EPA's onsite median co-disposal value was 2,170 cubic meters, or about 1,191 cubic meters less than the actual median waste volume, and about 21,709 cubic meters less than the average waste volume codisposed in these landfills during 1992.

Although there is no corresponding EPA table for offsite landfills, other sources of information can be utilized to obtain a rough approximation of offsite co-disposal rates. As discussed above, API reported that the largest volume of waste landfilled in 1992 by far was contaminated soils/solids. Two other waste categories landfilled in high quantities were "other inorganic residuals" and "other residuals". Together, these three waste categories accounted for 1,177,810 of the 1,664,887 tons of refinery wastes landfilled during 1992, or over 70% of all landfilled wastes.[32]

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[31] The value can be derived by multiplying the capacity remaining in 1992 and the percentage of capacity used in 1992, and converting the resulting number from cubic yards to cubic meters, the volume unit typically used by EPA in the groundwater modeling.

[32] API 1992-1993 Residual Report, p. B-18.

API also reports that 92% of the contaminated soils, 92% of the other inorganic residuals, and 72% of the other residuals were disposed offsite during 1992.[33] Since according to API these wastes accounted for the vast majority of refinery wastes landfilled during 1992, the relative proportion of onsite/offsite management for these wastes can be reasonably extrapolated to all refinery wastes during 1992. Therefore, using a 75%/25% offsite/onsite ratio for these wastes, an offsite 1992 co-disposal contribution from wastes not included in this rulemaking can be calculated by multiplying the onsite differential specified above (1,191 cubic meters median value; 21,709 cubic meters average value) by three to reflect the greater percentage of refinery wastes managed offsite.

The result is an annual contribution of 3,573 cubic meters (median), or 65,127 cubic meters (average) of other wastes codisposed offsite with the wastes covered by the instant rulemaking. These quantities are greater than the volumes modeled by EPA in the offsite co-disposal scenario, therefore the lack of consideration to other codisposed wastes is a profound deficiency in the co-disposal modeling.

KGS considered the effect of "other" codisposed wastes in two sets of modeling runs. In the first set, KGS adjusted only the volume of the codisposed wastes to include the median value increment estimate for wastes not included in this rulemaking and a longer landfill active life (39.2 years onsite, 40 years offsite). Adjusting this parameter only did not significantly affect the modeling results.

However, in the composite run analyses discussed below, KGS also adjusted the volume-weighted TCLP value for the codisposed wastes to reflect the higher benzene values reasonably expected in soils contaminated with product and crude oil, "other oily sludges and organic wastes", turbo fuel filters, oily rags, laboratory wastes, paint waste, and other codisposed wastes not covered by this rulemaking.[34] (EDF, 00006, pg 34)

Response: EPA does not dispute the commenter's central point that both onsite and offsite landfills contain other refinery wastes in addition to those in the scope of today's listing determination and EPA's related study wastes. EPA disagrees that it should take such wastes into account because EPA believes that it's core objective in conducting this listing determination is to assess the incremental risk associated with these materials. As such, it is not appropriate to take into consideration pre-existing contamination or co-disposal with residuals outside of the scope of the consent decree. Such considerations are better measures of the risk associated with *landfills* containing the residuals of concern, rather than the residuals themselves.
In regard to the additional modeling conducted by the commenter, EPA believes the commenter's use of an adjusted volume-weighted TCLP concentration is arbitrary and disagrees that the result

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[33] API 1992-1993 Residuals Report, p. B-20.

[34] See API 1991 Residual Study, Appendix A, for the types of wastes incorporated into each of the API residual categories.

is useful for assessing co-disposal. In fact, the existing data available to EPA for the refinery wastes under study show that very few of these wastes contain such high levels of benzene. EPA has no valid reason to project that benzene levels in other codisposed wastes would be drastically different, as assumed by the commenter. EPA further addresses the commenter's modeling of co-disposal in Section I.A.3, comment 2 of this NODA comment response.

Comment 6: The Practice of Co-disposal Is Trivial in Both Potential Risks and Likely Occurrences

As part of the NODA, EPA evaluated the risks associated with disposal of two or more residuals in the same land treatment unit (LTU) or landfill. In all cases, incremental risks associated with the co-disposal scenario were trivial and did not affect EPA's proposed no listing determinations. Although API disagrees with the need to evaluate co-disposal as a management scenario, API agrees that the analysis clearly demonstrates that risks associated with co-disposal are insignificant. In the case of LTUs, API also supports EPA's use of site-specific data to identify and model those few facilities that accepted several different residuals. (API, 00009, pg 11)

Response: EPA acknowledges the commenter's support regarding the co-disposal analysis.

I.A.4. Capping Waste Analysis Results at TC Levels

Comment 1: Inappropriate TC Capped Modeling

In the NODA Groundwater Risk Assessment, EPA conducted additional groundwater monitoring analyses of individual wastes and the co-disposal scenario where the TCLP values for wastes disposed in the landfill were capped at the toxicity characteristic threshold concentrations of 0.5 mg/l for benzene and 5.0 mg/l for arsenic. The implication of this modeling is the existing toxicity characteristic may be an appropriate alternative to listing the wastes covered in this rulemaking. For the host of reasons discussed in this portion of the comments, this TC capped modeling and the underlying implication are both technically and legally flawed.

First, the modeling approach presumes the TCLP is an accurate and consistent indicator of benzene availability, but as discussed at length above, the TCLP is ineffective on many oily and tarry wastes in this rulemaking and fails to account for potential free-phased flow in the landfill. Therefore, TCLP results are not an accurate or consistent indicator of benzene availability, whether or not a particular TCLP benzene concentration is above the characteristic threshold. The more relevant arbiter of risk is the total concentration of benzene in the waste.

Second, the approach is based upon the unsupported assumption that all wastes exhibiting the toxicity characteristic for either benzene or arsenic are properly managed as hazardous wastes. However, as EPA is well aware, federal law does not require testing of wastes to determine whether they exhibit a hazardous waste characteristic. Instead, a generator may simply apply

"knowledge" to render a hazardous waste determination. See 40 CFR 262.11. In EPA's own words, this regulation -

> ...has hindered enforcement efforts aimed at regulating generators and potential generators or individual waste streams under RCRA. There is no self-reporting mechanism to identify generators who claim to eliminate characteristic hazardous waste from the RCRA system. To take an enforcement action, EPA or the state must sample and analyze the waste to determine if the waste exhibits any of the hazardous waste characteristics.[35]

Moreover, EPA also acknowledged at the same time that very few generators receive inspections at all, let alone on a routine basis.[36] Under these circumstances, where EPA admits enforcing appropriate hazardous waste characteristic determinations generally is extremely problematic, and TCLP results on oily wastes are erratic at best thereby exacerbating the inaccuracies resulting from applying generator "knowledge",[37] EPA is obligated to consider the plausible mismanagement scenario of the disposal of characteristic refinery wastes in nonhazardous waste landfills. Specifically, it is quite plausible that refinery generators may render inaccurate hazardous waste determinations on oily refinery wastes, and regulatory authorities will not identify these inaccurate determinations for years, if at all.[38]

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[35] The Nation's Hazardous Waste Management Program at a Crossroads, EPA OSWER, July 1990, p. 39. In response to this problem with enforcing toxicity characteristic hazardous waste determinations, EPA staff recommended increasing funds for targeted sampling and analysis, and/or modifying the rules to require waste testing on either an industrial sector or case-by-case basis. Neither of these recommendations were implemented, thus the shortcomings of 40 CFR 262.11 remain in place today.

[36] Id. at 60. Only one-third of all RCRA generators received one inspection during the first 10 years of the federal regulatory program. Moreover, sampling and analysis of wastes for compliance purposes likely occurred at only a small fraction of such inspections.

[37] The wide variation in TCLP benzene recovery deficiencies displayed by the waste samples in this rulemaking demonstrate the difficulties of extrapolating waste to TCLP concentrations in the absence of actual testing.

[38] From an enforcement prospective, the TCLP deficiencies also raise serious issues since it will be extremely difficult to apply sampling results taken at a certain time and place to wastes generated prior to such sampling. Accordingly, regulatory officials may be unable to demonstrate previous wastes of the same type also exhibited the toxicity characteristic. In addition, refinery management practices may further complicate the issue. For example, one oil refinery mixes its sludge with soil before landfilling the waste. Under these circumstances, how will EPA know whether the sludge exhibited the toxicity characteristic prior to mixing? See Survey Response of Facility 182, p. 25. Moreover, even if such a demonstration could be made, it is also plausible,

Evidence of this plausibility is reflected in the docket for the NODA and other sources. Included in the docket is a Criminal Information filed by the U.S. Attorney for the District of New Jersey describing the improper shipment and management of 617,980 pounds of characteristic refinery waste as nonhazardous. The waste involved sometimes exhibited a hazardous waste characteristic, and in this instance, the company improperly relied upon a TCLP sample of a previous batch of wastes to wrongly conclude the offending wastes was nonhazardous when, in fact, it exhibited the toxicity characteristic for benzene.[39]

In addition, RCRA noncompliance resulting from inadequate hazardous waste determinations is apparently an important enough problem in EPA Region VI that the Region commenced roundtable discussions with the industry, and posted guidance on the Internet, to try to begin addressing the problem.[40]

Significantly, the plausibility of inaccurate hazardous waste determinations is an expressed consideration in hazardous waste listing determinations. One of the criteria for listing a hazardous waste is whether the waste exhibits a hazardous waste characteristic. See 40 CFR 261.11(a)(1). This criterion would be unnecessary and duplicative if there were no advantage or benefit to listing an otherwise characteristic waste. However, the principal advantage of a listing is the ease of implementation, including enforcement.[41]

In the enforcement case discussed above, if the waste was listed as hazardous, the issue of whether TCLP results were appropriate for a particular batch of wastes would not have arisen, because the application of hazardous waste requirements would have been clear and uncontrovertible from the outset. The fact that the criminal investigation and prosecution took years to bring is evidence of the fact that thousands of documents had to be reviewed to overcome the inconsistencies of the TCLP and the defendant's "knowledge" of the waste necessary to prove the violation of law. Therefore, in reaching the listing determination, EPA must consider the plausible enforcement shortcomings of maintaining a characteristic-based approach for the refinery wastes.

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indeed most likely, that the previously generated wastes will not be exhumed from the nonhazardous waste landfill in which they were placed.

[39] See Document # PRA-S0037.

[40] Enforcement and Compliance Assurance Accomplishment Reports, EPA-300-R-97-003, May 1997, p. 3-4.

[41] See 55 FR 11805-6 (March 29, 1990).

It is also reasonable to interpret the term "improper management" found in the statutory definition of hazardous waste,[42] and EPA's regulatory listing criteria, as inclusive of management not in accordance with applicable law, particularly where such law contains acknowledged opportunities for generators to reach inappropriate or inaccurate hazardous waste determinations when acting in good faith or otherwise. Webster's definition of "improper" includes "not accordant with fact or right procedure" as well as "not suited to the circumstances, design, or end," and therefore incorporates both elements of fact and law in defining "improper" behavior.[43] Management practices contrary to law is certainly plausible in the refinery industry.[44] EPA's Office of Enforcement and Compliance Assurance recently indicated the petroleum refinery industry was one of three highest priority national sectors for enforcement activities due to the "high noncompliance" associated with this industry.[45]

Third, in light of the documented practice of "air drying" wastes as discussed above, reliance on the toxicity characteristic in lieu of listing raises the specter of refineries storing wastes on pads until such time as the benzene volatilizes and the waste no longer exhibits a hazardous waste characteristic. While such volatilization should constitute treatment of an as generated hazardous waste, the enforcement difficulties associated with demonstrating the waste was hazardous after the fact, and the complexity of enforcing the RCRA land disposal restrictions on wastes that have been rendered nonhazardous solely through evaporation, argue strongly for listing the wastes as hazardous.

Fourth, the characteristic approach does not take into account the high PAH content of some refinery wastes. Like the groundwater modeling in the instant rulemaking, the toxicity characteristic concentration thresholds were developed assuming no preexisting contamination that would increase the risk associated with waste releases and/or facilitate the transport of hazardous constituents, in direct contrast to the gross contamination present at many refinery sites.[46] This inapplicability of characteristic development assumptions to refinery sites, combined

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[42] See Section 1004(5) of RCRA.

[43] See Webster's Seventh New Collegiate Dictionary, based upon Webster's Third New International Dictionary, 1967, p. 421.

[44] See e.g., Enforcement and Compliance Assurance Accomplishment Reports, EPA-300-R-97-003, May 1997, pp. B-47, B-48.

[45] See Memorandum on Final 98/99 OECA Memorandum of Agreement Guidance from Steven A. Herman, Assistant Administrator, June 5, 1997, p. 6.

[46] Reliance on characteristic thresholds to regulate wastes containing benzene and arsenic is also inappropriate insofar as the TC threshold concentrations assume no preexisting groundwater contamination contributing to exposed population risks. While such an assumption may be justified in the context of developing generic nationwide concentration thresholds, listing

with the ineffectiveness of the TCLP to measure PAH leachability discussed above, and the lack of coverage in the TC of the relevant PAHs, would inappropriately result in the management of wastes containing high PAH concentrations outside of the Subtitle C regulatory program.

Fifth, <u>by listing the wastes as hazardous, EPA can encourage pollution prevention activities associated with refinery waste management</u>.[47] A hazardous waste listing can bring refinery wastes under a variety of state and federal pollution prevention, waste minimization, and reporting requirements, which could trigger careful scrutiny of the way the wastes are generated and managed. Unfortunately, while it is incumbent upon EPA to consider the source reduction benefits of a hazardous waste listing as part of this rulemaking,[48] EPA simply lists waste minimization practices applicable to refinery wastes.[49] This lackadaisical approach toward pollution prevention is particularly disappointing given EPA's waste minimization policy. (EDF, 00006, pg 45)

Response: Responses to the above comments are as follows:

This modeling is technically flawed because (1) it assumes that TCLP is an accurate and consistent indicator of benzene availability,

See Response to Comments 6 and 7 in Section I.C.1. The Agency believes that the TCLP data are appropriate and are reasonable indicators of benzene availability for these wastes.

(2) Because the approach is based on the unsupported assumption that all wastes exhibiting the toxicity characteristic for either benzene or arsenic are properly managed as hazardous wastes,

In regard to the commenter's criticisms of TC cap modeling, EPA notes that it conducted this analysis for crude oil tank sludge, spent hydrotreating catalyst, spent hydrorefining catalyst, and unleaded tank sludge. EPA is finalizing its decision to list all but one of the above wastes, and notes that the TC cap risk results for unleaded tank sludge are not significantly different than the risk results for the non-TC cap analysis (see Table 5.1 of Supplemental Background Document

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determinations require consideration of plausible mismanagement for the particular industrial sectors, which should include the condition of sites that may contribute to risks posed by the improper management of the wastes covered by the listing determination.

[47] For example, the volume of tank sediments can be reduced through the use of mixers to suspend solids in the product. Yet according to EPA, only 50% of unleaded gasoline storage tanks, 28% of CSO tanks, and 68% of crude oil tanks have mixers. See 1995 Listing Background Document at 41; 60 FR 57782 (November 20, 1995).

[48] See Section 13103(b)(2) of the Pollution Prevention Act.

[49] 60 FR 57781-2 (November 20, 1995).

for Groundwater Pathway Risk Analysis (1997) and Tables 5.2 and 5.3 of the Additional Groundwater Pathway Risk Analyses, Supplemental Background Document (1998). Therefore, the commenter's criticisms regarding the TC cap analysis are moot.

(3) Refineries' use of "air drying" volatilizes benzene.
EPA responds to the commenter's concerns regarding air drying in NODA Section I.C.1, comment 1.

(4) The characteristic approach does not take into account the high PAH content of some refinery wastes and would underestimate the total risk associate with a particular waste.

EPA responds to the commenter's concerns regarding the effectiveness of the TCLP towards PAH containing wastes in NODA Section I.C.1 Comment 7. However, EPA agrees that the TC will not control risks from PAHs because PAHs are not TC constituents.

(5) By listing the wastes as hazardous, EPA can encourage pollution prevention activities associated with refinery waste management.

EPA agrees with the commenter that listing wastes as hazardous provides financial incentives to refineries to minimize the quantity of waste generation. EPA disagrees that it has a "lackadaisical" approach towards waste minimization, and instead believes that a critical component of waste minimization is communication. In this regard, EPA has published an inventory of waste minimization practices used for the listed wastes and other refinery wastes (see "Waste Minimization for Selected Residuals in the Petroleum Refining Industry," F-95-PRLP-S0064).

Comment 2: EPA Incorrectly Evaluated High-End TC-Capped Risks from Landfilling

In its revised groundwater risk assessment for landfilling of spent catalysts, EPA appropriately acknowledged that TCLP measurements that exceeded the toxicity characteristic (TC) for benzene and arsenic should not be included in the evaluation of potential risks. Residuals containing these elevated concentrations are currently, and will continue to be, managed in hazardous landfills and thus should not be included in the calculation of the average constituent concentrations. Although EPA's intention to remove the TCLP concentration data exceeding the TC is appropriate, the methodology used by EPA in the calculation of revised TCLP concentrations is flawed. Instead of either removing these concentrations or substituting the TC cap value, EPA used the TC cap value as the TCLP concentration when the average and maximum values proved to be greater than the TC cap. In essence, this approach causes the TCLP value to be a third high-end parameter for spent hydrotreating catalyst, and causes the waste concentration value to be greater even than the maximum reported value for spent hydrorefining catalyst. This approach is also inconsistent with the Monte Carlo analysis, where EPA used the original TCLP data, substituting the TC-capped value for individual values that exceeded the TC. Using the approach applied by EPA in the Monte Carlo analysis, API has

calculated that the average TCLP concentrations decreased to 60 percent of the TC cap, as shown in Table 1. (API, 00009)

Table 1. Revised TC Cap Waste Concentration for COTS and Spent Catalysts

Crude Oil Tank Sediment Benzene TCLP Data

Sample	Benzene Concentration (mg/L)	TC Capped[a] Benzene Concentration (mg/L)
R6B-CS-01	1.7	0.5
R8C-CS-01	1.6	0.5
R4B-CS-01	0.13	0.13
R10-CS-01	0.5	0.5
R19-CS-01	0.032	0.032
R22-CS-01	0.05	0.05
Median	0.32	0.32
Value used in RA	0.68	0.5
Average	0.67	**0.29**

Hydrotreating Catalyst Benzene TCLP Data

Sample	Benzene Concentration (mg/L)	TC Capped[a] Benzene Concentration (mg/L)
R1-TC-01	39	0.5
R8A-TC-01	0.17	0.17
R11-TC-01	3.7	0.5
R3B-TC-01	0.048	0.048
R18-TC-01	4.2	0.5
R22-TC-01	0.25	0.25
Median	1.98	0.38
Value Used in RA	7.90	0.50
Average	7.89	**0.33**

Hydrorefining Catalyst Benzene TCLP Data

Sample	Benzene Concentration (mg/L)	TC Capped[a] Benzene Concentration (mg/L)	Arsenic Concentration (mg/L)	TC Capped[a] Arsenic Concentration (mg/L)
R5-TC-01	0.1	0.1	0.23	0.23
R7B-RC-01	4.2	0.5	34	5
R21-RC-01	0.16	0.16	6.9	5
Median	0.16	0.16	6.90	5.00
Value Used in RA	1.49	0.50	13.71	5.00
Average	1.49	**0.25**	13.71	**3.41**

Notes:
a: TC Cap substituted for values greater than the cap.

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Revised average TCLP value in bold.
TC cap for benzene = 0.5 mg/L
TC Cap for arsenic = 5.0 mg/L
(API, 00009)

Response: The Agency does not completely agree with the commenter on this issue. If some samples of a particular waste stream have been shown to meet or exceed the TC rule limit, then a conservative TC Capped analysis will include the possibility that some of that waste disposed in Subtitle D landfills will have leachate levels as high as 99% of the TC limit. Therefore, the Agency does not believe that TCLP data above the TC Rule value should simply be eliminated. Rather the data should be treated as it was in the Monte Carlo Analysis.

Furthermore, EPA notes that even if the Agency used the TC-capped input values suggested by the commenter's approach, these values would be lowered for benzene by only 42% (crude oil sediment), 34% (hydrotreating catalyst), and 50% (hydrorefining catalyst). Because a decrease in the TCLP input to the model for these wastes did not result in a proportionate decrease in the level of benzene projected at the receptor well (e.g., the decrease is less as shown by the differences in the central tendency modeling results for crude oil sediment using the average TCLP data versus the TC-capped level), the impact of using the commenter's approach would be minimal (i.e., decreases would be well below a 50% decrease). Concerning the catalysts, the impact of TC-capping for the arsenic risks for the hydrorefining catalyst is negligible, showing that the modeling results are not impacted significantly by small changes in the arsenic input data. (This lack of sensitivity to arsenic input data is due to the relatively slow movement of this constituent via groundwater from the landfill to the receptor well). The arsenic risks for hydrotreating catalysts do not change under any TC-capping assumption because *none of the samples exceeded the TC level for arsenic.* As such, the risks would remain the same as the uncapped analysis.

Comment 3: EPA Incorrectly Evaluated High-End TC-Capped Risks from Landfilling

In its revised groundwater risk assessment for landfilling of spent catalysts, EPA appropriately acknowledged that TCLP measurements that exceeded the toxicity characteristic (TC) for benzene and arsenic should not be included in the evaluation of potential risks. Residuals containing these elevated concentrations are currently, and will continue to be, managed in hazardous landfills and thus should not be included in the calculation of the average constituent concentrations. Although EPA's intention to remove the TCLP concentration data exceeding the TC is appropriate, the methodology used by EPA in the calculation of revised TCLP concentrations is flawed. Instead of either removing these concentrations or substituting the TC cap value, EPA used the TC cap value as the TCLP concentration when the average and maximum values proved to be greater than the TC cap. In essence, this approach causes the TCLP value to be a third high-end parameter for spent hydrotreating catalyst, and causes the waste concentration value to be greater even than the maximum reported value for spent hydrorefining catalyst. This approach is also inconsistent with the Monte Carlo analysis, where

EPA used the original TCLP data, substituting the TC-capped value for individual values that exceeded the TC. (Phillips, 00014)

Response: See Response to Comment 2 above.

I.A.5.   Waste-Specific Comments

Comment 1: Crude Oil Storage Tank Sludge

While EPA proposed a negative listing determination (at the behest of OMB),[50] crude oil storage tank sludge warrants listing as a hazardous waste. After correcting only a few of the flaws in EPA's groundwater modeling methodology, KGS demonstrated this waste poses a human health risk of at least 1.6×10^{-4} via the groundwater pathway, substantially above EPA's presumptive listing risk level.

Significantly, the KGS modeling understates the risk posed by crude oil storage tank sludge because the KGS modeling revisions do not reflect:

1. The true leachability of the waste since the modeled TCLP value still assumes no primary leachate generation and no free-phased contaminant migration, and thus understates actual benzene availability;

2. High-end waste volumes that incorporate high-end landfill active life duration, since all volumes were derived using only an average landfill life assumption;

3. Appropriate offsite median and high-end landfill areas;

4. The contribution of pre-existing contamination at refinery sites to receptor well contamination;

5. Risks posed by contaminants other than benzene in the waste.

The average total concentration of benzene in this waste exceeds the comparable concentration of benzene in F037 and F038 that caused those wastes to be listed in 1990.[51] Similarly, the benzo(a)pyrene concentration in crude oil storage tank sludge exceeds the comparable concentration in F038 and K145 which caused those wastes to be listed in 1990 and 1992.[52] And

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[50] See Document S0002 (the Redlined Version of the 1995 Proposal Preamble).

[51] 55 FR 46365 (November 2, 1990).

[52] 55 FR 46365 (November 2, 1990); 57 FR 37289 (August 18, 1992).

the average total concentration of indeno(1,2,3-cd)pyrene exceeds comparable concentrations in K145 which caused that waste to be listed in 1992.[53]

EPA's proposed negative listing determination is also unjustified because the Agency has not evaluated the risks posed by management practices such as landfill daily cover, road spreading, and other uses constituting disposal. The NODA LTU Risk Assessment is not an adequate surrogate risk evaluation because the volume of waste modeled, coupled with other modeling shortcomings, rendered that assessment inapplicable to most plausible mismanagement scenarios. (EDF, 00006)

Response: EPA responds to the five ground water modeling issues in the above comments in other sections of this response to comments. Specifically, EPA presents a "roadmap" of where these five issues are discussed in response to comment 3, Section I.

The commenter also compares the concentration of benzene and PAHs between crude oil tank sludge and currently listed wastes. EPA does not agree with the commenter's comparisons because, in general, listing decisions are based on multiple reasons including, but not limited to, constituent concentrations. EPA recognizes that crude oil tank sediment may contain concentrations of some constituents comparable to previously listed wastes, including the F037 and F038 refinery residuals. However, direct comparison of these concentrations to previous listing benchmarks is not an adequate basis for listing. Listing determinations consider many factors beyond the concentrations of constituents in a waste, including the waste volume, constituent mobility, management practices, damage cases, other regulatory controls, etc. (see 40 CFR 261.11). Over the years since the rule cited by the commenter was promulgated, EPA's risk assessment process has evolved, due in part to public comments, EPA policy and research developments, the review of the Science Advisory Board, and others. The Agency has developed a more sophisticated set of risk assessment tools than were available for listing determinations in 1990. As a result, EPA believes that it is better able to measure and predict risk. Therefore EPA does not agree with the commenter's implication that wastes in the scope of this rulemaking should be listed if constituent concentrations exceed levels in previously listed wastes. In any case, EPA notes that it has decided to list crude oil tank sediment as hazardous, making the commenter's argument that EPA was inconsistent moot.

Section V.C of this NODA comment response presents comments and responses regarding risks posed by waste management practices cited by the commenter, including landfill daily cover and road spreading.

Comment 2: CSO Sludge

While EPA proposed to list this waste as hazardous, the Agency also sought comment on whether the listing should be limited to land treatment of the waste. As demonstrated by the KGS

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[53] 57 FR 37289 (August 18, 1992).

modeling, the purportedly low risks attributed to EPA's landfill modeling is a function of the ineffectiveness of the TCLP to leach PAHs from this oily waste. Therefore, the waste should be listed as hazardous without limitation.

CSO sludge contains very high percentages of oil and grease, and when not inappropriately air dried or mixed with cement kiln dust prior to sampling, the oil "seeps out when squeezed". It has all the properties of wastes for which EPA and its contractor previously found the TCLP grossly ineffective, particularly for PAHs.

According to the KGS modeling, if only 0.008% of the benz(a)anthracene or 0.005% of the chrysene in the average CSO sample leaches from the waste, the resulting risk in the receptor well equals 10^{-4}, EPA's presumptive listing risk level. When the combined risks of these chemicals are considered, the required leaching values are even lower. Moreover, the KGS modeling does not reflect:

1. High-end waste volumes that incorporate high-end landfill active life duration, since all volumes were derived using only an average landfill life assumption;

2. The contribution of pre-existing contamination at refinery sites to receptor well contamination;

3. The potential for facilitated PAH transport at refinery sites due to the cosolvency effects of oil and other compounds in the landfill, and/or the presence of NAPLs in the subsurface.

On the basis of EPA's previous positions regarding the efficacy of the TCLP on oily wastes, EPA's contractor report demonstrating the poor performance of the TCLP on PAHs in oily wastes, EPA's data indicating CSO sludge contains high percentages of oil and grease, and the potential for CSO sludge to contain free liquid, it is more than plausible that these wastes will leach PAHs at the extremely small concentrations which pose a substantial risk to human health and the environment.

The average total concentration of chrysene in the CSO residuals substantially exceeds the comparable concentrations in F037, F038, K143, K144, and K145 that caused those wastes to be listed in 1990 and 1992.[54] Similarly, the average total concentration of benzo(a)pyrene in the CSO residuals substantially exceeds the comparable concentrations in F038, K143, K144, and K145 that caused those wastes to be listed in 1990 and 1992.[55] In addition, concentrations of benz(a)anthracene and benzofluoranthene (total) in the CSO residuals exceed comparable concentrations in K143, K144, and K145 that caused those wastes to be listed in 1992.[56] And the

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[54] 55 FR 46365 (November 2, 1990); 57 FR 37289 (August 18, 1992).

[55] 55 FR 46365 (November 2, 1990); 57 FR 37289 (August 18, 1992).

[56] 57 FR 37289 (August 18, 1992).

concentration of indeno(1,2,3-cd)pyrene in the CSO residuals exceeds the comparable concentration in K145 which caused that waste to be listed in 1992.[57] (EDF, 00006)

Response: EPA responds to each of the commenters concerns in other sections of this response to comment document. The commenter's concerns regarding the ineffectiveness of the TCLP are discussed in comment 7 of Section I.C.1, concerns regarding sampling and the appropriateness of using samples that underwent treatment is discussed in comment 1 of Section I.C.1, leaching of PAHs is discussed in comment 7 of Section I.C.1, and a comment response "roadmap" for the three modeling issues is presented in Section I, comment 3. Finally, for the reasons presented in Section I.A.5 comment 1 (immediately above), EPA believes it is inappropriate to compare constituent levels in CSO sludge to constituent levels in previously listed wastes as a basis for a listing decision.

Finally, EPA notes that the final listing for CSO tank sediment will not incorporate an conditional limitations (e.g., disposal as nonhazardous in a landfill would not be allowed). Therefore, the commenter's concerns about the potential risks from landfilling of this waste will be addressed in any case.

Comment 3: Unleaded Gasoline Storage Tank Sludge

While EPA proposed a negative listing determination for this waste, unleaded gasoline storage tank sludge warrants listing as a hazardous waste. After correcting only a few of the flaws in EPA's groundwater modeling methodology, KGS demonstrated this waste poses a human health risk of at least 8.8×10^{-5} via the groundwater pathway.

Significantly, the KGS modeling understates the risk posed by unleaded gasoline storage tank sludge because the KGS modeling revisions do not reflect:

1. The true leachability of the waste since the modeled TCLP value still assumes no primary leachate generation and no free-phased contaminant migration, and thus understates actual benzene availability;

2. High-end waste volumes that incorporate high-end landfill active life duration, since all volumes were derived using only an average landfill life assumption;

3. Appropriate offsite median and high-end landfill areas;

4. The contribution of pre-existing contamination at refinery sites to receptor well contamination;

5. Risks posed by contaminants other than benzene in the waste; and

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[57] 57 FR 37289 (August 18, 1992).

6. A comprehensive set of two high-end sensitivity analyses for the revised modeling parameters.

The average total concentration of benzene in unleaded gasoline storage tank sludge exceeds the comparable concentrations in F037 and F038 that caused those wastes to be listed in 1990.[58]

Given the risk levels are at the high-end of the 10^{-4} to 10^{-6} risk range, unleaded gasoline storage tank sludge warrants listing as a hazardous waste based upon the factors EPA considers under its listing determination policy when risks from improper waste management fall within that range.[59] First, the waste characterization is extremely uncertain because EPA's sampling consists of two only valid samples (the third was inappropriately air dried), and the modeled leaching values are still based upon the ineffective TCLP with no primary leachate generation for wastes containing "one inch of free liquid". Second, even the KGS modeling results are uncertain because it may significantly underestimate potential risks for the reasons discussed above. Third, the risks posed by the co-disposal of this waste and other refinery wastes are also significant. Fourth, other regulatory programs will not address the risks posed by the mismanagement of these wastes.

EPA's proposed negative listing determination is also unjustified because the Agency has not evaluated the risks posed by management practices such as landfill daily cover, road spreading, and other potential uses constituting disposal. The NODA LTU Risk Assessment is not an adequate surrogate risk evaluation because the volume of waste modeled, coupled with other modeling shortcomings, rendered that assessment inapplicable to most plausible mismanagement scenarios. (EDF, 00006)

Response: EPA responds to the KGS modeling results for this waste in comment 2 of Section I. EPA also notes that the total oil and grease levels in the available samples for this specific waste were well below 1 percent. Thus, the commenter's concern about problems with oily waste are clearly unfounded for this waste.

EPA disagrees with the commenter's assertion that groundwater risks require listing of this waste. For the reasons given elsewhere (as noted above) earlier in today's notice, EPA does not agree with most of the changes incorporated into the alternative groundwater modeling. However the Agency did modify its modeling analysis to reflect comments on: increased active life of landfills, increased unit area corresponding to municipal landfills, and consideration of noningestion routes of exposure to groundwater by receptors (e.g., via inhalation during showering). EPA also performed a full sensitivity analysis, which had not been done for the NODA analysis. These changes resulted in a somewhat higher groundwater risk for this waste (3E-05; see NODA Table 1) than previously calculated. While this risk is of some concern to the Agency, EPA has decided not to list this waste after considering other factors. First, the waste is lacking the characteristics shown by other wastes that EPA decided to list in this rule. It does not present the additional risk

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[58] 55 FR 46365 (November 2, 1990).

[59] 59 FR 66077-78 (December 22, 1994).

factors due to arsenic and pyrophoricity exhibited by the spent hydrotreating and hydrorefining catalysts, nor does it show the high oil and PAH content found in the CSO and crude oil tank sediment. In addition, the unleaded gasoline sediment appears to consist largely of tank scale and rust, as evidenced by the high level of iron found in EPA's samples (average level of 41% measured as elemental iron). The relatively small annual volume generated (3,583 MT) and sent to landfills (22 MT average per facility) are significantly lower than volumes of the other storage tank sediments, e.g., crude oil tank sediment (generated, 22,017, average landfilled, 123 MT). Finally, EPA notes that recent Clean Air Act regulations will result in the decrease of benzene in gasoline (see Reformulated Gasoline Rule, February 16, 1994; 59 FR 7716). EPA also believes that the groundwater risk results suggest that its analysis was quite conservative, i.e., the DAF corresponding to the high-end risk for off-site landfills was only 4.4. Thus, EPA has decided not to list this waste.

EPA believes it is inappropriate to compare constituent levels in unleaded gasoline tank sediment to constituent levels in previously listed wastes as a basis for a listing decision as discussed in Section I.A.5, comment 1. EPA addresses the commenters concerns regarding sampling in Section I.C.1, comment 1. Comments regarding EPA's approach to assessing co-disposal are in Section I.B.3.

Section V.C of this NODA comment response presents comments and responses regarding risks posed by waste management practices cited by the commenter, including landfill daily cover and road spreading. EPA also notes that, in this case, no refineries reported use of unleaded gasoline tank sediment as landfill cover or in road spreading. The Agency has no data supporting these management scenarios and therefore does not see the need to model this pathway. Furthermore, EPA believes that nongroundwater risks are unlikely to be significant for this waste under any scenario, because unleaded gasoline tank sediment has none of the carcinogenic PAHs that were of concern for other wastes.

Finally, the risks posed by mismanagement of this waste has been assessed in the proposed rule, and on the basis of these results (and other listing criteria in 40 CFR Section 261), EPA has decided to list or not list specific refinery wastes; therefore EPA believes the commenter's concerns regarding the absence of regulatory programs for waste mismanagement are fully addressed by today's listing decisions.

Comment 4: Off-Spec Products and Fines from Thermal Processes

While EPA proposed a negative listing determination for this waste, off-spec products and fines warrants listing as a hazardous waste for two separate but equally important reasons. First, when this waste is landfilled, the KGS modeling indicates the risk posed by the groundwater pathway is at least 5×10^{-5}, after correcting only some of the flaws in EPA's modeling methodology.

Given the risk levels are at the high-end of the 10^{-4} to 10^{-6} risk range, off-spec products and fines warrants listing as a hazardous waste based upon the factors EPA considers under its listing

determination policy when risks from improper waste management fall within that range.[60] First, the waste characterization is extremely uncertain because the modeled leaching values are still based upon the ineffective TCLP. Second, even the KGS modeling results are uncertain because it may significantly underestimate potential risks for the reasons discussed below. Third, the risks posed by the co-disposal of this waste and other refinery wastes are also significant. Fourth, other regulatory programs will not address the risks posed by the mismanagement of these wastes.

In addition, KGS determined that if only 0.01% of the benz(a)anthracene or 0.5% of the chrysene in the average off-spec products and fines sample leached from the waste, the resulting risk via the groundwater pathway would be 10^{-4}, EPA's presumptive risk level for listing hazardous wastes. When the combined risks of these chemicals are considered, the required leaching values are even lower. Moreover, the KGS modeling does not reflect:

1. High-end waste volumes that incorporate high-end landfill active life duration, since all volumes were derived using only an average landfill life assumption;

2. A comprehensive set of two high-end sensitivity analyses for the revised modeling parameters, and associated high-end offsite volumes consistent with Table B.2 of the NODA Groundwater Risk Assessment and Section II.E.1 of these comments;

3. The contribution of pre-existing contamination at refinery sites to receptor well contamination;

4. The potential for facilitated PAH transport at refinery sites due to the cosolvency effects of oil and other compounds in the landfill, and/or the presence of NAPLs in the subsurface.

On the basis of EPA's previous positions regarding the efficacy of the TCLP on oily wastes, EPA's contractor report demonstrating the poor performance of the TCLP on PAHs in oily wastes, and EPA's data indicating off-spec products and fines contains high percentages of oil and grease, it is more than plausible that these wastes will leach PAHs at the extremely small concentrations which pose a substantial risk to human health and the environment.

Second, off-spec products and fines warrant hazardous waste listing when placed on piles for potential use as a fuel (with petroleum coke), irrespective of whether EPA considers this material a coproduct when managed in this fashion. The only meaningful risk assessment associated with this practice to date (performed in 1995) indicated risks exceeding 1×10^{-3} to nearby human receptors, and this evaluation was based on waste volumes only 1/7 of the values associated with high-end pile usage.[61] The NODA LTU Risk Assessment does not alter these previous results because the quantities of waste modeled in the NODA assessment were more than several orders

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[60] 59 FR 66077-78 (December 22, 1994).

[61] EPA 1995 Non-groundwater Risk Assessment, Appendix Q, Table 3a.

of magnitude lower than the relevant pile volumes, among other shortcomings discussed in Section III of these comments.

Even assuming arguendo the off-spec products and fines are coproducts when stored with petroleum coke, EPA must still render a listing determination for this residual when stored on piles. Pursuant to 40 CFR 261.33, EPA lists a variety of products, including off-specification products, when discarded or intended to be discarded. Therefore, if the off-spec products and fines are not actually reused when placed on the pile, because the residual blows off the pile or migrates via runoff, such off-spec products and fines would be regulated as hazardous wastes.

Pursuant to the Consent Decree in EDF v. Browner, EPA is required to issue a listing determination for off-spec products and fines. Given that an estimated 169,986 MT of the total 194,262 MT (87%) off-spec products and fines generated in 1992 was placed in piles and subject to discard,[62] and that such discard presents a substantial risk to human health and the environment based upon the assessments conducted to date, the required listing determination must include a decision as to whether off-spec products and fines warrants listing pursuant to 40 CFR 261.33.

Finally, EPA's proposed negative listing determination is inappropriate because the Agency has not considered the risks posed by managing off-spec products and fines in surface impoundments. As discussed above in Section IV of the [EDF] comments, EPA has documented the use of surface impoundments for this waste, yet EPA has failed to evaluate this plausible mismanagement scenario. (EDF, 00006)

Response: Many of the commenter's concerns are discussed in detail in other sections of this background document. EPA responds to the KGS modeling results for this waste in comment 2 of Section I.A. The effectiveness of the TCLP and PAH leaching is discussed in comment 7 of Section I.C.1. Comments regarding EPA's approach to assessing co-disposal are in Section I.B.3. EPA responds to the KGS modeling results for this waste in comment 7 of Section I.C.1 and presents a comment response "roadmap" for the four modeling issues in Section I, comment 3. The commenter's concerns regarding the placement of this waste in surface impoundments is discussed in Section IV.C in response to comment 3.

Regarding the commenters' criticism of EPA's use of TCLP results as input values to the landfill groundwater modeling for this material because it is "oily", the Agency points out that off-specification product and fines are generally not oily. The Agency conducted total oil and grease analyses on 4 samples and the average level of oil and grease (measured as Total Oil and Grease, i.e., not truly "free" oil) was 2 percent (3 samples were below 1 percent). Furthermore, the data from the §3007 Questionnaire show that the typical material has relatively low oil content (90th% value was 5%). Therefore, EPA believes that the use of the TCLP was valid.

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[62] 1995 Listing Background Document, Table 3.7.1.

With regards to commenter concern over free-phase flow of contaminants from off-specification product and fines due to oil content, the Agency concluded that free phase flow is unlikely with these residuals as discussed earlier in today's notice. It is particularly important to note that none of the six off-spec product and fines from thermal processes samples exhibited a separate oil phase, and that the measured and oil content was low, as noted above.

For the reasons noted elsewhere in this document (and referenced above), EPA does not agree with most of these changes employed in the alternative KGS analysis. The Agency did modify its modeling analysis to reflect suggestions on increased active life of landfills, increased unit area corresponding to municipal landfills, and consideration of noningestion routes of exposure to groundwater by receptors (e.g., via inhalation during showering). These changes tended to increase the projection well concentration of constituents. However, EPA notes that the risk results presented in the NODA was in error because it was based on an erroneous health-based number (carcinogenic risk factor) for the PAH, benz(a)anthracene (BaA). EPA had used the correct number in the nongroundwater risk analysis used to estimate releases from land treatment units, as noted in the 1995 background document (*Assessment of Risks from the Management of Petroleum Refining Wastes: Background Document, 1995*), but inadvertently did not use the correct number in the groundwater analysis. Use of the correct health-based number results in a 30-fold decrease in the risks calculated from the well concentration of BaA. Using the revised groundwater model inputs (active life and area) and the correct health-based number yields a high-end risks of 5E-7 and 2E-6, and Monte Carlo risks are 9E-7 and 1E-6 (see *Additional Groundwater Pathway Analysis*, 1998).

Furthermore, as EPA described in the NODA, the estimated groundwater risks were based entirely on one chemical (BaA) that was detected in only one out of six TCLP samples, at a level 8-fold below the quantitation limit. The risk of 2E-06 was calculated using this one TCLP input, along with two other high-end parameters in the sensitivity analysis.   EPA also calculated a risk of 5E-07 (based on the correct health-based number) by assuming that this one detected TCLP level should reflect one of the high-end parameters, and then running the sensitivity analysis to calculate risk based on one other high-end parameter. While EPA believes it is probably more appropriate to assign the one detected value to be one of the high-end parameters, even the high-end groundwater risk of 1E-06 clearly supports a decision not to list off-specification product and fines.

Concerning the placement of coke fines on coke piles, EPA considers this to be co-management of product with larger coke particles, and not waste. EPA discussed this issue in the NODA, and noted that only particle size distinguishes coke fines from other coke product. The majority of coke is removed from the coker by hydraulic drilling, and coke fines are generated as smaller pieces of coke during this process. In addition, EPA explained there is a jurisdictional distinction between coke fines produced from non-hazardous materials and coke fines produced from hazardous wastes (waste-derived fines). Fines generated from non-hazardous materials are simply coke product, as would be expected since they are produced from the same coking drum. In the case of waste-derived fines, so long as the fines are legitimate coke product, they are exempt from

RCRA regulation unless the material exhibits a characteristic as provided in 40 CFR 261.6(a)(3)(v). (See also RCRA section 3004(q)(2)(A)). EPA had no information that waste-derived coke fails any hazardous waste characteristic. The Agency invited comment or data to the contrary but received none.

The commenter is attempting to use a bounding analysis EPA undertook for nongroundwater risks for coke fines disposed in landfills as an indication that air releases from piles containing this material would present similar risks. In response, EPA first notes that bounding estimates are used as an initial screening estimate that overestimates the exposure or dose for the purpose of screening out exposures of little concern. The purpose of the bounding analysis is simply to determine what pathways and scenarios require further evaluation and does not represent an assessment of risks. The bounding analysis included worst-case assumptions (no cover or dust suppression, highest constituent levels, largest waste volumes and landfill area, worst climate, etc). Furthermore, the levels near 1E-3 arose from indirect pathways (ingestion of beef, dairy, fish, and plant products); the direct pathway of soil ingestion, even in the bounding analysis was on the order of 1E-06. Most importantly, the biotransfer factors used in the bounding analysis for beef, dairy, and plant indirect paths have been determined to overestimate risks by at least two orders of magnitude; likewise the apparent problem from mercury was also traced to an error in units for the bioaccumulation factor used. Thus, EPA believes that the bounding analysis was flawed and grossly overestimated risks.

EPA notes that the subsequent high-end analyses for nongroundwater risks from landfill disposal of off-spec product and fines did not show significant risk. While the high-end analysis included the assumption of daily cover for the landfill, and thus may not be a the best surrogate for air releases from piles, the scenario did consider windblown dust from on-site roads and particulate release caused by traffic (i.e., dump trucks), loading, unloading, etc. The high-end analysis showed risks no higher than 2×10^{-6} for any receptor (see U.S. EPA, Assessment of Risks from the Management of Petroleum Refining Wastes Background Document (F-95-PRLP-S0006), page 10-3). Therefore, some of the possible release mechanisms that could occur in a waste pile scenario (e.g., unloading/loading, traffic) were addressed in the risk assessment supporting the proposal and the pathway was not significant.

EPA also points out that some important characteristics of the coke pile and details of management practices used by refineries would tend to mitigate potential risks. The piles are not comprised simply of coke fines, but are mixtures of much larger pieces of coke product that are drilled from coker units; the larger chunks of coke would make up the bulk of the pile. Furthermore, coke is drilled out of the coker approximately once a day with hydraulic drills, thus new wet coke/fines from drilling are added to the coke pile, making air releases of dry particulates less likely. EPA has also found that coke piles are managed using various practices to control release of dust, including: (a) contained product storage areas; (b) dust-suppression water spray systems; © covered conveyor systems; and, (d) direct loading from coke-drums into railcars (see NODA response to comment document for a summary of these practices). In addition, EPA

expects that particulate releases from these areas would be controlled by Federal, State, or local air regulations and permit programs.

Finally, EPA disagrees with the commenter's argument that, pursuant to the Consent Decree in EDF v. Browner, EPA is required to issue a listing determination for off-spec products and fines that must include a decision as to whether off-spec products and fines warrants listing pursuant to 40 CFR 261.33. The Agency has, in fact, made a listing decision for the off-spec product and fines that are known to be discarded by refineries, i.e., the volumes of wastes that are disposed. EPA believes it has fulfilled the requirements of the Consent Decree. Furthermore, at this time the Agency has no valid assessment that indicates these wastes present a "substantial risk", and believes that the available data suggest otherwise.

Comment 5: EPA Has Dealt Reasonably With The Issue Of Risks Associated With the Disposal of Off-Specification Coke and Fines

EPA proposed not to list off-specification coke and fines, in part because so much of this material is sold as a product. *See* 62 Fed. Reg. 16750 (April 8, 1997). As discussed elsewhere in these comments, EPA is correct that off-specification coke and fines sold as product are not wastes and are beyond EPA's RCRA jurisdiction. *See id.*

As to coke and fines that are disposed of, EPA's original risk assessment projected high-end individual risk of 1×10^{-5}, and its revised risk assessments projected individual risks varying from 5×10^{-6} to 2×10^{-5}, depending upon the method used. *Id.* at 16751. Although the projected risks "are within the Agency's initial risk level of concern," *id.*, the Agency determined that the risks are, in fact, *not* significant.[63]

The Agency reasoned that (1) the risk estimates are unrealistically high, because they are based on one constituent (benz(a)anthracene) that was detected in only one of six samples, and even then, at a level 8-fold below the quantitation limit; (2) benz(a)anthracene is not very soluble in water, so its aqueous concentration may be expected to be low; and (3) benz(a)anthracene is not very mobile in groundwater. *Id.* API agrees with and supports this very reasonable approach to dealing with individual risk estimates that are on or near the borderline of EPA's range of risk levels of concern, especially under circumstances that indicate the projected risks are inflated. Thus, EPA is fully justified in finding that the listing of off-specification coke and fines is not warranted. (API, 00009, pg 17)

Response: The Agency acknowledges the support of the commenter.

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[63] Although EPA found that "it is highly unlikely that this waste would present a significant risk in a groundwater scenario," 62 FR 16751 (April 8, 1997), the listing criteria actually use the term "substantial . . . hazard." 40 CFR § 261.11(a)(3). Obviously, a waste that does not pose a significant risk also does not pose a *substantial* hazard.

Comment 6: HF Alkylation Sludge

While EPA proposed a negative listing determination for this waste, HF alkylation sludge warrants listing as a hazardous waste. After correcting only a few of the flaws in EPA's groundwater modeling methodology, KGS demonstrated this waste poses a human health risk of at least 2.3×10^{-5} via the groundwater pathway.

Significantly, the KGS modeling understates the risk posed by HF alkylation sludge because the KGS modeling revisions do not reflect:

1. The true leachability of the waste since the modeled TCLP value still assumes no primary leachate generation and no free-phased contaminant migration, and thus understates actual benzene availability;

2. High-end waste volumes that incorporate high-end landfill active life duration, since all volumes were derived using only an average landfill life assumption;

3. Appropriate offsite median and high-end landfill areas;

4. The contribution of pre-existing contamination at refinery sites to receptor well contamination;

5. Risks posed by contaminants other than benzene in the waste; and

6. A comprehensive set of two high-end sensitivity analyses for the revised input parameters.

Given the risk levels are at the middle of the 10^{-4} to 10^{-6} risk range, HF alkylation sludge warrants listing as a hazardous waste based upon the factors EPA considers under its listing determination policy when risks from improper waste management fall within that range.[64] First, the waste characterization is extremely uncertain because the modeled leaching values are still based upon the ineffective TCLP with no primary leachate generation for wastes that are "black and oily with a high liquid content", "yellow with a consistency of wet oily sludge", and have a consistency of "oil-in-water emulsion".

Second, even the KGS modeling results are uncertain because it may significantly underestimate potential risks for the reasons discussed above. Third, the risks posed by the co-disposal of this waste and other refinery wastes are also significant. Fourth, other regulatory programs will not address the risks posed by the mismanagement of these wastes.

EPA's proposed negative listing determination is also inappropriate because the Agency has not considered the risks posed by managing HF alkylation sludge in surface impoundments. EPA has

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[64] 59 FR 66077-78 (December 22, 1994).

documented the use of "pits" for this waste, yet EPA has failed to evaluate this plausible mismanagement scenario.

Response: EPA responds to each of the commenter's concerns in other sections of this response to comment document. EPA responds to the KGS modeling results in comment 2 of Section I.A. EPA presents a "roadmap" of the six enumerated issues in response to comment 2 of Section I.A. EPA disagrees that listing of this waste is appropriate. For the reasons noted in other sections in this document, EPA does not agree with most of the changes incorporated into the alternative groundwater modeling, however the Agency did modify its modeling analysis to reflect comments on: increased active life of landfills, increased unit area corresponding to municipal landfills, and consideration of noningestion routes of exposure to groundwater by receptors (e.g., via inhalation during showering). These changes resulted in a somewhat higher groundwater risk for this waste (1×10^{-5}; see *Additional Groundwater Pathway Analysis, 1998*) than previously calculated. While this risk is of some concern to the Agency, EPA has decided not to list this waste after considering other factors. Most importantly, the risk was due to the presence of one constituent, benzene, that was detected in only one of the five samples. Thus, the constituent does not appear to appear frequently in this wastes. Further, the level of benzene in this waste (0.076 mg/L average, 0.180 mg/L maximum) is considerably lower than the other petroleum wastes EPA is listing in this rule based on groundwater risks (e.g., for crude oil tank sediment the average and maximum benzene TCLP levels were 0.68 and 1.7 respectively). EPA also believes that the groundwater risk results suggest that its analysis was quite conservative, i.e., the DAF corresponding to the high-end risk for off-site landfills was only 2.8. Thus, EPA has decided not to list this waste.

Comment 7: Hydrotreating/Hydrorefining Catalyst

EPA proposed to list these spent catalysts as hazardous wastes. When EPA revises the NODA risk assessments to correct the modeling flaws discussed throughout these comments, the case for listing will be even more dramatic.

As discussed in Section II.J of this [EDF's] document, the listing mechanism for these and other refinery wastes is necessary and appropriate, notwithstanding the potential reach of the toxicity characteristic. In this context, listings would substantially improve the implementation and enforcement of RCRA requirements as they apply to refinery wastes. A characteristic-based approach for refinery wastes is extremely problematic, due to the inconsistencies of the TCLP on oily wastes, the acknowledged difficulties associated with determining the validity of "knowledge-based" hazardous waste determinations especially where TCLP results may vary from one batch of the same wastes to the next, the practice of air drying wastes thereby allowing benzene to volatilize, and other management practices such as mixing the waste with soil to dilute the benzene concentration. The implementation and enforcement advantages of a listing outweigh whatever disadvantages EPA perceives in listing the wastes as hazardous. (EDF, 00006, pg 76)

Response: EPA acknowledges the commenter's support of EPA's decision to finalize the listing of spent hydrotreating and spent hydrorefining catalysts.

Comment 8: EPA Should Not List Hydrotreating Catalyst

EPA's revised risk assessment in the NODA for landfilling hytrotreating catalysts clearly indicates that hydrotreating catalyst should not be listed as a hazardous waste. The individual risks calculated are within the discretionary range, even with the conservative assumptions employed. Similar risk levels for landfilling crude oil tank sediment were not determined by EPA to warrant listing. Moreover, the Monte Carlo analysis, which is the appropriate analysis on which to base a decision, has a risk finding that would support a no list decision. Such a decision can be further supported by EPA's evaluation of population risks which predicts only three cases of cancer in 100,000 years, a clearly negligible result. (Mobil, 00002, pg 2)

Response: EPA disagrees with the commenter. EPA continues to believe that the risks from the high-end analysis fully support listing this waste, and the somewhat higher revised groundwater risks further support EPA's decision. As a result of the revised risk analysis, the off-site landfill groundwater risks increased further. As shown in Table IV-2, the revised off-site risks for hydrotreating catalyst are 1E-04 for benzene and 8E-05 for arsenic; the TC-capped results for this waste showed lower risk for benzene (3E-05), but arsenic was unchanged. Similarly, the revised off-site risks for hydrorefining catalyst are 7E-05 for benzene and 6E-04 for arsenic, and the TC-capped analysis for this wastes lowered the benzene risks (3E-05) but had no impact on arsenic risk. The revised Monte Carlo risks for hydrotreating catalyst (benzene 3E-5, arsenic 2E-5 at the 95th percentile) and hydrorefining catalyst (benzene 2E-5, arsenic 4E-4 at the 95th percentile) were somewhat lower, but still well above the listing benchmark of 1E-5. As in the NODA analysis, the high-end and Monte Carlo risks for arsenic were not lowered by the TC-capped analysis. While the TC-capped risks for both catalysts were somewhat lower in the high-end (both at 3E-5) and Monte Carlo analysis (9E-6 and 8E-6 for the hydrotreating and hydrorefining risks respectively), EPA believes that the overall results are strongly supportive of listing both spent catalysts. Even in the TC-capping results, both catalysts present risks in off-site landfills that exceed 1E-5. Specifically, for both hydrotreating and hydrorefining catalysts, the TC-capped arsenic risks exceed 1E-5 for the Monte Carlo and high-end evaluations, and the benzene risks exceed this benchmark in the high-end evaluation and approaches this level in the Monte Carlo analysis.

In addition to the groundwater risks posed by these materials, the pyrophoric and self-heating nature of these catalysts also support EPA's conclusion that these materials present a substantial hazard. Therefore, after EPA considered the risks posed by these materials, along with the possibility that characteristically hazardous waste may not always be handled as hazardous, the Agency has decided to list hydrotreating and hydrorefining catalysts as hazardous wastes.

EPA responds to comments on population risk in Section IV.B.

Comment 9: EPA's Record Now More Strongly Supports Its Decision Not to List Eleven of the Fourteen Refinery Residuals Considered for Listing

In its original proposal of November 20, 1995 (60 FR 57747), EPA determined not to propose for listing crude oil storage tank sediment, unleaded gasoline storage tank sediment, off-specification product and fines from thermal processes, catalyst from reforming, catalyst from sulfuric acid alkylation, sludge from sulfuric acid alkylation, HF alkylation sludge, spent caustic from liquid treating, catalyst and fines from catalytic cracking, catalyst from sulfur complex and H_2S removal facilities, and sludge from sulfur complex and H_2S removal facilities. Some of these residuals are used in ways that prevent them from being solid wastes and are thereby inappropriate candidates for listing. Others are solid wastes and EPA based its proposal not to list these residuals on sound data and conservative risk analysis. The NODA adds additional information to support these "no list" decisions, with new analyses evaluating co-disposal and additive risk scenarios that found no basis to warrant listing. (Mobil, 00002, pg 2)

Response: EPA is finalizing its decision to not list all but one of the above wastes (EPA is listing crude oil tank sediment). EPA appreciates the commenter's support for its no-list decisions.

Comment 10: The EPA's Decision to Not List 11 Residuals as Hazardous Waste was Correct

Sun supports EPA's decision not to list the 11 residuals originally proposed for non-listing. Although Sun agrees that some re-evaluation of the listing process was needed, the results of the re-evaluation did not justify changing the listing decision on the 11 residuals. (Sun, 00008)

Response: EPA is finalizing its decision to not list all but one of the above wastes (EPA is listing crude oil tank sediment). EPA appreciates the commenter's support for its no-list decisions.

Comment 11: As part of the NODA, EPA conducted several new analyses, including evaluation of co-disposal and additive risk scenarios. EPA's proposal not to list sludge from H_2SO_4 alkylation catalyst from H_2SO_4 alkylation, crude oil storage tank sludge, Claus catalyst and SCOT-like catalyst, catalyst from catalytic reforming, catalyst and fines from catalytic cracking, sludge from sulfur complex and H_2S removal facilities, spent caustic from liquid treating, unleaded gasoline storage tank sediment, off-specification product and fines from thermal processes, and sludge from HF alkylation is adequately supported by these new analyses as well as the original listing proposal. The new analyses clearly support EPA's no list decision for the eleven residuals listed above. In the case of additive risks, EPA could find no circumstance which would suggest that individuals would be simultaneously exposed to residuals managed in an on-site landfill or land treatment unit. Similarly, for co-disposal, EPA's analysis emphasizes the fact that these eleven residuals do not warrant listing, even with great conservatism in the risk assessment. EPA's Monte Carlo analysis strengthens these results and further highlights the conservatism in EPA's risk assessment for both the high-end analysis and the co-disposal scenario. In fact, the Monte Carlo analysis illustrates that the risks predicted by the high-end analysis clearly fall at levels greater than the 95th percentile. (Phillips 00014)

Response: EPA is finalizing its decision to not list all but one of the above wastes (EPA is listing crude oil tank sediment). EPA appreciates the commenter's support for its no-list decisions.

I.A.6. Other Groundwater Modeling Issues

I.A.6.a. Distance to Well

Comment 1: EPA Fails to Follow its Own Policy Decision by Using the OSW Data Base to Determine the Distance Between Landfills and Groundwater Wells.

In the development of EPA's technical approach for this rulemaking, many discussions were held between EPA, industry, and NPRA and API. During these early discussions, industry proposed several alternatives to conducting the RCRA 3007 Survey including the use of existing EPA data such as the OSW data base. EPA strongly objected to the approach of using existing data and stated that the RCRA 3007 Survey and site sampling were the only acceptable approaches to obtaining data for this rulemaking. EPA stated that existing data such as the OSW data base was not acceptable because it was too general and was not specific enough for the refining sector which is the focus of this rulemaking.

EPA's decision to use the OSW data base is in direct contradiction to its earlier determination to base its rulemaking on the RCRA 3007 Survey and site sampling. EPA's subsequent decision to use the OSW data base because the 3007 Survey did not have enough respondents is not defensible.

- EPA had limited resources to conduct the site sampling and implement the RCRA 3007 Survey. In discussions with industry, adjustments were made to both the survey and the site sampling to fit into EPA's budget. It was understood that EPA would use the RCRA 3007 Survey to get a representative sample of the information needed to conduct the analysis. It was never expected to obtain a 100 percent response rate to questions such as distance to the nearest well.

- The 50 percent response rate to the RCRA 3007 Survey is a statistically sufficient sample. There is no reason to believe the missing 50 percent is statistically different than the 50 percent sample responses and a sample of 50 percent is consistent with EPA goal of getting a representative refining industry sample from the survey.

- The selection of the distance to the nearest drinking water well to be used in the ground water analysis will influence the results of the modeling and therefore the listing decision. Because the difference between the RCRA 3007 survey results and the OSW data base is significant (median distance to the nearest well of 2600 meters vs 430 meters, respectively), EPA must use data that most accurately represents the refineries which would be affected by a listing. As EPA stated at the start of this rulemaking, OSW data base is not refinery specific and

residuals being considered for listing. On the other hand, the 3007 survey is specific to refineries and to this rulemaking. (NPRA, 00004, pg 2)

Response: The RCRA Section 3007 questionnaire response for well distances was incomplete. Of the 172 3007 Questionnaires returned, 27 facilities reported the presence of nonhazardous on-site landfills used for the disposal of any waste in the survey in any year. Of these 27, EPA found that only 15 reported the distance to the nearest drinking water well with any reliable documentation (e.g., well location maps, groundwater flow gradients, company survey of nearby wells). ~~(only 10 out of 21 refineries with onsite landfills reported well distances) and inadequate with few supporting documents.~~ Furthermore, wells may be placed closer in the future. Therefore, EPA relied on well distances obtained from the EPA Office of Solid Waste (OSW) Municipal Solid Waste (MSW) Survey. The survey is statistically valid for the nationwide analyses. The Agency believes that the information on distances to drinking water wells in the Municipal survey can be used reasonably reliably compared to the limited number of responses from the refineries. The distribution of domestic drinking water wells around petroleum refining facilities should be reasonably similar. Furthermore, EPA notes that the Questionnaire data only provides well location information for evaluating *on-site* landfills, and even if used, would have not impact the modeling results for off-site landfills. Because the risks from off-site landfills were higher or comparable to risks calculated for on-site landfills, any change in the results for on-site landfills is unlikely to alter any listing decisions.

Comment 2: EPA Has Greatly Underestimated the Distance Between Landfills and Groundwater Wells

The evidence further supports the fact that facility boundaries and residential wells are not closely located. In fact, the results of the additive risk analysis provides further support to the contention that EPA has greatly overestimated the well distance location in the groundwater analysis.

Using the well distances calculated from the RCRA 3007 survey data, API recalculated the 9-year peak groundwater concentration at the receptor well. Based on these values, and using the same high-end parameters, risks from benzene in hydrorefining and hydrotreating catalysts decreased by a factor of three for off-site disposal and a factor of five for on-site disposal. Risks from on-site and off-site disposal of arsenic in hydrorefining catalyst decreased by an order of magnitude.

In the groundwater analysis for hydrotreating and hydrorefining catalysts for the NODA, EPA used the OSW Industrial Subtitle D Waste Management Facility Database to determine the distance to the nearest well for both on-site and off-site landfills. API strongly disagrees with EPA's decision to use the OSW database in place of the data available from the RCRA 3007 Survey. By EPA's own admission, the well distance reported in the OSW database (median of 430 meters) is significantly smaller than the well distances reported by landfills receiving petroleum residuals (median of 2600 meters). Although EPA argues that only 50 percent of the refineries with landfills responded to this survey question, by using the OSW database, EPA has made the assumption that the well distance for all of the non-reporting refineries is less than the

currently reported median. EPA has no basis for making such an assumption; in fact, it seems more likely that a facility did not report a well distance because a well was not located close enough to the facility to be a concern. In the absence of data to the contrary, EPA should use the data available in the RCRA 3007 survey. In fact, in the Background Document for the Groundwater Pathway Analysis, EPA stated that "[a]vailable information [from the RCRA 3007 Survey] was restricted to waste unit size (area and depth), geographical location of the unit, and depth to groundwater and distance to the nearest drinking water well, for on-site waste management units".[65] According to this statement, the results of the RCRA 3007 survey were sufficient to evaluate groundwater risks in the proposed rule.[66]

EPA asserts that the OSW Survey yields data more appropriate well distance data for its on-site landfill risk model than the RCRA 3007 Survey because the RCRA 3007 Survey well distance data had a 50 percent response rate, which EPA believes is too low for yielding useable data. However, API believes that the RCRA 3007 Survey is a more appropriate database for onsite well distance because this database reflects actual well distances for the on-site landfills at refineries. The OSW Survey included all types of RCRA landfills and thus is less directly applicable. At this time, API has no information that suggests that non-respondent facilities from either survey are different from respondent facilities in any way that would affect parameters estimated from their data.

Although the 50 percent response rate for the RCRA 3007 Survey well distance data is not ideal, response rate as an isolated measure does not indicate whether parameters derived from survey results are representative of the target population. In any instance where response rates are less than 100 percent, it is appropriate to question whether the non-respondents differ in some meaningful way from the respondents such that results derived for the respondents are not representative of the survey's target population. For this reason, non-respondent follow-up, whereby a separate sampling effort is devoted to contacting and collecting information from a sample of non-respondents, is a part of some survey protocols. Because the RCRA 3007 Survey did not include non-respondent follow-up, there is some unavoidable uncertainty about the applicability of the data to its target population.[67]

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[65]U.S. EPA, 1995. Listing Background Document. p. 11.

[66] A review of the distance to well values used in the groundwater model indicate that EPA did, in fact, use the OSW database in the 1995 proposal, although information in the background document suggests otherwise.

[67] Non-respondent follow-up efforts allow both reduction of uncertainty and, if necessary, refinement of parameters estimated from the respondent survey data. In the case where non-respondents differ from respondents only in their choice of whether to respond to the survey, then no adjustments are necessary and the respondents' data appropriately represent the target population.

The well distance parameters estimated from the RCRA 3007 and OSW Survey data are different enough to indicate that the OSW Survey does cover landfills with characteristics different from those of the refineries. The OSW Survey yields a high-end (10th percentile) well distance value of 120 meters and a median of 430 meters. The 10th percentile and median well distance from the RCRA 3007 Survey are 625 meters and 2310 meters, respectively. In order for the RCRA 3007 Survey and the OSW Survey to indicate the same parameters for the risk model, only five percent of the non-respondent facilities could have well distances that are within the range established by the respondents. A full 75 percent of the non-respondents from the RCRA 3007 Survey would have to have well distances between 120 meters and 305 meters, and 20 percent would need to be 120 meters or less. It is highly unlikely that the non-respondent facilities from the RCRA 3007 Survey are this different, as a group, from the facilities that responded. The facilities covered by the OSW survey and the RCRA 3007 survey are of different types and have different characteristics.

Thus, the OSW Survey clearly includes landfills that are different from the on-site refinery landfills. Therefore, the parameters derived from the RCRA 3007 Survey are more appropriate for this rulemaking than those from the OSW Survey. Concerns over response rate for the RCRA 3007 Survey are outweighed by the strong possibility that the well distance data derived from the OSW Survey are not applicable to the on-site refinery landfills.

Using the available well distance data from the RCRA 3007 survey, API modeled the distribution of well distances. API performed statistical tests to determine the distributional form of the data and determined that the data were log normally distributed. A Monte Carlo simulation of the distribution using the mean and standard deviation of the available data and the @RISK software (Palisade Corp.; 5,000 iterations) revealed a median well distance of 2339 meters and a high-end distance (10th percentile) of 847 meters. These values contrast significantly with the median and high-end distances from the OSW database (430 meters and 120 meters, respectively). The results of this analysis clearly indicate that the data from the OSW survey greatly underestimate the distance from on-site refinery landfills to the nearest receptor wells. Use of such unrealistic data in the rulemaking would violate the requirement in 40 CFR § 261.11(a)(3)(vii) that EPA consider "plausible" mismanagement scenarios.

Using the well distances calculated from the RCRA 3007 survey data, API recalculated the 9-year peak groundwater concentration at the receptor well. Based on these values, and using the same high-end parameters, risks from benzene in hydrorefining and hydrotreating catalysts decreased by a factor of three for off-site disposal and a factor of five for on-site disposal. Risks from on-site and off-site disposal of arsenic catalyst decreased by an order of magnitude. (API, 00009, pg 14)

Response: See Response to Comment 1 above for the issue of why the RCRA §3007 well distances are not appropriate. As for the issue of what was reported in the 1995 Groundwater Pathway Background Document "A review of the distance to well values used in the groundwater model indicate that EPA did, in fact, use the OSW database in the 1995 proposal, although information in the background document suggests otherwise", the document said well distances

were available in the survey responses it did not explicitly say well distances from the RCRA §3007 were used in the data sources section (Section 3.2) of the background document. EPA also notes that the commenter's suggestion that the Questionnaire well distance data be used for *of-site* landfills is not logical. The Questionnaire provides information only about well distances for *on-site* landfills and would not be appropriate to use for off-site landfills under any circumstances

Comment 3: RCRA 3007 database information should be utilized to assess distances from onsite management units to down gradient drinking water wells

Phillips call your attention to another aspect of the modeling effort used to support EPA's proposed hazardous waste listing for selected petroleum refining residuals. This is of particular importance to use because several of our refineries have onsite waste management facilities long distances form potential receptor wells. We believe this to be a common situation as many refineries are sited on large tracts of land, often purchased expressly to provide a buffer between plant operations and human inhabitants. EPA therefore should not discount the differences in distance to down gradient wells between onsite and off-site waste management units.

In the groundwater analysis for hydrotreating and hydrorefining catalysts for the NODA, EPA used the OSW Industrial Subtitle D Waste Management Facility Database to determine the distance to the nearest well for both on-site and off-site landfills. Phillips strongly disagrees with EPA's decision to use the OSW database in place of the data available from the RCRA 3007 Survey. By EPA's own admission, the well distance reported in the OSW database (median of 430 meters) is significantly smaller than the well distances reported by landfills receiving petroleum residuals (median of 2600 meters). Although EPA argues that only 50 percent of the refineries with landfills responded to this survey question, by using the OSW database, EPA has made the assumption that the well distance for all of the non-reporting refineries is less than the currently reported median. EPA has no basis for making such an assumption; in fact, it seems more likely that a facility did not report a well distance because a well was not located close enough to the facility to be a concern or interpreted the 3007 questionnaire to limit responses to distances less than 500 feet. In the absence of data to the contrary, EPA should use the data available in the RCRA 3007 survey. In fact, in the Background Document for the Groundwater Pathway Analysis (p. 11), EPA stated that "Available information [from the RCRA 3007 Survey] was restricted to waste unit size (area and depth), geographical location of the unit, and depth to groundwater and distance to the nearest drinking water well, for on-site waste management units." According to this statement, the results of the RCRA 3007 survey were sufficient to evaluate groundwater risks in the proposed rule, however before publication of the proposed rule had been rejected in favor of the OSW survey data.

EPA believes that the OSW Survey yields data more appropriate well distance data for its on-site landfill risk model than the RCRA 3007 Survey because the RCRA 3007 Survey well distance data had a 50 percent response rate, which EPA believes is too low for yielding useable data. However, API believes that the RCRA 3007 Survey is a more appropriate database for onsite well distance because this database reflects actual well distances for the on-site landfills at refineries.

We were surprised to learn form the March 1997 Supplemental Background Document (Groundwater Pathway Risk Analysis (pages 2-10)) that the nearest drinking water well distance parameter was derived exclusively from the OSW Industrial Subtitle D Waste Management Facilities Database for receptor well distances from both onsite and offsite waste management facilities. This distance was determined to be 430 meters (median) and 102 meters (10th percentile). Data from the RCRA 3007 database were not used to represent well distances from onsite units ostensibly because the database was incomplete. Documentation of this discrepancy includes:

- Review of the preamble to the subject proposed rule (60 FR 57747) and its supporting (August 1995) background documents, found mention of the use of the same receptor well distances (i.e., 102/ 430) but specifically indicated that they were derived from the RCRA 3007 database (e.g., August 1995 background documents for ground waste pathway analysis (pg. 30)). In the preamble (60 FR 57759 col. 2) to the proposed rule EPA explicitly states, "For onsite landfills and LTU's the Agency used the data submitted by the industry in the Section 3007 questionnaire to characterize the units in terms of the waste quantities disposed, surface area of disposal units, and distances to receptors. For offsite landfills and land treatment facilities from the Agency's "Industrial D Facility Study" (October 20, 1986)."

- The subject preamble additionally indicates that EPA completed an "exhaustive engineering review" of each 3007 response and "believes that the data are reliable and represent the industry's current residual generation and management practices" (60 FR 57751 col. 3). This does not track with the well receptor distance for onsite facilities as this particular 3007 survey data was rejected by EPA without explanation until the issuance of the March 1997 Background Document referenced above. Furthermore, there is no mention of this disparity, that we could fine. In the text of the NODA (62 FR 16747) although a short explanation is provided in a March 1997 supporting background document.

Having participated in the review of the RCRA 3007 survey as it was being prepared, we recall that the technical review time for Section IX and X was severely limited as EPA was anxious to distribute the survey. Section IX.C requests the "closest unit boundary to the nearest down gradient (public) private drinking water well (feet)." A similar question in Section X.7, requests a map locating all public and private drinking water wells with in 1 mile of the facility boundary. This could explain why some refiners reported >1 mile rather than an exact distance. EPA could easily solicit an actual distances from these refiners to improve the database information. Any other 3007 responses could similarly be addresses by a simple phone call, EPA did significant follow up with individual refiners to verify information contained in other portions of the 3007 survey, why should potential data concerns for this parameter be handled differently? EPA's decision to jettison all the well distance data for onsite management units appears to be extreme when follow-ups with refiners can be easily accomplished. If EPA is uncomfortable with the response rate it has ample time to contact the facilities that report having onsite landfills to seek clarification or correction of submitted data. These follow-ups would allow EPA to substitute the data for onsite units into modeling runs well in advance of a final decision for the hazardous waste

listing of petroleum refining residuals. EPA as previously made inquiries of individual refiners to correct errors, omissions, and misunderstandings in other areas of the 3007 questionnaire. Obtaining clarification on this specific issue is not likely to tax EPA resources as the number of responses that require reevaluation is likely small, perhaps on the order of 25-30.

Using the available well distance data from the RCRA 3007 survey, [i.e., as reported in correspondence from Vierow (SAIC) to Saleem (EPA)] API modeled the distribution of well distances. API performed statistical tests to determine the distributional form of the data and determined that the data were log normally distributed. A Monte Carlo simulation of the distribution using the mean and standard deviation of the available data and the @RISK software (Palisade Corp.; 5,000 iterations) revealed a median well distance of 2339 meters and a high-end distance (10th percentile) of 847 meters. These values contrast significantly with the median and high-end distances from the OSW database (430 meters and 120 meters, respectively). The results of this analysis clearly indicate that the data from the OSW survey greatly underestimate the distance from on-site refinery landfills to the nearest receptor wells. (Phillips, 00014)

Response: The reference for well distances on page 30 (Table 4.4) of the August 1995 Background Document for the Groundwater Pathway analysis was incorrectly listed as the RCRA Section 3007 survey, the correct citation should have been the OSW Survey of Municipal Landfills. See response to Comment 1 in this section for a response to this issue.

I.A.6.b.   Active Life

Comment 1: EPA assumed a 20-year active life for off-site landfills. See Supplemental Background Document; Listing Support Analyses; Petroleum Refining Process Waste Listing Determination; Active Lives of Landfills Used for Disposal of Petroleum Refining Wastes. This factor is very important because the waste volume inputs to the EPACMTP model are derived by multiplying the annual waste volume by the unit active life. Therefore, an inaccurate assumption of a short active life for landfills will cause the groundwater model to significantly underestimate risks.

The sole basis for EPA's assumption of a 20-year life for landfills is a 1986 survey of municipal landfills. See National Survey of Solid Waste (Municipal) Landfill Facilities, EPA/530-SW88-034 (Sept. 1988). This report states: "The average age of a facility is estimated to be 18.6 years." Id. at 8-1. In context, however, it is clear that this statement refers to the age of the facility at the time the survey was conducted, not the anticipated active life of the landfill unit. EPA simply misconstrued this report.

In an appendix to the survey report, the year that waste was first placed in the surveyed landfill and the year all active and planned units were expected to be completely filled were reported. The average responses were 1967 as the year the landfilled began operation and 2007 as the year of expected closure. Thus, the average active life actually reported in the survey was 40 years.

Moreover, the survey reported that 59% of national capacity remained at the surveyed landfills to be utilized. This means that 41% of the average landfill capacity was used over 18.6 years, so that all landfill capacity would be utilized in approximately 45.4 years. Clearly, an average life of 40-45 years is an accurate assumption -- not 20 years.

KGS conducted revised modeling runs using a 40-year life for off-site landfills. Based on KGS's evaluation of this factor and other parameters, the risks posed by potential mismanagement of most petroleum refinery wastes covered by the NODA require a hazardous waste listing, as discussed in Section II below. (ETC, 00005, pg 4)

Response: EPA reexamined the report cited and concluded that the assumed active life of 20 years may be an underestimate. Using the data in the report, however, the Agency calculated that an average active life of 30 years is accurate, rather than the 40 year life suggested by the commenter. EPA believes that the commenter simply summed the reported average age of the landfills (19 years) and the average remaining life (21 years) to obtain 40 years. This calculation, however is not accurate, because it would overestimate the active life for existing units. This is because the average age in the report included *closed* units, not only existing units, and thus does not reflect merely the average life for units still in operation. Likewise, the average remaining life in the report included *planned* units, as well as existing units, and this also would tend to increase the apparent active life for existing units. Correcting for this, EPA calculated a 30 year active life, based on corrected values of 16.5 years for the average age of active units, and 13.3 years for the average remaining life (see *Additional Listing Support Analysis,* 1998 in the docket to today's rule for full calculations). EPA has used the revised active life (and correspondingly larger volumes) to calculate the new risk numbers.

Comment 2: In the groundwater risk assessments performed in support of the 1995 proposal, EPA assumed a 20 year active life for both landfills and surface impoundments. The assumption is extremely important, because the waste volume inputs into the groundwater model are derived by multiplying the annual volume (based upon 1992 data) and the length of the unit active life. Therefore, an arbitrarily short unit active life can substantially understate the very important waste volume parameter used in the groundwater modeling.

EDF challenged the 20 year active life assumptions in its comments on the 1995 proposal, based upon the lack of factual basis for the assumptions and the large number of surface impoundments and landfills identified in refinery RCRA Facility Assessments (RFAs) that had active lives substantially beyond 20 years. In the NODA materials, EPA attempts to provide additional justification for the 20 year landfill assumption.[68] As explained in this portion of the comments, the information provided in the NODA actually supports at least a 40 year active life assumption for landfills.

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[68] No attempt was made in the NODA to justify the 20 year surface impoundment active life assumption.

With respect to offsite landfills, the sole basis for EPA's 20 year assumption is a survey of municipal waste landfills conducted by an EPA contractor beginning in 1986 and published in a 1988 report.[69] The target population of the survey was municipal landfills with at least one active unit as of November 1, 1986. Closed and planned units were included in the survey only where the facility had at least the one active unit.[70]

The age of the surveyed facilities is summarily discussed in one sentence of the body of the report. The sentence reads "The average age of a facility is estimated to be 18.6 years."[71] Apparently, EPA has interpreted this sentence to signify landfill age at closure, when in fact it is clear from the rest of the document that the 18.6 years simply pertains to age at the time the survey was conducted. Therefore, the 18.6 years bears no relationship to the length of the entire active life of the landfill.

Elsewhere in the report, the contractor notes 59% of the national capacity at the surveyed landfills still remains to be utilized. Moreover, this percentage may be under reported since it does not reflect new units to be built at the surveyed facilities when the additional capacity is required.[72] On a per facility basis, the average remaining capacity was also approximately 59%.[73]

Thus, more relevant to the instant rulemaking than the 18.6 year value are the responses to questions reported regarding the year waste was first placed in the landfill and the year all active and planned units are expected to be completely filled, as provided in the appendix. The average responses to those questions were 1967 and 2007 respectively, meaning the average active life actually reported in the survey was 40 years.[74]

Further, and of paramount importance, this 40 year period is largely confirmed when considering the remaining capacity in these facilities. If 41% of a facility's average landfill capacity was used over 18.6 years, all of the landfill capacity would be utilized in approximately 45.4 years, assuming historic landfilling rates continue to apply. Again, as the contractor noted, even this 40-45 year average active life is an understatement of the actual active life of the surveyed landfills,

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[69] See NODA Groundwater Risk Assessment at 2-4.

[70] National Survey of Solid Waste (Municipal) Landfill Facilities, EPA/530-SW88-034, September 1988 (hereafter "EPA Offsite Landfill Survey"), p. iv.

[71] EPA Offsite Landfill Survey at 8-1.

[72] EPA Offsite Landfill Survey at 7-3.

[73] EPA Offsite Landfill Survey, Table 7.5.

[74] EPA Offsite Landfill Survey, pp. A-2, A-3.

since the reported values do not fully reflect units to be built when additional capacity is required.[75]

In the case of onsite landfills, EPA calculated the active life of refinery waste landfills using the 1992 data provided as part of the industry surveys in the instant rulemaking. The active life was calculated using three different methodologies, and not surprisingly, EPA chose the methodology that produced the shortest active life irrespective of its relationship to reality.

Under Method 1 ultimately chosen by the Agency because it is "simple", EPA generally assumed the active life equaled the dates in the survey provided for facility start-up and projected closure.[76] Under Method 2, the projected closure date was based upon the remaining capacity of the landfill, assuming the rate of capacity usage in 1992 held true in the future. Under Method 3, the projected closure date was based upon the percentage of capacity used historically (not just 1992).

Method 1 produced a median value of 21.5 years, which EPA uses as the basis for the onsite 20 year active life value in the groundwater model.[77] Before addressing the validity of Method 1, several points should be noted from the outset. First, none of the EPA methods take into account additional capacity to be built when needed, therefore all methods reflect the same under reporting noted by the EPA contractor for the offsite landfill survey. Second, the <u>average</u> value calculated using Method 1 is 26.2 years.[78] Since EPA uses the average value for its offsite facility assumption (see NODA Groundwater Risk Assessment at 2-4), the average value should be used for onsite facilities as well.

More importantly, however, simplicity is an absurd justification for relying upon Method 1 in the first place. It is the least accurate of the three options developed by EPA, because it is the only option wholly unrelated to the remaining landfill capacity. Unlike the offsite landfill survey where the average survey responses and remaining capacity roughly correlate, the onsite landfill survey responses and remaining capacity analyses are not consistent. In the face of such inconsistency, it is far less accurate for EPA to rely solely upon unverified closure projections, the basis for which

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[75] Approximately 150 landfill expansions per year were approved in the years preceding the survey. See Solid Waste Disposal in the United States, Volume II, EPA/530-SW-88-011B, October 1988, p. 4-14.

[76] NODA Background Document, Ch. 6, p. 2.

[77] NODA Background Document, Ch. 6, p. 1.

[78] This average was derived from the "Calculated Active Life" values reported in Attachment 1 to Chapter 6 of the NODA Background Document.

is entirely unknown to the Agency,[79] when compared to the more objective inquiry based upon remaining capacity.

The fallacy of Method 1 is perhaps best illustrated by its application to landfill numbers 45 and 101 in EPA's data base. Under Method 1, landfill 101 would have an active life of 30 years. Yet since the landfill opened in 1983, about 4% of the landfill capacity has been utilized, including the 0.5% capacity used in 1992.[80] If this landfill closed in 2013 as predicted using Method 1, it would close about 87% empty based upon usage in the 1983-1992 time period. Similarly, landfill 45 would have an active life of 39 years using Method 1, but since it opened in 1991, much less than 1% of the landfill was utilized. Under Method 1, the landfill would close in 2020 almost completely empty based upon usage in the 1991-1992 time frame.[81] Therefore, Method 1 results in the irrational closing of landfills well before the vast majority of the capacity is utilized.

Methods 2 and 3 are both superior to Method 1 because they rely upon a reasonable and objective basis for projecting landfill closures instead of an unknown, unverified subjective basis for such projections. Ironically, Method 2 is also the option most consistent with assumptions used by the Agency in other parts of the rulemaking. The assumption inherent in Method 2 that the data reported for 1992 is an accurate indicator of future management practices is precisely the assumption used by EPA in other parts of its risk assessments, including the waste volume parameter. For example, EPA assumes the volumes of waste generally annually are the same amounts generated in 1992, and as noted above, it is this annual volume rate that is multiplied by the active life to derive the volume input into the groundwater model. It is arbitrary and capricious for EPA to base much of its modeling upon 1992 data, including one of two values used to derive volume projections, only to reject a methodology based upon the identical assumption because another option producing a less protective result is "simpler".[82]

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[79] Indeed, it is not merely a coincidence that for 15 of the 22 landfills whose active life was calculated using Method 1, either the projected closure year or the projected active life reported in the survey response was a number ending in five or zero. EPA should assume the projected closure dates that form the basis for Method 1 are nothing more than rounded guesstimates offered to the Agency in response to a survey question requiring projections years or decades into the future.

[80] NODA Background Document, Ch. 6, Attachment 1.

[81] Id.

[82] Moreover, it is far from clear why Method 1 is "simpler" since EPA has already calculated the numbers for all three methods. Also, since when does simplicity justify compromising on accuracy?

Significantly, the average active life of an onsite landfill using Method 2 is 39.2 years.[83] Accordingly, use of Method 2 more closely tracks the results of the offsite municipal landfill survey discussed above. It would be illogical to expect onsite landfills to have a shorter active life than municipal waste landfills due to the economic advantages of operating onsite landfills for facilities such as refineries generating substantial quantities of wastes, and the greater regulatory attention devoted to municipal waste landfills subject to federal technical requirements and mandatory state permitting.[84]

In fact, given the likelihood that refineries will be landfilling wastes onsite for the indefinite future, and building capacity as needed to engage in that practice, the Method 3 average active life of 56.6 years may be the most accurate number of the three methods. Accordingly, while the Method 2 average active life would essentially double EPA's assumed active life, it is certainly "plausible" that an onsite refinery waste landfill would be operating for a much longer period of time. See 40 CFR 261.11(a)(3)(vi). Since the high-end volumes in the groundwater modeling do not reflect longer active lives than the average values (i.e., only the annual generation rate is adjusted, not the multiplier reflecting the landfill active life), EPA's current modeling methodology does not account for longer than average active lives, irrespective of the ultimate value chosen.

In isolation, the volume parameter understatement is most important in the case of unleaded gasoline storage tank sludge. Adjusting only the volume parameter for the onsite landfilling scenario to reflect 39.2 years of median annual waste volume, KGS determined the resulting groundwater pathway risk doubled when compared to EPA's results. Perhaps more importantly, when the volume parameter is modified in conjunction with other necessary changes as discussed below in Section II.I [of EDF's comment], the risk results are more widely and profoundly increased.

To summarize, there is no factual basis for the assumed 20 year active landfill life in EPA's groundwater modeling. The report and data relied upon by EPA for the assumption actually demonstrate an active life at least twice that length is required. Moreover, by not adjusting the high-end volume parameters to reflect longer than average active lives, EPA high-end volume parameters do not reflect actual high-end conditions. (EDF, 00006, pg 26)

Response: See response to Comment 1 above for off-site landfill life. In the NODA, EPA provided an analysis of the data for on-site landfills for refineries from the §3007 Questionnaire, showing a calculated median of about 21 years for on-site landfills. The commenter disputed the

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[83] The average is derived based upon the Calculated Active Life values provided for Method 2 in the NODA Background Document, Ch. 6, Attachment 1. Where the Method 2 values are greater than 100 for individual facilities, Method 3 values were used consistent with EPA's approach for Method 1. See NODA Background Document, Ch. 6, p. 3.

[84] See Section 4005© of RCRA.

20 year calculation, and cited an alternative method presented by EPA in the NODA to calculate a 39 year average (i.e., mean) active life, which the commenter argued EPA should use.

In the NODA analysis, the Agency used the projected date for closure of on-site landfills reported by refineries in the Questionnaire to estimate active lives. EPA also examined alternative methods to calculate on-site landfill life for use when facilities did not report the projected date of closure. These alternative methods required EPA to use the remaining capacity reported for the units, assume disposal rates for all wastes in the landfills would remain constant, and thereby estimate when the landfill may reach full capacity. EPA believes the direct method chosen is most appropriate because it uses the actual landfill lives reported in the questionnaire, rather than relying on estimating remaining active life by projecting past waste disposal rates into the future. This alternative approach is especially uncertain when the landfill are relatively new, thereby requiring the extrapolation of a small percentage of used landfill capacity into the far future, which means that small variations or errors in the used capacity of a landfill may lead to widely varying landfill life projections. Thus, EPA did not revise its modeling for on-site landfills to reflect a longer landfill life. EPA also used the median active life, rather than the average suggested by the commenter, because the median value lessens the impact of widely variable data and outliers. EPA notes that the only data available for off-site municipal landfills were average values, not medians, so the Agency had no choice but to use the average estimate active life for the off-site landfills.

Comment 3: EPA has appropriately determined the active lives of landfills. In Section 6.0 of the Listing Support Analyses, EPA describes three methods for calculating the active life of a landfill. EPA has appropriately selected Method 1 as the optimal approach. In both Method 2 and 3, assumptions regarding prediction of future disposal patterns and landfill closure only after the landfill reaches capacity are likely to overestimate the life of the landfill, as supported by EPA's calculations. (API, 00009, pg 24)

Response: The Agency appreciates the commenter's supportive comment; EPA tends to agree with this comment and believes that Method 1 provides the best estimate of active life for on-site landfills. EPA further notes that Methods 2 and 3 are extremely sensitive to the reported volumes disposed in 1992, and/or the capacity of the landfill used to date. If the capacity remaining is large, or the volume of waste disposed in 1992 very small, this will lead to potentially large overestimates in active life.

I.A.6.c.   Landfill Size

Comment 1: In previous listing determinations, EPA has always used a standard size for both off-site and on-site landfills. In this rulemaking, however, EPA decided to vary the size of on-site landfills depending upon the waste streams evaluated. The landfill sizes varied from 24,594 square meters for CSO sludge to 145,692 square meters for off-spec thermal products and fines for the median value, and from 28,329 square meters for HF alkylation sludge to 202,350 square meters for FCC fines and catalysts for the high-end value. Of course, landfill size is an important

variable in the EPACMTP model, and manipulation of this variable can significantly affect the risk results.

EPA has provided no explanation for its decision to vary on-site landfill sizes to such a significant degree for different petroleum wastes when projecting plausible mismanagement scenarios. Since any petroleum waste can be disposed in any on-site landfill, plausible mismanagement should assume that any waste will be disposed in units representing the largest median and high-end values. It simply makes no sense for EPA to evaluate the risks posed by disposal of HF alkylation sludge in a landfill that is arbitrarily just 1/5 the size of the landfill used to evaluate disposal of off-spec thermal products and fines. It also makes no sense to assume a standard size for off-site landfills, but to dramatically vary the size of on-site landfills for various petroleum wastes.

For this reason, KGS conducted its modeling using the CSO waste median and high-end landfill area and depth for all on-site landfill disposal scenarios. Based on KGS's evaluation of this factor and other parameters, the risks posed by potential mismanagement of most petroleum refinery wastes covered by the NODA require a hazardous waste listing, as discussed in Section II below. (ETC, 00005, pg 6)

Response: The selection of landfill sizes in the on-site landfill modeling was strictly based on data reported in the RCRA §3007 survey. Onsite landfill sizes were wastestream-specific because the disposal data were wastestream-specific. An explanation was provided in the groundwater pathway background document, that areas were used as reported in the survey. The area distributions were not manipulated in any way. Wastestream-specific landfill size data were not available for offsite landfill sizes, therefore, standard sizes were used in the case of offsite landfills. EPA believes its approach of calculating different unit areas for different wastes was reasonable because they are reflective of actual operating practices, and another approach may result in unrealistic or unreasonable assumptions regarding waste management practices.

Furthermore, for a fixed waste volume, larger landfill areas may not necessarily produce greater groundwater risk. In addition, for fixed values of waste quantity, landfill area within a certain range is not necessarily a significantly sensitive parameter and, in fact, very large landfill areas can actually produce lower receptor well concentrations. See the Additional Groundwater Pathway Risk Analyses, Supplemental Background Document (1998). See also response to NPRM comment 1 of Section III.J. To support this, EPA notes that unit area was a high end parameter for only one of the scenarios (hydrorefining catalyst). Conversely, waste volume is a high end parameter for most of the scenarios.

Table I.A.1, below, summarizes unit area data from various sources in the docket. Specifically, the landfill areas used for each waste are presented in conjunction with the "universe" of landfills (as suggested by the commenter). In comparing waste-specific unit areas to the statistics generated when considering all landfills (in the first row), median areas for most wastes are higher while high end areas for most wastes are lower.

Table I.A.1. Landfill Sizes Used for Onsite Landfilling of Petroleum Refining Wastes [a]				
Waste Type	Number of Landfills	Area, 50th %ile, sq. meters (acres)	Area, 90th %ile, sq. meters (acres)	High End Parameters [e]
Contingent Management for CSO sediment [b]	33 [b]	25,000 (6.166)	202,000 (50)	Waste volume, well centerline
Co-disposal analysis [c]	8 [c]	---	202,000 (50)	---
CSO sediment	2	62,000 (15.2)	121,000 (30)	Waste volume, receptor distance
FCC catalyst and fines [d]	16	29,000 (7.13)	202,000 (50)	---
Hydrotreating catalyst	5	30,000 (7.38)	121,000 (30)	Waste volume, receptor distance (for benzene); infiltration rate, receptor distance (for TC cap benzene); waste volume, TCLP concentration (for arsenic)
Hydrorefining catalyst	4	31,000 (7.7)	121,000 (30)	Waste volume, unit area (for benzene); infiltration rate, receptor distance (for TC cap benzene); waste volume, receptor distance (for arsenic)
Unleaded gasoline tank sludge [d]	6	30,000 (7.5)	146,000 (36)	---
Off-spec product and fines from thermal processing	3	146,000 (36)	202,000 (50)	Waste volume, receptor distance
HF alkylation sludge	1	28,000 (7)	28,000 (7)	Infiltration rate, receptor distance

a. Unless otherwise indicated, all surface area data are from Appendix C of "Supplemental Background Document for Groundwater Pathway Risk Analysis," 1997. The number of landfills used as the bases for these statistics is from "Listing Background Document for the 1992-1996 Petroleum Refining Listing Determination," 1995 (e.g., page 92). The above table shows, for example, that four onsite landfills were used for the disposal of hydrorefining catalyst in any year reported (not necessarily in 1992). The median value and 90th percentile unit areas for this subset of refinery landfills is as indicated (because of the small number of data points in this case, the 90th percentile value corresponds to the maximum value). This information describing how area statistics were determined is presented as footnotes on page 92 of "Listing Background Document," and in similar tables throughout that report.
b. Thirty three landfills accepted any of nine petroleum refining wastes in any year and were used for the active life calculations described in the NODA. (These nine wastes, representing those listing wastes for which onsite landfill modeling was conducted, are CSO sediment, FCC catalyst and fines, hydrotreating catalyst, hydrorefining catalyst, Claus and SCOT catalyst, unleaded gasoline tank sludge, off-spec product and fines from thermal processing, HF alkylation sludge, and sulfur sludge. Onsite modeling for crude oil tank sludge was not conducted.) Surface area data from these 33 landfills were used for developing the unit area assumptions of the contingent management scenario.
c. From Appendix D of Supplemental Groundwater Pathway Risk Analysis. Unit area determined by distributing 90th percentile unit areas for 8 wastes and taking the 90th percentile value of this array.

d. Data for unleaded gasoline tank sediment and FCC catalyst and fines are taken from "Listing Background Document," 1995, pages 39 and 68, respectively.

e. From Table C.1 of Supplemental Groundwater Pathway Risk Analysis. Sensitivity analyses were conducted for each waste, scenario (i.e., onsite or offsite) and constituent. For each model run, two of the following parameters were set to high end: unit area, waste volume, TCLP concentration, infiltration rate, receptor well distance, distance from centerline, and receptor well depth. For each waste, a different combination of high end parameters results in the highest receptor well concentration.

Comment 2: Landfill area size continues to be one of the most important modeling parameters in EPA's groundwater risk assessment. In fact, landfill area is one of two default high-end parameters used by EPA where it failed to conduct a sensitivity analysis in the NODA Groundwater Risk Assessment (i.e., co-disposal, unleaded gasoline storage tank sludge).

In previous listing determinations, EPA employed a standard onsite and offsite landfill for the purpose of setting landfill area, depth, and other relevant modeling inputs.[85] Unfortunately, once again EPA has diverted from past practice and used modeling inputs that cannot be sustained.

Onsite landfill areas vary substantially depending upon the waste stream. The areas range from 24,594 square meters (contingent CSO sludge) to 145,692 square meters (off-spec thermal products and fines) for the median area value, and from 28,329 (HF alkylation sludge) to 202,350 square meters for the high-end value (FCC fines and catalysts).

EPA offers no explanation in the 1995 proposal or the NODA as to why the onsite landfill sizes should vary so much when projecting plausible mismanagement scenarios, other than the varying sizes simply reflect those landfills that happened to be used for particular wastes in 1992. In fact, since there is no technical or legal bar for any onsite nonhazardous refinery landfill to receive any nonhazardous refinery waste, the groundwater modeling is not indicative of the plausible types of mismanagement the Agency is required to consider pursuant to 40 CFR 261.11(a)(3)(vii). Indeed, since EPA is required by law to assess present and potential hazard, EPA's failure to assess the potential risks posed by the disposal of HF alkylation sludge in a landfill larger than 1/5 the area of the median off-spec thermal products and fines onsite landfill is arbitrary and capricious.[86]

EPA implicitly acknowledges the lack of validity in its onsite landfill area projections by using standard offsite landfills for all wastes, and a standard onsite landfill when evaluating the co-disposal scenario. In the offsite case, EPA uses the standard landfill for all wastes

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[85] See e.g., Assessment of Risks from the Management of Carbamate Wastes: Background Document, EPA, January 1994, pp. 39-40.

[86] For the same reason, it is also absurd that the median and high-end landfill areas values are the same for HF alkylation sludge. See NODA Groundwater Risk Assessment, Table C.29.

notwithstanding the fact that individual wastes were disposed at different landfills in 1992.[87] Thus, EPA employed a wholly different approach for offsite landfills v. onsite landfills, despite the similarity in circumstances presented. In both instances, landfills may receive individual refinery wastes in one year but not the next.

As discussed above, EPA modeled the onsite landfill co-disposal scenario using a high-end area value, and as Table D.2. of the NODA Groundwater Risk Assessment reveals, chose one landfill area size to represent the co-disposal scenario.[88] Therefore, in assessing the co-disposal scenario, EPA projected the landfilling of HF alkylation sludge and other wastes in larger landfills than it projected when evaluating the wastes individually. Since the same onsite landfills are the subject of both analyses, the co-disposal modeling recognizes that refinery wastes may be managed in any onsite landfill in a given year. It is arbitrary and capricious for EPA to project potential mismanagement of refinery wastes in the larger onsite landfills under one scenario but not the other.

KGS conducted its individual waste composite modeling using the CSO median and high-end landfill area and depth characteristics for all onsite scenarios. In this manner, the KGS composite modeling is more reflective of refinery waste plausible mismanagement. The composite modeling results are discussed below in Section II.I of the comments.

Finally, while the Agency employed a standard offsite landfill area for all wastes, the areas values selected were arbitrarily small (2,020 square meters median; 162,000 square meters high-end). EPA purportedly derived these area sizes from an industrial landfill survey database, but to the best of our knowledge, neither the survey report or the methodology used to gather the data has been provided and subject to notice and comment in this rulemaking. Instead, the NODA Groundwater Risk Assessment simply references other documents, which in turn, simply reference the database without providing comprehensive information on survey methodology.[89] Therefore, we are unable to comment on the validity of the database except to note it appears to be the same database prepared for the HWIR rulemaking where the Agency candidly admitted the survey "was not designed to collect accurate estimates" on landfill area. See 60 FR 66358 (December 21, 1995).

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[87] See NODA Groundwater Risk Assessment, Table B.2.

[88] EPA chose the highest high-end value of 202,350 square meters.

[89] See NODA Groundwater Risk Assessment, p. 2-3; EPACMTP Background Document for Finite Source Methodology, 1996, p. 41. Moreover, the 1997 model user's guide also referenced in the NODA Groundwater Risk Assessment indicates the median industrial landfill area is 16,188 square meters, approximately eight times larger than the median value used in the instant rulemaking. See EPA's Composite Model for Leachate Migration with Transformation Products - User's Guide, EPA, 1997, Table 4.2.

Furthermore, insofar as EPA is relying upon the same survey referred to in HWIR, this survey was taken of <u>onsite</u> industrial waste landfills, and therefore does not even apply to the relevant universe of facilities.[90] Moreover, since municipal solid waste landfills can receive the refinery wastes at issue in this rulemaking,[91] and EPA relies upon municipal landfill data for other offsite facility attributes (i.e., length of active life), it is arbitrary and capricious for EPA to rely upon irrelevant onsite landfill data when offsite landfill data is available (see discussion immediately below) from the same source EPA uses for other assumptions in its modeling.

The average area reported for active municipal waste landfills in EPA's 1986 survey was 131,527.50 square meters (32.5 acres), an area many times greater than EPA's median value in the instant rulemaking, and almost equal to the high-end value. See EPA Offsite Landfill Survey at A-12. Moreover, <u>more than half of the municipal waste landfills surveyed in 1986 were between 10-100 acres, or at least 40,470 square meters (over 20 times larger than EPA's median value)</u>. Over six percent of the municipal waste landfills were larger than 100 acres, or greater than 404,700 square meters.[92]

As EDF noted in its comments on the 1995 proposal, the high-end value in the instant rulemaking is about 1/6 the area of the high-end 949,317 square meter Subtitle D landfill modeled in the carbamates risk assessment. The central tendency value in the carbamates risk assessment was 451,262 square meters, approximately 223 times larger than the median offsite landfill are value used in the NODA.

In summary, onsite landfill area sizes inappropriately vary by waste stream based on data for 1992 only that is not determinative of practices in any other given year, consequently for some wastes the modeled areas are arbitrarily small. For offsite landfills, EPA correctly employed standard landfill area values, but the values selected are also arbitrarily small because they lack any relevant factual basis, and are inconsistent with previous rulemakings. (EDF, 00006, pg 41)

Response: EPA addressed the reasons for using different onsite landfill areas for different wastes in comment 1 (immediately above). EPA also wishes to correct the commenter's statement that EPA only considered landfills used during 1992. In fact, EPA considered all landfills used in any year for the disposal of each waste, therefore significantly increasing the number of landfills used

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[90] 60 FR 66358 (December 21, 1995). Not only are the onsite survey data irrelevant, but they are also inaccurate as far as onsite refinery landfills are concerned. The median area value of 2,020 square meters purportedly derived from the onsite industrial landfill survey is between 12-72 times smaller than the median areas EPA modeled for onsite refinery landfills. See NODA Groundwater Risk Assessment, Table 3.3.

[91] See 40 CFR 258.2.

[92] Solid Waste Disposal in the United States, Volume II, EPA/530-SW-88-011B, October 1988, Table 4-5.

in the assessment. Comment 1 above also addresses the commenter's concerns regarding the different approach used for onsite versus offsite landfills.

In response to the commenter's concern regarding the unit area used for the co-disposal analysis, EPA notes that a deterministic analysis forces the use of a single landfill area. EPA believes its approach was appropriate because it could no longer use waste-specific landfill areas in assessing the disposal of multiple wastes in a single landfill. Because so many different waste combinations were being disposed, the complexity of assessing the plausibility of each waste combination with each landfill would make any co-disposal analysis difficult or impossible. EPA also notes that it considered all landfills in its co-disposal Monte Carlo analysis, as described in NODA Supplemental Background Document for Groundwater Risk Assessment.

In response to the commenter's concern regarding the small median value for offsite landfills, EPA acknowledges that the cited area is in error. The median and high-end waste unit areas used in the two high-end parameter analysis were derived from the industrial subtitle D landfill survey database as stated in the background document; however, the median landfill area was incorrectly interpreted to be 2,020 square meters, when in fact it should have been 20,200 square meters. The high-end landfill area of 162,000 was correctly represented in the analysis. The Agency has conducted revised two high-end parameter modeling analyses with a median landfill area of 20,200 meters. The Agency has also expanded the analysis to also include municipal landfill area distributions. The revised and expanded analyses are presented in the new 1998 background document, *Additional Groundwater Pathway Analysis*.

EPA agrees that the median area used by EPA in this analysis was in error, and believes that the data for off-site municipal landfill area appear more relevant to the modeling of off-site landfills than the industrial data base used by the Agency. Therefore, EPA revised the groundwater modeling for off-site landfills to reflect the larger areas associated with municipal landfills, and the risk results incorporate the revised landfill areas. EPA notes that the use of municipal landfill data is entirely appropriate because the §3007 Questionnaire results showed that at least one-third of all landfills used for refinery wastes are municipal waste landfills ("Offsite Subtitle D Landfills Used for Disposal of Petroleum Refining Wastes," in "Additional Listing Support Analyses," 1998).

I.A.6.d.   Biodegradation

Comment 1: EPA Should Incorporate Biodegradation Rates Into the EPA Composite Model with Transformation Products (EPACMTP) Groundwater Model

The EPACMTP model, used to predict the fate and transport of constituents in the saturated zone, is designed to include first-order biodegradation rate constants in its derivation of dilution and attenuation factors (DAFs). For purposes of this listing determination, however, EPA has chosen to set biodegradation rates for all constituents equal to zero (no biodegradation) even though biodegradation of many of the constituents of concern is known to occur.

Although API is pleased that OSW has reviewed the Krumholz et al. (1996) paper, we are concerned that its interpretation of that summary of anaerobic BTEX biodegradation tends to focus only on those statements which lend support the Agency position on not including a biodegradation rate constant for benzene in this modeling. For example, Krumholz et al. observe that while there are numerous studies that do not show evidence of anaerobic biodegradation of benzene, they also observe that those results may be biased because they 1) have not been carried out for a very long period of time (e.g., only a matter of a few months), and 2) that most have been performed on mixtures of BTEX, and that there is evidence that the TEX compounds may be preferentially biodegraded (i.e., benzene biodegradation might only occur once the TEX was depleted). They also conclude that " the biodegradation of BTEX hydrocarbons has been clearly demonstrated to occur under a variety of anaerobic conditions." (Section 3.7, p. 91). Further, regarding the field studies summarized in the Krumholz article, it is important to note that 3 of the 10 field sites (not 2 of 15 as noted in the NODA) have documented field anaerobic biodegradation of benzene. Also, and importantly, a closer review of the individual papers for those sites (and other papers describing research at those sites) shows that at least 8 of those sites have shown aerobic biodegradation of benzene.

We agree with the Agency that the geochemical and biological processes responsible for mineralization of benzene under these conditions are indeed complex. It is a difficult task to isolate and characterize the response of indigenous microorganisms to the presence of benzene under either anaerobic or aerobic conditions. However, from a practical perspective, it may be most valuable to consider the results of two recent synoptic studies of the characteristics and behavior of hundreds of BTEX plumes from gasoline release sites (Rice et al., 1995; Mace et al., 1997). Collectively these studies include data from over 800 field sites, and they confirm the contribution of biodegradation processes (either anaerobic or aerobic or both) in limiting the mass transport of dissolved benzene in virtually all cases. In California, 90% of the benzene plumes studied decreased to less than 5 ppb at a distance of less than 260 feet from the source. In Texas, 90% of the benzene plumes decreased to less than 10 ppb in under 380 feet.

In conjunction with the Hazardous Waste Identification Rule (HWIR) for Process Wastes, API is currently working closely with the Agency to evaluate potential biodegradation rates which can be input in the EPACMTP. EPA will assemble a Peer Review Panel of outside biodegradation experts (scheduled to meet August 25-26, 1997) to evaluate criteria for including appropriate biodegradation rate constants for selected organic constituents. EPA is also assembling a team of in-house technical experts which will make a final determination on inclusion of any biodegradation factors in the EPACMTP model used for HWIR.

API, in conjunction with other members of the regulated community, will submit data shortly regarding anaerobic biodegradation rate constants of numerous constituents for consideration in the HWIR rulemaking. Given the recent delays in promulgation of the petroleum residual listings rule (currently scheduled for May 29, 1998), EPA should have ample time to reach a conclusion on incorporation of anaerobic and aerobic biodegradation rates in the HWIR context, and then incorporate those rates in the risk analyses for this rulemaking in accordance with the listing

criterion in 30 CFR § 261.11(a)(3)(v). API will continue to provide further documentation of this phenomenon to EPA during its development of this and related rules affecting the petroleum industry. (API, 0009, pg 11)

Response: EPA conducted an evaluation of all submitted data and the documented anaerobic biodegradation studies of benzene suggest that in-situ anaerobic biodegradation of benzene rates may be strongly dependent on site-specific conditions *(e.g.* availability of electron acceptors, availability of nutrients, temperature, etc.). Furthermore, the necessary conditions for anaerobic benzene biodegradation are poorly understood. The absence of biodegradation can be caused by the presence of competing substrates, such as, toluene, xylenes, and ethylbenzene, as well as inadequate geochemical conditions and lack of proper electron acceptors (nitrate, sulfate, iron, etc.). Therefore, because of the lack of information to correlate site-specific controlling factors to biodegradation, the limited number of field data, and the field and laboratory evidence that benzene tends to be recalcitrant to anaerobic biodegradation, biodegradation of benzene was not considered in the 1995 analysis or in the current groundwater pathway analysis[93]. Also, as EPA noted in the proposed rule, preliminary modeling suggests that incorporation of possible biodegradation rates into the analysis would result in relatively minor decreases in risk (i.e., reductions of 2 to 44%, depending on the waste; see proposed rule at 60 FR 57761).

The Agency is currently examining the protocol for the measurement of anaerobic biodegradation rates for chemicals in subsurface environments, and has organized a workshop as a forum for distinguished researchers in the area of anaerobic biodegradation to discuss and debate various issues related to anaerobic biodegradation, with a view to evaluate and improve the protocol, and to develop criteria for the evaluation of anaerobic biodegradation rate data for organic chemicals in the subsurface environment.

Furthermore, biodegradation of benzene was included in the 1995 groundwater pathway analysis, using lag times of 1, 5, and 10 years and considering biodegradation rates of $1.E-5$ and $1E-4$ day^{-1} [94]. Results of that analysis indicated that receptor well concentrations might be reduced to between 98% to 57% of their value with no biodegradation. A reduction to 57% is not sufficient to reduce Crude Oil Tank Sediment, Hydrotreating Catalyst, or Hydrorefining Catalyst benzene risk in the current analysis to below $1E-5$.[95]

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[93]U.S. EPA, 1997. Supplemental Background Document, Groundwater Pathway Risk Analysis.

[94] U.S.EPA, 1995. Background Document for Groundwater Pathway Analysis, Petroleum Refining Waste Listing Determination,   Office of Solid Waste, Washington, DC, 20460.

[95]U.S. EPA. Additional Groundwater Pathway Risk Analysis, Supplemental Background Document, 1998.

Comment 2: EPA Should Incorporate Biodegradation Rates Into the EPACMTP Groundwater Model

The EPACMTP model, used to predict the fate and transport of constituents in the saturated zone, is designed to include first-order biodegradation rate constants in its derivation of dilution and attenuation factors (DAFs). For purposes of this listing determination, however, EPA has chosen to set biodegradation rates for all constituents equal to zero (no biodegradation).

This issue is not unique to this rulemaking. In conjunction with the Hazardous Waste Identification Rule (HWIR) for Process Wastes, API is currently working closely with the Agency to evaluate potential biodegradation rates which can be input in the EPACMTP. EPA will assemble a Peer Review Panel of outside biodegradation experts (scheduled to meet August 25-26, 1997) to evaluate criteria for including appropriate biodegradation rate constants for selected organic constituents. EPA is also assembling a team of in-house technical experts which will make a final determination on inclusion of any biodegradation factors in the EPACMTP model used for HWIR. EPA should have ample time to reach a conclusion on incorporation of anaerobic and aerobic biodegradation rates in the HWIR context, and then incorporate those rates in the risk analyses for this rulemaking. This is especially true since any rates incorporated into the EPACMTP for HWIR purposes will have been thoroughly studied, and be very conservative, with little or no additional staff time needed for technical reevaluation of the selected biodegradation rates in the risk analysis for this rule. (Phillips, 00014)

Response: See Response to Comment 1 above.

I.A.6.e.   Plume Centerline

Comment 1: In the supplemental groundwater pathway risk analysis, EPA has continued to vary the location of receptor wells across the width of the contaminant plume in both the deterministic and Monte Carlo modeling runs, rather than fixing the receptor well on the plume centerline as is consistent with Agency policy and precedent. See 59 FR 9830 (March 1, 1994), 59 Fed. Reg. 66,086 (Dec. 22, 1994). By varying the location of the receptor well, the risk analysis assumes that wells will often receive significantly lower concentrations of contaminants at a given distance from the source area, and this results in underestimating the risks posed by land disposal mismanagement.

As discussed more fully in the KGS report, the EPACMTP model assumes that the groundwater contaminant plume has a bell-shaped concentration distribution, with the highest concentration occurring along the plume centerline. This distribution assumes that the waste is uniformly distributed and the geology is homogeneous which, of course, is not a realistic reflection of real-world waste disposal and geologic conditions. As EPA knows, many geological settings are heterogeneous and contain preferential pathways for contaminant migration such as sand channels or fractured zones. EPA's efforts to compensate for this model limitation actually contribute to a greater underestimate of risks. KGS Report at 3. Even locating the receptor well on the plume

centerline, thereby capturing the highest concentrations produced by the model, does not necessarily reflect the highest concentrations the modeled leachate could generate if released in a heterogeneous geologic setting.

For these reasons, EPA has previously located the receptor well on the plume centerline as a reasonably conservative assumption when evaluating groundwater risks for listing determinations. KGS evaluated the sensitivity of this parameter and conducted its analysis fixing the receptor well on the plume centerline. Based on KGS's evaluation of this factor and other parameters, the risks posed by potential mismanagement of most petroleum refinery wastes covered by the NODA require a hazardous waste listing, as discussed in Section II below. (ETC, 00005, pg 4)

Response: There are two main technical issues in the above comment: locating receptor well along the plume centerline; and effects due to local heterogeneity in conjunction with the placement of receptor well.

The first concern is addressed below in response to comment 2. In essence, the main objective of the probabilistic placement of receptor well is to mimic the real-world distribution of well locations as realistically as possible.

The issue of heterogeneity is addressed below.

The Agency currently uses homogeneous flow and transport models to simulate contaminant migration in the vadose and saturated zones. In these models, average or 'effective' properties are utilized. By using the effective properties, the plume geometry is symmetric about the centerline with the maximum concentration occurring along the centerline. However, it also represents the expected plume geometry or the geometry with the maximum probability of occurrence. With local heterogeneity, the plume geometry may no longer be symmetric and the maximum concentrations do not necessarily occur along the plume centerline.

However, with the receptor well being probabilistically located, the probabilistic distribution of relative distance between the plume centroid (where the maximum concentration occurs) and the well location remains relatively unchanged. From the Monte-Carlo simulation standpoint, with adequate number of Monte-Carlo realizations, the distribution of contaminant concentrations at receptor well should not be affected by local heterogeneity.

Some of the heterogeneity effects have been incorporated into the model, e.g, macro-hydrodynamic dispersion due to hydraulic conductivity contrast. For the macro-dispersion phenomenon which reflects the dependency of hydrodynamic dispersion on spatial scales, the Gelhar's scale-dependent relationship for the hydrodynamic dispersivity is employed in the model.

It must be pointed out here also that local heterogeneity could affect the distribution of contaminant concentration in both positive (concentration increases) and negative (concentration decreases) ways. For instance, the presence of hydraulically conductive fractures could cause

contaminant to travel faster due to an increase in groundwater velocity, along certain preferential pathways. However, there are several types of fractures, and not all fractures are hydraulically conductive. Old faults tend to be clay-filled, and some old fractures with history of hydrothermal processes could have been plugged by mineralization. These faults and fractures tend to function as flow barriers. Furthermore, retardation due to matrix diffusion into background rock matrix is also possible. Transport through fractured rocks could be retarded by contaminant absorption via matrix diffusion. Because of the narrowness of most preferential pathways, the probability of these pathways being intercepted by receptor wells also diminishes.

Because the local heterogeneity could affect the distribution of contaminant concentration in both the positive and negative manners, with an adequate size of sample population, the two opposite effects are likely to negate each other so that the final concentration distributions based on Monte-Carlo analyses are not significantly influenced by the presence of local heterogeneity.

In addition, a certain amount of conservativeness has been incorporated into the analyses. For example: the assumption that the saturated thickness remains constant (which causes the groundwater to be faster, the peak concentration to arrive at the receptor well more quickly, and contaminant concentration to be greater due to less dilution; and the exclusion of biodegradation. In the case of high-end parameter analyses, the contaminant concentrations are in most cases between the 95th and 99th percentiles of the corresponding distributions based on Monte-Carlo analyses.

Comment 2: Notwithstanding both precedent and expressed Agency policy to locate the receptor well in the plume centerline in listing determination risk assessments,[96] EPA continues to vary the location of the receptor well in the NODA deterministic and Monte Carlo modeling runs. Like the 1995 proposal, no explanation or justification is provided regarding this violation of Agency policy, and the approach is inconsistent with RCRA's underlying goals of preventing pollution in the first instance, and protecting human health <u>and</u> the environment. <u>Whether or not a receptor well happens to be located in the plume centerline at a particular point in time is irrelevant to the presence of the plume itself and thus the protection of the resource for present and future uses.</u> By juggling well locations, EPA is forsaking its mission of groundwater protection by playing geographic roulette with the health of the resource and its potential users.

In addition to the legal and policy bases for locating the receptor well in the plume centerline, there are also technical reasons as well. As noted in the KGS report, the large Dispersivity values used in EPA's groundwater model do not reflect reality. Further, as the model user's guide notes, the scientific literature does not provide a strong basis for choosing Dispersivity values when modeling generic scenarios. Therefore, locating the receptor well in the plume centerline reduces the overall level of technical uncertainty in the model predictions. (EDF, 00006, pg 40)

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[96] 59 FR 9830 (March 1, 1994); 59 FR 66086 (December 22, 1994).

Response: The Agency disagrees with the commenter on this issue. While EPA may have placed the receptor well on the plume centerline in modeling analyses in some past rulemakings, the Agency's risk assessment methodology has evolved. In the two high-end parameter analysis for the Petroleum Refining Waste Listing Determination placement of the well on the centerline was chosen to be a high-end parameter, and for the median value for this input parameter, the well was placed one-half the distance from the centerline to the edge of the plume.

EPA believes that well placement should be evaluated in the sensitivity analysis, rather than placing the well on the plume centerline in all cases, because this would result in arbitrarily choosing this as one of the high-end parameter assumption.

I.A.6.f.   Dispersivity

Comment 1: EPA's Dispersivity Value in the Groundwater Model Is Inaccurately Calculated

As reported in both the 1995 and 1997 Background Documents for the Groundwater Pathway, EPA has used the same values of Dispersivity (longitudinal, transverse, and vertical) for all simulations. EPA's calculation is based on the longitudinal, transverse, and vertical dispersivity values of 6.44 m, 0.805 m, and 0.04 m, respectively, which represent approximately median values for a receptor well 102 m beyond of the down gradient edge of a 2020 m^2 landfill. These values are inappropriate for the other combinations of receptor-well distance and source area used in the high-end analysis. Instead, the values of dispersivity should be a function of the down gradient distance from the center of the landfill. Using scale-dependent dispersivities, API estimates that receptor-well concentrations would be reduced by 40 to 60 percent. (API, 00009, pg 16)

Response: The commenter incorrectly assumed that the saturated zone dispersivity values used in the groundwater modeling analysis represent values for a landfill area of 2020 m^2 and a receptor well distance of 102 meters. That was neither stated in the 1995 Background Document nor in the 1997 Background Document. The dispersivity values used in the model were the median values based on a scale dependent dispersivity distribution given by Gelhar (1992), adjusted for landfill size and well distance. The landfill areas varied from 40.5 m^2 to 3.12 x 10^6 m^2 (the range of 790 Subtitle D Industrial landfills in the U.S.). Well distances varied between 0 and one mile.

Nevertheless, the Agency acknowledges that using the same dispersivity values for different combinations of landfill areas and well distances is not consistent with previous modeling analyses. To respond to this comment, dispersivity values in the current sensitivity analysis (see *Additional Groundwater Pathway Analysis, 1998*) were properly adjusted for landfill area and well distance. However, EPA does not believe this had much impact on the modeling results.

I.A.6.g.   Existing Groundwater Contamination

Comment 1: Failure to Consider Impacts of Existing Groundwater Contamination

June 29, 1998   I-71

As acknowledged by EPA,[97] and as further documented in this portion of the comments, the subsurface under many refineries is grossly contaminated. This existing contamination affects the instant rulemaking in two important ways. First, both the EPA and KGS modeling assumes there is no existing groundwater contamination contributing to the risks at the receptor wells. This assumption is completely contrary to available information on refinery sites.

Under a plausible mismanagement scenario, particularly where EPA is basing its onsite facility groundwater modeling on hydrogeologic conditions at refinery sites,[98] EPA is obligated to consider all the important and relevant subsurface conditions at these sites that may affect the risks posed by improper waste management. Such conditions include existing groundwater quality, especially where the Agency can make generally applicable observations based upon the extensive contamination throughout the industry. Therefore, since existing subsurface contamination can contribute significantly to groundwater risks at the modeled receptor well near refinery sites, EPA must include the cumulative risks in its assessments and base its listing determinations on such cumulative risks.[99] This identification and consideration of cumulative risks is not only necessary to satisfy RCRA's protection of human health mandate, but is otherwise required by Section 3-301(b) of Executive Order 12898.

Second, the existing refinery groundwater contamination is often in the form of NAPL, colloids, or other conditions that can greatly facilitate the transport of contaminants at refinery sites, including but not limited to the PAHs in refinery wastes. Organic compounds exiting a waste management unit can become dissolved into the more mobile NAPL phase in the subsurface, and subsequently migrate with the NAPL body.[100] While the rate and extent of NAPL migration can

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[97] "There are numerous instances of groundwater contamination at petroleum refineries and in some cases the contamination is of such magnitude that oil is recovered from the aquifer. Environmental damage at refineries is extensive and is generally attributable to the following contaminant sources (or combination thereof: crude oil spills, refined product spills, and waste management....This provides additional evidence of the mobility of oily materials and oily wastes." See 55 FR 46369 (November 2, 1990).

[98] See NODA Groundwater Risk Assessment, pp. 2-5, 2-6, 2-11.

[99] Indeed, the determination of whether a waste is capable of posing a substantial present or potential hazard when improperly managed, based upon factors such as hazardous constituent toxicity and concentration, requires consideration of whether the waste-related exposures will occur in conjunction with other plausible toxic contaminant exposures within the relevant pathways. See 40 CFR 261.11(a)(3)(I) and (ii).

[100] As a result, even assuming arguendo the free-phased flow of contaminants will not occur as a result of cosolvency in the landfill, the contaminants can nevertheless become part of an existing NAPL plume upon exiting the unit. The NODA materials do not address this NAPL migration scenario.

depend upon site-specific circumstances, NAPL migration often results in lower dilution and attenuation, and therefore higher receptor well concentrations, than the typical dissolved-phased flow of contaminants. The Agency recognized this phenomenon in the previous refinery waste listing by assessing the mobility of benzo(a)pyrene (one of the toxic PAHs of concern in the instant rulemaking too) as "high" when released into substantially contaminated subsurface environments, but "low" under other circumstances.[101]

However, EPA's groundwater model in the instant rulemaking assumes the contaminants are mobile in the dissolved phase only. Therefore, the model predicts contaminants are subject to substantial dilution and attenuation in the subsurface through the sorption of organic compounds. This sorption is a function of a chemical specific soil-water partitioning values indicative of conditions free of cosolvents, surfactants, and NAPLs. Accordingly, EPA's modeling substantially understates the concentrations of contaminants that can reach receptor wells under the groundwater conditions that prevail at many refinery sites.

This failure to consider the free-phased flow of contaminants due to existing NAPL and other contamination at refinery sites is inconsistent with EPA's criteria for listing hazardous waste. Under these criteria, the Agency must consider the potential of hazardous constituents to migrate under the plausible types of mismanagement to which the waste could be subjected.[102]

EPA regards onsite landfilling and land treatment as plausible mismanagement scenarios. Since the subsurface under many refineries is contaminated with NAPLs and other contaminants that can facilitate transport, and the onsite units will not prevent the hazardous constituents from reaching that subsurface contamination, EPA must take into account the hydrogeologic conditions known to the Agency that can facilitate transport of hazardous constituents.

While there is no formal nationwide survey of NAPL occurrence at refinery sites, there is ample evidence that NAPL contamination is frequent and severe. For example, in October and November 1990, API conducted a survey to determine the extent of free hydrocarbon recovery in the refining, marketing, and transportation sectors of the petroleum industry. Among the 42 refinery survey respondents, there were 64 facilities with active hydrocarbon recovery programs, indicating that sufficient free-phase hydrocarbon existed in the subsurface at these sites to warrant recovery by trenching or extraction.[103]

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[101] 55 FR 46368 (November 2, 1990).

[102] See 40 CFR 261.11(a)(3)(iii), (vii).

[103] Letter to EPA RCRA Docket Officer from Terry F. Yosie regarding Hazardous Waste Management System: Identification and Listing of Hazardous Waste Toxicity Characteristic (Docket No. F-90-PRAS-FFFFF); Appendix C: Survey of Free Hydrocarbon Recovery Cleanups, December 21, 1990.

The Los Angeles area provides another example of the magnitude of LNAPL occurrence at refineries. Southern California has numerous refineries, representing over 80 years of operations. These refineries were constructed in the 1920s after oil was discovered on the Los Angeles Coastal Plain. The location was considered convenient because it was both near shipping corridors and near major oil production fields. When the refineries were first constructed, they were located in a Spanish Land Grant District that was largely agricultural. Major population centers have now encroached on these locations, and refinery releases in the region pose a high risk of exposure.

In early 1985, oil droplets were evident on sand near a high-priced beach just west of one refinery. The product had leaked from the refinery and moved laterally toward the ocean, following the regional groundwater gradient. About 6,000,000 barrels of product were estimated to exist beneath the refinery.

Following this discovery, the California Department of Health Services (DOHS) designated 17 refineries in the area as health hazards. This designation reflected the potential and actual subsurface occurrence of leaked hydrocarbon product derived from these facilities.[104]

As part of California Regional Water Quality Control Board Order No 85-17, these refineries were ordered to delineate LNAPL pools, including chemistry, areal extent, and total volume. All 17 refineries were subsequently found to have NAPL pools in the subsurface.[105] Some of these pools are extremely large, as indicated in the tables below.

A review of literature sources, materials from other RCRA dockets, and RCRA Facility Assessments (RFAs) provided additional information on NAPL contamination at seven refineries. This information is briefly summarized in the following tables and the subsequent three paragraphs.

Extent of LNAPL Contamination at Selected Petroleum Refineries

Refinery Name	Estimated Areal Extent of LNAPL (acres)	Minimum Estimated Quantity* of LNAPL in the Subsurface (barrels)	Minimum # of LNAPL Pools	Maximum Length of an LNAPL Pool (feet)
Chevron El Segundo	1,000	6,000,000	1	2,400
Rancho San Pedro	265	601,000	16	4,000
Amoco Mandan	120	NA	11	2,100
Navajo Artesia	NA	>110,000	2	3,000

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[104] Testa, S.M. and D.L. Winegardner, 1991. Restoration of Petroleum Contaminated Aquifers. Lewis Publishers, Chelsea, MI.

[105] Telephone conversation with David Hung, Los Angeles Regional Water Quality Control Board, July 10, 1997.

Conoco Ponca City	52	NA	1	1,500
Sinclair Oil Tulsa	NA	> 130,000	1	na
Chevron El Paso	1,192	2,922,930	2	3,700

*where applicable, includes both LNAPL removed from subsurface and estimate of volume remaining in subsurface.

Nature of LNAPL Contamination Beneath Solid Waste Management Units

Refinery Name	Minimum # SWMUs Under- lain by LNAPL*	Min. Depth to LNAPL beneath a SWMU (ft)	Max. Apparent Thick-ness of LNAPL beneath a SWMU (ft)	Types of LNAPL Beneath SWMUs
Chevron El Segundo	All	20	7	gasoline, kerosene diesel
Ranch San Pedro	21	NA	NA	naptha, kerosene, light gasoline
Amoco Mandan	12	5	10	naphtha, gasoline, diesel
Navajo Artesia	1	10	up to several feet	gasoline, diesel
Conoco Ponca City	2	NA	15	gasoline
Sinclair Oil Tulsa	1	17	4-5	rude oil
Chevron El Paso	14	85	20	gasoline range

* Since wells were not always drilled directly through SWMUs, units with LNAPL encounters nearby and upgradient are included in this category.

Several additional observations from these source materials are worth noting.[106] In several instances, LNAPL has migrated offsite. At the Sinclair Oil refinery, LNAPL contamination was discovered when drinking water wells for nearby residences were found to contain gasoline, thereby dramatically illustrating the potential for NAPL plumes to reach receptor wells. In all cases for which the extent of LNAPL was mapped, the length of the pools extends further than EPA's modeled median and high-end distances to the receptor wells.

At the AMOCO refinery, the LNAPL plume was moving at the rate of 584 ft/yr, and at the Chevron refinery, the plume was moving at a rate of 182.5 ft/yr. These velocities can be compared to the 36.4 ft/yr groundwater velocity in EPA's deterministic run modeling. Nearly all of the extensive floating hydrocarbon plumes were composed of lighter petroleum fractions (gasoline, kerosene) which are more mobile in the subsurface than the heavier fractions such as crude oil.

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[106] The sources for the tables and these observations are as follows:

Chevron USA, Inc, 1991. Comprehensive Investigation and Feasibility Study Final Report, Chevron El Paso Refinery.

Kearney, A.T., Inc., 1986. Preliminary Assessment Report, Navajo Refining Company, Aretesia, NM.

Kearney, A.T., Inc., and Pope-Reid Associates, Inc. Final RFA Report, Conoco, Inc. Ponca City, OK.

Navajo Artesia Refinery, Groundwater Remediation and Monitoring Report, April 1977 and 1996 Annual Groundwater Report, March 1997.

Remediation Technologies, Inc., 1996. Draft Phase II RCRA Facility Investigation Report for the Amoco Mandan Refinery.

Remediation Technologies, Inc., 1993. Current Conditions Report for the Amoco Mandan Refinery.

Testa, S.M., and D.L. Winegardner, 1991. Restoration of Petroleum-Contaminated Aquifers. Lewis Publishers, Inc. Chelsea, MI.

Texas Water Commission, 1987. RCRA Facility Assessment for Chevron USA, Inc., El Paso.

USEPA, 1992. Industry Study, 1992-1996 Petroleum Refining Listing Determinations. Site Selection Strategy. Appendix 2. Population of Refineries. RCRA Docket #PRLP-S0073.B.

Significantly, the largest LNAPL pools are often composed of many separate spills that have coalesced in the subsurface. Accordingly, compounds that are initially part of a small pool with insufficient volume to travel very far may still be transported great distances as pools coalesce.

Finally, the literature also contains studies of abandoned refineries. Since the activity periods of many operating refineries extend back over the same time period as the abandoned facilities, the practices that occurred at abandoned refineries are also likely to have been historical practices at active refineries. The prevalence of LNAPL in groundwater at abandoned refineries can therefore provide additional evidence regarding the plausibility of LNAPL at active refineries.

Spruill investigated the impact of waste areas at an abandoned oil refinery site in the Arkansas River Valley and noted that nonaqueous-phase organic liquids associated with petroleum sludge wastes were distributed heterogeneously throughout the site. An oil sheen was observed in most of the monitoring wells, suggesting that floating oil and gasoline hydrocarbons occur beneath much of the site. Oil-soaked soil was noted during the drilling of all but one monitoring well.[107]

Battermann and Meier-Lohr discuss an abandoned refinery site at which 2,000 metric tons of gasoline-range LNAPL covered an area of about 17 acres.[108]

The 215-acre Old Citgo Refinery, located near Shreveport, Louisiana, is an abandoned oil refinery on the National Priorities List. This refinery has a floating, free-phase hydrocarbon plume with an areal extent that is not yet fully defined. In 1990, 47 families were evacuated from nearby apartments due to health hazards from methane and hydrocarbon fumes originating from the groundwater zone beneath the refinery. Other area residents have reported hydrocarbon seepages in their yards.[109]

At an unnamed 60-acre abandoned refinery in the Midwest, an LNAPL zone 2,400 feet in lateral extent and about 0.5 ft thick was found floating on a water table only 5 feet below ground surface. The floating hydrocarbon was discovered during the late 1970s when adjacent residents began to complain that shallow irrigation wells smelled like gasoline. Despite long-term efforts to

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[107] Spruill, T.B., 1990. Preliminary Evaluation of the Effects of an Abandoned Oil Refinery on Chemical Quality of Water in the Arkansas River Valley, Arkansas City, Kansas, 1985-86. USGS Water Resources Investigations Report 89-4190.

[108] Batterman, G. and M. Meier-Lohr, 1995. Nitrate as an Electron Acceptor in In Situ Abandoned Refinery Site Bioremediation, in Hinchee, R.E., J.A. Kittel, and H.J. Reisinger, eds. Applied Bioremediation of Petroleum Hydrocarbons.

[109] NPL Project Site Summary available on the Internet.

remove this hydrocarbon from the subsurface, the off-site portion of the LNAPL plume is still extensive and mobile.[110]

At an unnamed abandoned refinery in the southern U.S., product seeps to a canal led to a comprehensive groundwater investigation. Numerous distinct pools of different petroleum products, including gasoline, kerosene, and crude oil, were found to be floating on the water table beneath the refinery, suggesting multiple releases. The LNAPL thickness was highly variable and ranged from a light sheen to several feet of product.[111]

In summary, extensive groundwater contamination at many refinery sites is well documented. Where EPA's onsite groundwater modeling is intended to reflect hydrogeologic conditions at refinery sites, EPA cannot ignore hydrogeologic conditions such as existing subsurface contamination that can both contribute significantly to risks at receptor wells and facilitate the transport of toxic contaminants released from waste management units or practices. Even assuming arguendo refinery waste co-disposal management practices will not result in the free-phased flow of contaminants from the disposal unit, EPA is obliged to consider the plausible scenario of subsurface free-phased flow arising from waste constituents reaching existing NAPL pools and other migratory enhancing contamination conditions in the vadose zone or the water table. (EDF, 00006, pg 19)

Response: (See also Response to Comments 7.a (no free oil in waste) and 8.f (incremental risks assessed) in Section I.C.1.)

The Agency believes that its groundwater pathway analysis is adequate and that the release of constituents via NAPLs is not a concern for two reasons:

(1)   A modeling analysis conducted in support of the 1995 Proposal (see Background Document for Groundwater Pathway Analysis, USEPA, 1995) showed that crude oil tank sediment was not likely to exit the landfill, and
(2)   An analysis of crude oil tank sediment samples conducted by EPA showed that even without deoiling, no free liquids were reported by the samplers or the laboratory (see response to comment 7a in Section I.C.1.)

EPA agrees that there are no doubt petroleum refineries at which significant LNAPL contamination from product spills exist, however the Agency does not believe this would necessarily have a significant impact on its listing decisions for several reasons. EPA cannot conclude that LNAPLs would be present at the precise sites where these wastes are likely to be disposed and potentially release constituents. As the commenter also noted, the rate and extent of

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[110] Testa, S.M. and D.L. Winegardner, 1991. Restoration of Petroleum Contaminated Aquifers. Lewis Publishers, Chelsea, MI.

[111] Id.

NAPL migration can depend upon site-specific circumstances. The proper consideration of existing contamination would call for the full analysis of many other site-specific factors as well, some of which may tend to reduce constituent release from landfills, subsurface transport, and human exposure. Such factors would include the possible lack of potable groundwater near the site. Further, if LNAPL or other contamination exists, there may well be on-going remediation, perhaps involving groundwater interception or pumping that would significantly limit groundwater flow. The Agency believes that a more site-specific assessment would be more appropriately carried out by State or Federal programs related to remediation of sites, and that such an approach would be quite difficult to follow in pursuit of an industry-wide listing determination.

EPA also notes that it is not likely that aquifers so widely contaminated so as to have floating hydrocarbons would be a continuing source of drinking water. Such contamination should be easily detected and avoided, and would be unlikely to lead to the multiple-year transport and exposure scenario that is the basis for EPA's risk assessments. Furthermore, the level of benzene in likely sources of LNAPLs, gasoline (1.6 % average, or 16,000 ppm), would dwarf any potential risk that might arise from the leachable levels of benzene in wastes under consideration in this rule, making any concept of cumulative risk difficult to apply in any meaningful way in a listing determination. As noted above, the commenter's approach also presumes a number of additional worst-case assumptions (regarding the presence of critically placed NAPLs) that cannot be considered in a vacuum.

The Agency notes that the practical impact of considering LNAPLs and facilitated transport, even if this could be done, is not likely to be significant for most wastes of concern. EPA has decided to list the wastes with higher oil content (CSO tank sediment and crude oil tank sediment), as well as the spent catalysts. Thus, the wastes for which this comment is most relevant are being listed, leaving unleaded gasoline tank sediment and HF alkylation sludge as the only other wastes that showed any groundwater risk of concern to the commenter. EPA notes that the effective dilution and attenuation factors for benzene resulting from the modeling (DAF; calculated by dividing the TCLP input at the point of release from the landfill by the projected concentration at the receptor well) for both of these wastes were on the order of 2 to 4 (see *Additional Groundwater Pathway Analysis,* 1998). These low DAFs approach the theoretical limit of one, which mean that benzene released from the landfill is estimated to reach the receptor well at concentrations that approach the levels in leachate released. Therefore, it is highly unlikely that EPA's assessment significantly understates groundwater risks for these wastes, and any further considerations in the modeling (such as "facilitated transport" due to existing contamination) are unlikely to significantly alter the modeling results.

I.B.   Supplemental Background Document: Non-groundwater Pathway Risk Analysis: Petroleum Refining Process Waste Listing Determination

Comment 1: The NODA Non-groundwater pathway risk assessment consists of land treatment modeling only. Further, as discussed in this portion of the comments, the modeled wastes and their associated quantities, and the land treatment unit sizes, were so arbitrarily small that the risk assessment does not reflect plausible mismanagement scenarios consistent with industry practices.

The underlying methodology flaw with the NODA LTU Risk Assessment is the assumption that the land treatment practices reported on the EPA surveys for 1992 are the only plausible mismanagement scenarios the Agency should evaluate, because such practices will not vary over time. In most cases, EPA mechanically applies the data reported for 1992 without consideration of whether the data adequately represent the only plausible mismanagement scenarios that warrants modeling.

Consequently, the waste volumes modeled in the NODA LTU Risk Assessment are grossly inappropriate for at least three reasons. First, the median and high-end individual waste volumes modeled for land treatment are typically much smaller than modeled for landfilling, sometimes orders of magnitude smaller, even though there is no legal or technical bar for the wastes to be managed in either fashion during any given year.[112] The fact that refineries relied upon land treatment less in 1992 is not necessarily indicative of future practices.

API's residual reports demonstrate the fallacy of EPA's reasoning in this regard. At least 30 refineries practice land treatment. While some use the practice for only one waste, others rely on land treatment to dispose of 9-12 waste streams. An average of 581,666 MT of refinery wastes were land treated annually from 1991-1993.[113]

API reported 11,612 MT of nonleaded tank bottoms and 4,098 MT of "other oily sludges and organic wastes" were land treated in 1992, the year covered by EPA's Section 3007 survey. However, in 1993, API reported 14,215 MT of nonleaded tank bottoms and 35,837 MT of other oily sludges and organic wastes were land treated in 1993.[114] Significantly, these sharp increases

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[112] The median annual HF alkylation sludge modeled onsite landfill volume is 1,448 MT, versus 6 MT for onsite land treatment. Compare NODA Groundwater Risk Assessment, Table 3.3 with NODA LTU Risk Assessment, Table 2.1. Similarly, the high-end annual modeled onsite landfill volume for off-spec products and fines is 659 MT, versus 34 MT for onsite land treatment. Id. In the case of CSO sludge, EPA's annual high-end waste volume is 3,143 MT, as compared to 2,520 MT for land treatment. Compare NODA Groundwater Risk Assessment, Table C.2 with NODA LTU Risk Assessment, Table 2.1.

[113] API 1991 Residual Report, p. 3-33; API 1992-1993 Residual Report, p. 34.

[114] API 1992-1993 Residual Report, pp. B-14, B-34.

in land treatment volumes from one year to the next were associated <u>with the same number of reporting entities</u> for "other oily "sludges" and only one additional reporting entity for nonleaded tank bottoms.[115] Therefore, on a per facility basis, substantially larger volumes of these wastes were land treated in 1993, as compared to EPA's 1992 survey year.

Ironically, API recognizes the large potential for land treatment volume fluctuation for individual refinery wastes from year to year. Explaining these "fluctuations that occur among the various streams", API stated:

> It appears that fluctuations in streams undergoing land treatment is not a simple trade-off among residual streams within a facility that has an operating land farm, but <u>a more complex and dynamic phenomenon that involves the type of residual, its regulatory status and the facility's resources</u>.[116]

Therefore, given the importance of land treatment as a management practice generally, and the recognized annual land treatment volume fluctuation from year to year, EPA must reasonably project that median and high-end waste generation values can be managed via land treatment or landfilling, and model both practices accordingly.[117] (EDF, 00006, pg 50)

Response: The Agency used data collected for this rulemaking as a basis for waste stream quantity assumptions. There is no reason to assume that 1992 is an aberrant year and not representative of the industry over time. EPA conducted an extensive survey of the industry, characterizing waste generation and management practices in 1992. Based on EPA's assessment of the economic factors affecting the refining industry and practices observed during the Agency's field investigation,[118] 1992 was a typical year for refinery operations (e.g., the National economy was not in extremis, capacity rates were high, plant closings and openings were within normal

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[115] Id., pp. B-15, B-35.

[116] API 1991 Residual Report, p. 3-33.

[117] Vacillations in waste quantities are especially likely in the case of tank sludges because the cyclical nature of tank cleanouts may extend across survey cycles or calendar years. See Management of Residual Materials: 1994, API Publication Number 336, September 1996, p. 24 (hereafter "API 1994 Residual Report"). Indeed, tank bottom generates rates have fluctuated from 107,000-227,000 tons annually from 1987-1994, with periods of both rising and falling generation rates within that period. Id., p. C-1.

[118] As discussed further in the 1995 Listing Background Document, EPA conducted its field investigation at 25 randomly selected refineries. This sample of the petroleum refining industry reflects the variety of U.S. refineries with respect to factors such as throughput, complexity, crude slate, integration, geography.

ranges[119]).   See also Section III.I, Comment 2 of the Proposal Response to Comments Background Document.

For a discussion of the interchangeability of landfills and land treatment units, see the Agency's response to Comment 5 in Section III.I of the Proposal Response to Comments Background Document.

Comment 2: Moreover, <u>EPA's modeling assumes no background or other contaminant exposure to nearby receptors, an unjustifiable assumption given the other waste management occurring at these facilities, the pre-existing contamination at many refineries, and the routine and accidental releases associated with refinery operations</u>. The Agency's failure to consider cumulative exposures to nearby human populations violates RCRA and Executive Order 12898. (EDF, 00006, pg 55)

Response: The Agency does not believe this is appropriate to consider for this listing for several reasons. First, EPA does not have the type of specific information on off-site contamination that would be required, nor did the commenter provide any. Furthermore, without extensive site-specific data, EPA cannot conclude that existing soil contamination would occur at the same off-site locations that might be impacted by releases from landtreatment units containing the wastes under study. The proper consideration of existing contamination would call for the full analysis of many other site-specific factors, some of which may reduce constituent release, transport, bioaccumulation, and exposure. Such factors include specific landtreatment unit design, the direction of any slope from the unit, the existence of downgradient residential receptors, and corrective action requirements that may lead to clean up of any release.

Comment 3: In summary, the NODA LTU Risk Assessment does not evaluate meaningful plausible mismanagement scenarios because the wastes volumes, landfill areas, and hazardous constituents modeled were far too limited to reach any conclusions about potential risks to human health and the environment. The risk assessment should be completely revised to reflect projected mismanagement scenarios that include waste volumes that could be land treated in any given year (including codisposed wastes not covered by the listing determinations or the associated study), standard unit dimensions based upon existing and potential unit sizes, an evaluation of all hazardous constituents that may pose significant risks, and cumulative risks contributed by land treatment and other sources at the facilities. (EDF, 00006, pg 55)

Response: The approach of realistic management scenarios has been adopted in response to the court decision reached in the suit contesting the carbamate listing determination (U.S. Court of Appeals for the District of Columbia Circuit. 1996 Dithiocarbamate Task Force, Petitioner v. Environmental Protection Agency, et al., Respondents; Uniroyal Chemical Company, Inc., and Zeneca, Inc., Intervenors Consolidated with Nos. 9501251, 95-1253, 95-1255. Nov.1).

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[119]DOE/EIA Petroleum Supply Annual 1992, Volume 1. May 1993.

I.B.1. Eliminating Wastes Managed as Hazardous

Comment 1: EPA "recalculated" the waste volumes to exclude wastes managed as hazardous in 1992.[120] This recalculation affects crude oil storage tank sludge most dramatically. The NODA modeled land treatment high-end annual volumes for this waste were 181 MT onsite and 100 MT offsite, as compared to 632 MT in the landfill modeling, and 1,839 MT in the 1995 LTU risk assessment (onsite).[121]

These "recalculations" are apparently based upon the unverified and insupportable assumption that simply because certain wastes were managed in 1992 as hazardous, they will always be managed as hazardous. This assumption is insupportable because EPA is unaware of whether the waste actually exhibited the toxicity characteristic for benzene, or the refinery operator simply chose to manage the waste as hazardous that year for convenience or some other reason.[122] Moreover, assuming arguendo that the waste exhibited the toxicity characteristic that year, EPA has no information that this same sludge will <u>always</u> exhibit the toxicity characteristic for benzene, particularly given the inconsistency of the TCLP on oily wastes and the highly varied deoiling techniques applied to these wastes as discussed above.[123]

In addition, by excluding wastes that happened to be managed as hazardous in 1992, EPA is implicitly relying on the existing toxicity characteristic in lieu of listing the waste, and therefore making the same policy errors as the landfill TC Capping modeling discussed above in Section II.J of the comments. The enforcement difficulties associated with this approach, the implications for air drying wastes, and other factors strongly favor listing these wastes rather than relying upon the existing characteristic to protect human health and the environment. (EDF, 00006, pg 52)

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[120] NODA LTU Risk Assessment at 3.

[121] See NODA Groundwater Risk Assessment, Table C.8; NODA LTU Risk Assessment, Table 2.1. EPA's landfill modeling included only the offsite scenario for this waste.

[122] The fact that a refinery operator simply chooses to manage a waste in a hazardous waste facility is not sufficient justification for a negative listing determination for several reasons. First, lacking a legal bar from doing so, the refinery operator may manage the waste as nonhazardous sometime in the future, and in this rulemaking, EPA has not presented any evidence indicating otherwise. Second, wastes regulated as hazardous would be subject to land disposal restrictions and applicable treatment standards prior to placement in a land treatment unit. Therefore, voluntary land treatment in a hazardous waste unit is not "proper management" where the waste is placed in the unit without prior substantial reductions in the toxicity and/or mobility of the waste.

[123] As noted by API above, land treatment practices can fluctuate annually based upon the "regulatory status" of the waste.

Response: EPA does not agree with this comment. The wastes that were excluded for the revised land treatment modeling were, in fact, reported to be hazardous in the §3007 Questionnaire. While there may well be variability for some wastes in when they exhibit a characteristic, EPA has no reason to believe that 1992 was not a typical year, or that such variation would lead to significant changes in the risk analysis. While excluding these volumes does rely on the characteristic as the commenter noted, this reliance seems justified because these waste did, in fact, exhibit the characteristic and were managed as hazardous. Furthermore, EPA notes that the waste of concern to the commenter, crude oil tank sediment, is being listed as hazardous in any case due to groundwater risks from landfill disposal. For the other wastes modeled in landtreatment units, removal of volumes regulated as hazardous did not alter the risk results significantly, i.e., the median and 90th percentile volumes were only slightly different (see Table 2.1 in the NODA nongroundwater risk assessment background document, *Supplemental Background Document; Nongroundwater Pathway Risk Assessment,* March, 1997).

I.B.2. Model Modifications Regarding Release and Transport of Soil to Off-site Receptors

Comment 1: EPA Failed to Adequately Respond to NPRA's Comment That the Runoff from a Land Treatment Unit Can Not Physically Reach the Home Gardens as Described in the Technical Documents.

In the original comments, NPRA stated that the methodology for computing runoff from landfills containing Clarified Slurry Oil Sediment which would reach "home gardeners" was faulty. NPRA asserted that it was not physically possible for the runoff to reach the home garden and therefore there was no direct or indirect risk to the home gardener.

In the NODA, EPA responded to NPRA's and others comments by revising its overland transport equations. Although EPA made changes to these equations, it did not address the fundamental question how can the runoff reach the home garden given the roads, ditches, buildings, and management controls between the land treatment unit and the home garden?

In response EPA merely:

• Acknowledged that the USLE equation it used "estimates only soil erosion to waterbodies." Then, EPA defined a new setting for the analysis so that it could use the equation. The new setting for the analysis is shown in Figure 3-11 of the document Supplemental Background Document for the Non-groundwater risk Assessment for the Petroleum Waste Listing Interim Notice of Data Availability. It places the receptor (home garden) between the landfill and the waterbody.

• Assumed that "the sediment not reaching the waterbody is considered to be deposited evenly over the area of the subbasin." This assumption means that EPA believes that water runoff acts like air and therefore it is reasonable to assume that a uniform deposition of runoff over the nonwater body subbasin is possible.

NPRA's response to EPA's revision of the methodology is as follows:

- Although EPA made changes to the methodology, it did not address a fundamental principle of the methodology which concerns the flow (i.e. pathways) of runoff from the land treatment units to the residential plot or vegetable garden.

- The assumption that runoff from landfills can be modeled as uniform deposition over the subbasin has no scientific, technical or legal basis. Any runoff (water) from the land treatment area would flow in channels and would not flow uphill over ditches, roads, and other natural barriers to reach the residential plots or vegetable gardens. (NPRA, 00004, pg 4)

Response: Listing determinations estimate risk that might reasonablely be expected to occur at existing or future petroleum refining locations. Fate and transport modeling conducted for this purpose must estimate risk for present conditions and predict risk for future conditions. Soil erosion models that include channeling require extremely detailed site descriptions. Simplifying assumptions are required to estimate soil erosion applicable to many possible scenarios. With the USLE it is possible to estimate the quantity of contaminated soil which leaves the LTU and the quantity of soil expected to be delivered to the nearby water body. Using the equations used in this risk assessment it is assumed that the soil that does not arrive at the waterbody is spread evenly over the area between the LTU and the water body. Until a better model is available this estimation will be used to estimate the risk due to soil erosion. Significantly, due to an error in the air dispersion modeling conducted for the proposal, the percentage of total risk attributable to the air pathway has increased and is now the same order of magnitude as the risk due to soil erosion. Upon reexamination of the air pathway analysis, it was discovered that a unit conversion from $\mu g/m^2$ to g/m^2 was performed in two locations, creating an underestimation of risk due to air deposition of 6 orders of magnitude. The unit conversion correction makes the risk due to air deposition from windblown soil from the LTU the same order of magnitude as the risk attributed to soil erosion in each case; i.e., if the risk due to soil erosion is in the range of 10E-6, the risk due to air deposition is also. Consequently, USLE modeling is less critical for the listing determination. Table I.B.1 compares the contribution to the media concentrations of contaminants from air and erosion as investigated in the uncertainty analysis conducted in support of this listing determination.

Table I.B.1. Risk to Home Gardener from the Onsite Disposal of Clarified Slurry Oil Sludge
Comparison of Risk Due to Air Deposition and Soil Erosion Pathways

Constituent	Total Risk	Risk Due to Air Deposition	Risk Due to Erosion
Benzo(a)pyrene	6.6E-06	3.3E-07	6.3E-06
Dibenz(a,h)anthracene	1.9E-05	1.5E-05	4.5E-06

Methylcholanthrene, 3-	1.9E-05	1.4E-06	1.8E-05
Benz(a)anthracene	2.6E-06	1.8E-07	2.5E-06
Dimethylbenz(a)anthracene, 7,12-	5.3E-05	5.2E-05	7.9E-07
Indeno(1,2,3-cd) pyrene	9.5E-08	2.2E-09	9.3E-08
Benzo(b)fluoranthene	1.0E-06	2.0E-07	8.3E-07
Benzo(k)fluoranthene	6.2E-07	2.4E-08	6.0E-07
Chrysene	3.3E-07	1.8E-08	3.2E-07
Total Risk for Gardener	**1.0E-04**	**6.9E-05**	**3.4E-05**

Comment 2: EPA Failed to Include Management of Refinery Residuals in its Revised Estimates of Population Risks.

EPA did not consider the controls at the land treatment areas described by NPRA in its original comments as examples of management at land treatment areas. EPA specifically requested this information from NPRA. The management of these land treatment units significantly reduces the potential for any runoff to flow from the site. There is no justification for ignoring this relevant information by EPA. (NPRA, 00004, pg 4)

Response: EPA did not ignore this information. However, it is not possible to confirm the effectiveness of controls. Therefore, EPA has assumed that in most cases (central tendency) controls are present and 50 percent effective and in a few cases (95th percentile) no controls are present or are ineffective in preventing erosion.

Comment 3: EPA Has Overestimated Population Risks for Land Treated Residuals

EPA also has overstated the size of the potentially exposed populations for land treated residuals. EPA estimated the population of adult residents and home gardeners based on the assumptions that 38 percent of the population are home gardeners and that the total population lives within 300 meters downhill of a land treatment unit. As discussed earlier, the adult resident and home gardener exposure scenarios are fundamentally flawed and lead to substantial overestimates of individual and population risks. Specific flaws in the model that result in overestimates of risks are described in the following five comments. (API, 00009)

Response: Population risk is attributable to air dispersion and deposition pathways as well as soil erosion. Since, as discussed in comment 1 of this section, the risk from the air pathways has

increased by 6 orders of magnitude, population risk can be assumed to have increased significantly. EPA responds to comments on population risk in Section IV.B.

Comment 4: EPA Assumes that Run-on/Run-off Controls Do Not Adequately Capture Sediments Originating at a Land Treatment Unit (LTU)

LTUs are essentially bioreactors using soil as a support medium for bacterial populations that degrade oily materials. Proper operation of LTUs requires the control of run-on in order to prevent unwanted sediments and water from entering the units. As discussed <u>Supra</u>, data provided by EPA clearly document that run-on/run-off a controls are in place at all LTUs currently receiving refinery residuals (Vierow, 1995). This assumption is not appropriate when calculating population risks. In the case of population risks, any reduction in run-off from adequate controls will result in an overall reduction in the population risk.

The implausibility of the assumption that run-on controls do not adequately capture sediments is also demonstrated by the fact that run-on/run-off controls at LTUs do not have to achieve 100 percent control of sediments in order to reduce risks from the resident or home gardener exposure scenarios. Because run-off from a LTU is likely to be quickly channelized, the controls only have to direct the run-off into a common ditch and produce a channelized flow to prevent the scenario modeled by EPA, as discussed below. As a result, the vast majority of LTUs are not expected to affect off-site residential properties. (API, 00009)

Response: There are no data to document the presence, design parameters, and effectiveness of run-off controls. The Agency has assumed for central tendency that controls are present and reduce run-off by 50 percent and for the high end (95th percentile) case that controls are absent or do not reduce run-off at all. Again, the revised air pathway results make assumptions regarding the adequacy of run-off controls less critical to the listing determination.

Comment 5: EPA Assumes that Run-off from the Land Treatment Unit Does Not Channelize, but Uniformly Flows Across a Residential Plot

As discussed <u>Supra</u>, EPA's assumptions for overland flow are clearly unsupportable. EPA has assumed that run-off moves by overland or sheet flow from the LTU directly to a residential plot or vegetable garden. Overland sheet flow is unlikely to occur because surface water run-off rapidly channelizes into streams that travel by means of swales or gullies and are intercepted by other bodies of water.

Overland or sheet flow is a phenomenon that occurs only over short distances. In the population risk assessment, EPA assumes that overland flow continues for 300 meters. In addition, the analysis implicitly assumes that overland flows crosses multiple properties and reaches all individuals 300 m down gradient of the LTU. In reality, properties will be generally separated by fences or landscaping, which can effectively prevent significant run-off from moving between

properties. At most, therefore, the size of the population exposed should be limited to the one or two properties directly adjacent to the LTU. (API, 00009)

Response: Listing determinations estimate risk that might reasonably be expected to occur at existing or future petroleum refining locations. Because fate and transport modeling conducted for this purpose must estimate risk for present conditions and predict risk for future conditions, sufficiently detailed site descriptions are not available for use in soil erosion models that include channeling. Simplifying assumptions are required to estimate soil erosion applicable to so many possible scenarios. With the USLE it is possible to estimate the quantity of contaminated soil which leaves the LTU and the quantity of soil expected to be delivered to the nearby water body. Using the equations used in this risk assessment it is assumed that the soil that does not arrive at the waterbody is spread evenly over the area between the LTU and the water body. Until a better model is available this estimation will be used to estimate the risk due to soil erosion.

Comment 6: EPA Assumes that Residences Are Directly Down gradient of Land Treatment Units

As discussed above, overland transport of run-off will revert to channelized flows when the overland flow reaches any obstruction or if the ground surface begins to go uphill (creating a small valley or swale). Thus, the existence of a ditch, fence, wooded area, road, building, or swale between the landfarm and a garden eliminates this pathway of exposure. In the assessment, however, EPA assumed that no such obstructions occurred between any of the nearest residential properties and the LTU. Again, this assumption has likely led to an overestimate of the population potentially exposed. (NPRA, 00004)

Response: Due to the scope of the rulemaking, it is not possible for the Agency to make site specific findings regarding the existence of actual obstructions that may reduce or prevent soil erosion to a receptor. In the absence of specific data regarding the use and effectiveness of run-off controls, EPA believes that its default assumptions that controls are 50% effectiveness for central tendency and are not present or ineffective for high-end are reasonably conservative.

Comment 7: EPA Assumes that Run-off Reaching Residential Properties Will Reach Vegetable Gardens

EPA has assumed that overland flows reaching a residence will also reach a vegetable garden (if one exists on the property). However, even if a property receives overland flow directly from a LTU and a vegetable garden exists at the property, EPA should not assume that run-off will always reach the garden. The garden may be located on the opposite side of the property from the LTU, the presence of buildings or paved surfaces may divert the run-off, or the garden itself may have run-on controls to avoid flooding during rain events. For these reasons, a substantial fraction and perhaps a majority of gardens would not be affected by any run-off reaching the residential property. It is not appropriate to ignore this issue when estimating population risks. (API, 00009)

Response: See response to comment 4 regarding run-off control assumptions. EPA responds to comments on population risk in Section IV.B.

Comment 8: EPA Assumes that there Is No Loss of Contaminants During Preparation of Produce (i.e., Washing, Peeling, or Cooking)

EPA's assessment does not consider the loss of PAHs during food preparation. It is well known that uptake of lipophilic compounds by root crops is limited to the surface of the crops (EPA, 1994). Therefore, any peeling or cleaning of root vegetables would reduce or even eliminate contaminant levels in the produce, thereby limiting exposures via this pathway.

It is highly unlikely that EPA's assumptions hold true for any of the land treatment units receiving refinery residuals, since all of them reported some form of run-on/run-off controls, and uniform sediment flow over any substantial distance is improbable, if not impossible. For these reasons the estimates of population risk contained in the proposed rule are likely to be significant overstatements of the actual number of individuals at risk from landfarm releases. In addition, as stated earlier, the overland flow models used by EPA are fundamentally flawed and cannot provide meaningful estimates for either population or individual risks for the adult resident or home gardener scenarios.

In sum, population risk estimates associated with existing management of the residuals indicate that there is no basis for listing CSO sediment or the hydroprocessing catalysts. In contrast to EPA's findings in the proposed rule and NODA, API believes most of the exposure pathways evaluated by EPA are incomplete for the majority, if not all, of the populations evaluated. While EPA's calculations of population risks overstate potential risks, they are still extremely low and clearly support a no-list decision for CSO sediment and spent hydrotreating and hydrorefining catalysts. Not only has EPA greatly overestimated the actual risks, but the risks calculated are as low or lower than previous policy decisions not to regulate. When the risk estimates at issue are properly adjusted to eliminate flaws in EPA's assumptions, the population risk is even lower than EPA has calculated and makes the case for no-listing even more compelling. (API, 00009)

Response: The uncertainty analysis conducted in support of the listing determination have included distributions of consumption rates for exposed fruits and vegetables and root vegetables. The cooking losses estimated for exposed fruits, vegetables, and root vegetables are approximately 20 percent as shown in Table I.B.2. This loss will not affect the listing decision.

Table I.B.2. Percent Loss During Preparation for Exposed Fruits Vegetables and Root Vegetables.

Fruit	% Loss	Vegetable	% Loss	Root	% Loss
Apple	22	Asparagus	22	Beets	28

Pear	22	Cucumber	18	Carrots	19
Peaches	24	Sweet pepper	13	Onions	5
Strawberries	10	Tomato	15	Potatoes	0
Average	**20**	Broccoli	14	**Average**[124]	**17**
		Cabbage	11		
		Lettuce	22		
		Snap beans	18		
		Average	**17**		

See previous discussions of soil erosion and run-off controls. EPA responds to comments on population risk in Section IV.B.

Comment 9: The Model Used To Evaluate Overland Flow Should Not Be Used To Evaluate Runoff from Land Treatment Units

To evaluate overland flow from LTUs, EPA used the Universal Soil Loss Equation (USLE) and a simple model of sediment transport. API has evaluated this model, although there is clear evidence that current run-on/run-off control practices effectively limit overland flow. As implemented by EPA, the model inaccurately evaluates run-off and is likely to greatly overestimate the amount of residual that enters a residential garden. First, the model assumes that surface water run-off from a LTU moves by sheet flow to a residential vegetable garden. Sheet flow of surface water and the associated erosion typically occurs when rain falls on sloping ground that naturally or artificially has been smoothed. Such flow however, is inherently unstable and occurs only over short distances (Brooks et al., 1991). In most cases, surface water rapidly channelizes and even on smooth slopes tends to form gullies if the substrate has minimal vegetation. Channelization of sheet flow will also occur when the sheet flow encounters the first swale or ditch. Once surface water is channelized it does not tend revert back to sheet flow under non-alluvial conditions.

Second, the model assumes that the fraction of the soil delivered to the garden is based on the percentage of the sheet flow that is intercepted by the garden plot. Because the water is likely to be channelized, the percentage of runoff intercepting the garden plot should be extremely small. As currently modeled by EPA however, all runoff is expected to be sheet flow. The interception of runoff between the LTU and the residential garden, therefore, is mostly a function of the size difference between the two areas. Finally, EPA has failed to acknowledge that once the soil from the landfarm enters a stream it potentially becomes mixed and diluted with sediment and runoff

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[124] Zerovalue was not used to calculate the average.

from other areas. Such dilution would greatly reduce the residual concentration reaching the residential garden located within the discharge area. For these reasons, the model used to estimate overland transport of sediment must be viewed as implausible and should be replaced by a more realistic model, in accordance with 40 CFR § 261.11(a)(3)(vii). (API, 00009, pg 17)

Response: EPA agrees that the USLE model is simplistic and a more realistic model would be desirable. However, no other model is currently available. It is impossible to obtain sufficient site specific data for using models that predict runoff using rills and channels and to have that data be applicable to all current and future management units. Although EPA realizes the short comings of the USLE, it is the only model that can be used to predict erosion from a field to a water body in a non-specific setting as required for listing decisions. It is unlikely that this change in the model would change the listing decisions for CSO sludge however, because, the contribution by the air pathway alone is sufficient for listing CSO sludge.

Comment 10: Clarification of the Sediment Delivery Ratio Derivation Is Required

The descriptive information provided in Section 3 of the *Supplemental Non-groundwater Background Document* concerning the derivation of the sediment delivery ratio (SDR) is limited, with Table A-1.4 suggesting that the equation and its variables were empirically derived. The model implies that the value for (SDR) should be between 0 and 1, which is also consistent with the definition of SDR from the Universal Soil Loss Equation:

> "The sediment delivery ratio …is the ratio of sediment delivered at a watershed outlet to the gross upland erosion." *(Corbett, 1995, p. 7.67).*

Recalculation of the SDR using the equation provided in A-1.4 yielded values greater than 1. Review of the input values for this equation showed that units were not specified for sub-basin areas in the table labeled "Values for Empirical Intercept Coefficient, a" (although the footnote suggests that the units are in square miles).

When different sub-basin areas are substituted into the equation for sub-basin areas less than 10 mi^2, the calculated SDR values exceeded 1. Such a finding is not consistent with the definition of this term. The equation should be re-examined since it affects the soil load runoff (e.g., $SL_{0,F}$ and SL_{BF}) which in turn affects the off-site soil concentrations (C_R) used in the risk assessment. (API, 00009, pg 18)

Response: The units for the area of the subbasin are in square meters not in square miles. The equation may indeed fail for areas smaller than 10 square meters, but LTUs will always be larger than this.

Comment 11: Incorrect Calculation of the Unit Soil Loss from the Source ($X_{e,s}$) May Result from the Inappropriate Use of Default Values

In several instances, the parameter values used by EPA to evaluate soil loss are inconsistent between and within equations. First, values used for the USLE rainfall factor (R_S) are not consistent with central tendency or high end distributions. The R_S values used for central tendency and high end (428 and 50 yr^{-1}, respectively) in Table A-1.3 do not appear to represent these descriptors. The value of 428 yr^{-1} is consistent with heavier rainfall areas such as the southeastern US, while the value of 50 yr^{-1} is consistent with central and western central U.S. (Corbett, 1995). Since there is a direct relationship between unit soil loss and the rainfall factor, at a minimum these values appear to be reversed. In addition, some consideration should be made to use a value from 100 to 150 yr^{-1} for central tendency.

Second, the USLE supporting practice factor (P_S) used to compensate for different erosion control measures employed at the source area (Table A-1.3 ($P_S = 1$)) (Corbett, 1995) is extremely conservative and assumes that no erosion control measures (e.g., berming) are employed at the LTU. Berming and other runoff control measures would reduce the amount of solids transport from the LTU since the soil concentration at the receptor (C_R) is proportional to the value of P_S. Typical engineering control practices should yield a Ps value of 0.25 (since run-on and run-off controls are assumed to reduce risks by approximately 75 percent) for most LTUs. (API, 00009, pg 18)

Response: The central tendency location was originally chosen on the basis of air modeling considerations. For this reason Houston was selected as the central tendency location. However, when soil erosion was considered it became a high end location for soil erosion. This is no longer an issue because for the uncertainty/variability analysis conducted in support of the risk assessment each nonhazardous LTU managing each waste stream is modeled using meteorologic data, soil data, and USLE parameter values specific to the geographical location. This analysis has shown that the listing decision is not altered by the use of more site -specific data.

See previous responses for reply to comments on runon/run-off controls.

Comment 12: The Constituent Loss Constants May Be Incorrectly Calculated Due to Selection of Variable Terms

In the selection of constituent loss constants, EPA has incorrectly selected sample specific characteristics for different soil types within the same equation in Table A-1.3. The value used for the USLE soil erodability factor K_S (0.37) was consistent with a silty clay loam containing 0.05% organic matter (Corbett, 1995). In Table A-1.17, the value for K_S (saturated hydraulic conductivity) was consistent with a silty sand, while the value for b (soil-specific exponent representing water retention) was consistent with a loam, and θ_S was consistent with a sandy clay (EPA 1988). None of the recommended values represented the median value for their respective distributions. The soil type should be consistent (e.g., silty clay loam) for all input values. (API, 00009, pg 19)

Response: The uncertainty/variability analysis conducted in support of the non-groundwater risk assessment modeled each nonhazardous LTU using meteorologic data, soil data, and USLE parameter values specific to the geographical location of the unit. This analysis has shown that the listing decision is not altered by the use of more site-specific data. See Table B.1.3 for soil parameter values at these nonhazardous land treatment units.

Table I.B.3. Soil Parameter Values for Geographic Locations of Concern

State	County	Site	Soil Texture	Layer Depth (cm)	Bulk Density[a] (g/cm³)	Sat. Vol. Water Content[b]	Sat Hydraulic Conduct[c]	Soil Specific Exp (b)[d]	Range in Organic Matter %	Fraction Organic Carbon[e]
WA	Sakit	Anacortes	Silt Loam	18	1.02	0.45	0.45	5.3	5.0-10.0	0.035
WA	Whatcom	Blaine	Loam	18	1.00	0.43	1.04	5.39	4.0-10.0	0.040
LA	Iberville	White Castle	Clay	64	1.35	0.38	0.20	0.42	0.9-3.5	0.015
TX	Nueces	Robstown	Clay	66	1.35	0.38	0.20	11.40	0.9-3.5	0.015
NM	San Juan	San Juan	Sandy Loam	10	1.48	0.41	4.42	4.9	1.0	0.01
UT	Salt Lake	Salt Lake City	Loam	20	1.26	0.43	1.04	5.39	2.0-4.0	0.015

[a] MUIR Database

[b] Carsel and Parrish (1988) Table 2

[c] Carsel and Parrish (1988) Table 4

[d] Clapp and Hornberg (19) Table 4

[e] Calculated using equation 6.5 in MULTIMED documentation (p. 92)

Comment 13: EPA Has Not Fully Accounted for the Use of Run-on/Run-off Controls at LTUs

In response to comments, EPA appropriately removed hazardous waste land treatment units receiving CSO sediments from the risk assessment. The risks to home gardeners and adult residents were instead based on potential runoff from land treatment units receiving only nonhazardous wastes. Although the analysis was restricted to nonhazardous land treatment units, EPA has inappropriately assumed that the nonhazardous LTUs do not use run-on/run-off controls. This assumption is in clear contrast to EPA's own data (Vierow, 1995) which reports that all LTUs responding to the RCRA 3007 survey use some type of run-on/run-off control.   As a result, the risk assessment conducted by EPA clearly overestimates potential risks to adult residents and home gardeners. (API, 00009, pg 19)

Response: See previous responses for discussion of soil erosion and air deposition risks.

Comment 14: EPA Has Not Yet Addressed the Numerous Errors in Its Risk Assessment for Home Gardeners

EPA's revised risk assessment does not address a number of comments API made regarding the risk assessment for consumption of vegetables assumed to be impacted by land treatment unit runoff. EPA's revised assessment appropriately included biodegradation of PAHs at the receptor site and corrected the root vegetable consumption rate; however, EPA did not correct its estimates of PAH uptake by above-ground and root vegetables. Among the errors documented by API in the comments on the proposed rule are: overestimated vegetable lipid content used to calculate PAH uptake from air; use of an inappropriate regression to estimate the uptake of PAHs from air; failure to use EPA's own recommended empirical reduction factors for chemicals with low solubility and for root vegetables other than barley; failure to incorporate an adjustment from dry weight concentration to wet weight concentration for above ground produce; and overestimation of the fraction of consumed vegetables that are home-produced. Addressing these errors would result in at least a 100-fold net decrease in risks from vegetable consumption. (API, 00009, pg 19)

Response: The uncertainty/variability analysis conducted in support of this listing decision has addressed many concerns expressed by this commenter. Measured air-to-plant biotransfer factors are used for all constituents where values are available. When measured values are not available, a factor of 40 is included in the denominator of the equation used to estimate the bioaccumulation factor as suggested in the dioxin document (US EPA, *Estimating Exposure to Dioxin-Like Compounds, volume III,* 1994). In addition, a quantitative uncertainty analysis has been conducted using a Monte Carlo simulation to address the variability in consumption rates for fruits and vegetables as presented in the 1996 *Draft Exposure Factors Handbook* (U.S. EPA, *Uncertainty Analysis: Nongroundwater Pathway Risk Assessment; Petroleum Refining Waste Listing Determination*, 1998). An adjustment factor (VG_{AG}) has been incorporated into the equations to address the overestimation for lipophilic compounds ($K_{OW} > 4$). In this analysis, VG_{AG} has been assigned a value of 0.01 for all exposed fruits, exposed vegetables, and root vegetables intended for human consumption or a value of 1 for all forage crops. The VG_{AG} term was included in the original analysis as well.

Comment 15: EPA Did Not Revisit Population Risks in the Revised Assessment

EPA's revised risk assessment for both groundwater and Non-groundwater pathways did not include revised estimates of population risks associated with management of refinery residuals. As stated in previous API comments on the proposed rule and in our present population risk discussion (supra), EPA should consider population risks in its final listing determination for refinery residuals. An elevation in the theoretical individual cancer risk should not be considered grounds for listing a waste as hazardous if the incremental cancer cases avoided are essentially zero (as demonstrated by proper population risk assessments). (API, 00009, pg 20)

Response: The increase in the proportion of risk due to air dispersion and deposition will significantly increase the population risk in the non-groundwater risk analysis. EPA responds to comments on population risk in Section IV.B.

Comment 16: Sun Does not Believe that EPA has Justified Listing of any Residuals.

The minor incremental cancer risks calculated using still very conservative models do not justify a listing determination for the three wastes proposed for listing. In particular, as detailed in NPRA's comments, the EPA's Home Gardener model assumptions are totally unreasonable and are not based on actual land treatment unit design and operational practices. (Sun, 00008)

Response: In the uncertainty/variability analysis conducted in support of the non-groundwater risk analysis each nonhazardous LTU managing the waste streams of concern are modeled using meteorologic data, soil data, and USLE parameter values specific to the geographical location where the unit is located. In addition, variation in constituent concentration, distance to receptor, and the distributions of consumption rates and exposure durations presented in the *1996 Draft Exposure Factors Handbook* are included in the analysis. This analysis has shown that the listing decision is not altered by the use of more site-specific data.

I.B.3. Co-disposal

Comment 1: For the co-disposal land treatment scenarios, the volumes modeled were minuscule compared to industry practice. In the onsite case, three wastes were codisposed with a combined annual total volume of 149 MT. In the offsite case, two scenarios were modeled, and the highest volume for these scenarios covered two wastes with a combined annual total of 75.1 MT.[125]

Incredibly, these waste quantities are lower than the median volumes for HF alkylation sludge (offsite) alone, and lower than the corresponding high-end volumes for crude oil storage tank sludge, HF alkylation sludge, and sulfuric acid alkylation sludge, both onsite and offsite. Accordingly, EPA's modeling presumes these wastes will never be codisposed in a land treatment facility at these quantities or higher.

More importantly, the modeled co-disposal volumes should be compared to the vast majority of the average annual 581,666 MT of land treated wastes reported by API from 1991-1993 that is managed onsite at 30 refineries, where 9-12 different refineries wastes are managed in the same units as discussed above.[126] Clearly, EPA's volume input parameters do not reflect actual co-disposal on either a total volume or number of wastes basis. By modeling only those facilities

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[125] NODA LTU Risk Assessment, pp. 25-27.

[126] API 1992-1993 Residual Report, p. B-36.

reporting co-disposal in EPA's 1992 Section 3007 survey, omitting wastes that happened to be managed in hazardous waste units that year, and omitting refinery wastes other than those covered by this listing determination (and an associated study), <u>EPA's land treatment co-disposal modeling fails to account for the vast quantities of co-disposal actually taking place</u>.

It should be noted that EPA's landfill modeling included a co-disposal scenario for all wastes covered by the instant rulemaking and the associated study. As discussed above, EPA employed median annual landfilling volumes in the co-disposal scenario, <u>irrespective of whether that scenario was observed at a particularly facility in 1992</u>. While there are still flaws in the landfill co-disposal modeling, as discussed in Section II.E of the comments, at least in that effort EPA attempted to project a plausible mismanagement co-disposal scenario that was not so constrained by the 1992 survey data that the result was a completely meaningless exercise. The inconsistency between the landfill and land treatment co-disposal modeling approaches is additional evidence of the inadequacy of the NODA LTU Risk Assessment.

With respect to land treatment unit dimensions, the Agency once again assumes the units that happened to receive certain wastes in 1992 represent the only plausible mismanagement scenario for these wastes. Accordingly, median onsite land treatment unit sizes are different for <u>every</u> waste modeled, with areas ranging from 3 acres for HF alkylation sludge to 15.8 acres for CSO sludge.[127] In the co-disposal scenarios, the onsite unit is 7.5 acres, based upon the size of the one unit modeled.

Since there is no legal or technical bar for an existing land treatment unit to receive any refinery waste, or for a refinery to construct a new land treatment unit to receive any refinery waste, this modeling approach fails to account for both present and potential mismanagement. There is simply no legal or factual basis for EPA to conclude that the only plausible mismanagement scenario for the land treatment of HF alkylation sludge is in a unit 1/5 the size of a unit used for CSO sludge. Under these circumstances, EPA must project a standard onsite land treatment unit to cover the full range of plausible mismanagement at refinery sites. See also Section II.G of these comments for a discussion of the analogous landfill modeling issue.

Indeed, EPA recognizes the absurdity of its position when modeling the offsite co-disposal scenario. In this instance, EPA did <u>not</u> rely upon the size of the 1992 units receiving certain wastes, but instead projected a standard area size based upon all offsite units "because it would be possible for refiners to send wastes to any nearby offsite facility".[128] Unfortunately, EPA did not apply this logic to all the other onsite and offsite land treatment scenarios modeled, even though in all cases future practices are similarly not restricted in accordance with the data reported for one year, particularly when the potential construction of new units is considered.

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[127] NODA LTU Risk Assessment, Table 2.2.

[128] NODA LTU Risk Assessment at 25.

Two other flaws in the NODA LTU Risk Assessment warrant scrutiny. Only PAH constituents were considered in the risk assessment co-disposal scenarios, therefore inhalation risks posed by benzene and other volatile chemicals, or direct and indirect risks posed by arsenic and other metals, were not evaluated.[129] The risks posed by these constituents may be particularly important in a reasonable co-disposal scenario, since the waste types and volumes involved would be much larger than evaluated in the NODA effort. (EDF, 00006, pg 57)

Response: All chemicals in each waste stream and major management practice combination were evaluated initially in the bounding analysis in which all model parameters are set at high end values. The bounding analysis showed PAH compounds to be constituents of concern in land treatment units and all other waste stream constituents and other management practices to be below a level of concern. Only those constituents and management practice combinations showing risk in the bounding analysis were evaluated in the more detailed risk analyses. PAH compounds in land treatment units were identified as the constituents to be evaluated in detail in this analysis.

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[129] NODA LTU Risk Assessment, at 24.

I.C.   **Supplemental Background Document: Listing Support Analyses: Petroleum Refining Process Waste Listing Determination**

I.C.1.   **Analyses Regarding Leaching of Oily Waste**

Comment 1: TCLP Leachate Concentrations are Fundamentally Inaccurate

Comment 1.a: EPA used TCLP results from waste samples as input parameters for leachate concentrations to the EPACMTP groundwater model. However, some of the field sampling procedures that are documented in the NODA reveal that volatile contaminants, such as benzene, were lost prior to analysis. For example, Field Sampling Descriptions for R4-S0-01 (CSO sludge) and R16-US-01 (unleaded gasoline storage tank sludge) reveal that these waste samples were allowed to "air dry" prior to collection. Thus, a basic sampling protocol for wastes containing volatile constituents was violated. As a result, the CSO sludge sample registered the lowest concentration of all volatile organics among the four samples collected by EPA. The unleaded gasoline storage tank sludge sample also contained ludicrously low concentrations of volatiles compared to other samples (e.g., 2.7 ppm benzene compared to 43 ppm and 110 ppm benzene in the other two samples). As EPA should have realized, the concentrations of benzene in the air dried samples used as inputs to the groundwater model were grossly inaccurate. (ETC, 00005)

Response: The commenter states that EPA improperly allowed the samples to be air dried prior to sampling. The commenter has misunderstood the sampling descriptions in the Agency's *Supplemental Background Document* which described the *refineries'* practices of air drying the storage tanks either after draining or water-washing. This procedure was also described in the 1995 Listing Background Document, which stated that occupational benzene levels must be decreased before confined space entry into the tank.[130] This is typically done to allow for safe entry into the tank by refinery employees for sludge removal and tank integrity inspections. Until the air drying process had been completed, no refinery personnel (or EPA sampling teams) could enter the tanks without violating standard occupational safety practices. Current health and safety regulations specify that prior to entry in a confined space (e.g., storage tank), the space must be ventilated (e.g., air dried) and tested for oxygen levels and the combustible gases (e.g., benzene) must be below the lower explosive limit (LEL) and/or chemical of concern (e.g., benzene).[131]

In both cases (i.e., with respect to Samples R4-SO-01 and R16-US-01), as described in the respective *Analytical Data Reports*, refinery personnel shoveled the sludge from the tanks. The unleaded storage tank sludge was placed in drums, from which EPA sampled. The CSO sludge was placed on the refinery's state-permitted "stained soils pad" from which EPA sampled. In no case did EPA's sampling procedure allow for additional "air drying." The tank air drying

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[130]"Listing Background Document for the 1992-1996 Petroleum Refining Listing Determination." EPA. October 31, 1995. Pg. 35.

[131] 29 CFR 1910.146. "Permit Required Confined Space Entry."

conducted by the refineries prior to EPA's arrival onsite for sampling represents standard operating practices designed to comply with basic occupational safety practices. EPA thus concludes that these samples are representative of residuals generated throughout the industry.

Note that EPA was limited by scheduling constraints to collect samples as they were available at the time of EPA's site visits. The sampling program targeted residuals generated from a wide range of process units and storage tanks which generally were operated on independent schedules. The sampling teams attempted to schedule sampling trips when the greatest number of samples would be available. In some cases, facilities had completed preparation of the residuals for disposal and in others the residuals were still being removed from the process units. Where possible, EPA's priority was to sample the residual after removal from the units in question to optimize the value of the sample as a modeling input. This included after any drying or dewatering conducted by the facility, based on the assumption that wastes with high liquid content would not be land disposed.

In some cases, however, the samples were collected directly from the process unit or tank rather than after scheduled dewatering or deoiling. These samples were collected prior to the refineries' residual processing steps either (1) to avoid cross-contamination with other refinery residuals, or (2) when the residual processing step was not scheduled at a time when EPA expected to be able to collect the sample. As an example, in the case of one of the unleaded gasoline tank sludges not discussed by the commenter (R8A-US-01), the facility had not completed residual removal procedures at the time of the sampling event and the sludge was sampled directly from the tank bottom. This refinery reported general use of vacuum trucks to remove tank sludges and transport them to their sludge de-oiling unit prior to recycle to the coker or landfilled. Note that the refinery must have ventilated this tank prior to sampling to allow for tank entry. The refinery did not want the sample collected from the truck because of cross-contamination with other oily residuals. This sample contained higher levels of volatile and semivolatile organics than the other two unleaded gasoline tank sludge samples which possibly could be due to reduced residual handling steps prior to sampling. EPA agrees with the commenter that the concentrations in residuals sampled from the process unit are likely to be higher than those sampled closer to the point of disposal, such as the samples referenced by the commenter. The inclusion of these samples in the data set as inputs to the groundwater model may be somewhat conservative.

However, in response to the commenter's comparison of detected benzene levels among the three unleaded gasoline samples (i.e., 2.7, 43 and 110 ppm), EPA notes that the Agency's entire sampling data set demonstrates concentration ranges of multiple orders of magnitude. Table I.C.1 illustrates the range of detected benzene levels within each of the residual categories. Ranges of at least three orders of magnitude were observed in total benzene concentrations for crude oil tank sludge, unleaded gasoline tank sludge, hydrotreating catalyst, hydrorefining catalyst, reforming catalyst, and spent caustic. EPA concludes that this variability is normal and is attributable to many factors in addition to the place and time of sampling, such as the various residual generation practices employed among refineries, differences in process operating conditions, crude and product slate characteristics, etc. This variability was anticipated at the

outset of this listing determination and was the basis for EPA's stratified random selection of refineries for the field study, as well as the basis for the collection of as many samples as possible (up to six of each listing residual category) given budget, availability and schedule constraints. Thus the Agency is neither surprised nor concerned with the range of benzene levels detected in the unleaded gasoline storage tank samples.

Table I.C.1. Range of Total Detected Benzene Concentrations (ug/kg)		
Residual Category	Concentration	Sample Number
Crude oil tank sediment	220,000	R8C-CS-01
	69,000	R6B-CS-01
	52,000	R10-CS-01
	8,200	R4B-CS-01
	660 J	R19-CS-01
	<2,500	R22-CS-01
Unleaded gasoline storage tank sediment	110,000	R8A-US-01
	43,000	R6B-US-01
	2,700 J	R16-US-01
Clarified slurry oil sediment	1,200 J	R9-SO-01
	<12,500	R4-SO-01
	<12,500	R1B-SO-01
	<2,500	R20-SO-01
Hydrotreating catalyst	500,000	R1-HT-01
	160,000	R18-HT-01
	24,000	R11-HT-01
	9400 J	R8A-HT-01
	2,900	R22-HT-01
	2,000	R3B-HT-01
Hydrorefining catalyst	100,000	R7B-RC-01
	27,000	R21-RC-01
	4,200	R5-RC-01
SCOT® catalyst	60	R11-SC-01

Table I.C.1. Range of Total Detected Benzene Concentrations (ug/kg)		
Residual Category	Concentration	Sample Number
	<625	R7B-SC-01
	<5	R5-SC-01
Reforming catalyst	26,000	R15-CR-01
	2,300	R7B-CR-01
	1,000 J	R5-CR-01
	570	R14-CR-01
	430	R11-CR-01
	<25	R2-RC-01
Hydrofluoric acid alkylation sludge	14,000	R8B-HS-01
	6,100 J	R3-HS-01
	<650	R7C-HS-01
	<625	R9-HS-01
	<313	R15-HS-01
Off-specification product and fines from thermal processing	1,500	R11-TP-01
	<625	R3B-TP-01
	<625	R12-TP-01
	<25	R8A-TP-01
	<23	R14-TP-01
	<5	R6-TP-01
Spent caustic	2,200	R12-LT-01
	420 J	R13-LT-01
	90 J	R3-LT-01
	<5,000	R6-LT-01
	<500	R3-LT-02
	<50	R22B-LT-01
Sludge from H_2S removal and sulfur complex	420 J	R18-ME-01
	96	R5-ME-02
	<2,500	R1-ME-01

Table I.C.1. Range of Total Detected Benzene Concentrations (ug/kg)		
Residual Category	Concentration	Sample Number
	<1,250	R6-ME-01
	<625	R14-ME-01

Reference: 1995 Listing Background Document.

The commenter correctly states that sample R4-SO-01 generally has lower concentrations of volatile constituents, and has more constituents reported as "not detected" in comparison to the other three CSO sediment record samples[132]. However, an important fact is omitted by the commenter that influences the risk analysis. The levels of benzene (the most important volatile compound from a risk standpoint for these refinery wastes) across all samples are uniform in both the total and leachate analyses such that the calculated average and maximum values are equal for the total analyses and within 40 percent for the leachate analyses. Table I.C.2 summarizes the benzene levels in CSO sediment from page 52 of the October 1995 Listing Background Document.

Table I.C.2. Benzene Concentrations in CSO Sediment		
CSO Sediment Record Sample Number	Benzene Concentration, μg/kg	TCLP Benzene Concentration, μg/L
R4-SO-01	<1,250	<50
R9-SO-01	1,200 (estimated)	84 (estimated)
R1B-SO-01	<1,250	<50
R20-SO-01	<2,500	<50
Average of Four Samples	1,200	59
Maximum of Four Samples	1,200	84

EPA concludes that benzene was not "lost" during the management of this particular waste at this particular refinery because benzene levels from other samples (including samples that were not described by the refineries as "air-dried") were present at similar levels, in both total and leachate analyses.

Comment 1.b: As another example, one sample of CSO sludge was mixed with cement kiln dust prior to sampling, thus the sample did not reflect the liquid content of the CSO itself. Two HF alkylation sludges were "dewatered" prior to analysis. See Listing Background Document at 135.

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[132]See page 52 of the October 1995 Listing Background Document.

In addition, many of the waste sample descriptions reveal that the sludge samples were composited. The compositing process can result in substantial loss of volatile compounds, and the background document does not reveal whether careful procedures were followed. (ETC, 00005)

Response: In response, EPA notes that it collected samples of CSO sediment wastes that were available after tank clean out, and in this specific case, it had been mixed with cement kiln dust. This was done by the facility prior to landfilling of the waste. While this treatment may have altered some properties of this sample, the oil content (16% TOG) was relatively low, compared to the other three samples of CSO sediment collected by EPA (see Table 3.1.18 in the *Listing Background Document,* 1995). Even if the Agency discounted entirely the analytical results for the one sample mixed with cement kiln dust, it would not impact the risk assessment significantly, because this would only raise average levels of some critical PAHs slightly (approximately 10-20%). In any case EPA is listing this waste, so inclusion of this sample had no material impact on EPA's final decision.

Dewatering, drying, oil recovery, and mixing with cement kiln dust or diatomaceous earth are all common techniques used by the refinery industry to prepare residuals for disposal. These techniques serve a number of valid purposes, including recovery of valuable hydrocarbons; reduction of combustible gases to their lower explosive limit (LEL) as well as benzene concentrations, so that refinery personnel are not subjected to explosive or dangerous benzene exposure conditions; waste minimization goals; and compliance with state restrictions on liquids in landfills.

EPA researched the frequency that onsite stabilization was reportedly used by refineries according to the RCRA §3007 survey results (and presents the results here for the first time). In the survey, the respondents were required to list all interim management practices, using codes, for each waste. "Onsite stabilization" was one such code (see page 22 of survey, document number F-95-PRLP-S0023). The survey mechanism did not provide a method for refineries to describe their stabilization process. EPA found that 26 refineries reported onsite stabilization (14 percent of the 185 refineries operating in 1992) for treating various wastes[133]. Of these, 17 treated some type of sludge (i.e., crude oil tank sludge, unleaded tank sludge, CSO sludge, residual oil tank sludge, desalting sludge, or alkylation sludge) while six treated CSO sludge in particular. Table I.C.3 summarizes these findings. The survey results do not identify the stabilization media or the proportions of waste-to-media; EPA concludes from these data that the collection of a record sample of kiln dust-stabilized CSO sludge is representative of management methods used by other refineries and therefore represents a logical sample collection and use in the risk assessment.[134]

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[133]These data were compiled from the petroleum refining database, for any waste generated in any year.

[134]Note that Sample R1B-SO-01 is the only stabilized sample in EPA's data set.

Table I.C.3. Use of Onsite Stabilization for Treating Refinery Wastes in Any Year		
Residual Type	# of Refineries	# of Wastes
Crude Oil Tank Sediment	7	7
Unleaded Tank Sludge	3	6
CSO Sludge	6	7
Residual Oil Tank Sludge	6	12
Desalting Sludge	3	5
HF or H_2SO_4 Alkylation Sludge	2	2
Number of Refineries Represented in Above	17	39
Number of Refineries Conducting Stabilization of Any Listing or Study Waste	26	67

With respect to the commenter's concern regarding the two de-watered HF sludge samples, the commenter is referring to Sample R7C-HS-01, which was sampled after the refinery had processed the sludge in a centrifuge, and Sample R3-HS-01, which EPA sampled from the neutralization tank and filtered in the laboratory at the refinery's request to better simulate the characteristics of the waste as it would actually be generated. As discussed in response to the previous comment, EPA's samples represent the range of conditions between the process (e.g., in the neutralization pit) and disposal (e.g., after de-watering). The inclusion of the samples from the process unit in the modeling data set may be conservative. The generation of HF sludge is described further below.

As described in the *1995 Listing Background Document*, HF neutralization sludge is generated when HF acid is neutralized by aqueous KOH and soluble potassium fluoride (KF) is produced. Some facilities employ KOH regeneration. Periodically some of the KF-containing neutralizing solution is withdrawn to the KOH regenerator. In this vessel KF reacts with a lime slurry to produce insoluble calcium fluoride (CaF_2) and thereby regenerates KF to KOH. The regenerated KOH is then returned to the system, and the solid CaF_2 is routed to the neutralizing tank. The KF, at facilities that do not have a regenerator, is sent directly to the neutralizing tank, where it is reacted with lime to form sludge.

Spent caustic, KOH scrubbers, acidic waters from acid sewers and, in some cases, CBM (an azeotropic mixture of the HF acid and water) are charged to neutralization tanks, which neutralize effluent to the WWTP. Neutralizing controls fluoride levels to the WWTP. Neutralizing agents (sodium, calcium, and potassium hydroxide) are selected based on the refineries' WWTP permits. Effluent to the tank is neutralized, generally with lime which forms sludge (calcium fluoride) that collects on the bottom of the pit. This sludge is the sampled residual of concern.

As described above, the HF neutralization sludge sampled for this listing determination is an aqueous-based sludge which may have small amounts of oil from carryover from the reactor or

the neutralized catalyst. This does not represent an "oily" sludge but an aqueous sludge which may contain small amounts of oil. The laboratory personnel described a strong hydrocarbon odor which was the very distinct odor of neutralized HF acid or calcium fluoride. (Note that R9-HS-01 appears to be an exception with 31% oil and grease. See response to Comment 7.a for additional discussion of this sample.)

Field compositing procedures, when necessary, in all cases were performed for the non-volatile analytes only. Compositing consisted of throughly mixing the contents of all non-volatile grab sample aliquots into a homogeneous mixture prior to sample containerization. This procedure was performed for all non-pyrophoric samples regardless of whether the grab samples were collected from a single source such as a storage pad or multiple locations such as roll-off boxes or bins. Conversely, a separate grab sample for volatiles analysis was collected and maintained in as undisturbed a state as possible prior to placing the contents into sample containers without headspace to minimize volatiles loss. For some samples such as the catalysts that were potentially pyrophoric, field compositing procedures were not possible. Instead, these samples were composited using all the non-volatile sample contents in the laboratory after they were screened for pyrophoricity. The volatiles aliquot, as was the case for all residuals sampled, was maintained in a separate sample container and was never composited or combined with other sample aliquots prior to analysis. The laboratory performed all volatiles analyses using only those containers that were designated for this analysis by the field sampling personnel. These sampling and analysis protocols were consistent with the guidance outlined in Chapter Nine of SW-846, and were documented in each site-specific Sampling and Analysis Plan (SAP). Compliance with each SAP was documented in the site-specific Analytical Data Reports (ADRs). Thus, careful procedures were followed in sampling conducted for volatile analyses and loss of volatiles was minimized.

In conclusion, EPA disagrees with the commenter's assertion regarding the representativeness of the samples and the appropriateness of the sample compositing procedures.

Comment 1.c: In addition, in EPA's TCLP analyses the percentage of benzene captured varies widely, even within a given waste stream. In comparing benzene capture efficiencies for different waste streams, it is apparent that the average benzene capture rate for wastes with high oil content is significantly lower than for non-oily wastes. (ETC, 00005)

Response: The commenter's concern is addressed in response to Comments 6 and 7 below.

Comment 1.d: Prior to this rulemaking, EPA consistently acknowledged that the TCLP test substantially underestimates leachate concentrations of toxic constituents for oily wastes. See Response to Comments Background Document for the Listing of Primary and Secondary Oil/Water/Solids Separation Sludges From the Treatment of Petroleum Refinery Wastewaters (Aug. 29, 1990). Oily wastes tend to clog the filter during initial filtration causing inadequate expression of contaminants and otherwise interfere with the TCLP test method. See 57 FR 37294 (Aug. 18, 1992); Evaluation and Modification of Method 1311 for Determining the Release Potential of Difficult-to-Filter Wastes, Vol. 1, RTI, April 1990, page 23. (ETC, 00005)

Response: The RTI report referenced by the commenter, titled "Evaluation and Modification of Method 1311 for Determining the Release Potential of Difficult-to-Filter Wastes" dated April 1990, studied the effectiveness of the TCLP with oily wastes and an alternative filtration procedure designed to simulate the movement of oily wastes into soils beneath landfills. The study evaluated four oily wastes, two of which were obtained from operating petroleum refineries: slop oil emulsion, and API separator sludge. These two residuals are derived from refining wastewater treatment systems, as are F037/F038 wastes. The similarities between F037/F038 sludges and API separator sludge is discussed in detail in the final sludge rulemaking (55 FR 46376, November 2, 1990).

The petroleum wastes studied by RTI were described as multiphasic and extremely heterogeneous and difficult-to-filter. This contrasts with all of the current petroleum listing residuals except for two HF alkylation samples that were identified as single phased without free liquid and considered to be homogeneous wastes (see response to Comment 3.b below for a discussion on the HF samples liquid phase). For this reason the RTI report is not considered directly applicable to the petroleum listing residuals. Note that the RTI report concluded that the TCLP method was adequate for filterable "oily" wastes, such as the samples of API separator sludge tested (see pages 79 and 155 of RTI's report). This issue is explored further in Section III.H of the Proposal RTC.

Comment 2: <u>Improper Consideration of Air Dried Samples</u>

The NODA materials include field sampling descriptions for some of the wastes covered by this rulemaking. In at least two instances, the samples were taken of wastes that were allowed to "air dry" prior to collection.[135] Therefore, the volatile contaminants in these wastes were allowed to escape prior to sample collection, in violation of basic sampling protocols for handling wastes containing volatile chemicals.

The effect of this air drying is clear and profound, upon comparing the analytical results of these samples to the other waste samples. The CSO sludge sample registered the lowest concentration of <u>all</u> volatile organics among the four samples collected by the Agency. The concentrations of total xylenes and naphthalene in this sample are more than an order of magnitude lower than the maximum concentration detected in the other three samples.[136]

The unleaded gasoline storage tank sludge sample was one of three waste samples collected by the Agency. This sludge sample contains 2.7 ppm benzene, as compared to 43 ppm and 110 ppm

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[135] See Field Sampling description of R4-SO-01 (CSO sludge) and R16-US-01 (unleaded gasoline storage tank sludge) in NODA Background Document, Ch. 2, Table 1.

[136] 1995 Listing Background Document, Table 3.1.18.

benzene found in the other two samples.[137] Therefore, for the principal constituent of concern in EPA's groundwater modeling, the volatile contaminant concentration in the air dried sample is 6.2% and 2.4% of the comparable benzene concentration in the two other samples.

Given the substandard field sampling procedure utilized and its resulting analytical impacts, these air dried samples must be discarded.[138] Once the air dried unleaded gasoline tank sludge sample is discarded, the mean benzene waste concentration for this waste rises from 51.9 ppm to 76.5 ppm, significantly altering the modeling results for this waste. Where KGS adjusted only the TCLP value to reflect EPA's purported 53% benzene leaching efficiency (as discussed above) of the revised mean value, the TCLP value increases from 0.75 mg/l to 2.03 mg/l, and the resulting groundwater pathway risk increases by more than half. More importantly, when the modification is considered in conjunction with parameter revisions, the resulting risk estimates are substantially modified, as discussed immediately below. (EDF, 00006, pg 41)

Response: See responses to Comment 1, above. EPA fundamentally disagrees with the commenter that Samples R4-SO-01 and R16-US-01 should be removed from the Agency's modeling input data set and with the commenter's contention that EPA's sampling procedures were "substandard".

Comment 3:   Field and Laboratory Observations of Wastes

Comment 3.a: EPA states that the petroleum refinery wastes are not "oily." In making this statement the Agency's point of reference is used oil which is "essentially all oil." However, a waste does not have to be "essentially all oil" in order to be subject to the OWEP test. A review of the "Scope and Applicability" section of Method 1330A of SW846 (the OWEP) shows that this method is applicable to materials that are not "essentially all oil" including sludges, emulsions, rags, and "other oil wastes derived from petroleum refining." The OWEP is not intended to be applied only to samples that contain "all oil" but to any sample with significant amounts of oil that would preclude use of merely the aqueous-based TCLP test extraction medium. (ETC, 00005)

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[137] 1995 Listing Background Document, Table 3.1.12. The concentrations of other volatile contaminants are also demonstrable lower for the air dried sample. For example, the concentration of toluene is almost an order of magnitude lower than the maximum value detected in the other two samples, and the level of ethyl benzene is about 13% of the maximum value.

[138] It should also be noted that air drying constitutes "treatment", as that term is defined in 40 CFR 260.10. See e.g., EPA Permit Policy Compendium, Section 9432.1987(03), where EPA interpreted the definition of treatment to include evaporation, particularly where hazardous constituents are released into the environment. Accordingly, the air dried samples are not reflective of the risks posed by the wastes as generated. Significantly, in other RCRA contexts, the mere evaporation of hazardous constituents is not considered appropriate management. See 40 CFR 268.4(b).

Response: The commenter has taken EPA's statement out of context. In Section 2.1 of the Supplemental Listing Background Document (March 1997), EPA introduced its description of "Field Laboratory Observations of Wastes" with the following:

> "Commenters suggested that the wastes evaluated in the proposed listing determination are oily in nature and have <u>high levels of "free" oil</u>. EPA contends that the residuals are not oily <u>in the manner anticipated by the commenter</u>." (Emphasis added)

EPA has not disputed that the residuals of concern contain oil, but that they contain <u>free</u> oil which could move in the environment as a separate phase or pose the types of TCLP filtration problems described in RTI's 1990 study.

As described in SW-846, Third Edition, the OWEP was designed to address the mobility of metals from oily matrices. The commenter is implying that it is an alternative to the TCLP for oily wastes; this is not accurate. OWEP was developed in 1986 for the Delisting Program and modified in 1992, as an alternative to the EP Toxicity Test Method, to address concerns regarding the EP's ability to measure metals mobility in oily matrices. The method was actually designed to liberate through a traditional leaching protocol, the metals present in the solid phase after solvent removal of the oily-organic phase and thereby predict the metals leachability from waste streams in which the organic phase could potentially inhibit the metals mobility (see response to Comment 5.a below for additional details). More specifically, the OWEP method was developed for samples with a distinct solid phase and liquid component consisting of oil. The OWEP method differs from both the EP and TCLP methods in that the solid phase is first extracted with tetrahydrofuran and then toluene to remove the oily organic layer prior to the solids leaching procedure. Therefore, this leaching method is only applicable to the inorganic metals constituents since any organic constituents originally present in the solid phase would be removed with the solvent extraction.

Comment 3.b: EPA then goes on to state that "[c]rude oil tank sediment, the oiliest of all the residuals of concern, <u>was never observed by the Agency during the sampling and analysis to exhibit an oily phase</u>" (emphasis added). A review of Tables 1 and 2 of the Listing Support Document quickly contradicts EPA's assertion. Table 1 contains descriptions of the samples of Crude Oil Tank Sediments. Descriptions such as "oily black appearance" and "black and oily" are seen for several of the samples. Table 2 shows the analysis results for percent oil and grease performed on samples of Crude Oil Tank Sediments. The results averaged 20.7% oil and grease with results as high as 41%. This is a substantial oily phase, and the data contradicts EPA's statement quoted above from the text of the Listing Support Document. (ETC, 00005)

Response: While the sampled residuals obviously *contain* oil, this observation is not equivalent with concluding, as the commenter does, that a discrete *"substantial oily phase"* is present in these residuals. During EPA's observation and handling of crude oil tank sediment during sampling and laboratory analysis, a discrete oily phase was not observed. This held true even for Sample R8C-CS-01 which was collected by refinery personnel directly from the bottom of the

drained crude oil storage tank. This sampling approach for R8C-CS-01 deviated from the anticipated procedures as specified in the Sampling and Analysis Plan in which the sample was expected to be collected from roll-off boxes after the tank bottoms had been centrifuged. The refinery requested this modification in the planned sampling procedure because the refinery was concerned that the samples would be contaminated by other materials processed in the refinery's de-oiling unit. Since this refinery de-oils their tank sludge prior to landfilling it, the Agency agreed to remove any free liquid phase prior to analysis.[139] The laboratory was instructed to filter this sample prior to any total or TCLP analyses, however, the laboratory reported there was no free liquid present in any of the sample container aliquots and that pressurized filtration as specified for the TCLP did not produce a liquid phase.

Similarly, a discrete oil phase was not observed for Sample R10-CS-01, which was sampled directly after removal from the storage tank. No separate oil phase was observed by the samplers or the analytical laboratory despite the facility's intention to recycle this residual to the refinery's cracking unit and to add additional liquid hydrocarbon to the sludge to make it more pumpable to the refining process.[140] If the refinery had instead intended to landfill this sludge, it likely would have been a good candidate for hydrocarbon recovery via deoiling given its high oil and grease content (41 percent) and to ensure that it passed the paint filter test.

While collecting Sample R19-CS-01, the sampling team observed a "small amount of water present" in the storage bin. The field log stated that "the weather at the site was cloudy and damp, rain having fallen steadily overnight," accounting for the sampling team's observation of water in the storage bin. A water phase was not present in the sample or observed by the laboratory personnel.[141] As stated in the Supplemental Background Document, "crude oil tank sediment, the "oiliest" of all the residuals of concern, was never observed by the Agency during sampling and analysis to exhibit an oil phase."[142]

An integral step in the TCLP is filtration when multiple phases are anticipated. The contract laboratory determined through sample screening and preparation that filtration was unnecessary for all residuals listed in Table 1 of the 1997 Supplemental Listing Background Document except for two HF alkylation samples R8B-HS-01 and R9-HS-01. These samples were filtered and found to contain 50% and 24% liquid, respectively. In both cases the filtrate was compatible with the leachate and was combined for the semivolatiles and metals analyses. Volatiles analyses were

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[139] Supplemental Background Document. 1997. Pg. 3

[140] Supplemental Background Document. 1997. Pg. 3

[141] EPA. Sampling and Analytical Data Report for Record Sampling and Characterization under the 1992-1996 Petroleum Refining Listing Determination and Industry Study: Pennzoil Atlas Refinery, Shreveport, LA. Sample date October 12, 1994. CBI

[142] Supplemental Background Document. 1997. Pg. 2.

performed separately on the filtrate and leachate portions with the detected target analyte concentrations combined mathematically based on the ratio of filtrate and leachate to total volume. It is also important to note that the % oil and grease for sample R9 was 31% and was the third highest level reported for samples listed in Table 1, however, there were no reported difficulties encountered during the filtration procedure. The Agency also notes that among all of the samples analyzed via the TCLP in its petroleum refining investigation, none of the samples for which filtration was required generated a phase that was immiscible with the TCLP leachate. In other words all generated filtrates were soluble with the TCLP extraction fluid according to Section 7.2.13.2 from Method 1311; these filtrates did not represent separate oil phases.

It is possible to conclude based on a superficial reading of the 1990 RTI report that no separate phase was generated from these residuals because of the shortcomings in the TCLP filtration step identified in the RTI report. However, a closer examination of this study shows that the samples of concern to EPA in the current petroleum refining investigation differ dramatically from the samples studied in the RTI report titled "Evaluation and Modification of Method 1311 for Determining the Release Potential of Difficult-to-Filter Wastes" dated April 1990, which were fluid in nature and obviously contained multiple phases. Further, RTI documented that one of the two refinery wastes, API separator sludge, in fact did not pose a filtration problem and thus was only studied during the first phase of RTI's study.[143]

Comment 3.c: Another key point is the tarry consistency of several of the waste samples. Tar is a petroleum based material consisting of higher molecular weight fractions of petroleum products. Although these fractions are solid at room temperature, they still constitute an immiscible oily phase when subject to the TCLP test. An OWEP is more effective at extracting metals from this phase then a TCLP extraction medium. Tarry waste is derived from petroleum refining and does constitute an oily type material that is subject to the OWEP test, per the applicability section of Method 1330A. (ETC, 00005)

Response: In response to comments on the 11/20/95 proposal, EPA did conduct the OWEP. EPA does not believe, however, that the results of these analyses are appropriate for use as input to the groundwater model (see response to Comment 5 below).

Comment 4: Reported Oil and Grease Levels of Wastes Sent to Landfills

EPA attempts to make the argument that the petroleum refinery wastes that currently are disposed of in landfills are low in oil content. However, EPA concludes its NODA discussion with a statement that the average oil content of a significant subset of these wastes is 19%, after eliminating outlier data points. This represents a significant volume of waste with a substantial oil phase. When considering the hundreds of thousands of tons of these petroleum refinery wastes,

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[143] "Method 1311 did, however, accurately estimate the mobile and immobile fractions of API separator sludge determined in the column experiments, suggesting that it is suitable as written for oily wastes that are not difficult-to-filter". RTI. 1990. p. 79.

19% of this tonnage represents a substantial amount of waste with essentially an all oil phase. This fraction needs to be adequately characterized using the OWEP, since it represents a major amount of oily waste phase that is highly mobile in a land disposal scenario. (ETC, 00005)

Response: EPA reiterates that the March 1997 Supplemental Listing Background Document, and more specific data supplied to EDF through a FOIA, incorporates a great deal of information concerning oil and grease data of refinery wastes. The commenter has incorrectly construed EPA's evaluation of the oil and grease data. Specifically, EPA evaluated the available data to conclude that the median oil and grease content of landfilled residuals is less than 1 percent (see Appendix A to the Supplemental Listing Background Document). EPA did find a limited number of cases where the oil and grease content was greater than 10 percent. The commenter refers to text which is in reference to residuals that were landfilled off-site. EPA specifically verified the data for these residuals and determined that the average oil and grease content of *these six wastes* is 19 percent. This "subset" referred to by the commenter cannot appropriately be assumed to be representative of all such residuals landfilled offsite; it merely shows that the average oil and grease values of the *six most oily* waste is a certain value. EPA never implied that these particular samples with oil and grease levels above 10 percent are representative of other wastes, as stated by the commenter, and disagrees with the commenter's interpretation of these data.

EPA points out that the Agency's percent oil and grease results presented in the NODA were obtained using Method 9071A. This method involves extracting a chemically dried waste sample with a strong organic solvent, trichlorotrifluoroethane (Freon-113), in order to solubilize the heavy petroleum fraction constituents associated with oil and the nonvolatile greases. While EPA measured levels of "total oil and grease" (TOG) that appeared high for some wastes (e.g., up to 25 percent for crude oil storage tank sediment), the Agency stresses that the TOG method used measures all extractable organic material, including waxes, greases, and other large molecular weight substances. The TOG method does not measure, in any sense, "free" oil (i.e., oil that might migrate from the waste as a separate phase).   This procedure will also recover any organic materials soluble in trichlorotrifluoroethane that are not volatilized during the solvent extract evaporation step. Thus, Method 9071A oil and grease results will likely overestimate free flowing oil content and wastes with apparently high oil and grease concentrations may not contain mobile oil.

Further, as described above, there is an important distinction between *oil content, percent oil and grease,* and a *discrete oil phase*. EPA believes that refineries have a number of important incentives to minimize the potential for free oil phase development in wastes destined for land disposal, including state restrictions on liquids in landfills, waste minimization practices, and recovery of valuable feedstock. Furthermore, *percent oil and grease* results encompass many materials other than free oil since all nonvolatile compounds soluble in trichlorotrifluoroethane are potentially recovered and quantitated. EPA thus fundamentally disagrees with the commenter that these data point toward the conclusion that these residuals commonly exhibit "essentially an all oil phase," or that the OWEP is the appropriate leaching model.

Comment 5: <u>Alternative Leaching Methods</u>

Comment 5.a: EPA argues that the OWEP is not appropriate, and then discusses the results of re-analysis of 27 samples using the OWEP and testing the extraction medium for metals. EPA concludes that the oil content of the relevant wastes does not affect the metals mobility.

First, the ETC does not agree with EPA's remark that the OWEP would "over estimate" the leaching potential of metals from the waste. The point of using the OWEP test is to properly and accurately account for metals tied up in the oily phase that would be mobile in a land disposal scenario. Failure to use the OWEP results in erroneous under-estimation of the true mobile metals content of the waste stream. EPA then states that use of the OWEP "may cause drastic changes in the original sample matrix." This is also a misleading statement. The OWEP <u>is</u> the test applied to the original un-altered waste matrix. It does not change the waste matrix, just as the TCLP test does not change the waste matrix. In conclusion, both of EPA's qualitative arguments against use of the OWEP are totally wrong, misleading, and invalid defenses against EPA's failure to use the proper extraction procedures to characterize oily waste. (ETC, 00005)

Response: The commenter correctly quotes page 10 of the March 1997 Listing Support Analysis for Petroleum Refining Wastes. The commenter, however, is incorrect in stating that the purpose of the OWEP method is to "properly and accurately account for metals tied up in the oily phase". The method was actually designed to liberate (through a traditional leaching protocol) the metals present in the solid phase after solvent removal of the oily-organic phase. Although the solvent and leachate phases are analyzed separately with the results combined mathematically to obtain the OWEP result, the metals concentrations in the solvent phase in some cases are greater than the leachate values due to the smaller volume of solvent (0.6 L) compared to the leachate volume (1.3 L). However, the metals solvent concentration contributes very little to the final OWEP result since the concentration in (mg) associated with 0.6 L is diluted by a combined total volume of 1.9 L (see response to Comment 7.g for the equation for calculating the final leachate concentration that is also provided in Method 1330).

One disadvantage of the OWEP is that it changes or alters the original sample matrix to render it unusable for the determination of the leaching potential for the organic constituents. Any organics present prior to the solvent extraction step will likely be removed after Soxhlet extraction similar to the total analysis procedures for volatiles and semivolatiles using methanol and methylene chloride. Leaching the solvent extracted solid phase for organic target analytes would result in the inaccurate and falsely low determination of leachate constituents while the analysis of the solvent phase would more closely resemble the total concentration.

Further, EPA continues to maintain that the initial extraction with organic solvents in the OWEP may well drastically alter the waste matrix. The aqueous extraction that follows the initial organic extraction will see a very different waste matrix from that in the original sample, i.e., most organic material (as well as oil) is removed. While the test was designed to do this, the Agency recognizes that this approach may not be fully representative of possible leaching from a landfill.

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Comment 5.b: Secondly, EPA's re-analysis of the samples using the OWEP is not an accurate indication of the degree of underestimation of the leaching potential of these wastes for metals. The TCLP tests performed on the samples were conducted in 1993. EPA merely retrieved these aged samples from storage, and ran the OWEP analysis on these archived samples. Three years separate the TCLP analysis and the OWEP analysis. Holding times of metals have been exceeded, and therefore the results are not valid to draw definitive conclusion. (ETC, 00005)

Response: EPA agrees that the OWEP analysis was conducted outside of the optimal holding times for these samples. As such, EPA believes that the exact measurements should not be relied on for risk assessment evaluations. However, EPA undertook the OWEP analysis in an attempt to fully consider the commenter's concerns. The purpose of adhering to the method-specified holding times of 180 days to leaching and an additional 180 days for analysis is to minimize the loss of potential contaminants through volatilization or biodegradation. The primary concern with exceeding holding times is that the sample matrix would degrade, changing leaching properties and that volatilization would reduce contaminant levels. However, given the archiving practices of the laboratory in which samples were stored in their original sealed containers at room temperature (note that the unrefrigerated storage of samples designated for metals analysis is considered an acceptable practice) without exposure to light. Under these conditions any potential sample decomposition was minimal and with the seven constituents of concern, non-volatile metals, EPA believes that the results are of some use for comparative purposes in response to the commenters' concerns.

Comment 5.c: A review of Tables 1 through 7 of Appendix B of the report reveals further problems. The metals detection limits used for the 1993 TCLP analysis are higher than the reported detection levels for the 1996 OWEP analysis, making the two data sets impossible to compare. The 1993 TCLP analysis could have been performed to the same lower detection levels as the 1996 OWEP analyses; the laboratory simply did not choose to analyze to a lower level. Samples for which non-detect levels were reported in 1993 cannot be compared with the OWEP analysis performed in 1996. (ETC, 00005)

Response: EPA agrees that the differences in the TCLP and OWEP detection levels limits the extent of subsequent comparisons, however, insufficient sample leachate volumes were available for the re-analysis of the TCLP. Given the time constraints of the rulemaking schedule and the significant logistical constraints associated with sampling many of these residuals, it was not possible to collect additional sample volumes.

The 1993 TCLP analyses referred to by the commenter were predominantly performed in mid to late 1994. These TCLP metals samples were leached according to SW-846, Method 1311, and the resulting leachate acid digested according to Method 3010A. The TCLP digestates were then analyzed for the same target analytes as the total metals constituent list with the exception of sodium. The determinative methodology included CVAA, Method 7470A for mercury; GFAA, Methods 7060A, 7421, 7740, and 7841 for arsenic, lead, selenium, and thallium, respectively; and ICP, Method 6010A for all other analytes.

Prior to digestion, all leachates for both the original TCLP and the OWEP determinations were diluted by a factor of ten to minimize the effect of the acetate buffer leaching solution on the target analyte quantitation. The analytical detection limits for the OWEP leachates are approximately five times below those reported for the TCLP leachates due to the increased sensitivity obtained with the OWEP analytical technique (ICP-MS) compared to the TCLP analyses using ICP and GFAA. In addition, the OWEP solvent detection limits were substantially less than the TCLP detection limits since these extracts were digested directly without dilution.

Comment 5.d: Samples with detected levels in 1993 for the TCLP, when compared to the 1996 OWEP analyses, show substantial differences in leachable metals. For example, measured mobility of arsenic is as much as 5 times higher, and chromium and lead are 10 times higher. This dramatic difference between the TCLP and OWEP metals results does support the concern that the oil content of the waste significantly impacts the mobility of metals in the wastes. (ETC, 00005)

Response: In evaluating the arsenic OWEP/TCLP data, the only residual where OWEP appears to consistently demonstrate higher leaching levels is for hydrotreating catalysts. For two samples, OWEP results are two to three times higher than the detection limits reported for the TCLP results. For two other samples, OWEP results are an order of magnitude lower than the TCLP detection limits. For the fifth sample, OWEP results are 4.4 times greater than TCLP results. These results, however, do not support the commenter's claim that leaching behavior of oily wastes is better measure with the OWEP because none of the hydrotreating catalysts samples had detectable levels of oil and grease. Thus, the OWEP results for these samples could be considered irrelevant as far as evaluating the impact of oil and grease content on metal leachate.

Furthermore, as discussed earlier, there is greater variability within each waste category (e.g., hydrotreating OWEP results range from 0.02 to 6.69 mg/l arsenic as documented in Appendix B of *Supplemental Background Document Listing Support Analysis*, 1997) than between the OWEP and TCLP results. EPA disagrees that any measurable differences, of which there are few, in OWEP/TCLP results are dramatic or particularly meaningful. In addition, EPA does not give the OWEP results the same weight of validity as the TCLP results because the samples were old and subjected to aggressive pre-leaching with organic solvents. Moreover, the OWEP results for arsenic in hydrotreating catalysts suggest that the aggressive organic leaching in the OWEP procedure may be disturbing the waste matrix sufficiently to cause some increase in mobility (without oil content being a concern). However, the data are too limited to draw any conclusions.

The following table reduces the results of the OWEP analyses to the most relevant results: those samples with oil and grease content greater than 1 percent, as well as detected results for either OWEP or TCLP analyses.

Table I.C.4. OWEP / Original TCLP Data Comparison				
Residual	Sample ID	%Oil & Grease	OWEP	TCLP
Arsenic mg/L				

Table I.C.4. OWEP / Original TCLP Data Comparison						
Crude Sludge	R8C-CS-01	25		0.03	<	0.10
CSO Sludge	R9-SO-01	70		**0.11**	<	0.10
	R20-SO-01	24		0.04	<	0.10
Barium mg/L						
Crude Sludge	R8C-CS-01	25		**3.33**		2.4
	R4B-CS-01	15		0.22	<	2
	R10-CS-01	41		0.88	<	2
	R22-CS-01	4.9		0.38	<	2
	R19-CS-01	14		0.64		2.7
CSO Sludge	R9-SO-01	70		0.33	<	2
	R1B-SO-01	16		0.70	<	2
	R20-SO-01	24		0.39	<	2
HF Alkylation Sludge	R9-HS-01	31		0.29	<	2
	R15-HS-01	6.8		0.08	<	2
Sulfur Complex Sludge	R1-ME-01	1.0		0.11	<	2
Off-Spec Product	R12-TP-01	8.4		0.64	<	2
Chromium, mg/L						
Crude Sludge	R8C-CS-01	25		**0.89**	<	0.10
	R4B-CS-01	15		0.02	<	0.10
	R10-CS-01	41		0.01	<	0.10
	R22-CS-01	4.9		0.01	<	0.10
	R19-CS-01	14		0.06	<	0.10
CSO Sludge	R9-SO-01	70		0.01	<	0.10
	R1B-SO-01	16		0.04	<	0.10
	R20-SO-01	24		0.03	<	0.10
HF Alkylation Sludge	R9-HS-01	31		0.02	<	0.10
	R15-HS-01	6.8		0.06	<	0.10
Sulfur Complex Sludge	R1-ME-01	1.0		0.02	<	0.10
Off-Spec Product	R12-TP-01	8.4		0.01	<	0.10
Lead, mg/L						
Crude Sludge	R4B-CS-01	15		**0.07**	<	0.03
	R10-CS-01	41		0.03	<	0.03
	R22-CS-01	4.9		0.01	<	0.03
	R19-CS-01	14		**0.30**	<	0.03
CSO Sludge	R9-SO-01	70		0.01	<	0.03
	R1B-SO-01	16		0.01	<	0.03
HF Alkylation Sludge	R9-HS-01	31		0.00	<	0.03
Selenium, mg/L						
Sulfur Complex Sludge	R1-ME-01	1.0		**0.11**	<	0.05

Shaded rows in Table I.C.4 represent samples with both OWEP and TCLP detected values, of which there are only two. These two barium results in crude oil tank sediment samples are within

one order of magnitude, and show no pattern as one pair has higher OWEP results while the other has higher TCLP results.

Bolded OWEP results in Table I.C.4 indicate samples where OWEP leaching is greater than TCLP (using TCLP detection limits). Only six of the 35 data pairs show this pattern. (It is not possible to determine whether the OWEP or TCLP indicated greater leaching for the remainder of the relevant samples because of the TCLP detection limits.) EPA does not believe that these results demonstrate any dramatic differences in leaching.

Comment 5.e: A fundamental problem with the report is that three years separate the OWEP and TCLP analyses, and holding times of metals have been greatly exceeded. These results are invalid and violate EPA's own QA/QC requirements. Furthermore, the difference in levels of detection used by the laboratories in 1993 compared to 1996 makes most of the data useless. Finally, nowhere is the total metals content of the waste reported. Assessment of leachability of metals must be made relative to the total metals composition of the waste. The comparison between the TCLP and OWEP methods on the waste samples does not show the initial total metals content of the samples before the extraction procedures. Since the objective is to compare the two extraction procedures with regard to impact of the oil phase on metals mobility, the initial metals content is a critical point that must be known. To properly demonstrate this point, EPA needs to obtain new samples of the various waste streams, and run concurrent total, TCLP and OWEP metals analyses. (ETC, 00005)

Response: EPA agrees that the holding times were exceeded, but as discussed above, believes that the results are useful in exploring the commenter's concerns. EPA realized that the data are only of limited utility due to the problems noted. However, despite these limitations, EPA believes that the data support the overall conclusion that the oily content of the waste has no significant effect on the leachability of the metals. Thus, EPA disagrees that it is necessary to conduct further sampling and analysis.

Note that total metals levels were reported with TCLP results in the 1995 Listing Background Document and are not expected to change between the times of the TCLP and OWEP analyses.

Comment 5.f: The report, as it stands now, does not support EPA's assertion that failure to use the OWEP would not have affected the listing determination with regard to mobility of metals. If anything, the handful of useful data points, for which detection levels were obtained on TCLP in 1993, show substantially higher mobility of metals under the OWEP for arsenic, lead and chromium. The data further supports the concerns raised by commenters that the oily phase of the waste does significantly impact the mobility of arsenic, chromium and lead. (ETC, 00005)

Response: See responses above to Comment 5.a through 5.e.

Comment 6: Analysis of Leaching Efficiency

Comment 6.a: EPA evaluated the efficacy of the TCLP for organic constituents for waste samples relevant to the listing determination. Only one organic constituent, benzene, was evaluated. EPA presented the results of TCLP analysis for benzene on the various samples and calculated the percent of benzene extracted by comparing this to the total benzene content of the waste. EPA concluded that the TCLP test adequately predicts the mobility of organic hazardous constituents, and is not adversely impacted by the oil content of the waste.

This conclusion is absurd in that EPA did not bother to analyze the benzene leachability using the OWEP. There is no side-by-side comparison of percent benzene extracted for the TCLP vs. the OWEP procedures. The OWEP procedure would likely have leached a higher percentage of benzene, and would be expected to more accurately predict the mobility of organic hazardous constituents in an oily waste matrix. EPA simply chose not to run the OWEP analyses. EPA's conclusion is based on a subjective and biased opinion that the TCLP extractions give a high enough percentage of benzene extracted. For example, for the crude tank sludge sample number R10-CS-01 in Table 2, 22% of the benzene is leached using the TCLP for a sample with 41% oil phase. It is highly likely that the OWEP would yield a significantly higher percentage of benzene mobility. EPA cannot conclude that the TCLP procedure is sufficient for organic constituents without also running the OWEP. (ETC, 00005)

Response: As stated previously (see response to Comment 5.a), the commenter has a fundamentally flawed understanding of the OWEP method. The OWEP was designed to measure metals mobility in oily matrices, as an aggressive alternative method to the EP. Currently, EPA does not have a comparable method for the TCLP since the OWEP method would inaccurately determine leachable organic constituents that would more closely resemble the total sample concentration. Moreover, toluene used as an OWEP solvent is also a volatiles target analyte and would interfere in the low level detection of other volatile constituents. For this reason, EPA limited its OWEP analyses to metals.

Comment 6.b: Finally, a key flaw in EPA's conclusions regarding organic constituents is that EPA only ran one constituent, benzene. Other organic hazardous constituents found in significant concentrations in these petroleum refining wastes should have also been evaluated. Constituents with different polarities and solubility properties, such as halogenated organics, phenolics and polynuclear aromatic compounds should have been evaluated. The report proves nothing regarding the mobility of hazardous organic constituents under the TCLP test compared to the OWEP procedure, and the impact of the oily nature of the waste on this mobility. (ETC, 00005)

Response: The leaching potential of benzene was evaluated because this compound was consistently detected in all samples listed in Table 2 of the 1997 Supplemental Listing Background Document. Furthermore, due to its high toxicity and relative mobility, benzene was a key constituent in the risk analysis results. Based on the commenter's concern for the lack of reviewing a full range of organic constituents, the Agency further evaluated the leaching potential of four additional target analytes (m,p-xylenes, naphthalene, 3/4-methyl phenol, and phenanthrene). These leaching potential data are presented in Tables I.C.4 through I.C.7 below.

A review of these data indicates there is no significant discernable trend with respect to lower leaching values associated with higher oil and grease content. The constituents generally leached in similar proportions for all residual types and any differences may be attributed to sample homogeneity, pH, particle size distribution and other factors that have some impact on the leaching rate and the final leachate target analyte concentration. In addition, even though phenanthrene and several other PAH compounds were detected at significant levels in the total analyses, the concentrations are expected to be low in the leachates due to the relative insolubility of these compounds in water.

Table I.C.4. PETROLEUM REFINING RESIDUALS LEACHING POTENTIAL				
Sample ID	Total m,p-Xylenes (µg/kg)	TCLP m,p-Xylenes (µg/L)	% m,p-Xylenes Leached[1]	% Oil & Grease
CRUDE TANK SLUDGE				
R6B-CS-01	320,000	1,500	9	24
R8C-CS-01	830,000	1,300	3	25
R4B-CS-01	180,000	450	5	15
R10-CS-01	390,000	560	3	41
R19-CS-01	1,400	< 50	<70	14
R22-CS-01	18,000	120	13	4.9
UNLEADED GASOLINE TANK SLUDGE				
R6B-US-01	1,400,000	3,200	5	NA
R8A-US-01	1,300,000	6,100	9	0.09
R16-US-01	610,000	2,700	9	< 0.09
CSO SLUDGE				
R4-SO-01	11,000	250	45	NA
R9-SO-01	19,000	100	11	70
R1B-SO-01	69,000	250	7	16
R20-SO-01	100,000	160	3	24
HYDROTREATING CATALYST				
R1-TC-01	550,000	13,000	47	< 0.05
R8A-TC-01	280,000	520	4	NA
R11-TC-01	34,000	150	10	< 0.05
R3B-TC-01	30,000	130	9	< 0.05

Table I.C.4. PETROLEUM REFINING RESIDUALS LEACHING POTENTIAL				
Sample ID	Total m,p-Xylenes (μg/kg)	TCLP m,p-Xylenes (μg/L)	% m,p-Xylenes Leached[1]	% Oil & Grease
R18-TC-01	12,000	46	8	< 0.05
R22-TC-01	99,000	3,000	61	< 0.2
HYDROREFINING CATALYST				
R5-TC-01	4,100	< 50	<24	NA
R7B-RC-01	78,000	530	14	NA
R21-RC-01	23,000	< 50	<4	0.08
REFORMING CATALYST				
R2-CR-01	220	< 50	NA	NA
R5-CR-01	170,000	1,500	18	NA
R7B-CR-01	9,000	340	76	NA
R11-CR-01	< 25	500	NA	NA
R14-CR-01	4,500	140	62	NA
R15-CR-01	69,000	NA	NA	NA
HF ALKYLATION SLUDGE				
R3-HS-01	352,000	480	3	NA
R8B-HS-01	< 625	< 50	NA	0.3
R9-HS-01	3,700	< 50	<27	31
R15-HS-01	460	< 50	NA	6.8
R7C-HS-01	1,200	< 50	<83	0.08
SULFUR COMPLEX SLUDGE				
R1-ME-01	54,000	630	23	1.0
R5-ME-01	480	< 50	NA	NA
R6-ME-01	1,000	340	100	NA
R14-ME-01	790	< 50	NA	0.2
R18-ME-01	< 600	< 50	NA	< 0.05

1   Percent m,p-xylenes leached calculated based on a 20 to 1 ratio of leaching fluid to sample mass. The total
    concentration was multiplied by 0.05 to obtain the total theoretical leachate concentration. This value was divided

into the reported TCLP concentration to obtain the percent leached. Calculated percentages greater than 100% are listed as 100% leached.

Table I.C.5. PETROLEUM REFINING RESIDUALS LEACHING POTENTIAL				
Sample ID	Total Naphthalene (μg/kg)	TCLP Naphthalene (μg/L)	% Naphthalene Leached[1]	% Oil & Grease
CRUDE TANK SLUDGE				
R6B-CS-01	180,000	65	1	24
R8C-CS-01	150,000	77	1	25
R4B-CS-01	280,000	500	4	15
R10-CS-01	150,000	59	1	41
R19-CS-01	< 11,500	< 50	NA	14
R22-CS-01	6,100	14	5	4.9
UNLEADED GASOLINE TANK SLUDGE				
R6B-US-01	190,000	780	8	NA
R8A-US-01	400,000	830	4	0.09
R16-US-01	49,000	640	26	< 0.09
CSO SLUDGE				
R4-SO-01	62,000	32	1	NA
R9-SO-01	88,000	23	1	70
R1B-SO-01	180,000	170	2	16
R20-SO-01	360,000	< 50	<1	24
HYDROTREATING CATALYST				
R1-TC-01	1,000	< 50	NA	< 0.05
R8A-TC-01	250,000	170	1	NA
R11-TC-01	< 165	< 50	NA	< 0.05
R3B-TC-01	1,100	< 50	NA	< 0.05
R18-TC-01	400	< 50	NA	< 0.05
R22-TC-01	140	< 50	NA	< 0.2
HYDROREFINING CATALYST				
R5-TC-01	< 165	< 50	NA	NA
R7B-RC-01	3,000	< 50	NA	NA

Table I.C.5. PETROLEUM REFINING RESIDUALS LEACHING POTENTIAL				
Sample ID	Total Naphthalene (μg/kg)	TCLP Naphthalene (μg/L)	% Naphthalene Leached[1]	% Oil & Grease
R21-RC-01	580	170	100	0.08
REFORMING CATALYST				
R2-CR-01	2,100	59	56	NA
R5-CR-01	8,900	27	6	NA
R7B-CR-01	190	< 50	NA	NA
R11-CR-01	130	< 50	NA	NA
R14-CR-01	< 165	< 50	NA	NA
R15-CR-01	6,900	< 50	<14	NA
HF ALKYLATION SLUDGE				
R3-HS-01	110,000	320	6	NA
R8B-HS-01	< 1,000	< 50	NA	0.3
R9-HS-01	4,800	< 50	<21	31
R15-HS-01	< 2,063	< 50	NA	6.8
R7C-HS-01	2,000	22	22	0.08
SULFUR COMPLEX SLUDGE				
R1-ME-01	13,000	93	14	1.0
R5-ME-01	18,000	42	5	NA
R6-ME-01	< 16,500	< 50	NA	NA
R14-ME-01	< 2,063	< 50	NA	0.2
R18-ME-01	< 165	< 50	NA	< 0.05

1   Percent naphthalene leached calculated based on a 20 to 1 ratio of leaching fluid to sample mass. The total concentration was multiplied by 0.05 to obtain the total theoretical leachate concentration. This value was divided into the reported TCLP concentration to obtain the percent leached. Calculated percentages greater than 100% are listed as 100% leached.

Table I.C.6. PETROLEUM REFINING RESIDUALS LEACHING POTENTIAL				
Sample ID	Total 3/4-Methylphenol (μg/kg)	TCLP 3/4-Methylphenol (μg/L)	% 3/4-Methylphenol Leached[1]	% Oil & Grease
CRUDE TANK SLUDGE				
R6B-CS-01	< 10,313	< 50	NA	24
R8C-CS-01	< 4,125	< 50	NA	25
R4B-CS-01	12,000	490	82	15
R10-CS-01	< 49,500	< 50	NA	41
R19-CS-01	< 11,500	< 50	NA	14
R22-CS-01	< 413	< 50	NA	4.9
UNLEADED GASOLINE TANK SLUDGE				
R6B-US-01	< 1,650	160	> 3	NA
R8A-US-01	< 1,650	180	> 4	0.09
R16-US-01	< 165	< 50	NA	< 0.09
CSO SLUDGE				
R4-SO-01	41,000	< 50	<3	NA
R9-SO-01	< 20,625	< 50	NA	70
R1B-SO-01	< 20,625	32	3	16
R20-SO-01	< 41,250	76	4	24
HYDROTREATING CATALYST				
R1-TC-01	4,200	< 50	<24	< 0.05
R8A-TC-01	< 4,125	75	36	NA
R11-TC-01	2,100	790	100	< 0.05
R3B-TC-01	< 165	< 50	NA	< 0.05
R18-TC-01	950	12	25	< 0.05
R22-TC-01	< 165	96	100	< 0.2
HYDROREFINING CATALYST				
R5-TC-01	< 165	< 50	NA	NA
R7B-RC-01	1,800	150	100	NA
R21-RC-01	170	65	100	0.08
REFORMING CATALYST				

Table I.C.6. PETROLEUM REFINING RESIDUALS LEACHING POTENTIAL				
Sample ID	Total 3/4-Methylphenol (μg/kg)	TCLP 3/4-Methylphenol (μg/L)	% 3/4-Methylphenol Leached[1]	% Oil & Grease
R2-CR-01	< 165	< 50	NA	NA
R5-CR-01	24,000	490	41	NA
R7B-CR-01	330	< 50	NA	NA
R11-CR-01	< 165	< 50	NA	NA
R14-CR-01	< 165	< 50	NA	NA
R15-CR-01	4,500	< 50	<22	NA
HF ALKYLATION SLUDGE				
R3-HS-01	32,000	1,200	75	NA
R8B-HS-01	< 1,000	< 50	NA	0.3
R9-HS-01	< 5,157	< 50	NA	31
R15-HS-01	< 2,063	< 50	NA	6.8
R7C-HS-01	< 165	< 50	NA	0.08
SULFUR COMPLEX SLUDGE				
R1-ME-01	< 6,600	< 50	NA	1.0
R5-ME-01	< 3,300	< 50	NA	NA
R6-ME-01	< 16,500	< 50	NA	NA
R14-ME-01	< 2,063	66	48	0.2
R18-ME-01	< 165	< 50	NA	< 0.05

1   Percent 3/4-methylphenol leached calculated based on a 20 to 1 ratio of leaching fluid to sample mass. The total
     concentration was multiplied by 0.05 to obtain the total theoretical leachate concentration. This value was divided
     into the reported TCLP concentration to obtain the percent leached. Calculated percentages greater than 100% are
     listed as 100% leached.

Table I.C.7. PETROLEUM REFINING RESIDUALS LEACHING POTENTIAL				
Sample ID	Total Phenanthrene (μg/kg)	TCLP Phenanthrene (μg/L)	% Phenanthrene Leached[1]	% Oil & Grease
CRUDE TANK SLUDGE				
R6B-CS-01	73,000	< 50	<2	24
R8C-CS-01	76,000	< 50	<2	25
R4B-CS-01	380,000	27	<1	15
R10-CS-01	47,000	< 50	<2	41
R19-CS-01	< 11,500	< 50	NA	14
R22-CS-01	7,300	< 50	<14	4.9
UNLEADED GASOLINE TANK SLUDGE				
R6B-US-01	2,400	< 50	<42	NA
R8A-US-01	1,400	< 50	<71	0.09
R16-US-01	< 165	< 50	NA	< 0.09
CSO SLUDGE				
R4-SO-01	200,000	< 50	< 1	NA
R9-SO-01	1,000,000	12	< 1	70
R1B-SO-01	320,000	< 50	< 1	16
R20-SO-01	680,000	< 50	< 1	24
HYDROTREATING CATALYST				
R1-TC-01	< 660	< 50	NA	< 0.05
R8A-TC-01	400,000	16	< 1	NA
R11-TC-01	< 165	< 50	NA	< 0.05
R3B-TC-01	< 165	< 50	NA	< 0.05
R18-TC-01	< 165	< 50	NA	< 0.05
R22-TC-01	300	< 50	NA	< 0.2
HYDROREFINING CATALYST				
R5-TC-01	< 165	< 50	NA	NA
R7B-RC-01	2,200	< 50	< 45	NA
R21-RC-01	1,200	< 50	< 83	0.08

Table I.C.7. PETROLEUM REFINING RESIDUALS LEACHING POTENTIAL				
Sample ID	Total Phenanthrene (µg/kg)	TCLP Phenanthrene (µg/L)	% Phenanthrene Leached[1]	% Oil & Grease
REFORMING CATALYST				
R2-CR-01	14,000	35	5	NA
R5-CR-01	< 660	< 50	NA	NA
R7B-CR-01	< 165	< 50	NA	NA
R11-CR-01	< 165	< 50	NA	NA
R14-CR-01	690	< 50	NA	NA
R15-CR-01	160	< 50	NA	NA
HF ALKYLATION SLUDGE				
R3-HS-01	4,500	< 50	< 23	NA
R8B-HS-01	< 1,000	< 50	NA	0.3
R9-HS-01	< 5,157	< 50	NA	31
R15-HS-01	< 2,063	< 50	NA	6.8
R7C-HS-01	2,400	< 50	< 42	0.08
SULFUR COMPLEX SLUDGE				
R1-ME-01	27,000	12	1	1.0
R5-ME-01	< 3,300	< 50	NA	NA
R6-ME-01	< 16,500	< 50	NA	NA
R14-ME-01	14,000	< 50	< 7	0.2
R18-ME-01	580	< 50	NA	< 0.05

1   Percent phenanthrene leached calculated based on a 20 to 1 ratio of leaching fluid to sample mass. The total concentration was multiplied by 0.05 to obtain the total theoretical leachate concentration. This value was divided into the reported TCLP concentration to obtain the percent leached. Calculated percentages greater than 100% are listed as 100% leached.

Comment 6.c: In short, EPA's report is not credible and does not address the comment that EPA's listing determination is flawed due to the failure to use the OWEP in evaluation of the wastes. The report is inaccurate in its statements and is biased in its analysis of the data presented in the Tables and the appendices. The report reflects poor science and improper experimental design in the manner in which the analytical work was planned in an attempt to prove a pre-conceived and erroneous conclusion.

In order to account for these problems, KGS modeled various refinery wastes by applying the average TCLP efficiency of 53% for benzene to the waste streams with a substantial oil content. For many wastes, the TCLP concentration mean input value increased significantly as a result, and the groundwater exposure pathway risk posed by landfilling the wastes also increased significantly. Based on KGS's evaluation of this factor and other parameters, the risks posed by potential mismanagement of most petroleum refinery wastes covered by the NODA require a hazardous waste listing, as discussed in Section II [of ETC's comments] below. (ETC, 00005)

Response: EPA fundamentally disagrees with the commenter. The OWEP is not appropriate for use in this listing determination because: 1) this leaching method is only applicable to the inorganic metals constituents since any organic constituents originally present in the solid phase would be removed with the solvent extraction; and 2) leaching the solvent extracted solvent phase for organic target analytes would result in the inaccurate and falsely low determination of leachate constituents while the analysis of the solvent phase would more closely resemble the total concentration. Furthermore, EPA believes that using an average TCLP leaching efficiency of 53 percent for benzene is inappropriate and disregards useful information on individual wastes.

Comment 7: <u>Ineffectiveness of the TCLP on Oily/Tarry Wastes</u>

Comment 7a: Notwithstanding EPA's repeated and longstanding acknowledgment that the TCLP substantially understates the leaching potential of oily wastes, the Agency relied on TCLP results as the input value to the groundwater model for landfill disposal in the 1995 proposal.[144] EDF commented extensively on this glaring inconsistency and the inadequacy of the TCLP for this purpose. Now, instead of admitting error in this regard, EPA attempts to justify use of the TCLP through additional analyses in the NODA. As explained in this section of the comments, the NODA analyses actually prove the inadequacy of the TCLP for the oily wastes in this rulemaking.

First, EPA produced lab preparation sample descriptions for some wastes, and based upon the purported "tarry or granular consistency" sample descriptions, contends TCLP results are appropriate because no liquid phase in the waste sample was observed.[145] However, contrary to EPA's generalization, many sampling descriptions indicate liquid phases <u>were</u> observed. One crude oil tank sludge sample was described as "black and oily with a consistency of cake icing.

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[144] "The Agency has consistently maintained that the EP test as well as the TCLP test underestimate releases of hazardous constituents from oily wastes." Response to Comments Background Document for the Listing of Primary and Secondary Oil/Water/Solids Separation Sludges from the Treatment of Petroleum Refinery Wastewaters - F037 and F038, EPA, August 29, 1990, p. 171.

[145] NODA Background Document, Chapter 2, p. 1.

Compared to other centrifuge cakes from other refineries this sludge was more fluid."[146] A second sample had "a small amount of water present," even after filtering. Similarly, a CSO sample "seeped oil out of the filter if squeezed"; an unleaded storage tank sludge sample had "approximately one inch of free liquid on the surface"; and three HF alkylation sludge samples were "black and oily with a high liquid content," "yellow with a consistency of wet oily sludge," and had "the consistency of oil-in-water emulsion".[147] These seven samples comprise 38% of the 18 EPA samples of crude oil tank, CSO, HF alkylation, and unleaded storage tank sludges combined.[148] (ETC, 00006)

Response: The commenter misconstrued the Agency's discussion in the referenced NODA document (see also response to Comment 3.b).

The commenter misinterpreted EPA's description of its field and laboratory observations when EDF implied that EPA's statement that "no liquid phase in the waste sample was observed" was a "generalization" that applied to "many sampling descriptions". In fact, however, EPA's "no liquid phase" description was in specific reference to the crude oil tank sediment samples and was not a generalization that was meant to be applied to all other samples (in particular, this statement could not be applied to the HF sludge samples collected prior to dewatering). With respect to the specific samples called out by the commenter, EPA disagrees that the TCLP is inappropriate for their characterization. The TCLP filtration step was apparently problem-free with no instances of filter clogging that were reported by the laboratory. Any phase separation that occurred during this filtration step resulted in phases that were, in all cases, miscible with the TCLP extraction fluid (i.e., not free phase oil).

Sample R10-CS-01 was characterized in Table 1 of the 1997 Supplemental Listing Background Document [full quotation provided, in comparison with commenter's paraphrase above] as "black and oily with a consistency of cake icing. Compared to filter cakes or centrifuge cakes from other refineries this sludge was described as more fluid." This sample represented crude oil tank sludge, sampled directly after removal from the storage tank, which was slated for recycling to the cracking unit.[149] In order for this sludge to be landfilled, it likely would have been deoiled to recover the high level of observed hydrocarbons (41 percent oil and grease) and to ensure that the

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[146] Id. at 4.

[147] Id. at 5-6.

[148] In addition, the remaining crude oil storage tank sludge samples were described as "oily black", "black tar-like", "black sandy tar", and "black tar-like medium texture". Therefore, all the crude oil storage tank sludge samples exhibited the oily or tarry physical state associated with previous TCLP findings of ineffectiveness. See 57 FR 37294, 37296 (August 18, 1992).

[149] Supplemental Listing Background Document. 1997. Pg. 3

residual passed the paint filter test. Even <u>without</u> deoiling, however, no free liquids were reported by the samplers or the laboratory.

While collecting sample R19-CS-01 ("a small amount of water present"), the sampling team observed a "small amount of water present" in the storage bin due to the rain that had fallen steadily through the night, however a water phase was not present in the sample or observed by the laboratory personnel.[150] (If water had been observed by the laboratory, this should not have aroused the commenter's concern regarding free oil content.) As stated in the Supplemental Listing Background Document, "crude oil tank sediment, the "oiliest" of all the residuals of concern, was never observed by the Agency during sampling and analysis to exhibit an oil phase."[151] As discussed in response to Comment 1.d, these samples are fundamentally different from the multi-phase, fluid, difficult-to-filter wastes characterized in the 1990 RTI report.

Sample R9-SO-01, which was observed by the sampling team to seep oil while being torn into pieces prior to placement in sampling containers, was a sample of in-line filters which remove FCC fines from the slurry oil between the FCC fractionator and the CSO storage tank. Only 3 refineries in the U.S. reported performing this type of in-line filtering prior to storage in the 1992 survey. Each facility reported storing their filters in drums generating an annual total quantity of 1.074 MT.

The paper filter media subsample of R9-SO-01 designated for non-volatile analyses was composited by manually tearing the material in order to obtain the most representative sample for analysis. This process, which also increases the sample surface area, undoubtedly resulted in the absorption of any free oil product that may have been present at the time of sampling. Since no free liquid was present in the composite mixture at the laboratory, the filtration step for the TCLP analysis was not necessary. There were no laboratory-reported method deviations for the TCLP preparation and analysis of sample R9-SO-01.

The HF alkylation sludges were taken directly from the in-ground tanks in which they are generated. HF alkylation sludge is primarily aqueous-based calcium fluoride sludge with minimal hydrocarbon contamination. Due to sample availability, the Agency collected four of the five samples prior to dewatering, their normal treatment prior to disposal. See response to Comment 1.b, above, for additional discussion on the generation of HF alkylation sludge.

The commenter quoted Table 1 of the 1997 Listing Background Document for R3-HS-01, R15-HS-01, and R9-HS-01. R3 and R15 were collected directly from the neutralization tank, thus their semi-liquid nature. R3 was decanted at the laboratory at the refinery's request to better

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[150] EPA. <u>Sampling and Analytical Data Report for Record Sampling and Characterization under the 1992-1996 Petroleum Refining Listing Determination and Industry Study: Pennzoil Atlas Refinery, Shreveport, LA</u>. Sample date October 12, 1994. CBI

[151] Supplemental Background Document. 1997. Pg. 2.

reflect the characteristics of the waste as it would have been generated for disposal. R9 was collected from four 55-gallon drums where it was being stored after removal from the neutralization tanks (no centrifugation or filtering had been conducted prior to sample collection). The fifth HF sample, R7C-HS-01, was collected after centrifuging and was of a very different consistency that the other "in process" or "as generated" samples. Thus, the more liquid nature of the four samples is a reflection of their "as generated" nature and their derivation from an aqueous neutralization system. Further, during the initial filtration step of the TCLP, the filtrate generated for each of these samples was easily miscible with the TCLP leachate and thus cannot be characterized as free oil.

Sample R9-HS-01 exhibited oil and grease levels of 31 percent, much higher than the other samples, and much higher than oil and grease levels reported for HF alkylation sludge landfilled in Subtitle D units in 1992 (see Table I.C.10 in response to Comment 8.b, below). The drums holding this residual were labeled as D001/D002 and EPA's analytical results confirmed the corrosive nature of this waste with pH=13. The facility reported in its survey response that its HF sludge is typically managed at an off-site hazardous waste landfill. The high oil content of this residual appears to be due to the refinery's practice of draining its Acid Soluble Oil (ASO)[152] from its HF regenerator to the neutralization pit, from which the HF sludge is generated. Thus, the inclusion of this sample within EPA's modeling input is conservative due to the already regulated nature of the waste.

In conclusion, EPA disagrees with the commenter's characterization of the samples and its subsequent conclusion that the TCLP is inappropriate for these wastes and notes in summary:

•   The crude oil sediment samples exhibited no liquid phase, despite the Agency's expectation that these would be the oiliest of all the RCs studied, aside from a small amount of rainwater observed in one waste storage bin.

•   The CSO filter sample was not filtered by the laboratory based on its observation of the sample.

•   The free liquid associated with the unleaded gasoline sludge was a characterization of the material remaining in the storage tank at the time of sampling and was not sampled.

•   The liquid nature of the HF sludge samples is a reflection of the collection of these samples directly from the neutralization tank (for 3 of the samples) or from dredged sludge held in drums which had not been dewatered. TCLP filtration of these samples resulted in a filtrate that was easily miscible with the TCLP leachate and there was no indication of filtration difficulties such as those encountered in the 1990 RTI study.

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[152]ASO is described further in the 1996 Study.

Comment 7.b: In other instances, the samples were inappropriately manipulated to discourage the detection of free liquids. For example, EPA allowed CSO sludge to be mixed with cement kiln dust prior to sampling, thus the sample did not reflect the liquid content of the CSO sludge itself. One other CSO sludge sample and an unleaded storage tank sludge sample were "air dried," thereby allowing the free liquid to evaporate and toxic contaminants such as benzene to volatilize. Two HF alkylation sludges were "dewatered" prior to sampling or at the laboratory prior to analysis.[153] Given these sampling practices, and the liquids observed in other samples, the sampling descriptions do not justify reliance on the TCLP in the instant rulemaking.[154] (EDF, 00006)

Response: As discussed in response to Comment 1.b, above, EPA attempted to collect samples that were representative of the wastes after the wastes were removed from the generating units. Stabilization with materials such as cement kiln dust is not uncommon. EPA did not "allow" the CSO sludge to be mixed with cement kiln dust, but did note that this had already occurred prior to arrival of EPA's contractor team on-site. In regards to the CSO sample, all of the samples were below the detection limit for benzene and many of the other volatile constituents demonstrating that air drying of the tank had little to no effect on the sample's volatile contents as discussed further in response to Comment 1.a.

The "air dried" unleaded gasoline and CSO sludge samples, as discussed earlier in response to Comment 1.a, were being managed in a manner consistent with practices designed to lower benzene and LEL levels to allow safe working conditions for the personnel conducting the tank turnarounds[155]. Further, this is a "typical" management practice for unleaded tank sediment as observed during site visits to other refineries and from RCRA §3007 survey results. At the sampled refinery, the tank was water washed, allowed to dry, and then solids were drummed. Water washing was reported to be a common technique reported during the site visits. Additionally, more than 85 percent of the facilities cleaning a tank since 1991 reported washing, most often with water (Waste Minimization for Selected Residuals in the Petroleum Refining Industry, December 1996). EPA has insufficient data to determine the frequency of "air drying".

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[153] 1995 Listing Background Document at 135.

[154] In evaluating risks associated with waste mismanagement scenarios, EPA must evaluate the risks posed by the waste <u>as generated</u>, absent a legal or technical basis for concluding all the waste will be pretreated in a certain manner prior to disposal. In the instant rulemaking, EPA has never claimed air drying, waste dewatering, or mixing waste with CKD are legally or technically compelled, or universally performed. See also discussion regarding sampling of "air dried" wastes in Section II.H of the comments.

[155] "Listing Background Document for the 1992-1996 Petroleum Refining Listing Determination." EPA. October 31, 1995. Pg. 35.

One HF sludge sample, R3-HS-01, was collected directly from the neutralization pit and was filtered by the lab to better characterize the waste as it was to be disposed. The liquid portion of the Exxon sample was decanted and analyzed for volatiles only. Based on the low volatile concentrations in the liquid phase, this phase appears to be an aqueous and not an oil phase. The other HF sludge sample of concern to the commenter, R7C-HS-01, was of sludge which had been centrifuged and was awaiting transport to an offsite municipal landfill. EPA believes that this sample is particularly representative of HF sludge as disposed.

Comment 7.c: Further, the lack of primary leachate generation during initial filtration recorded by EPA in the NODA Background Document for the above wastes, and the other "tarry" sludges, merely confirms EPA's previous findings that the TCLP understates the leachability of difficult-to-filter wastes, particularly for "tarry wastes".[156] In the 1990 EPA contractor report on the TCLP, the authors noted that no liquids were expressed from the filter before the specified filtration time had elapsed, because the filter "greatly over predicts percent solids".[157] The filter "was therefore deemed inaccurate for oily wastes".[158]

Significantly, when utilizing soil column results not limited by the TCLP filter, EPA's contractor found that large proportions of the oily waste were released as primary leachate.[159] Moreover, when the TCLP filter was modified, the results more closely resembled the soil column tests.[160] Since the primary leachate accounts for a large proportion of the toxic contaminants released from the waste in the soil column tests, the lack of primary leachate generation from the oily wastes in the instant rulemaking is part of the problem, not an indication the TCLP is appropriate. (EDF, 00006)

Response: See responses to Comments 1.d and 7.g.

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[156] 57 FR 37294, 37296 (August 18, 1992); Evaluation and Modification of Method 1311 for Determining the Release Potential of Difficult-to-Filter Wastes, Vol. 1, RTI, April 1990, p. 23 (hereafter "EPA TCLP Report"). The TCLP inadequacy can be attributed at least in part to the clogging of the filter causing inadequate expression of contaminants in the primary leachate, inaccurate liquid/solid leaching ratios causing excess dilution of the secondary waste leachate, and inadequate expression of the oil phase in the secondary waste leachate.

[157] EPA TCLP Report at 79.

[158] Id. The TCLP was also found to be "imprecise when applied to oily wastes", since subsequent efforts to filter replicate samples of the oily wastes produced a range of 28-100% solids. Id. at 81.

[159] Id., Figure 5-3.

[160] Id., Figure 5-4.

Comment 7.d: Second, EPA assessed the TCLP's benzene leaching efficiency for various wastes. Unlike mere sample descriptions, leaching efficiency calculations provide a direct measurement of TCLP benzene performance for various wastes.[161] According to EPA, the average leaching efficiency for the 27 samples where such an efficiency could be calculated was 53%, and this efficiency "was fairly consistent" regardless of the oil content of the waste.[162]

Again, a closer look at the actual data proves the inadequacy of the TCLP on the oily wastes in this rulemaking. Of the 27 samples in EPA's data base, only six have both an oil/grease content greater than 1% and a benzene efficiency value. Five of the six samples are crude oil storage tank sludge. Three of these five samples have a benzene leaching efficiency of 32% or less, with the lowest at 15%.[163] (EDF, 00006)

Response: The commenter correctly notes that six samples have both an oil and grease content above 1 percent and a calculated benzene leaching efficiency (see Supplemental Listing Background Document). The average of these six values is 53 percent, identical to the average leaching efficiency of other samples representing reforming catalyst, hydrotreating catalyst, hydrorefining catalyst, sulfur sludge, unleaded tank sludge, and hydrotreating catalyst. Further, several samples with low oil and grease also have low benzene leaching efficiency, such as samples of unleaded gasoline tank sludge and hydrorefining catalyst. Therefore, EPA reiterates

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[161] EPA also resampled some wastes using the OWEP (a more appropriate procedure for oily wastes) to compare OWEP and TCLP leaching results. However, this comparison was performed for metals only, and cannot be generalized to other hazardous constituents in the waste. In addition, while EPA claims the OWEP and TCLP results are "consistent", the OWEP detection levels differ from the TCLP analyses by an order of magnitude or more in some cases, therefore meaningful comparisons cannot be made for the vast majority of the samples. See NODA Background Document at 14-22. Indeed, EPA's claim that the higher OWEP results are all within an order of magnitude of the TCLP results is simply a function of the different levels of detection. For example, the OWEP leaching value for crude oil storage tank sludge sample R8C-CS-01 is 0.89 mg/l, compared to the TCLP detection limit of 0.10 mg/l (already almost an order of magnitude difference). If the TCLP testing was performed to the same level of detection as the OWEP analysis (at least one order of magnitude lower), the TCLP results may indeed be orders of magnitude below the OWEP levels. Id. at 19. In fact, practically every TCLP result for chromium is the detection limit of 0.10 mg/l, and EPA incorrectly regards each of these results as "higher" than the corresponding OWEP result where the measured OWEP value is less than this concentration. Finally, the samples on which the OWEP results were performed had been collected as far back as October 1993, and EPA does not address whether comparing the results of the leaching procedures based upon samples with substantially different ages is consistent with established quality assurance/quality control sampling procedures. Id. at 14.

[162] NODA Background Document, Chapter 2, p. 3.

[163] Id. at 9.

its conclusion that oil and grease is not an accurate predictor of leaching efficiency based on the analyses presented in the NODA.

The commenter is concerned that only six of 27 samples have both a benzene efficiency value and an oil and grease content greater than 1 percent. EPA notes that only 11 samples had oil and grease content above 1 percent to begin with, and that benzene leaching efficiencies could not be calculated for the remaining five samples because benzene was not detected in both total and leachate, and such a calculation would be meaningless.

With respect to the commenter's footnote regarding the OWEP analyses, see earlier responses to Comments 3.a and 5.

Comment 7.e: Indeed, crude oil tank sludge and hydrotreating catalyst samples comprise the vast majority of <u>all</u> samples with both an oil/grease % and a benzene capture efficiency value. In the case of hydrotreating catalysts, where the oil/grease % recorded in the data base is well below 1%, the average benzene capture efficiency is 72.8%. In contrast, the average crude oil tank sludge benzene leaching efficiency is 43%. Therefore, the TCLP was substantially less effective in capturing benzene on the waste with the higher oil/grease content.[164] (EDF, 00006)

Response: The average calculated benzene efficiency levels cited by the commenter are correct. Additional waste-specific benzene efficiencies can be calculated for each of the wastes, as follows (only wastes with three or more leaching efficiency data points are presented):

<table>
<tr><td colspan="5" align="center">Table I.C.8. Benzene Leaching Efficiencies</td></tr>
<tr><td>Residual</td><td>Average Benzene Efficiency (%)</td><td>PRDB Mean Oil &Grease (%)*</td><td>PRDB Median Oil % Grease (%)</td><td>EPA Analytical Data Mean Oil & Grease (%)**</td></tr>
<tr><td>Hydrotreating catalyst</td><td>73</td><td>3.6</td><td>0.35</td><td>ND</td></tr>
<tr><td>Reforming catalyst</td><td>69</td><td>1.0</td><td>0.01</td><td>NA</td></tr>
<tr><td>Hydrorefining catalyst</td><td>49</td><td>12.5***</td><td>2.5</td><td>0.08</td></tr>
<tr><td>Crude oil tank sludge</td><td>43</td><td>34.3</td><td>2.5</td><td>20.6</td></tr>
<tr><td>Unleaded tank sludge</td><td>33</td><td>10.81</td><td>6.1</td><td>0.09</td></tr>
</table>

    *1995 Listing Background Document
    **1997 NODA Supplemental Background Document, assumed non-detect values were equal to the detection limit.
    ***Corrected value.

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[164] Similarly, the average TCLP benzene capture efficiency of the three samples of unleaded storage tank sludge, another waste where oil/grease content is relatively high, is 32.6%.

Table I.C.8 shows that the benzene leaching efficiency of an "oily" waste (crude oil tank sludge) can be either higher or lower than wastes that are less oily.[165] Further, Table I.C.9 shows that within a residual category there is variability and no clear pattern:

Table I.C.9. Crude Oil Tank Sludge Leaching Efficiency		
Sample Number	Benzene Efficiency	Oil and Grease (%)
R19-CS-01	97	14
R6B-CS-01	49	24
R4B-CS-01	32	15
R10-CS-01	22	41
R8C-CS-01	15	25
R22-CS-01	NA, benzene not detected	4.87

EPA continues to conclude that leaching efficiency is not well correlated to oil and grease content. Even if the TCLP was less effective for oily samples of crude oil tank sediment, this is essentially no longer relevant because EPA has decided to list this waste as hazardous.

Comment 7.f: Furthermore, the actual benzene leaching efficiencies used in EPA's groundwater modeling, where the values matter the most, are much lower than 53% for most of the oily sludges EPA does not propose to list as hazardous in this rulemaking. In the case of crude oil storage tank sludge, the modeled TCLP benzene leaching efficiency was 23% for the median value, and 15% for the high-end value.[166] Similarly, when EPA modeled the landfilling of HF alkylation sludge, the TCLP benzene leaching efficiency was 35% and 25% for the median and high-end values respectively.[167] And the benzene leaching efficiency for unleaded storage tank

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[165]EPA disagrees with the commenter's footnoted characterization of unleaded storage tank sludge as having a relatively high oil and grease level.   As presented in the March 1997 Supplemental Listing Background Document, the oil and grease content of two of the three samples are well below 1 percent (the oil and grease content of the third sample could not be determined due to insufficient sample volume). The oil and grease contents of the unleaded sludge record samples are lower than the oil and grease content of these sludges reported in the RCRA §3007 survey. This discrepancy is irrelevant, however, in calculating benzene leaching efficiencies for comparison to other wastes because only the record samples were analyzed for total, TCLP, and oil and grease levels.

[166] NODA Groundwater Risk Assessment, Table C.8.

[167] Id., Table C.29.

sludge is 28% for the median value.[168] Accordingly, TCLP benzene capture efficiencies used in the groundwater modeling for these oily sludges are roughly one-half of EPA's so-called average value, and approximately one-third of the hydrotreating catalyst with lower oil/grease content.

KGS modeled various refinery wastes using all EPA value inputs, except the TCLP values were adjusted to reflect EPA's purported average benzene leaching efficiency. In the case of crude oil storage tank sludge, the TCLP concentration mean input value increased from 0.678 mg/l to 1.56 mg/l, and the resulting groundwater exposure pathway risk posed by landfilling this waste almost doubled from the 2.7×10^{-5} risk computed by EPA to the KGS result of 4.8×10^{-5}.

In the case of HF alkylation sludge (offsite), the TCLP value was a high-end parameter in EPA's deterministic modeling, consequently modifying the TCLP value to reflect the purported 53% average benzene leaching efficiency more than doubled the resulting risk. The TCLP value increased from 0.18 mg/l to 0.371 mg/l, and the resulting groundwater use risk increased from 6.4×10^{-6} to 1.5×10^{-5}. Therefore, had EPA's TCLP values for oily wastes actually reflected the benzene leaching efficiency EPA claims, the modeling results would be substantially different. (EDF, 00006)

Response: The commenter compared the TCLP and total benzene concentrations corresponding to 50th percentile and 90th percentile concentrations to calculate "actual" efficiencies. EPA disagrees with this assessment and believes that these are not actual efficiencies at all because they are not necessarily taken from the same sample. Such 50th and 90th percentile values were used in conducting 2 parameter high end modeling scenarios for both the NODA and the 1995 proposal. For the Monte Carlo analysis, however, paired total and TCLP values were used (see page 4-2 of the Groundwater Supplemental Background Document) and therefore the comment is not applicable to the Monte Carlo results.† EPA disagrees with the commenter's decision to change the concentration input data for the ground water modeling analyses. EPA does not observe a strong correlation between "oiliness" and extraction efficiency as discussed above. EPA fundamentally disagrees that the TCLP results are inappropriate for *these* residuals. EPA conducted record sampling to measure total and leachate values for use as risk assessment inputs and believes these values to be appropriate and representative of the population of wastes. The commenter does not provide a compelling reason to increase the leachate value used as the ground water modeling input. To incorporate the commenter's suggestion to maintain leaching efficiency as constant, EPA could increase the leachate concentration (as suggested by the commenter), or even decrease the total concentration while holding the measured leachate value constant. EPA does not see the benefit of either approach in assessing risks.

EPA also wishes to again correct the commenter's suggestion that unleaded gasoline tank sludge is an "oily" waste. As presented in the Supplemental Background Document, the oil and grease

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[168] NODA Groundwater Risk Assessment, Table D.5. No high-end value is provided because EPA did not conduct a sensitivity analysis of the waste. Instead, EPA assumed waste unit area and distance to the well were the most sensitive parameters in the modeling.

content of two of the three samples are well below 1 percent (the oil and grease content of the third sample was not reported).   This waste demonstrates a case where somewhat lower benzene leaching efficiencies are observed in wastes with low oil and grease contents.

Comment 7.g: More fundamentally, however, because of the lack of primary leachate generation using the TCLP, even these benzene leaching efficiencies understate the magnitude of the TCLP shortcomings in the instant rulemaking. Where primary leachate is generated, it is not subject to the 20-1 dilution factor that EPA used to calculate its "benzene leaching efficiency" values. Therefore, when EPA's contractor used a modified filter that would allow for primary leachate generation of oily wastes, 20.3% of the benzene in slop oil emulsion was extracted into the leachate. Only 0.8% of the benzene in the waste was recovered by the contractor in the leachate using the TCLP.[169] For comparison purposes, EPA's benzene recovery percentages (derived by dividing the TCLP value by the waste benzene concentration without applying the 20-1 dilution factor) in the instant rulemaking (using median values) are 1.2% for crude oil tank sludge, 1.8% for HF alkylation sludge, and 1.5% for unleaded storage tank sludge. Accordingly, the TCLP may understate the leachable benzene concentrations by a factor of 10 or more, an understatement not fully reflected by merely adjusting the TCLP benzene leaching efficiency to 53% after applying the 20-1 dilution factor. (EDF, 00006)

Response: For samples with >0.5% solids content, the commenter is incorrect in stating that "where primary leachate is generated, it is not subject to the 20-1 dilution factor that EPA used to calculate its "benzene leaching efficiency" values." Conversely, the concentration detected in the primary leachate or initial filtrate, assuming initial filtrate and leachate are not compatible and require separate analyses, are combined mathematically based on the following calculation as presented in Method 1311:

$$\textit{Final Analyte Concentration (mg/L)} \ = \ \frac{(V_1) \ (C_1) \ + \ (V_2) \ (C_2)}{V_1 \ + \ V_2}$$

where:

V_1 =   The volume of the first phase filtrate (L)
C_1 =   The concentration of the analyte in the first phase filtrate (mg/L)
V_2 =   The volume of the second phase leachate (L)
C_2 =   The concentration of the analyte in the second phase leachate (mg/L).

Using this equation it is apparent that any analyte concentration detected in the initial filtrate will be diluted by some factor depending of the sample solids content. For example, suppose a waste was found to contain a 40% liquid phase or 0.01 L of filtrate based on a 25 gram sample used for

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[169] EPA TCLP Report, Table 5-23.

the zero headspace extraction (ZHE) procedure. The solid phase or 15 grams would require 0.3 L of extraction fluid based on the method required 20 to 1 ratio of fluid to sample weight. Assuming a worst case scenario in which benzene is detected almost entirely in the filtrate at a concentration that resembles the total concentration of 100 m/L, the maximum theoretical solid phase leachate concentration would be 5 mg/L. Using these values with the equation listed above, the final leachate concentration would be 8.1 mg/L which represents a 12-fold dilution from the filtrate only concentration. Since the final leachate concentration for non-compatible phases is a combination of the initial filtrate and solid phase leachate concentrations, the amount of filtrate concentration dilution is directly proportional to the sample solids content. A higher solids content results in a greater filtrate concentration dilution. Only in situations where the sample % solids content was <0.5%, the filtrate volume would be considered the leachate and the detected concentration would represent the final leachate concentration.

As stated previously, the Agency does not believe the wastes described in the RTI document are comparable to the petroleum refining residuals. Furthermore, it should be noted that the leachate recoveries of benzene in the slop oil emulsion as presented in the RTI report are calculated differently since the values were calculated from the total concentration without considering the extraction fluid dilution.

Comment 7.h: Significantly, EPA did not even conduct comparable TCLP efficiency analyses for PAHs. The data available indicates if it had, the results would be even more dramatic. For many oily wastes, high concentrations of PAHs were detected in the waste, but none were detected in the TCLP extract.

For example, in the case of CSO sludge, even though waste samples contained up to 390 ppm benz(a)anthracene, 230 ppm benzo(a)pyrene, and 860 ppm chrysene, the TCLP values used for groundwater modeling purposes were non-detect.[170] Similar TCLP non-detects were observed for crude oil tank sludge, notwithstanding total concentrations of benz(a)anthracene of up to 49.5 ppm. Off-spec thermal product and fines is the underline only waste where PAHs were detected in the TCLP extract. For the sample containing the highest totals contaminant concentration (28 ppm), the TCLP benz(a)anthracene leaching efficiency (using the 20-1 dilution factor) is less than 4%.[171]

This phenomenon of measuring PAH non-detects in TCLP samples, but measuring substantial PAH leaching when using improved leaching procedures, is exactly what EPA's contractor

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[170] NODA Groundwater Risk Assessment, Table A.4; EPA 1995 Listing Background Document, Table 3.1.18.

[171] NODA Groundwater Risk Assessment, Table A.4. See also Table A.6 on benzo(a)pyrene, where the TCLP leaching efficiency for the waste containing the highest totals concentration (33 ppm) is 3%.

observed when assessing the shortcomings of the TCLP in 1990.[172] Indeed, the EPA contractor measured the leachability of semivolatiles (including PAHs) using the modified filter, and found them roughly comparable to the "real world" soil column testing results. Moreover, in those instances where the TCLP allowed primary waste generation, the TCLP results closely resembled the modified filter values.[173]

Using the modified filter, slop oil emulsion leached 4.06 mg/l benzo(a)anthracene from waste containing an average 20.5 ppm of the PAH. Similarly, 4.21 mg/l chrysene leached from the same waste containing an average 33.3 ppm of the toxic chemical.[174] Accordingly, the leachate recovered between 12.5-20% of these chemicals found in the wastes.[175] For comparison purposes, in the case of the one waste (off-spec products and fines) where PAHs were detected using the TCLP in the instant rulemaking, the benz(a)anthracene and benzo(a)pyrene leachate recovery rates were approximately 0.1% for the samples containing the highest concentration of these contaminants.[176] With respect to the other oily wastes in this rulemaking, notwithstanding greater concentrations of these and other contaminants, the leachate contaminant recovery rate was essentially zero.

KGS used EPA's groundwater model to compute the PAH leachate value necessary to produce a 1×10^{-4} risk level in the receptor well for CSO sludge and off-spec products and fines. Using modified input values for volumes (for the reasons discussed in Sections II.D and II.E), and locating the well in the plume centerline (as discussed in Section II.F), KGS determined leachate values of only 0.017 mg/l benz(a)anthracene or 0.026 mg/l of chrysene would yield the 10^{-4} risk at the receptor well for the CSO sludge onsite scenario. These leachate values represent 0.008% of the benz(a)anthracene and 0.005% of the chrysene in CSO sludge, when compared to average CSO sample results.[177] The PAH leaching efficiency analogous to EPA's benzene leaching efficiency calculation (i.e., applying the 20-1 dilution factor) necessary to produce these leachate values are 0.17% (benz(a)anthracene) and 0.11% (chrysene), again compared to average waste concentrations.

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[172] EPA TCLP Report, Table 5-30. Using the modified filter, 2.97 mg/l benzo(a)pyrene was measured in the leachate of slop oil emulsion containing an average 19.0 ppm of the contaminant, while using the TCLP none of the contaminant was detected.

[173] Id. at 117.

[174] Id., Tables 5-4, 5-30.

[175] The benzo(a)pyrene leachate recovery rate was 15.6%.

[176] NODA Groundwater Risk Assessment, Tables A.4, A.6.

[177] 1995 Listing Background Document, Table 3.1.18.

Similarly, KGS determined leachate values of 0.0115 mg/l benz(a)anthracene or 0.125 mg/l chrysene would produce a 10^{-4} risk at the receptor well for the off-spec product and fines offsite scenario. These leachate values represent 0.01% of the benz(a)anthracene and 0.5% of the chrysene in the waste, when compared to average off-spec product and fines sample results.[178] The leaching efficiency necessary to produce these leachate values (applying the 20-1 dilution factor) is 1.9% for benz(a)anthracene and 10.4% for chrysene.

Significantly, these leachate values corresponding to the 10^{-4} risk assume only that individual constituent will reach the receptor well. Since benz(a)anthracene and chrysene can be expected to travel at approximately the same velocity, even lower leachate values would produce the same unacceptable risk for the two chemicals combined.

Therefore, in the case of CSO sludge and off-spec products and fines, high groundwater risks will result from extremely small fractions of the PAH contaminants leaching from the landfill, without even accounting for facilitated transport associated with refinery sites as discussed in Section II.C below. On the basis of EPA's previous positions regarding the efficacy of the TCLP on oily wastes, EPA's contractor report demonstrating the poor performance of the TCLP on PAHs in oily wastes, EPA's data indicating CSO and off-spec products and fines contain high percentages of oil and grease,[179] and the potential for CSO sludge to contain free liquid,[180] it is more than plausible that these wastes will leach PAHs at the extremely small concentrations which pose a substantial risk to human health and the environment. (EDF, 00006)

Response: EPA notes that it is not possible to calculate extraction efficiencies for the PAHs because the TCLP results were generally non-detects for these compounds. This was not surprising considering the highly hydrophobic nature of these compounds and as illustrated below for the CSO sludge PAHs highlighted by the commenter:

PAH	Solubility (mg/L)*
Benz(a)anthracene	0.0128

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[178] 1995 Listing Background Document, Table 3.7.6.

[179] All the CSO sludge samples in EPA's database contain greater than 5% oil, one sample contains 20% oil, and another contains 70% oil ("oil seeped out of this sample if squeezed"). See NODA Background Document, Chapter 2, Table 1; Raw Data Used in Support of Appendix A. Similarly, two of four off-spec product and fines samples contained approximately 10% oil/grease or more. See Raw Data Used in Support of Appendix A.

[180] The Agency's assertion that CSO sludge would not contain free liquids is completely at odds with one sample description, and in two other cases, the samples were either "air dried" or mixed with CKD, therefore they do not represent the waste as generated or as plausibly mismanaged. See NODA Background Document, Chapter 2, Table 1.

| Benzo(a)pyrene | 0.00194 |
| Chrysene | 0.00194 |

* Solubilities as reference in the technical background document, *Nongroundwater Risk Assessment for Petroleum Refining Waste Listing Decision: Uncertainty Analysis.* 1997.

Significant risk for these compounds, however, was identified for the indirect exposure pathways, largely due to the same hydrophobic properties that minimize risk from a groundwater perspective.

Further, EPA notes that the PAH leachate levels associated with the coke fines are highly problematic as previously discussed in the April 1997 NODA at 62 FR 16751.

EPA notes an error in the comment. Of the four off-spec product and fines record samples analyzed for oil and grease content, three of the samples had oil and grease contents less than 1 percent and the fourth had a level of 8.4 percent. See NODA Supplemental Listing Background Document, Chapter 2, Table 1. This is in contrast to the commenter's claim that 2 of 4 such samples have oil and grease contents of 10 percent.

EPA refers the reader to the response to Comment 1 regarding the validity of the commenter's concern about free liquids in the CSO samples and the validity of the Agency's sampling protocols.

EPA disagrees that the commenter's purely hypothetical analysis indicates that PAHs present a substantial risk via leaching to groundwater. Based on the existing TCLP data, as well as the demonstrated insolubility of PAHs in water, EPA does not believe such an analysis is correct.

Comment 8: TCLP Fails to Account for Co-disposal with Solvents or Oils

Comment 8.a: In addition to the problems associated with the procedure itself, the TCLP is inappropriate because it measures only the dissolved phase flow of contaminants into the groundwater, and thus fails to consider the cosolvency effects of oil and other compounds in the landfill. In the previous petroleum refinery listing determination, the Agency expressly acknowledged the cosolvency effects associated with the solvents and oils present in petroleum refinery land disposal facilities.

> However, evidence exists that [PAHs] may move more rapidly in soil with low organic content or if codisposed with other solvents or oils. Because wastes listed today typically contain high concentrations of oils, the mobility of the PAHs is expected to be quite high due to cosolvent effects. Therefore, it is reasonable to

assume that wastes containing these constituents could pose a significant threat to human health via groundwater contamination if mismanaged.[181]

Response: EPA notes that the FRN passage quoted by the commenter reflects the Agency's views at that time regarding co-solvency effects <u>within</u> the F037/F038 sludges, as opposed to any co-solvent effects due to co-disposal with other mobilizing wastes. Further, the 1990 sludge listing was based primarily on the results of totals analysis; the TCLP analyses that were conducted in support of this effort were completed prior to the 1988 NODA for the sludge listings. The TCLP itself was not promulgated until March of 1990. Thus, the TCLP was essentially still a developmental method during the data collection phase of the sludge listing determination. In retrospect, if the TCLP had been available as an established method during data development, EPA could have argued that co-solvent effects within the wastes were measured by the TCLP. In fact, the 1990 RTI report concluded that API separator sludge, which is quite similar to F037/38, did not exhibit TCLP filtration problems:

> "Method 1311 did, however accurately estimate the mobile and immobile fractions of API separator sludge determined in the column experiments, suggesting that it is suitable as written for oily wastes that are not difficult-to-filter." p. 79.

Note: the commenter characterized the TCLP as measuring "only the dissolved phase flow of contaminants". As discussed further in response to Comment 7.g, the method also assesses any filtrate generated during the initial filtration step (issues of oily waste non-filterability aside), regardless of whether the filtrate is aqueous or nonaqueous.

Comment 8.b: Notwithstanding this express finding of co-disposal potential in refinery waste landfills made by the Agency in 1990, in the NODA materials the Agency minimizes the prospect of co-disposal with oil and solvents solely on the basis of oil and grease content data on selected waste samples. As discussed in this portion of the comments, the survey results are extremely incomplete and therefore provide an insufficient basis for overriding the express finding made in 1990.

First, oil and grease data were reported on only 48 of 127 residuals disposed in onsite landfills, and only 120 of 621 residuals disposed in offsite landfills, thereby accounting for only 22% of all landfilled residuals in EPA's data base.[182] Perhaps more importantly, <u>very few of the reported values involved the oily wastes in this rulemaking</u>.

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[181] 55 FR 46369 (November 2, 1990) (emphasis added).

[182] See Raw Data Used in Developing Statistics in Appendix A of the March 1997 Supplemental Background Document for Listing Support Analyses (hereafter "EPA Oil/Grease Landfill Data"), provided by EPA staff upon request, p. 1.

For onsite landfilling, no CSO sludge, and only two unleaded gasoline storage tank and one HF alkylation sludge values are included in the data base.[183] In contrast, 12 samples of treating clay from alkylation were included in the data base.

For offsite landfilling, only one crude oil tank sludge and unleaded storage tank sludge sample, five CSO sludge samples, and three HF alkylation sludge samples are included in the data base.[184] In contrast, EPA's data base contains 20 samples of off-spec product from sulfur complex and H_2S, and 14 samples of sulfur removal sludge.

Therefore, median or average values computed by EPA are virtually meaningless since the data base largely reflects nonoily wastes, and there is no rationale for the large variation in number of samples included for each waste type (other than availability of data). (EDF, 00006)

Response: EPA disagrees that the median oil and grease data presented in the NODA are meaningless. The NODA Supplemental Background Document for Listing Support Analyses summarizes the available oil and grease data for all wastes that were managed in nonhazardous landfills in 1992. For onsite landfills, the median oil and grease content of disposed wastes was less than 1 percent. For offsite landfills, the median oil and grease content of disposed wastes was approximately 1 percent. The commenter subsequently requested (and received) the supporting data for this summary, which itemizes the oil and grease content of each of the approximately 170 (120 offsite and 48 onsite) wastes used in the calculations. As discussed below, EPA believes that the data are representative of both oily and nonoily wastes, contrary to the commenter's conclusions.

The oil and grease data cited by the commenter are based on the results of the RCRA §3007 survey. Respondents were required to submit available data characterizing each residual and, if necessary, to use judgement in estimating properties (respondents were not required to conduct testing specifically for the survey). Therefore, the quality of the oil and grease data submitted, as well as other waste characterization data, was wholly dependent on the availability of data at each refinery. Given this situation, wide variability in the data base representativeness is expected, with oil and grease data availability much greater for some waste categories than for others. As the commenter correctly points out, approximately 22 percent of all the landfilled residuals had oil and grease data. These data are summarized in Table I.C.10. As explained further below, some landfilled wastes have relatively more oil and grease data than others.

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[183] Id., Table 2.

[184] Id., Table 2.

Table I.C.10. Oil and Grease Content of Refinery Wastes Disposed in Onsite and Offsite Subtitle D Landfills in 1992			
Residual	Number of Values	50th percentile, %	Maximum, %
Desalting Sludge	5	10	20
Residual Oil Tank Sludge	3	28*	99*
Treating Clay from Clay Filtering	18	1	23
Treating Clay from Isomerization/ Extraction	5	1	5
Hydrocracking Catalyst	2	5	9
Process Sludge from Residual Upgrading	1	5	5
Off-Spec Sulfur	23	0	13
Reforming Catalyst	1	1	1
Polymerization Catalyst	5	0	2
Treating Clay from Alkylation	17	0	1
CSO Sludge	5	5	20
Crude Oil Tank Sludge	1	80*	80*
Hydrotreating Catalyst	7	9	9
Sulfur Sludge	16	0	5
Off-spec Product and Fines from Thermal Processes	4	6	14
Unleaded Gasoline Tank Sludge	3	5	5
Hydrorefining Catalyst	1	9	9
FCC Catalyst	17	0	1
FCC Fines	19	0	1
Sulfur Complex Catalyst (Claus)	11	0	2
Sulfur Complex Catalyst (SCOT)	2	0	0
HF Alkylation Sludge	4	1	3

The commenter calls out nine specific residuals: crude oil tank sediment, CSO sediment, unleaded storage tank sludge, residual oil tank sludge, HF alkylation sludge, off-spec product from sulfur processes, sulfur removal sludge, treating clay from clay filtering, and treating clay from alkylation. The commenter notes that the number of wastes with oil and grease data varies widely. However, the number of wastes landfilled also varies widely. Comparing the number of

wastes with oil and grease data to the number of wastes landfilled allows for a more appropriate and normalized comparison. Such "coverage" of oil and grease data for these specific wastes are as follows (sources of all data are the "EPA Oil/Grease Landfill Data" cited by the commenter, the 1995 Listing Background Document, and the 1996 Study; all data are for 1992):

> • For HF alkylation sludge, 3 of 8 wastes disposed offsite and 1 of 1 waste disposed onsite reported oil and grease content (overall coverage is 4 of 9 wastes, or 44 percent).

> • For off-spec product from sulfur processes, 20 of 51 wastes disposed offsite and 3 of 3 wastes disposed onsite reported oil and grease content (overall coverage is 23 of 54 wastes, or 43 percent).

> • For treating clay from alkylation, 5 of 30 wastes disposed offsite and 12 of 18 wastes disposed onsite reported oil and grease content (overall coverage is 17 of 48 wastes, or 35 percent).

> • For CSO sediment, 5 of 16 wastes disposed offsite and 0 of 2 wastes disposed onsite had oil and grease content (overall coverage is 5 of 18 wastes, or 28 percent).

> • For treating clay from clay filtering, 14 of 91 wastes disposed offsite and 4 of 15 wastes disposed onsite reported oil and grease content (overall coverage is 18 of 106 wastes, or 17 percent).

> • For residual oil tank sludge, 3 of 17 wastes disposed offsite and 0 of 3 wastes disposed onsite had oil and grease content (overall coverage is 3 of 20 wastes, or 15 percent).

> • For sulfur removal sludge, 14 of 110 wastes disposed offsite and 2 of 18 wastes disposed onsite reported oil and grease content (overall coverage is 16 of 128 wastes, or 13 percent).

> • For unleaded storage tank sludge, 1 of 35 wastes disposed offsite and 2 of 3 wastes disposed onsite reported oil and grease content (overall coverage is 3 of 38 wastes, or 8 percent).

> • For crude oil tank sediment, 1 of 25 wastes (4 percent) disposed offsite had oil and grease contents (no wastes were disposed onsite).

The above comparison shows that, for the nine wastes mentioned by the commenter, four are "over-represented" in the data base while five are "under-represented," i.e., the actual number of landfilled samples with oil and grease content is higher ("over-represented") or lower ("under-represented") than 22 percent. Among the wastes with a higher than average number of landfilled samples with oil and grease data are HF alkylation sludge and CSO sediment, two of the four wastes claimed by the commenter to be lacking in data.

It is important to note that the commenter's evaluation differentiates between data reported for on-site and off-site landfilling; EPA's analysis presented above is based on the premise that data from these 2 subsets can be combined. Thus, while the commenter is concerned that oil and grease data for CSO sludge landfilled on-site is lacking (only 2 such sludges were reported to be landfilled on-site in 1992), overall, EPA's analysis shows that the industry reported O&G data for 28 percent of wastes landfilled (regardless of whether the management occurred on- or off-site).

The primary reason EPA believes that there are more oil and grease data for sulfur removal sludge than for HF alkylation sludge (for example) is, in part, simply that a greater number of sulfur removal sludge wastes than HF alkylation sludge wastes are landfilled. EPA acknowledges that some wastes are not well represented in the database (such as crude oil tank sediment and unleaded tank sediment), but believes that others of concern to the commenter are fairly well represented (such as treating clay from clay filtering) and others are very well represented (such as HF alkylation sludge) when the more appropriate normalized approach presented above is used.

Comment 8.c: Indeed, many of the wastes codisposed in highest volumes onsite (CSO sludge, HF alkylation sludge, treating clay from clay filtering) are the wastes with substantial oil content.[185] Similarly, in the case of offsite landfilling, the higher volume wastes contained a substantial percentage of oil (CSO sludge, HF alkylation sludge, residual oil sludge).[186] (EDF, 00006)

Response: As discussed above in response to Comment 8.b, the Agency's normalized assessment of the availability of O&G data is helpful in evaluating and alleviating the commenter's concerns. For CSO sediment, HF alkylation sludge, and treating clay from clay filtering, the "coverage" of oil and grease data in the data base is 28, 44, and 17 percent, respectively, which compares favorably with the overall coverage of 22 percent described in the NODA. The median oil and grease content of these wastes when landfilled was 5, 1, and 1 percent, respectively (as discussed in response to comment 8.b above). EPA disagrees that these results should be characterized as demonstrating "substantial oil content".

Overall, as previously argued by EPA in Section 2.2 of the March 1997 Supplemental Background Document, the residuals of concern infrequently exhibit oil and grease contents above 10 percent. EPA identified 13 wastes disposed in offsite Subtitle D landfills with oil and grease contents equal to or greater than 10 percent, and 1 waste disposed in an onsite Subtitle D landfill with an oil and grease content equal to or greater than 10 percent (in the Supplemental Listing Background Document to the NODA, EPA presented its evidence suggesting that 2 of these 14 residuals have actual oil and grease levels lower than reported). The median oil and grease content for residuals landfilled either off- or on-site was equal to or less than 1 percent. EPA believes, <u>especially within the context of the commenter's concern regarding co-disposal,</u>

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[185] NODA Groundwater Risk Assessment, Table D.2.

[186] NODA Groundwater Risk Assessment, Table D.5.

that these low median values demonstrate little potential for enhanced toxicant mobility. Furthermore, the Agency notes that higher levels of oil and grease, such as the 14 residuals landfilled with oil and grease levels equal to or greater than 10 percent do not necessarily correspond to high levels of free oil, as discussed at length earlier in this section.

As a means of addressing the commenter's concern regarding high volumes of oily wastes in the co-disposal scenario, EPA calculated a hypothetical volume-weighted oil and grease value. Specifically, if all 120 residuals managed in nonhazardous offsite landfills in 1992 with oil and grease data were disposed in a single landfill, the overall volume-weighted oil and grease level would be 4 percent. If two wastes were removed (i.e., the wastes with oil and grease values of 80% and 99% which the NODA background document state are not indicative of landfilled wastes), the overall volume-weighted oil and grease level decreases to 3 percent. This compares favorably to the overall median value (given in the NODA Background Document) of approximately 1 percent. The results for the onsite analysis are similar. If all residuals with oil and grease data managed in nonhazardous onsite landfills were disposed in a single landfill, the overall volume-weighted oil and grease level would be much less than one percent. If one waste was removed (i.e., a single low-oil waste generated in a very large volume), the overall volume-weighted oil and grease level increases to just 0.5 percent, which compares favorably to the overall median value given in the NODA Background Document, of less than 1 percent. EPA concludes that the higher volume wastes do not contain a substantial percentage of oil, as claimed by the commenter, because the above analysis shows similar results using a volume-weighted approach and the original, overall median approach.

Comment 8.d: For example, the third highest volume waste codisposed offsite - residual oil sludge - consistently exhibited oil content much greater than 10% in EPA's data base, ranging in oil content up to 99%.[187] The highest volume codisposed waste - HF alkylation sludge - has a mean oil/grease content of 2.68%, and a 90% value of 5% oil and grease.[188] EPA's sampling descriptions of HF alkylation sludge typically indicated the waste has a liquid phase or high liquid content.

Response: First, EPA reiterates that the 99 percent value of oil and grease cited by the commenter for residual oil tank sludge is incorrect for the reasons stated in Appendix A of the Supplemental Listing Background Document. EPA contends that it is completely inappropriate to assume that the 99 percent value represents the true oil content of a material that could be landfilled. For the residual in question, the survey clearly reports that this residual underwent de-oiling prior to landfilling. The facility did not estimate the oil and grease content of de-oiled sludge for this stream. The facility did estimate the liquid and solid quantities of material generated from the process. Specifically, the facility estimated the volume reduction at 2 percent (which is much lower than for other facilities and shows that the level of oil and grease following

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[187]   EPA Oil/Grease Landfill Data, Table 1.

[188]   1995 Listing Background Document at 135.

"de-oiling" is approximately the same as the level prior to de-oiling). EPA believes that the low level of free oil recovered shows that very little free oil is left in the waste to migrate. The low quantity of oil recovered shows that the material did not have much free oil to begin with. In regard to the high levels of reported oil and grease, EPA believes that this is due to the nature of the matrix (i.e., residual oil tank sludge). Because this waste is generated from the settling of heavy oil, it will generally consist of heavy hydrocarbons. Note that residual oil itself is very "heavy" and typically must be warmed to be pumped. The material settling from these tanks, therefore, can be expected to be very high molecular weight and solid.

De-oiling does produce wastes with varying oil contents; see, for example, the oil and grease content of record samples presented in the Supplemental Listing Background Document, where the oil and grease content of de-oiled crude oil tank sludge ranges from 5 to 15 percent. By definition, however, the oil content of a de-oiled sludge would be less than its oil content prior to that step. Therefore, for the wastes referenced by EPA in the Supplemental Background Document as representative of "pre-" de-oiling, the de-oiling process produces a volume-reduced sludge and recovered oil; the oil content of the volume-reduced sludge is obviously less than that in the starting material. Regardless of the frequency that de-oiling is used in the industry (see below), EPA knows that the wastes in question by the commenter did, in fact, undergo de-oiling.

Second, with respect to the HF alkylation sludge oil and grease data, the commenter is using data representing <u>all</u> HF sludges, rather than the more appropriate subset of sludges reported to be landfilled. As shown in the previous table, the median and maximum oil and grease values for landfilled HF sludges are 1 and 3 percent, respectively. These lower values reflect the common sense observations made earlier in this section that high levels of hydrocarbons are often reclaimed rather than disposed by the refining industry, and perhaps more importantly, refineries generally do not landfill materials with a measurable free oil or water content due to restrictions on liquids in landfills.

Third, while EPA's sampling description of HF alkylation sludge did indicate the presence of a liquid phase, three of the five samples were collected from the settling portion of the HF neutralization tanks. Standard refinery operating practice for such sludges destined for landfilling is centrifugation or filter press[189]. However, due to scheduling constraints EPA was not able to obtain these samples after dewatering. Thus the use of these samples as input to the groundwater model is conservative. With respect to the remaining two HF sludge samples, EPA collected these samples from the storage drums (no dewatering occurred) for R9-HS-01 (31 percent oil and grease) and after centrifugation for R7C-HS-01 (0.08 percent oil and grease). The refineries reported in their §3007 surveys that these residuals were destined for a Subtitle C landfill and a Subtitle D landfill, respectively, when generated in 1992, although it is unclear whether these practices were also used for the sludge samples collected in May 1994 and October 1994, respectively.

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[189]Just under half the refineries reported dewatering prior to Subtitle C landfilling and just over half reported dewatering for Subtitle D landfilling.

Comment 8.e: EPA attempts to minimize the importance of the high oil percentages found in the oily sludge samples by noting in some cases they "appear" to represent oil levels prior to deoiling, thereby suggesting the wastes actually landfilled would have a substantially lower oil content.[190] However, as the Agency previously noted, there is a wide range of effectiveness of deoiling techniques between refineries, and samples of deoiled wastes could contain more oil than as generated wastes of the same type. Largely as a result of these observations, the Agency determined that differentiating between oily and de-oiled sediments was inappropriate.[191] Further, only about 30% of crude oil storage tank sludge is even deoiled prior to final management.[192] No comparable percentages are provided for other wastes. (EDF, 00006)

Response: With respect to data regarding the prevalence of deoiling, EPA has previously published a report ("Waste Minimization in the Petroleum Refining Industry," Docket F-95-PRLP-S0064) and through the EPA's Internet server) specifically discussing pollution prevention practices of all residuals and wishes to correct the commenter's statement that de-oiling frequency is only provided for crude oil tank sediment. This waste minimization report presents volume reduction techniques on a waste-by-waste basis and states that based on the RCRA §3007 survey responses, residual oil tank sludge, crude oil tank sludge, and CSO tank sludge undergo volume reduction processes (such as de-oiling) in approximately 23-32 percent of the cases, depending on the specific waste.

Comment 8.f: Accordingly, contrary to EPA's assertions, the extremely limited data in the NODA indicate the co-disposal of oily wastes in refinery waste landfills is an actual or potential mismanagement scenario, consistent with the Agency's 1990 finding. Furthermore, since the NODA data base is limited to wastes landfilled in 1992 that are covered by this rulemaking or included in an accompanying study, it does not account for other wastes codisposed in refinery waste landfills.

These other wastes fall into several categories. The first category consists of wastes previously landfilled that are now hazardous wastes, including the very F037-F038 wastes EPA found to create cosolvent effects in 1990. Many refinery waste landfills, both onsite and offsite, operate for decades and contain numerous waste streams now considered hazardous (including solvents and oily wastes) that can facilitate the transport of contaminants in the wastes covered by the instant rulemaking. Indeed, F037- F038 wastes were legally disposed in nonhazardous refinery waste landfills until May 2, 1991, just prior to the 1992 calendar year upon which EPA's data in the instant rulemaking are based. Accordingly, if in 1990 EPA based its regulatory decision

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[190] NODA Supplemental Background Document, Appendix A.

[191] 1995 Listing Background Document at 29, 32, 46.

[192] 60 FR 57782 (November 20, 1995).

making on cosolvency for F037-F038 wastes, there is no reason to depart from that approach in the instant rulemaking because the <u>same</u> wastes are still present in the <u>same</u> landfills.[193]

The second category of other wastes includes nonhazardous wastes that are landfilled but not covered by the instant rulemaking or the accompanying study. These other wastes represent a substantial portion of the waste codisposed with the wastes at issue in this rulemaking.

As illustrated in EDF's comments on the 1995 proposal, refinery waste landfills are used for the broad array of nonhazardous wastes generated at refinery sites, including sludges and other wastes with a substantial oil component.[194] For example, API reports that over 1.6 million metric tons of refinery wastes were landfilled in 1992, most of which is nonhazardous waste.[195]

The highest volume refinery waste landfilled in 1992, according to API, was contaminated soils/solids, accounting for almost 600,000 tons of landfilled waste.[196] This category includes "soil (crude) contaminated" and "soil (product) contaminated".[197] Given the extensive environmental contamination at petroleum refineries from a combination of crude oil spills, refined product spills, and improper waste management, as previously documented by EPA and further described elsewhere in these comments, much of this contaminated soil and solids can be expected to contain a substantial amount of oil.[198] An additional 20,000 tons of "other contaminated soils"

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[193] Historic wastes facilitate free-phased flow by taking up a substantial portion of the oil-retention capacity of the landfill, thereby making it more likely that free-phased NAPL from the oily wastes covered by this rulemaking will exit the unit. EPA's analyses to date have not considered the impact of historic wastes on the potential for NAPL formation. See EDF's March 1996 Comments at 22-28.

[194] EDF March 21, 1996 Comments, pp. 17-19.

[195] Generation and Management of Residual Materials 1992-1993, API, February 1995, p. B-18 (hereafter "API 1992-93 Residual Report").

[196] <u>Id</u>.

[197] Generation and Management of Residual Materials 1991, API, May 1994, Appendix A (hereafter "1991 API Residuals Report").

[198] 55 FR 46369 (November 2, 1990). Oil-contaminated soils may be managed as nonhazardous wastes because of the ineffectiveness of the TCLP on oily wastes, the exemption from the toxicity characteristic for petroleum-contaminated media generated from underground tank cleanups (40 CFR 261.4(b)(10)), and the discretion inherent in the implementation of the contained-in rule.

which includes "oil spill debris, waste oil absorbents, and tank bottom absorbents," were also landfilled in 1992.[199]

Other landfilled wastes containing oil and possibly solvents include "other oily sludges and organic wastes". While only 9,922 metric tons of these wastes were landfilled in 1992, the amount landfilled in 1993 mushroomed to 180,221 metric tons, representing the second largest waste category landfilled (after contaminated soils/solids) that year.[200] This category of waste includes "PL pigging sludge, spent additive (gasoline), and other organic sludges and liquids".[201]

As explained in Section II.E of these [EDF's] comments, both EPA and API data demonstrate the importance of these other wastes in an evaluation of a co-disposal scenario. On an average basis, EPA data indicate 23,979 cubic meters of onsite landfill capacity was used in 1992, well beyond the annual median value of 2,170 cubic meters of co-disposal EPA calculates for the wastes covered by this rulemaking and the study wastes.[202] Since most of the high volume wastes are landfilled offsite, the effects of co-disposal offsite is even greater for offsite disposal.[203]

The plausibility of the free-phased flow of contaminants from a refinery waste management unit is further documented in the accompanying Waterloo NAPL Report. Twenty-five landfills receiving industrial wastes, and four waste management facilities associated with the petroleum sector, have either definitively caused DNAPL contamination of the groundwater, or caused contamination with a high probability of DNAPL based upon available indirect evidence.[204] Refinery wastes similar or identical to the wastes at issue in this rulemaking, such as refinery tank bottom sludge, were codisposed with other wastes in the units causing the DNAPL contamination.[205]

Significantly, the authors also found DNAPL contamination from waste management is a "national" problem associated with many hydrogeological regions, and many types of waste management units including landfills and surface impoundments.[206]

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[199] 1991 API Residuals Report, Appendix A; 1992-1993 API Residuals Report, p. B-38.

[200] 1992-1993 API Residual Report, p. B-38.

[201] 1991 API Residuals Report, Appendix A.

[202] NODA Groundwater Risk Assessment, Table D.5.

[203] 1992-1993 API Residuals Report, p. B-20.

[204] Waterloo NAPL Report, Table 1.

[205] Id., Table A-2.

[206] Id., pp. 8, 10.

In summary, EPA's 1990 finding regarding the potential for cosolvent effects from the variety of solvents and oils disposed at refinery sites remains valid today. The oil content of the wastes covered by this rulemaking and the related study, and the oil and solvents contained in other codisposed refinery wastes, can produce a free-phased flow of contaminants in a refinery landfill under a plausible mismanagement scenario. (EDF, 00006, pg 6)

Response: EPA has addressed the comments related to co-disposal in Section I.A.3, and comments related to existing contamination in Section I.A.6.g. In brief, EPA finds the commenter's suggested expansions of the co-disposal scenario are not appropriate. EPA believes that its core objective in conducting this listing determination is to assess the incremental risk associated with these materials. As such, it is not appropriate to take into consideration pre-existing contamination or co-disposal with residuals outside of the scope of the consent decree. Such considerations are better measures of the risk associated with *landfills* containing the residuals of concern, rather than the residuals themselves.

EPA does not disagree that specific conditions at specific landfills related to historical management of high oil content materials may create a more aggressive mobility scenario than that modeled in support of this rulemaking. However because the Listing Program is centered on waste-specific determinations, rather than unit-specific determinations, such scenarios are better controlled via corrective action and the identification of these landfills as solid waste management units (SWMUs).

In addition, EPA is making its listing determinations in the range of a 10^{-5} to 10^{-6} risk threshold. This low threshold is inherently conservative and is in keeping with the thresholds set in the BIF rule which assumed that other exposure routes had not been fully explored (hence the risk threshold was not set at 10^{-4}).[207]

EPA also believes that the Agency's decision making data base is sufficiently broad to address the types of variability in waste characteristics of concern to the commenter. As a result EPA does not believe it is necessary to project additional variability via consideration of further co-disposal scenarios. More specifically, EPA's database is based on a full census of the refining industry, capturing refining waste generations and management practices for 1992. EPA has no reason to believe that 1992 is not representative of normal refinery operation and waste practices. The industry is sufficiently large that, while individual refineries may change their practices from year to year (e.g., landfilling to land treatment), it is likely that these shifts will be offset by other refineries making the reverse changes (e.g., from land treatment to landfilling). Further, EPA's use of Monte Carlo modeling allows for the consideration of thousands of permutations of possible modeling parameters, taking into account variability at the 90th to 95th percentile. Consideration of this variability did not drive the modeled risk to lower levels.

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[207] 56 FR 71650 ~ 71699, February 21, 1991.

Furthermore, the commenter argues that contaminated soils codisposed with the residuals of concern are likely to contain "a substantial amount of oil". The basis for this conjecture is not on any hard data regarding actual measurements, but on the commenter's concern with the adequacy of the TCLP for oily wastes, the exemption from the TC for petroleum-contaminated media from UST cleanups and concerns regarding the implementability of the contained in rule.

EPA evaluated the Waterloo report on ground-water contamination. Given EPA's desire to assess the incremental risk associated with the residuals of concern, EPA first assessed whether this report provided any directly relevant damage cases not previously identified by EPA. Among the 14 landfills and 27 surface impoundments characterized in this report, none of these units were operated at petroleum refineries. In addition, it appears that none of these units were modeled as off-site landfills in support of the listing determination. EPA also assessed the wastes and toxicants reported to be managed in these units and was unable to confirm EDF's conclusions that these units managed any of the 29 refinery residuals of concern. Furthermore, these materials are unlikely to generate DNAPL due to the relative density of oil and water. The presence of benzene and PAHs in these materials confirms the conclusions already made by EPA with respect to it's own damage case assessment, that these contaminants are difficult to link to specific wastes or releases due to their commonness in refinery materials (both product and residual), but can be linked to environmental releases and damage cases.

Comment 9: Analysis of Leaching of Oily Waste

EPA correctly concludes that the residuals under consideration do not have high levels of free oil. EPA investigated the potential for release and migration of residual constituents via multiphase transport. The Agency modeled a high-end, worst case scenario using the residual (Crude Oil Tank Sludge) with the highest fraction of oil liquid. From the modeling results, EPA correctly concluded that there is little or no expectation of significant release of waste constituents via multiphase transport. (EPA, 1995c) Therefore, EPA is justified in its decision not to consider free oil in its risk assessment calculations for the petroleum residuals in question.

Further, EPA should also have a high level of confidence that the estimates of contaminant leaching generated using the TCLP are, if anything, conservative. API continues to maintain that the TCLP likely overestimates contaminant leaching. (Narquis, 1992) In addition, EPA's use of the alternative OWEP on refining residuals, conducted in response to comments, did not pose significantly different results than the TCLP. (API, 00009)

Response: EPA acknowledges the commenter's input.

I.C.2. Potential for Additive Risks from Multiple Sources

Comment 1: Potential For Additive Risks from Multiple Sources

In response to comments by EDF, EPA evaluated whether an individual could be simultaneously exposed to waste concentrations from both a LTU and a landfill. In principle, API disagrees with the need to evaluate cumulative risks from multiple wastes due to the implausibility of the scenario. The results of EPA's analysis confirms this position. Using site-specific data for on-site facilities, EPA determined that simultaneous exposure was not possible. The evidence evaluated by EPA clearly supports the conclusion that there is a sufficient distance between land disposal units and residential wells and homes to prevent simultaneous exposure. (API, 00009, pg 24)

Response: EPA agrees with the commenter that the available data indicate that simultaneous exposure to landfills and land treatment units is unlikely.