LOUISIANA MID-CONTINENT
OIL AND GAS ASSOCIATION

Richard T. Metcalf
DIRECTOR OF ENVIRONMENTAL AFFAIRS

730 North Boulevard
Baton Rouge, Louisiana 70802
E-mail: Metcalf@Lmoga.com

225-387-3205
Fax: 225-344-5502
Cell: 225-276-5385

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Informal Report

FINAL REPORT: RISK ASSESSMENT FOR PRODUCED WATER DISCHARGES TO LOUISIANA OPEN BAYS

Anne F. Meinhold, Michael P. DePhillips, and Seymour Holtzman

March 1996

Prepared for
Bartlesville Project Office
United States Department of Energy
Bartlesville, OK

BIOMEDICAL AND ENVIRONMENTAL ASSESSMENT GROUP

ANALYTICAL SCIENCES DIVISION

DEPARTMENT OF APPLIED SCIENCE

BROOKHAVEN NATIONAL LABORATORY
UPTON, LONG ISLAND, NEW YORK 11973

BROOKHAVEN NATIONAL LABORATORY

MEMORANDUM

DATE: June 12, 1996

TO: Holders of BNL Report #62975, FINAL REPORT: RISK ASSESSMENT FOR PRODUCED WATER DISCHARGES TO LOUISIANA OPEN BAYS

am

FROM: Anne Meinhold, Brookhaven National Laboratory, (516) 344-2019

SUBJECT: Revision

Enclosed is a revision to the above referenced BNL report. Please discard the earlier version.

The revision corrects some typographical errors and some errors made in the preliminary transcription of field data for the concentrations of radionuclides in the discharges for the coastal DOE study sites. These modifications resulted in minor changes to the dose estimates for ecological receptors described in chapter 7.

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ASSOCIATED UNIVERSITIES, INC.

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ACRONYMS

AHQ	Acute Hazard Quotient
BEAG	Biomedical and Environmental Assessment Group
BEDS	Biological Effects Database
BNL	Brookhaven National Laboratory
CHQ	Chronic Hazard Quotient
CORMIX	Cornell Mixing Zone Expert System Model
CSA	Continental Shelf Associates, Inc.
ERL	Effects Range Low
ERM	Effects Range Median
HEAST	Health Effects Assessment Summary Tables
HQ	Hazard Quotient
IAEA	International Atomic Energy Agency
ICRP	International Commission on Radiological Protection
IRIS	Integrated Risk Information System
LDEQ	Louisiana Department of Environmental Quality
LOAEL	Lowest Observed Adverse Effect Level
NCRP	National Council on Radiation Protection and Measurements
NOEL	No Observed Effects Level
NORM	Naturally Occurring Radioactive Material
PAH	Polycyclic Aromatic Hydrocarbon
RfD	Reference Dose
SEP	Sediment Equilibrium Partitioning
UF	Uncertainty Factor
USDOE	United States Department of Energy
USEPA	United States Environmental Protection Agency
WHQ	Water Quality Criteria Hazard Quotient

EXECUTIVE SUMMARY

Introduction

Potential human health and environmental impacts from discharges of produced water to the Gulf of Mexico are of concern to regulators at the State and Federal levels, the public, environmental interest groups and industry. Current and proposed regulations require a zero discharge limit for coastal facilities, based primarily on studies in low energy, poorly flushed environments. However, produced water discharges in coastal Louisiana include a number of open bay sites, where potential human health and environmental impacts are likely to be smaller than those demonstrated for low energy canal environments, but greater than the minimal impacts associated with offshore discharges.

Additional data and assessments are needed to support risk managers at the State and Federal levels in the development of regulations that protect human health and the environment without unnecessary cost to the economic welfare of the region and the nation. This project supports the *Natural Gas and Oil Initiative* objectives to:

- improve coordination on environmental research;
- streamline State and Federal regulation;
- enhance State, and Federal regulatory decision making capability;
- enhance dialogue through industry/government/public partnerships; and
- work with States and Native American Tribes.

This report is part of a series of studies of health and ecological risks from discharges of produced water to the Gulf of Mexico, supported by the United States Department of Energy (USDOE). These assessments are coordinated with a field study managed by USDOE, titled "Environmental and Economic Assessment of Discharges from Gulf of Mexico Region Oil and Gas Operations" (USDOE Field Study).

Human health and ecological risk assessments for produced water discharges to open bays in Louisiana were done to support risk managers in developing regulations for discharges of produced water to coastal open bays. The human health and ecological risk assessments were done in a tiered approach. The initial human health and ecological risk assessments consist of conservative screening analyses meant to identify potentially important contaminants and to eliminate others from further consideration. More quantitative assessments were done for contaminants identified in the screening analysis as being of potential concern.

Data used in the assessment are from two major sources:

- Data collected in the ongoing USDOE field study; and

- Data abstracted from the Louisiana Department of Environmental Quality (LDEQ) permit files for open bay sites in Louisiana that plan to continue to discharge produced water until January, 1997.

Risk Assessment

Risk assessment can be defined as the process of estimating magnitudes and probabilities of potential adverse effects on human health or the environment. Risk management involves the political, economic and social decisions and actions taken to accept, mitigate, or control potential risks. Risk assessments provide risk managers with the scientific information needed to balance the degree of risk permitted against competing risks and the cost of risk reduction.

The United States Environmental Protection Agency (USEPA) currently considers excess individual lifetime cancer mortality risks less than 1×10^{-4} (one in ten thousand) to 1×10^{-6} (one in one million) to be acceptable (Federal Register, 1991). No similar standard "acceptable risk" value is available for toxic effects -- estimated doses or intakes are usually compared to a chemical specific reference dose to determine if toxic effects are expected.

A tiered approach to human health and ecological risk assessment is logical and cost-effective. In a tiered approach, the initial analysis is a conservative (i.e. worst case) screening step, designed to eliminate from further analyses contaminants and pathways that are not of concern in terms of potential impacts to human health or ecological values. Further analyses are unnecessary when use of conservative models and assumptions yield estimated risks that are small (i.e. individual lifetime fatal cancer risk less than 1×10^{-6} or no toxic effects predicted). If a conservative analysis suggests that risks are high, a more detailed, comprehensive and realistic assessment is performed.

The state-of-the-science in risk assessment uses a probabilistic approach that explicitly considers uncertainties and variability in assumptions, data and results. Probabilities of effects, and uncertainties are explicitly considered in both the analysis and the expression of its result.

Hazard and Receptor Identification

Many contaminants in produced water have known or suspected human health and or ecological effects at high exposures. Contaminants of special concern include: toxic metals such as lead, mercury and cadmium; potentially toxic organic compounds such as phenol and PAHs; and known or suspected carcinogens such as benzene and radionuclides.

Ingestion of contaminated fish is expected to be the most important exposure route for people, because many of the contaminants found in produced water are known to accumulate in edible fish and shellfish. The important receptors for radium discharged in produced water are recreational fishermen and their families. The primary route of exposure was assumed to be ingestion of finfish, because most seafood taken near platforms by recreational fishermen are finfish rather than mollusks or crustaceans.

Potential ecological receptors for contaminants in produced water include recreationally and commercially important fish and shellfish species, benthic invertebrates living close to the platforms, and threatened and endangered species living in open Louisiana bays. Potentially important exposure pathways include direct exposure in water or sediment, and ingestion in food, water or sediment.

Risk Assessment Approach

The overall approach was to use available data and modeling analyses for continuing open bay discharges, in a tiered assessment of human health and ecological risk. The initial analysis consisted of conservative screening assessments meant to identify contaminants of potential concern. More quantitative, probabilistic risk assessments were performed for contaminants identified in the screening analyses.

The data that form the bases of the risk assessments presented here include:

- Data collected in the ongoing USDOE field study:

  -- PAH and metal concentrations in sediment near two open bay discharges;
  -- radium concentrations in edible biota near two open bay discharges;
  -- radionuclides in the effluent of two open bay discharges; and
  -- fish ingestion rates for recreational fishermen and their families.

- Data abstracted from LDEQ permit files for-open bay sites in Louisiana that plan to continue to discharge produced water until January, 1997:

    -- location, depth and discharge rate data;
    -- data describing chemical concentrations in the effluents;
    -- data describing radionuclide concentrations in the effluents;
    -- results of toxicity testing on effluents.

The modeling analyses used the USEPA CORMIX model (Doneker and Jirka, 1990) and Louisiana's mixing zones (acute: 50 feet; chronic and human health: 200 feet). These distances imply a risk management decision about the "acceptable" location for environmental impacts; and were used in the current risk assessment.

USDOE Field Study Preliminary Data

<u>Background</u>

Risk assessments are coordinated with a USDOE project titled " Environmental and Economic Assessment of Discharges from Gulf of Mexico Region Oil and Gas Operations" (referred to as the "USDOE Field Study"). Continental Shelf Associates, Inc. (CSA) was contracted to conduct the field study. The study includes 4 technical tasks, two of which are relevant to the risk assessment presented here:

- Task 4 - Monitoring of the Recovery of Impacted Wetland and Open Bay Produced Water Discharge Sites in Coastal Louisiana and Texas; and

- Task 6 - Synthesis of Seafood Catch, Distribution and Consumption Patterns in the Gulf of Mexico Region.

Steimle & Associates, Inc. were subcontracted by CSA to perform the two tasks relevant to the risk assessments presented here (Tasks 4, 6). Available preliminary results were used in the current analysis.

USDOE Open Bay Sites

The emphasis in the study of coastal sites is an assessment of the recovery of these sites from any impact from produced water discharges. Data were collected prior to the termination of discharge at three sites (including the two open bay sites discussed here), and several times after the discharge was terminated. The data used in the current risk assessments were limited to those collected before termination of the discharges. The open bay study sites were located at Delacroix Island and Bay de Chene.

The Delacroix Island Oil and Gas Field, located approximately 5.5 miles southeast of Delacroix, Louisiana, has been in production since the first well was drilled in the field in 1940. The area is part of a subsiding delta, which results in broken marsh and numerous small water bodies with few large open bays. The tank battery studied (Tank Battery #1) is located in approximately 4.9 feet (1.5 m) of water and discharges approximately 2,000 bbl/day. The Delacroix Island site is not located in a completely open bay, but was used in the assessment presented in this report with the understanding that impacts at the site may over-estimate impacts from true open bay discharges.

The Bay de Chene Field is located approximately 13 miles northwest of Grand Isle, Louisiana and is part of the Barataria Basin. The field has been in constant production since the first well was drilled in 1942. The tank battery studied (Tank Battery #5) is located in Hackberry Bay, a large open bay typical of the Barataria system. The discharge is located in about 7.5 feet (2.3 m) of water and discharges approximately 4,000 bbl/day.

Concentrations of radionuclides were measured in discharges. Radium concentrations were measured in tissues of fish and shellfish at reference stations and the discharge stations. Sediment PAH and metal concentrations were also available.

Both pre- and post-termination benthos were collected at the study sites, and preliminary data are available. The study (Mulino *et al.*, 1995; 1996) found depressed numbers of species and individuals at and near the discharge during the pre-termination sampling, suggesting an impact on the benthos between 0 and 100 meters from the platform.

Fishermen Survey

Commercial fishermen (including oystermen) and recreational fishermen were surveyed by personal interview from May through November 1993 to determine categories of seafood taken over the previous three months, types of license(s) held, and information on the number, gender and ages of individuals in the household and their seafood consumption habits. Respondents were also interviewed about locations fished, estimated distances from oilfield structures, and species caught (Steimle & Associates, Inc., 1995).

In this preliminary assessment, ingestion rates for recreational fishermen of fish caught near coastal platforms were derived from the reported data on meals per week. The data reported for meals per week had an arithmetic mean of 1.8, a standard deviation of 1.3, and a range of 0 to 15. A lognormal distribution of meals per week was used in calculating ingestion rates (g/d) of fish.

Characterization of Continuing Discharges

Louisiana Regulations (Title 33, March 20, 1991) required termination of all produced water discharges to natural or man-made water bodies located in intermediate, brackish or saline marsh areas after January 1, 1995, unless the discharge (s) have been authorized in an approved schedule for elimination or effluent limitation compliance. A variance through January, 1997 was granted (12/16/94) for permitted discharges located in open waters and at least 1 mile from any shoreline in Chandeleur Sound, Breton Sound, Barataria Bay, Caminada Bay, Timbalier Bay, Terrebonne Bay, East Cote Blanche Bay, West Cote Blanche Bay or Vermillion Bay. LDEQ identified produced water discharges in open bay areas that may qualify for this variance.

Information critical to an assessment of the environmental impact from a produced water discharge includes the depth of the platform and the rate of discharge. Water depths ranged from 4 to 18 feet with a mean of 9.1 feet (1.2 - 5.5 m; mean: 2.8 m). Discharge rates ranged from 1 to 37,113 bbl/day (mean: 4,527 bbl/day).

Chemical contaminants and radionuclides measured in open bay produced water discharges were abstracted from LDEQ permit files. Data describing effluent toxicity tests were also abstracted from LDEQ permit files.

The USEPA surface water transport model CORMIX (Doneker and Jirka, 1990) was used to estimate the dilution expected 50 and 200 feet from open bay discharges ($DF_{50\ ft}$ and $DF_{200\ ft}$). Eight feet (2.44 m) was chosen to represent the assumed depth of the receiving water body for continuing open bay discharges in Louisiana. A range of discharge rates was modeled to cover the range of discharge rates for open bay sites. The following empirical relationships were derived from the modeling results:

For discharge rates $\leq$ 5000 bbl/d

$DF_{50\ ft} = 10633 * (\text{DISCHARGE RATE})^{-0.867}$ (R=0.997)

$DF_{200\ ft} = 46303 * (\text{DISCHARGE RATE})^{-0.946}$ (R=0.9997)

For discharge rates > 5000 bbl/d

$DF_{200\ ft} = 36061 * (\text{DISCHARGE RATE})^{-0.762}$ (R=0.9997)

These empirical relationships were applied to distributions of discharge rates for open bay discharges to produce distributions of dilution factors for 50 and 200 feet. The dilution factor distributions were also used to develop distributions of percent effluent expected in the water column at 50 and 200 feet.

Human Health Risk Assessment for Radium

Screening and probabilistic human health risk assessments were done for open bay radium discharges in Louisiana.

In the conservative screening analysis, estimated risks for the ingestion of radium in fishes exceeded 1×10^{-6} in all cases. Estimated cancer risks for fish sampled at reference stations at Delacroix Island and Bay de Chene (pre-termination data) were similar to those for ingestion of fish caught near the discharges. Maximum predicted screening-level risks were greater than 1×10^{-3} for the modeled continuing discharges.

These results are from a conservative, screening level assessment, and do not represent best estimates of risk associated with radium discharged by open bay platforms. They do, however, suggest the need for a more detailed, probabilistic assessment.

A probabilistic risk assessment was done using distributions of: radium concentrations in fish (from field sampling and modeling); fish ingestion rates (from USDOE fishermen survey); and risk factors for cancer mortality.

The 95th percentile individual lifetime fatal cancer risks for both DOE study sites (Delacroix Island and Bay de Chene) were less than 1×10^{-5}. The 95th percentile individual lifetime fatal cancer risk for continuing open bay discharges was 4.3×10^{-6}, in good agreement with the DOE study site results.

These results suggest that the ingestion of radium in fish, caught near open bay produced water platforms, does not present an important risk to human health.

Ecological Risk Assessment for Radionuclides

This assessment used concentrations of radionuclides measured in the effluent at the two USDOE study sites, and radium concentrations reported in permit files for continuing open bay discharges, to assess potential ecological effects from radionuclides discharged in produced water. Worst-case water concentrations were predicted using a dilution factor that was similar to the most conservative factor derived from modeling analyses. Predicted water concentrations were compared to screening dose-rate factors developed by the International Atomic Energy Agency (IAEA, 1988) that relate exposure of an organism to the concentration of each radionuclide in the water in which the organism lives. Estimated doses were compared to reference dose rates suggested by IAEA (1988).

None of the predicted doses to aquatic animals exceeded the range of 0.1-24 mSv/d that IAEA (1988) associated with a potential for only minor effects on individual animals. Because of the conservative nature of this initial analysis, it can be concluded that no effects on aquatic animals from radionuclides discharged in produced water to open bays in Louisiana are expected.

Human Health Risk Assessment for Chemical Contaminants

A human health risk assessment screen was done for metals and organic compounds measured in continuing open bay discharges. This analysis followed the USEPA approach to estimating risks from toxic materials and carcinogens by applying RfD (reference dose) and slope factor values to estimates of chemical intake rates (USEPA, 1989). Predicted water concentrations were also compared to human health water quality criteria developed by USEPA and the State of Louisiana.

Arsenic, chromium, copper, silver, naphthalene, toluene and xylenes were eliminated from further consideration. This screening step identified antimony, benzene, cadmium, lead, mercury, nickel, zinc and phenol as contaminants of potential concern.

More realistic and quantitative assessments were performed for contaminants identified in this screening analysis. The results show that intakes of contaminants discharged to open bays in produced water pose a negligible hazard to human health.

The potentially toxic contaminants examined (antimony, cadmium, mercury, nickel, zinc and phenol; lead was analyzed separately) all had low risks of toxic effects. The only contaminant that marginally exceeded its oral RfD value was cadmium.

Because of the concern for lead exposure to children, and the current belief that the dose-response function for lead exposure does not have a threshold, lead was analyzed in a separate probabilistic risk assessment. Risk from ingestion of lead in fish caught near platforms only slightly exceeded risks from background intake of lead and was similar to risks from ingestion of lead in fish caught in the Gulf of Mexico but not near platforms.

For benzene (the only potential carcinogen of concern), the predicted distribution of values for incremental individual lifetime risk of cancer mortality had a mean value of 1.6×10^{-6} and a 95th percentile value of 7.4×10^{-6}. This is within the range accepted by USEPA (1×10^{-6} to 1×10^{-4}; Federal Register, 1991).

These analyses used several conservative assumptions. The first assumption was that all the fish spend all of their time living and feeding within the plume, although they probably spend only a fraction of time within a plume. The predicted concentrations represent values at the midline of the plume at 200 feet from the discharge. These values were generated by a model that underestimates dilution (Smith *et al.*, 1993). It was also assumed that all the fish eaten by a person were captured at the midline of a plume, while people may eat fish from several sources. Although contaminant concentrations in water should increase with decreasing distances from a discharge, bioaccumulation in fish would be offset by expected reduced residence of fish within the smaller plume volumes.

Ecological Risk Assessment for Chemical Contaminants and Effluent

Three ecological risk assessments were performed:

- a screening assessment of chemical toxicity to benthic biota;
- an assessment of potential toxicity of individual produced water components to fish and crustaceans in the water column; and
- an assessment of whole effluent toxicity to fish and crustaceans.

Screening Assessment Of Sediment Toxicity

Sediment metal and PAH concentrations measured at the USDOE study sites (pre-termination data) were compared to proposed sediment quality criteria (ERM: Effects Range Median; ERL: Effects Range Low; Long *et al.*, 1995).

None of the measured concentrations of metals in sediment samples exceeded their respective ERM values. In general, measured sediment concentrations were below the ERL, with the exception of arsenic and nickel. Each of these metals exceeded its ERL value in samples from at least one reference site, and each discharge site. There was no clear pattern of concentration with distance from a discharge.

With the exception of acenaphthene, individual and total PAH concentrations exceeded ERL criteria at, and 100 m from the discharge at Delacroix Island. Acenaphthene concentrations exceeded the ERL values at the discharge, 100, 300 and 500 m sample sites. Neither individual nor total PAH concentrations in sediment samples from Delacroix Island exceeded ERM criteria.

Individual and total PAH concentrations exceeded ERL criteria at the discharge site, and 100 m and 300 m from the discharge at Bay de Chene. Individual and total PAH concentrations in samples from the discharge site exceeded ERM criteria.

In preliminary results of the benthos sampling performed at the USDOE study sites, depressed numbers of individuals and numbers of species were found only at distances less than 100 m from the discharges (Mulino *et al.*, 1995; 1996). Although comparisons of PAH concentrations to sediment criteria were generally consistent with the results of benthos observations, they could not explain differences between the benthic biota at the two study sites. Mulino *et al.*, (1995; 1996) attributed the more severe impacts at Delacroix Island (smaller discharge) to hydrologic influences on salinity and oxygen content of the water.

These results are preliminary, and cannot be applied to all other open bay discharge sites with much confidence, but the discharge rates and depths of the Bay de Chene and Delacroix Island study sites are comparable (discharge rates are on high end of distribution) to those that are continuing to discharge.

Assessment Of Potential Toxicity Of Individual Contaminants In The Water Column

Concentrations of contaminants in plumes were predicted from worst-case measurements in continuing open bay discharges (LDEQ permit files). These water column concentrations were compared to USEPA and Louisiana water quality criteria.

In this screening analysis, predicted water concentrations exceeded acute water quality standards for copper, lead, nickel, silver and zinc. Chronic water quality criteria were exceeded for antimony, cadmium, copper, lead, mercury, nickel, silver, zinc and phenol. Arsenic, chromium, benzene, naphthalene and toluene were eliminated from further consideration.

A quantitative risk assessment was done for contaminants not eliminated by the initial screen. Distributions of predicted chemical concentrations were compared to acute and chronic toxicity criteria for marine biota.

None of the predicted chemical concentrations (at 200 ft) exceeded their respective acute toxicity criteria. Antimony, phenol, and zinc concentrations did not exceed any of their respective chronic toxicity criteria. Less than five percent of the concentrations of cadmium, copper, lead, nickel and silver, at 200 ft, are expected to result in chronic toxicity to biota. More than 90% of the predicted concentrations of mercury are expected to be below its chronic toxicity criterion. Since these all represent midline values for the plumes, the expectation would be that environmental impacts of the individual chemicals would be limited.

Assessment Of Whole Effluent Toxicity

Standard laboratory test organisms, a shrimplike mysid crustacean (*Mysidopsis bahia*) and the sheepshead minnow (*Cyprinodon variegatus*), were used in toxicity tests reported in LDEQ permits. Predicted water column concentrations of effluents were compared with reported results of acute and chronic toxicity tests on diluted effluent samples. For the results of each type of toxicity test, data were expressed in the same way as the predicted water column concentrations: as percent effluent.

For discharges reported to the LDEQ, modeled relationships between discharge (flow) rates and dilution factors were used to estimate concentrations of effluents at 50 m and 200 m from discharges.

Acute toxicity test data consisted of mortality responses, expressed as an effluent median lethal concentration for an exposure duration of 96 hrs (96-hr LC_{50}), or the effluent concentration which results in the mortality of 50% of the test organisms in a 96-hr exposure period. Acute toxicity ratios (AHQ) were calculated between the estimated percent effluent at 50 ft and 200 ft from the discharge and the available corresponding LC_{50} values (*M. bahia*; *C. variegatus*) for each platform). Ratios of one or greater indicate potential lethality.

At 50 ft, 15% of the modeled effluent concentrations exceeded their respective LC_{50} values *for M. bahia*, and 5% exceeded their respective LC_{50} values for *C. variegatus*. A 200 ft, 15% of the modeled effluent concentrations exceeded their respective LC_{50} value for *M. bahia* and 2.5% exceeded their respective LC_{50} value for *C. variegatus*. The results suggest a potential for lethal effects for some discharges at 50 and at 200 feet.

Chronic toxicity ratios were calculated for the estimated percent effluent at 200 ft and the available corresponding chronic NOEL values for survival and growth inhibition. Ratios greater than one suggest a potential for toxic effects.

At 200 ft, 37% of the modeled effluent concentrations exceed their respective survival NOEL values for *M. bahia*, and 19% exceed their respective survival NOEL value *for C. variegatus. At* 200 ft, 39% of the modeled effluent concentrations exceeded their respective growth-inhibition NOEL values for *M. bahia*, and 18% exceeded their respective growth-inhibition NOEL values for *C. variegatus*. Approximately two times more of the predicted effluent concentrations exceeded chronic NOEL values (both survival and growth-inhibition) for *M. bahia* than for *C. variegatus*.

The results suggest a potential for chronic effects within 200 feet of some discharges. These results should be taken only as an indicator of potential toxicity. The percent effluent values exceeded their respective NOEL values by small amounts.

Since the percent effluent values compared to the NOEL in this analysis represent the concentrations at the midline of the plume at 200 ft from the discharges, an organism would have to live totally in the plume, within 200 ft of the discharges for at least the period of the chronic test to be affected. This is unlikely because the plume is a relatively small fraction of the volume of water within 200 ft of a platform. That volume, in turn, is a small fraction of the body of water in which the discharge occurs. Therefore, major effects to local populations or to the ecology of the region around open bay discharges is not expected.

Conclusions

The tiered approach to risk assessment is a cost-effective way to provide information needed to make risk management decisions. This screening assessment for human health and ecological risks from open bay produced water discharges in Louisiana eliminated a number of contaminants from further consideration. More quantitative assessments were performed on contaminants of potential concern.

Human health risks from radium in produced water appear to be small. Ecological risks from radium and other radionuclides in produced water also appear to be small.

Intakes of chemical contaminants in fish caught near open bay produced water discharges are expected to posed a negligible toxic hazard or carcinogenic risk.

Potential impacts to benthic biota and fish and crustaceans in the water column are possible for some discharges within the 200 ft mixing zone. Permanent damage to populations of organisms and ecosystems are not expected, because mixing zones represent relatively small volumes and animals are not expected to remain continuously in the plume.

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1 INTRODUCTION

1.1 Problem

Produced water discharged to coastal waters in Louisiana can contain a number of contaminants, including organic compounds, metals and radionuclides. Many of these contaminants are toxic to marine organisms at high concentrations. Most contaminants discharged in produced water occur naturally in the geologic reservoir along with the oil and gas. Biocides or other chemicals that may be toxic to aquatic organisms are added to some effluents.

Potential human health and environmental impacts from discharges of produced water to the Gulf of Mexico are of concern to regulators at the State and Federal levels, the public, environmental interest groups and industry. This area supports economically important commercial and recreational fisheries, unique, socially- valued ecosystems, and several endangered and threatened species.

In offshore and other high energy environments, produced water is diluted so rapidly that contaminants cannot be detected in the water column or sediment even a few meters from the outfall. Effects on marine life are likely to be minimal. In shallower, low energy coastal canal environments, contaminants were detected in water, sediment and organisms several hundred meters from the discharge. Effects on benthic organisms in shallow coastal settings and on organisms in the biofouling mat close to discharge points have been documented (Boesch and Rabalais, 1989; Gallaway *et al.*, 1981).

Current and proposed regulations require a zero discharge limit for coastal facilities, based primarily on studies in low energy, poorly flushed environments. However, produced water discharges in coastal Louisiana include a number of open bay sites, where potential human health and environmental impacts are likely to be smaller than those demonstrated for low energy canal environments but greater than the minimal impacts associated with offshore discharges.

Additional data and assessments are needed to support risk managers at the State and Federal levels in the development of regulations that protect human health and the environment without unnecessary cost to the economic welfare of the region and the nation.

1.2 This Report

The United States Department of Energy (USDOE) has a program of research in the environmental aspects of oil and gas extraction. This program includes a project titled "Environmental and Economic Assessment of Discharges from Gulf of Mexico Region Oil and Gas Operations" (here called the USDOE field study). Part of this project involves a comprehensive sampling and analysis program for offshore and coastal platforms in the Gulf of Mexico. This sampling project will characterize the environmental impacts associated with the discharge of naturally occurring radioactive materials (NORM), metals and organics in produced water.

This report is part of a series of studies of the health and ecological risks from discharges of produced water to the Gulf of Mexico, supported by the USDOE. These assessments are being coordinated with the field study described above, using the collected data to perform human health and ecological risk assessments. These assessments will provide input to regulators in the development of guidelines and permits, and to industry in the development and use of appropriate discharge practices.

This project supports the *Natural Gas and Oil Initiative* objectives to:

- improve coordination on environmental research;
- streamline State and Federal regulation;
- enhance State, and Federal regulatory decision making capability;
- enhance dialogue through industry/government/public partnerships; and
- work with States and Native American Tribes.

This report presents human health and ecological risk assessments for produced water discharges to open bays in Louisiana. The risk assessments were done to support risk managers in developing regulations for discharges of produced water to coastal open bays. The human health and ecological risk assessments are presented in a tiered approach. The initial human health and ecological risk assessments consist of conservative screening analyses meant to identify potentially important contaminants, and to eliminate others from further consideration. More quantitative assessments were done for contaminants identified, in the screening analysis, as being of potential concern.

Data used in the assessment are from two major sources:

- Data collected in the ongoing USDOE field study:
  - -- contaminant concentrations in water, sediment and edible biota at two coastal discharges
  - -- ingestion rates for recreational fishermen

- Data abstracted from the Louisiana Department of Environmental Quality (LDEQ) permit files for open bay sites in Louisiana that plan to continue to discharge produced water until January, 1997:
  - -- location, depth and discharge rate data
  - -- chemical and radionuclide concentrations in the effluents
  - -- results of effluent toxicity testing

Section 2 gives an overview of human health and ecological risk assessment, to help put the analyses presented here in perspective. Section 3 provides the hazard assessment portion of the risk assessment, and identifies the important receptors and pathways of concern. Section 3 also outlines the approach taken to the risk assessments presented in the rest of the report. The remaining sections (4 through 9) present the human health and ecological risk assessments for discharges of produced water to open bays in Louisiana.

2 RISK ASSESSMENT OVERVIEW

2.1 Risk Assessment and Risk Management

Risk assessment can be defined as the process of estimating magnitudes and probabilities of potential adverse effects on human health or the environment. Risk management involves the political, economic and social decisions and actions taken to accept, mitigate, or control potential risks. Risk assessments provide risk managers with the scientific information needed to balance the degree of risk permitted against competing risks and the cost of risk reduction.

A risk assessment should be performed independently of risk management, but the needs and concerns of risk managers should be considered in the design of the risk assessment to ensure that the results are relevant, useable, and understandable to risk managers.

2.2 Human Health Risk Assessment

A health risk assessment for an environmental pollutant describes the discharge of the contaminant, its transport and fate in the environment, and the resulting human exposure. Human-health risks are then calculated based on data and models that relate exposures to health effects.

The most commonly used framework for human health risk assessment includes the following four phases (NRC, 1983):

- Hazard identification;
- Dose-response assessment;
- Exposure assessment; and
- Risk characterization.

Hazard identification involves the use of exposure and effects data from the laboratory and the field to determine whether the agent of concern can cause health effects and to identify what those effects are (NRC, 1983).

Dose-response assessment characterizes the relationship between administered dose and the incidence of an adverse effect. Dose-response information is usually derived from animal toxicology studies or from clinical studies or epidemiology studies of people exposed at high levels. Assumptions must be made about the comparability of the response in laboratory animals to that of humans. Statistical methods are usually necessary to extrapolate the dose-response function from high experimental doses to the generally much lower doses in the human population.

Exposure assessment estimates the magnitude, frequency and duration of exposure, and characterizes subgroups of the human populations subject to different levels of exposure. This phase includes estimating the source term, fate and transport of the contaminant(s) of concern, and subsequent human exposure.

Risk characterization integrates the results of the previous phases, estimates the incidence of an adverse human health effect under conditions defined in the exposure assessment, and describes the uncertainties in the data and assumptions. Human health risks are described as the probability of an adverse health effect (e.g., cancer death or toxic effect) in an individual of an exposed population (individual risk), or the number of health effects expected in the population (population risk) during a given time interval.

The United States Environmental Protection Agency (USEPA) currently considers excess individual lifetime cancer mortality risks less than 1×10^{-4} (one in ten thousand) to 1×10^{-6} (one in one million) to be acceptable (Federal Register, 1991). USEPA recently proposed standards for radionuclides in drinking water that the agency considers to be associated with an individual lifetime cancer fatality risk of 1×10^{-4} (Federal Register, 1991). No similar standard "acceptable risk" value is available for toxic effects -- estimated doses or intakes are usually compared to a chemical specific reference dose to determine if toxic effects are expected.

2.3 Ecological Risk Assessment

Early environmental decision-making was based on qualitative descriptions of effects of pollutant discharges on organisms and the environment, with some reliance on the assumption that protection of human health would ensure adequate protection of the environment. Current information and environmental regulations suggest a need for a risk-based approach to decision-making for environmental protection.

With some modifications, and addition of important uncertainties, the general paradigm for human health risk assessment is now being applied to estimation of risks to the environment. The field is new and definitions are not standardized. For the purposes of this report, "environmental risk assessment" refers to an assessment of the risks to man from contaminants in the environment (air, water, soil or food). "Ecological risk assessment" refers to an assessment of risks to the natural environment (Suter, 1993). The receptors or values of concern in an ecological risk assessment may range from individual organisms to entire ecosystems and fundamental ecological processes.

Because of the number of different species in a community and the complexity of inter-species interactions and basic ecological processes, the level of organization for which the assessment is performed can vary widely (individual, population, community, ecosystem), and the potential endpoints for the assessment are many (death, acute or chronic toxicity, reproductive or developmental effects, disruption of basic processes). USEPA (1992) proposed a framework for ecological risk assessment that includes three phases:

- Problem formulation;
- Analysis (exposure and effects assessment); and
- Risk characterization.

The problem formulation phase identifies the factors to be considered in the assessment, and determines the scope and objectives of the analysis. This phase includes the preliminary data gathering and conceptual development needed to define the problem. Specific steps in the problem formulation phase include planning, identification of stressor characteristics, description of the ecosystem potentially at risk, identification of potential ecological effects, endpoint selection, and development of a conceptual model for the assessment.

In exposure assessment, environmental concentrations of the contaminant are described, and exposure of the organisms and ecosystems of concern are estimated. The exposure assessment estimates the transport of the contaminant through the environment, including its transformation and uptake by organisms.

In effects assessment, a dose-response relationship between exposure and effects is developed. An effects assessment determines the relationship between exposure to the contaminant and effects on the measurement endpoint. An effects assessment is usually based on extrapolating results of toxicity studies on standard individual test organisms to effects on individuals of other species, populations, communities and ecosystems.

Risk characterization integrates the estimates of exposure and dose-response relationships developed in the analysis phase to produce an estimate of the risk to the identified assessment endpoint.

2.4 Tiered Approach

A tiered approach to human health and ecological risk assessment is logical and cost-effective. In a tiered approach, the initial analysis is a conservative (i.e. worst case) screening step, designed to eliminate from further analyses contaminants and pathways that are not of concern in terms of potential impacts to human health or ecological values. Further analyses are unnecessary when use of conservative models and assumptions yield estimated risks that are small (i.e. individual lifetime fatal cancer risk less than 1×10^{-6} or no toxic effects predicted). If a conservative analysis suggests that risks are high, a more detailed, comprehensive and realistic assessment is performed.

Ecological risk assessments may be more qualitative than human health assessments because of the many sources of uncertainty in assessing risks to ecological values (USEPA, 1992).

2.5 Probabilistic Analysis and Uncertainty

The current application of the National Research Council risk assessment paradigm (NRC, 1983) to estimation of human health and ecological risk requires explicit description of uncertainties in assumptions, models and parameters, and incorporation of these uncertainties into a final expression of risk. Until recently, the common practice in risk assessment was to use conservative assumptions in a "worst case" analysis rather than to estimate uncertainty. This approach: obscures recognition of the degree of conservatism and the uncertainties in risk estimates; allows for improbable scenarios and results; and ignores the potentially excessive costs of decisions made based on conservative assumptions (Burmaster *et al.*, 1990; Paustenbach *et al.*, 1991).

As discussed above (Section 2.4), a conservative, screening level assessment is an appropriate first step in an assessment. A more quantitative and realistic analysis can be performed when the threshold established in the screening process is exceeded. The state-of-the-science in risk assessment uses a probabilistic approach that explicitly considers uncertainties and variability in assumptions, data and results. Probabilities of effects, and uncertainties are explicitly considered in both the analysis and the expression of its result.

A commonly used tool in probabilistic, quantitative risk assessment is Monte Carlo analysis. In a Monte Carlo analysis, a sample from the distribution of an input parameter is placed into a simulation to interact in a model with samples from other input parameters. The frequency of sampling within an independent variable depends on the relative frequency of a value in the frequency distribution (Paustenbach *et al.*, 1991).

3 HAZARD ASSESSMENT AND RISK ASSESSMENT APPROACH

3.1 Background and Overall Approach

Screening-level assessments were performed to identify potentially important contaminants and ecological receptors, and to eliminate others from further consideration. Based on the results of this preliminary analysis, more quantitative risk assessments were done for specific contaminants.

Two sources of data were used in the risk assessments: data collected in the USDOE field study and data abstracted from LDEQ permit files. These data sets and associated modeling analyses were used to assess potential human health and ecological risks associated with continuing open bay discharges of produced water in Louisiana.

This section:

- presents the hazard identification step for the human health and ecological risk assessments;
- briefly describes the data and modeling analyses used in the risk assessments presented in this report (given in detail in sections 4 and 5 and Appendices A and B); and
- outlines the approach used in the human health and ecological risk assessments (presented in sections 6 through 9).

3.2 Hazard and Receptor Identification

Hazard identification involves the use of exposure and effects data from the laboratory and field to determine whether the agent of concern can cause health effects and to identify what those effects are (NRC, 1983). In the context of this report, hazard identification includes: identification of contaminants of potential concern in produced water, identification of important human receptors and exposure pathways, and a description of potentially important ecological effects and receptors.

3.2.1 Contaminants

Many contaminants in produced water have known or suspected human health and or ecological effects at high exposures. Contaminants of special concern include: toxic metals such as lead, mercury and cadmium; potentially toxic organic compounds such as phenol and PAHs; and known or suspected carcinogens such as benzene and radionuclides.

Radionuclides

Radionuclides known to occur in produced water above background surface water concentrations include ^{226}Ra, ^{228}Ra, and ^{210}Pb. Other decay products of radium (^{210}Po, ^{228}Th, ^{222}Ra) may also be expected in produced water.

The health effects of radionuclides can be attributed to their radioactive emissions. The alpha, beta and gamma radiation released by the decay of radionuclides cause ionization of cellular components which may result in the mutation or death of affected cells.

Current practice in radiation protection is to assume there is a cancer risk associated with even very small doses of radiation. Risk factors are derived from epidemiological data and extrapolated down to low doses to describe the cancer risk associated with small exposures. See Appendix C for a more detailed discussion.

Most of the available studies of the effects of radiation on aquatic organisms are concerned with the induction of deterministic, somatic effects. These effects include increases in mortality and pathophysiological, developmental and reproductive effects. There is little information available concerning induction of cancer and genetic effects, although a few studies of stochastic genetic effects in organisms are available (Anderson and Harrison, 1986).

The National Council on Radiation Protection and Measurements (NCRP) and the International Atomic Energy Agency (IAEA) reviewed the literature on the effects of ionizing radiation on aquatic organisms, and suggested reference levels that would protect aquatic populations (NCRP, 1991; IAEA, 1988). Effects on aquatic organisms are discussed in more detail in Appendix C.

Chemical Contaminants

USEPA publishes cancer slope factors, reference doses or other estimates in the IRIS data base (Integrated Risk Information System) and water quality criteria for many of the contaminants commonly found in produced water. As a first level screen, chemical contaminants with published water quality criteria, slope factors and reference doses were included in the analysis. Published reference values suggest a potential concern for human health effects.

Most chemical contaminants discharged in produced water present a potential human health hazard because of toxicity associated with ingestion in fish and shellfish. A few of the chemical contaminants found in produced water are suspected or known human carcinogens including benzene and arsenic.

Effects on aquatic organisms may be associated with a number of contaminants found in produced water discharges. Water and sediment toxicity studies, and water quality criteria are available for a few contaminants suggesting reasonable concern for potential ecological effects. Toxicity testing of produced water effluents using standard laboratory test animals has shown a range of acute LC_{50}s and NOELs, again suggesting the potential for concern about effects to fish and shellfish species.

Effects on sediment communities have also been demonstrated (Armstrong *et al.,* 1977; Rabalais *et al.,* 1991), but the relationship between effects on number of species and individuals and chemical contaminants in sediments were site specific and not consistent across all studies. These studies suggest a potential for toxic effects to benthic communities living close to platforms.

3.2.2 Exposure Pathways and Receptors

Ingestion of contaminated fish is expected to be the most important exposure route for people, because many of the contaminants found in produced water are known to accumulate in fish and shellfish. The important receptors for radium discharged in produced water are recreational fishermen and their families. Recreational fishermen are important receptors because they may fish close to a platform, return often to the same fishing spot, and ingest a large percentage of fish caught near a platform. Mollusks and crustaceans are commercially important in the Gulf of Mexico, but most of the seafood caught near platforms by recreational fishermen are fish.

There may be some commercial fishing near coastal platforms but the amount of fish and shellfish impacted by contaminants discharged in produced water will be small because of the dilution with distance from a platform. Commercially caught fishes are marketed widely, making the prediction of an individual's consumption from a single source difficult (USEPA, 1990). Because the catch of sports fishermen is not diluted in this way, they represent the population most vulnerable to exposure by consumption of contaminated fishes from one location (USEPA, 1990). Some sports fishermen may sell or give away the fish they catch, but an analysis of their consumption and risk will result in a more conservative estimate of risk than an assessment of risk for the general public. Recreational fishermen may also include commercial fishermen who fish near offshore platforms and eat some of their catch.

Potential ecological receptors for contaminants in produced water include recreationally and commercially important fish and shellfish species, benthic invertebrates living close to the platforms, and threatened and endangered species living in open Louisiana bays. Potentially important exposure pathways include direct exposure in water or sediment, and ingestion in food, water or sediment.

3.3 Risk Assessment Approach

The overall approach was to use available data and modeling analyses for continuing open bay discharges, in a tiered assessment of human health and ecological risk. The initial analysis consisted of conservative screening assessments meant to identify contaminants of potential concern. More quantitative, probabilistic risk assessments were performed for contaminants identified in the screening analyses.

3.3.1 Data and Modeling Analyses

The data that form the bases of the screening and probabilistic risk assessments presented here include:

- Data collected in the ongoing USDOE field study:

  -- PAH and metal concentrations in sediment near two open bay discharges;
  -- radium concentrations in edible biota near two open bay discharges;
  -- radionuclides in the effluent of two open bay discharges; and
  -- fish ingestion rates for recreational fishermen and their families.

- Data abstracted from LDEQ permit files for open bay sites in Louisiana that plan to continue to discharge produced water until January, 1997:

  -- location, depth and discharge rate data;
  -- chemical and radium concentrations in the effluents; and
  -- results of toxicity testing on effluents.

Data and modeling analyses that form the basis of the risk assessments are described in detail in sections 4 and 5. Section 4 describes the USDOE field study. Preliminary results of sampling conducted at the two coastal sites in Louisiana are summarized. The results of the survey of recreational fishermen in Louisiana are described and a distribution of fish ingestion rates derived. These data were used in the risk assessments presented in sections 6 through 9.

Section 5 summarizes the data abstracted from the LDEQ permit files for assumed continuing open bay discharges in Louisiana. Discharge rates and platform depths are summarized. Available chemical and radionuclide effluent data are described. Data summarizing acute and chronic toxicity studies are also presented. A surface water transport model was used to estimate dilution factors with distance from the discharge, and this modeling analysis is presented. These data and modeling results were used in the risk assessments given in sections 6 through 9.

3.3.2 Human Health and Ecological Risk Assessments

Human health and ecological risk assessments are presented separately. Risk assessments for radium and other radionuclides in produced water are presented separately from assessments for chemical contaminants.

The state of Louisiana has identified a standard acute mixing zone of 50 feet, and a standard chronic and human health zone of 200 feet from produced water discharges. These distances imply a risk management decision about the "acceptable" location for environmental impacts. These distances were used in the current risk assessment.

Human Health Risk Assessment for Radium

Screening and probabilistic human health risk assessments were done for open bay discharges of radium in Louisiana.

A screening assessment was performed using worst-case estimates of: concentrations in fish, ingestion rates, and dose-response factors to determine the need for a more quantitative analysis. Based on the results of these analyses, a probabilistic risk assessment was done using distributions of: radium concentrations in fish (from field sampling and modeling); fish ingestion rates (from USDOE fishermen survey); and risk factors.

Ecological Risk Assessment for Radionuclides

This assessment used concentrations of radionuclides measured in the effluent at the two USDOE study sites, and radium concentrations reported in permit files for continuing open bay discharges. Worst-case water concentrations were predicted using a dilution factor derived from the modeling analyses presented in section 5. Predicted water concentrations were compared to screening dose-rate factors developed by IAEA (1988) . These dose-rate factors relate the radiation exposure to an organism to a unit concentration of the radionuclide in the water in which the organism lives. Estimated doses were compared to reference dose rates suggested by IAEA (1988).

Human Health Risk Assessment for Chemical Contaminants

A screening human health risk assessment was done for metals and organic compounds measured in continuing open bay discharges. This analysis followed the USEPA approach to estimating risks from toxic materials and carcinogens by applying RfD (reference dose) and slope factor values to estimates of chemical intake rates (USEPA, 1989a). Predicted water concentrations were also compared to USEPA and Louisiana human health water quality criteria.

For contaminants that were identified as being of potential concern in the screening analysis, a more quantitative risk assessment was performed, using distributions of contaminant concentrations in the discharges.

Ecological Risk Assessment for Chemical Contaminants and Effluent

Three assessments were performed in the ecological risk assessment:

1. Screening assessment of sediment toxicity: sediment metal and PAH concentrations measured at the USDOE study sites were compared to proposed sediment quality criteria.

2. Assessment of potential toxicity of individual contaminants in the water column: Worst-case predicted water column concentrations of contaminants measured in continuing open bay effluents (LDEQ permit files) were compared to USEPA and Louisiana water quality criteria. A more quantitative analysis was done for contaminants identified in the screening analysis as being of potential concern.

3. Assessment of effluent toxicity: Predicted water column concentrations of effluent were compared to results of acute and chronic toxicity tests performed in the laboratory with standard test organisms.

Section 6 presents the screening and probabilistic risk assessments for the human health effects of radium. Section 7 gives the screening assessment for ecological effects of radium and other radionuclides. Section 8 presents the screening risk assessment for the human health effects from metals and organic contaminants. The risk assessments for the ecological effects of individual produced water contaminants and effects associated with the total effluent are presented in section 9.

4 USDOE FIELD STUDY PRELIMINARY DATA

4.1 Background

This report is part of a series of studies of the human health and ecological risks associated with discharges of produced water to the Gulf of Mexico supported by USDOE. These risk assessments are coordinated with a USDOE project titled "Environmental and Economic Assessment of Discharges from Gulf of Mexico Region Oil and Gas Operations" (referred to as the "USDOE Field Study").

Continental Shelf Associates, Inc. (CSA) was contracted to conduct the field study. The objective of the project is to increase the base of scientific knowledge concerning the following topics:

- The fate and environmental effects of contaminants found in produced water;
- The economic impacts of proposed regulations on offshore oil and gas producers of the Gulf of Mexico region; and
- The catch, consumption, and human use patterns of seafood species collected from coastal and offshore waters of the Gulf of Mexico.

The study includes 4 technical tasks, two of which are relevant to the risk assessments presented here:

- Monitoring of the Recovery of Impacted Wetland and Open Bay Produced Water Discharge Sites in Coastal Louisiana and Texas (Task 4); and
- Synthesis of Seafood Catch, Distribution and Consumption Patterns in the Gulf of Mexico Region (Task 6).

Steimle & Associates, Inc. were subcontracted by CSA to perform the two tasks (Tasks 4, 6) relevant to these risk assessments. Preliminary results from Tasks 4 and 6 are available, and were used in the current analyses. The following sections summarize the preliminary data available from the Task 4 and Task 6 work, and derive or summarize the data used in subsequent sections of the report.

4.2 Open Bay Sites

The data and descriptions of the study sites were abstracted from material provided by Steimle & Associates, Inc. The emphasis in the study of coastal sites is an assessment of the recovery of these sites from any impact from produced water discharges. Data were collected prior to the termination of discharge at three sites (including the two open bay sites discussed here), and several times after the discharge was terminated. The preliminary data presented in this section are limited to those collected before termination of the discharges.

4.2.1 Site Descriptions

Delacroix Island

The Delacroix Island Oil and Gas Field located approximately 5.5 miles southeast of Delacroix, Louisiana, has been in production since the first well was drilled in the field in 1940. The area is part of a subsiding delta, which results in broken marsh and numerous small water bodies with a few large open bays. The tank battery studied (Tank Battery #1) is located in approximately 4.9 feet (1.5 m) of water. The Delacroix Island site is not located in a completely open bay, but will be used in the assessment presented in this report with the understanding that the impacts from the site may over-estimate impacts from true open bay discharges.

Salinities in the Delacroix Field vary widely between seasons and years, with late summer/fall salinities being the most stable. Spring salinities are the lowest experienced during the year due to the influence of the Mississippi River. The influence of the Mississippi River is particularly noticeable in this area because of the proximity of the Caernarvon Diversion.

The bottom substrate in areas of subsiding marsh like the Delacroix Island area varies from soft, fine grained sediments in open water to old root mat which is firmer and may persist for many years.

The Delacroix Island area is typical of many brackish habitats in Louisiana inshore waters in that its inhabitants are eurytolerant opportunistic species. Commercially important species in this area include the American Oyster (*Crassostrea virginica*), the blue crab (*Callinectes sapidus*), brown shrimp (*Penaeus aztecus*) and white shrimp (*Penaeus setiferus*).

The area around the Delacroix Field is marginal for oysters, although during some years oyster crops can be successful. Crabs are harvested extensively year round. Commercial and recreational shrimping is conducted in this area. Recreational and commercial finfishing is also popular. Red drum or redfish (*Sciaenops ocellatus*) and speckled trout (*Cynoscion nebulosus*) are the most prized species in inshore areas. Both of these species are most available in the late fall and winter months. Flounder (*Paralicthys lethostigma*) are most abundant in the fall months and Croaker (*Micropogonias undulatus*), spot (*Leiostomus xanthurus*), sand seatrout (*Cynoscion arenarius*), black drum (*Pogonias cromis*) and sheepshead (*Archosargus probatocephalus*) are fished inshore year round.

Bay de Chene

The Bay de Chene Field is located approximately 13 miles northwest of Grand Isle, Louisiana and is part of the Barataria Basin. The field has been in constant production since the first well was drilled in 1942. The tank battery studied (Tank Battery #5) is located in Hackberry Bay, a large open bay typical of the Barataria system. The discharge is located in about 7.5 feet (2.3 m) of water.

Salinities in the Bay de Chene Field vary during the year with the lowest salinities occurring when the Mississippi influences the area. The bottom substrate in most open water areas is soft fine grain sediments. Portions of the bay have been altered by the planting of Rangia shell by the Louisiana Wildlife and Fisheries for oyster culture One of these planted areas on the west side of the bay was chosen as a reference site because no drilling was allowed on shell planting sites.

The Bay de Chene habitat is mesohaline (5 to 18 ppt) most of the year, and the organisms that characterize this habitat are euryhaline and opportunistic.

Commercially harvested species are identical to those harvested at Delacroix. The American Oyster (*C. virginica*) is cultivated on numerous leases in the area. Blue crab (*C. sapidus*) are harvested year round. Brown (*P. aztecus)* and white (*P. setiferus*) shrimp are harvested commercially and recreationally.

Recreational and commercial finfishing are also conducted in this area. Red drum or redfish (*S. ocellatus*) and speckled trout (*C. nebulosus*) are the most prized species in inshore areas. Both of these species are most available in the late fall and winter months. Flounder (*P. lethostigma*) are most abundant in the fall months and croaker (*Micropogonias undulatus*), spot (*Leiostomus xanthurus*), sand seatrout (*C. arenarius*), black drum (*P. cromis*) and sheepshead (*A. probatocephalus)* are fished inshore year round.

4.2.2 Discharge and Sampling Information

Delacroix Island Tank Battery #1

Discharge rates in LDEQ files (Discharge Monitoring Reports) for 1990-1992 average 1,741 bbl/d for this site. At the time of termination (April 1993) the volume of produced water fluctuated between 1,964 and 1,978 bbl/d for the period 26 March to 19 April 1993, when there were 11 wells in production. Discharge volumes from 19 to 25 March ranged from 2,246 to 2,256 bbl/d, with 12 wells in production.

Sampling at the Delacroix Island study site was conducted according to the station layout shown in Figure 4-1. Biota were collected using otter trawls, gill nets and crab traps at the two reference stations (R1 and R2) and the discharge station. Only species of commercial or recreational importance were retained. Animals were placed on ice and frozen within 12 hours of collection.

Bay de Chene Tank Battery #5

The LDEQ data base shows a one-time sampling record of 3,666 bbl/d. This discharge terminated on 15 October 1993. At the time of the pre-termination survey, data provided by Texaco indicated that the discharge was for four wells, with a discharge volume of 3,825 bbl/d.

Sampling at the Bay de Chene study site was conducted according to the station layout in Figure 4-2. Biota were collected using otter trawls, gill nets and crab traps at the two reference stations and the discharge station. Only species of commercial or recreational importance were retained. Animals were placed on ice and frozen within 12 hours of collection.

4.2.3 Radionuclides in Water and Biota

Average concentrations of radionuclides in the discharges are given in Table 4-1. Maximum concentrations of ^{226}Ra and ^{228}Ra measured in croaker, spot, sea trout, blue crab and shrimp at the discharge and highest of the reference stations for each site are given in Table 4-2. Preliminary results of tissue analyses for ^{226}Ra and ^{228}Ra are given in Appendix A.

Table 4-1. Concentrations of radionuclides measured in discharge at Delacroix Island and Bay de Chene study sites.

Radionuclide	Delacroix Island (pCi/l)	Bay de Chene (pCi/l)
^{210}Pb	60.3	78.0
^{210}Po	<2.0	<1.1
^{226}Ra	162.5	218.5
^{228}Ra	317.5	264.5
^{228}Th	15.0	20.5

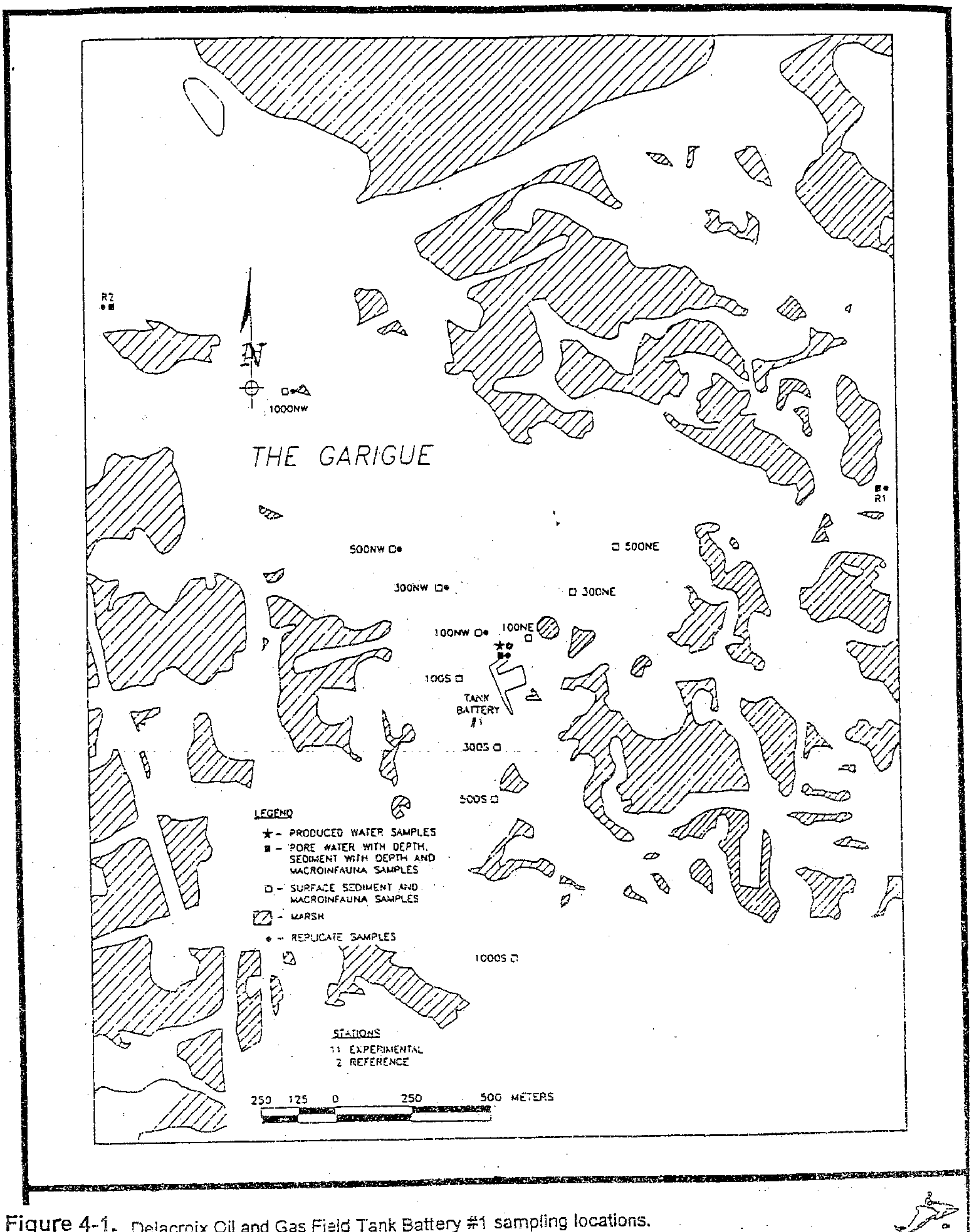


Figure 4-1. Delacroix Oil and Gas Field Tank Battery #1 sampling locations.

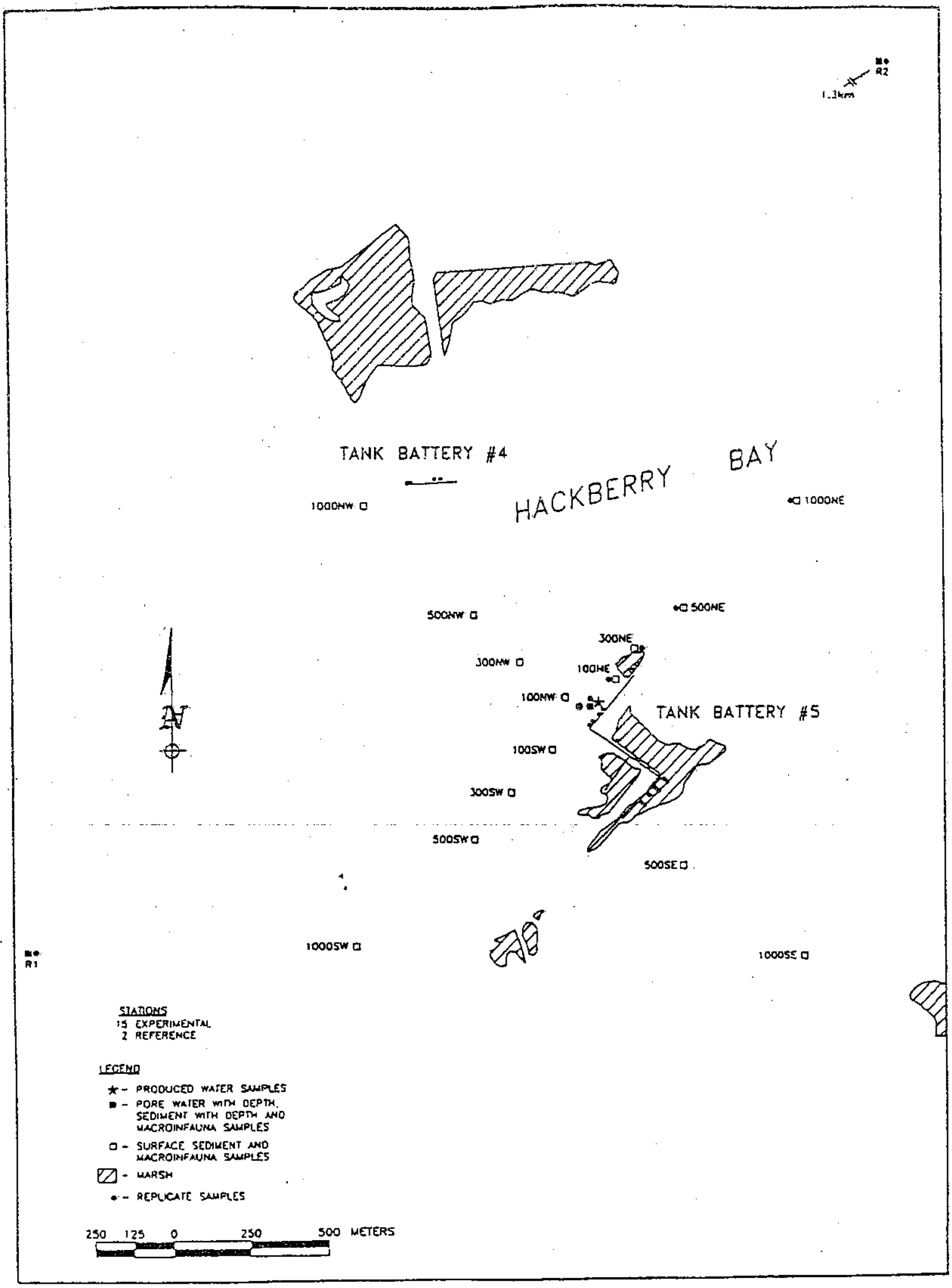


Figure 4-2. Bay De Chene Field Tank Battery #5 sampling locations.

Table 4-2. Maximum radium concentrations measured in biota from the Delacroix Island and Bay de Chene study sites (pCi/g) (pre-termination samples).

	Delacroix Island				Bay de Chene			
	Discharge		Reference		Discharge		Reference	
	^{226}Ra	^{228}Ra	^{226}Ra	^{228}Ra	^{226}Ra	^{228}Ra	^{226}Ra	^{228}Ra
croaker	0.025	0.037	0.018	0.112	0.024	0.094	0.032	0.05
spot	0.005	0.027	0.002	0.076	0.034	0.086	0.029	0.042
sea trout	NS	NS	NS	NS	0.021	0.159	0.016	0.036
blue crab	0.013	0.032	0.025	0.09	0.023	0.059	0.024	0.01
shrimp	NS	NS	NS	NS	0.011	0.026	0.027	0.124

NS = no sample

4.2.4 Chemicals in Sediment

Preliminary results of the chemical analyses (PAHs and metals) of sediments are given in Appendix A.

4.2.5 Benthos Sampling

Both pre- and post-termination benthos were collected at the study sites, and preliminary data are available. The study (Mulino *et al.,* 1995; 1996) found depressed numbers of species and individuals at and near the discharge during the pre-termination sampling, suggesting an impact on the benthos between 0 and 100 meters from the platform.

4.3 Fishermen Survey

4.3.1 Survey and Overall Results

The following material and data from the fishermen survey were abstracted from Steimle & Associates, Inc.(1995).

Commercial fishermen (including oystermen) and recreational fishermen were surveyed by personal interview from May through November 1993 to determine categories of seafood taken over the previous three months, types of license(s) held, and information on the number, gender and ages of individuals in the household and their seafood consumption habits. Respondents were also interviewed about locations fished, estimated distances from oilfield structures, and species caught.

To determine the distribution of the catch, all fishermen were asked to estimate by species the percentage sold, the percentage given away to others, and the percentage kept for personal consumption. Fishermen were also asked to estimate the frequency of seafood consumption and cooking methods employed.

Processing plants and wholesalers in Texas and Louisiana were surveyed to determine their sources of seafood (i.e. in-state vs. out-of-state), and the origin of the seafood sold (i.e. fishing zones and ports of commercial fishermen). Site surveys of seafood retailers were conducted to determine the types of shellfish and saltwater finfish sold, the parts of the seafood sold, and the types of prepared seafood sold. Restaurant surveys asked respondents about the source, quantities and method of preparation of seafood sold/served by the restaurant.

Finfishing was the most popular form of recreational fishing (95%) with most fishermen possessing an in-state license (92%). The majority of respondents fished from a private boat inshore (62%), often near an oilfield structure, and most commonly caught speckled sea trout and red snapper.

On average, fishermen reported keeping 80% of the finfish; 97% of the blue crab catch; and 83% of shrimp for personal consumption. They reported serving seafood 1.8 times per week on average. Their preference was to consume the meat only from the fish over 90% of the time, and the most popular cooking method was frying (30%).

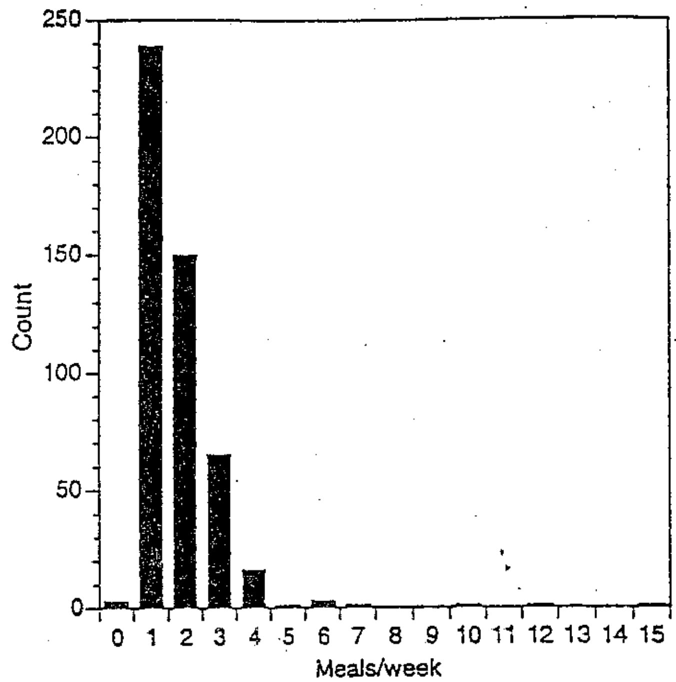
4.3.2 Estimation of Intake Rates

Variables needed for the human health risk assessment include those that contribute to an estimate of the ingestion rate of fish caught near (less than 1,000 ft; 300 m) a coastal platform in Louisiana. Data collected by the survey (Steimle & Associates, Inc., 1995) include the following:

- amount of fish caught per trip
- number of seafood eaters in fishermen's family
- number of trips near structures
- number of trips inshore vs. offshore
- fraction of catch kept
- number of days since last seafood meal
- number of times per week fish served

In this assessment, ingestion rates for recreational fishermen of fish caught near coastal platforms were derived from the reported data on meals per week (Figure 4-3). The original data set contained a single value of 22 meals/week that was excluded as an outlier. A lognormal distribution was assumed for meals/week (arithmetic mean of 1.8, a standard deviation of 1.3, and a range of 0 to 15).

Figure 4-3. Number of times per week recreationally caught fish served.



The ingestion rate distribution for recreational fishermen and their families was derived as follows:

$$I_{fish} = \frac{M \times MS}{7d \times week^{-1}} \quad (4.1)$$

where:
I_{fish} = derived ingestion rate (g/d)
M = meals per week
MS = meal size (150 g/meal; USEPA, 1989a).

The resulting lognormal distribution (Table 4-3) was used to estimate exposures to recreational fishermen and their families. For some contaminants (lead in particular), the subpopulations with highest susceptibility to adverse health effects are infants and young children. USEPA (1990) reported data for intake rates of seafood by the population consuming seafood, obtained in a survey conducted over a period of one year (1973-1974). For juveniles (0-9 years of age), the rate of seafood ingestion was approximately 43% that of the general population. The intake rate distribution derived for recreational fishermen and their families was multiplied by a factor of 0.43 to estimate the rate of juvenile ingestion of fish (Table 4-3).

Table 4-3. Derived lognormal intake distributions for fish caught near open bay platforms.

	Intake (g/day)	
	Recreational Fishermen and Families	**Children**
mean	38.4	16.6
median	31.5	13.6
standard deviation	26.4	11.6
minimum	3.3	1.3
maximum	228.6	115.7
95th percentile	89.5	38.5

5 CHARACTERIZATION OF CONTINUING DISCHARGES

5.1 Identification Of Continuing Discharges

Louisiana regulations (Title 33, March 20, 1991) required the termination of all produced water discharges to natural or man-made water bodies located in intermediate, brackish or saline marsh areas after January 1, 1995, unless the discharge (s) were authorized in an approved schedule for elimination or effluent limitation compliance. A variance through January, 1997 was granted (12/16/94) for permitted discharges located in open waters at least 1 mile from any shoreline in Chandeleur Sound, Breton Sound, Barataria Bay, Caminada Bay, Timbalier Bay, Terrebonne Bay, East Cote Blanche Bay, West Cote Blanche Bay or Vermillion Bay.

The Louisiana Department of Environmental Quality (LDEQ) identified produced water discharges in open bay areas (Table B-1 in Appendix B) that may qualify for this variance.

In August, 1994, a telephone survey of the operators was conducted to determine if they would take advantage of an extension of the phase-out rule for coastal Louisiana produced water discharges. Most operators indicated that they would continue to discharge through 1997 if allowed. Discharges that planned re-injection or had been shut-in were not included in the current assessment (Table B-1, Appendix B). Some operators could not say what company policy would be if an extension were granted. These discharges were assumed to continue discharging, although they may have since been terminated. Therefore, the list of continuing open bay discharges used in the current assessment may include wells that are no longer active.

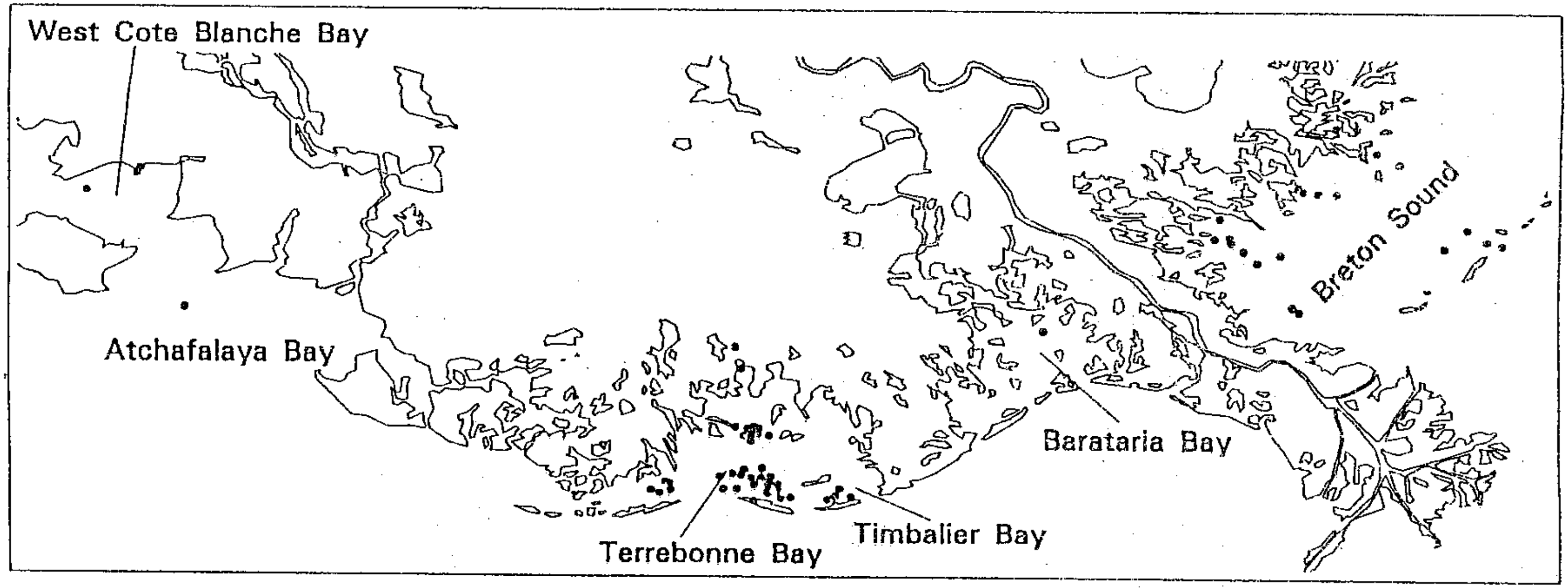
Figure 5-1 shows the locations of the assumed active discharges in open Louisiana bays. More detailed maps are given in Appendix B.

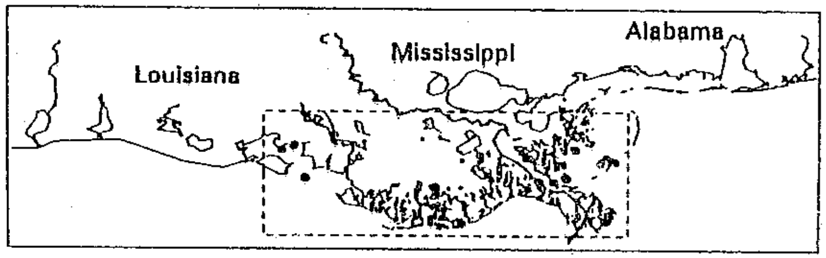
5.2 Characterization Of Discharges

5.2.1 Data Sources

Data describing the assumed continuing discharges listed in Table B-1 (Appendix B) and shown in Figure 5-1 were abstracted from LDEQ permit files. Table B-2 in Appendix B summarizes the data available for each discharge. A few permit files were not available.

Figure 5-1. Assumed active discharges in open Louisiana bays.





5.2.2 Depths and Discharge Rates

Information critical to an assessment of the environmental impact from a produced water discharge includes the depth of the platform and the rate of discharge. Higher rates of discharge in shallower waters can be expected to have more impact in terms of both human health and ecological effects than smaller discharges in deeper waters where dilution is greater.

Table 5-1 summarizes the data for platform depths and discharge rates. The total discharge rate data set is described in Table 5-1. High (>5,000 bbl/d) and low (≤5,000 bbl/d) discharge rates (Table 5-1) were described as lognormal distributions (Figure 5-2). Figure 5-3 shows the distribution of platform depths in the data set. Table B-2 in Appendix B gives the depth and discharge rate for each discharge point included in the analysis.

Table 5-1. Platform depths and discharge rates.

	Depth (feet)	Discharge Rate (bbl/d)		
		All discharges	≤ 5,000	> 5,000
number	29	62	46	16
mean	9.1	4,527	999	13,865
standard deviation	2.3	7,166	1,249	7,991
minimum	4	1	1	5,364
maximum	18	37,113	4,914	37,113

Note that the two coastal sites in the USDOE study are reasonably representative of these discharges, falling on the high end of the distribution for low discharge rates, and the low end of water depths (2,000 and 4,000 bbl/day; 5 and 7.5 feet).

5.2.3 Contaminants in the Effluent

Chemical contaminants measured in open bay produced water discharges and reported in LDEQ permit files are summarized in Table 5-2. Data abstracted from LDEQ permit files for each discharge site are given in Appendix B, Table B-3. These data are for contaminants that were above the detection limit only, and overestimate the mean concentration in the data set. These data are the most current measurement data for each discharge. These data are uncertain because many permits have more than one discharge, and it was often difficult to relate the chemical concentration data to the correct discharge point. They are also uncertain because concentrations change over time, and a single sample may be of limited value.

Radium concentrations measured in the discharges are given in Table B-4 in Appendix B, and are summarized in Table 5-3. This data set suggests no clear relationship between ^{226}Ra and ^{228}Ra concentrations in the effluent (Figure 5-4).

Figure 5-2. Lognormal Tests: discharge rates of continuing open bay discharges; A, 1 to 5,000 bbl/day (r = 0.8049); B, >5,000 bbl/day (r = 0.9514).

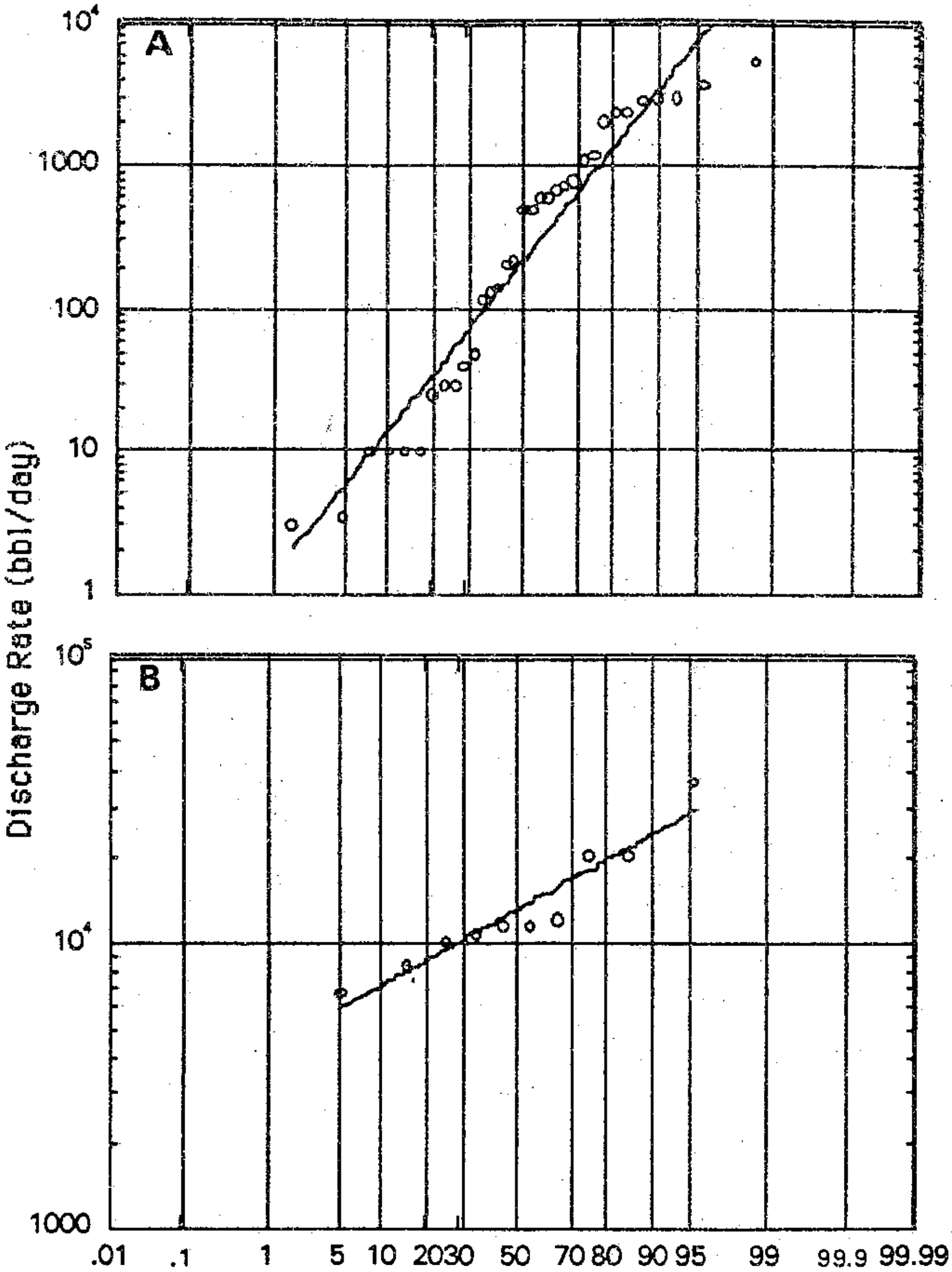


Figure 5-3. Depths of platforms, continuing open bay discharges.

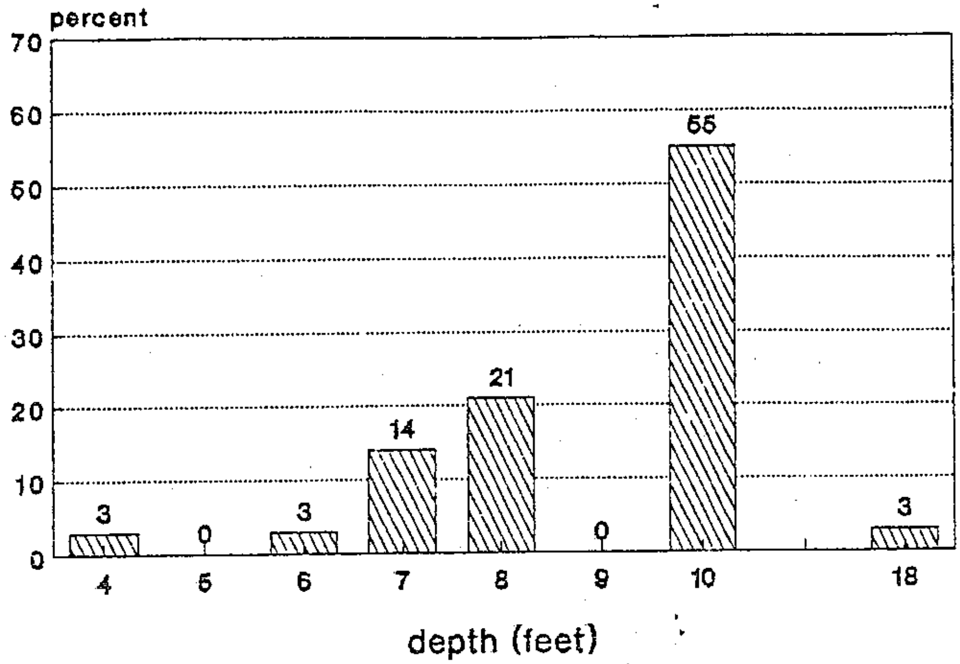


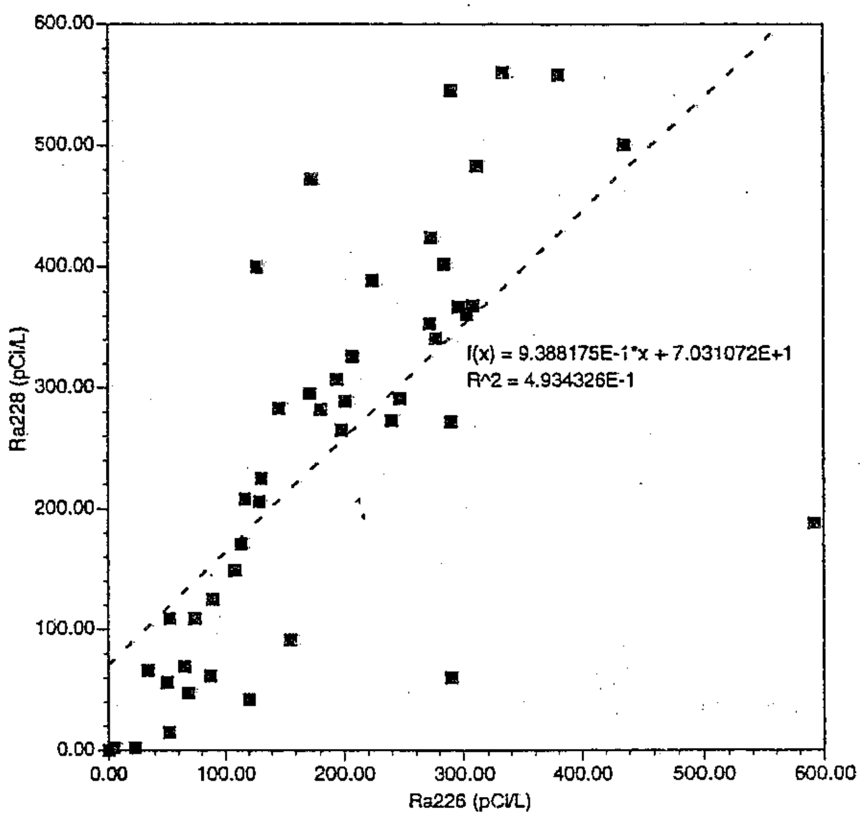
Table 5-2. Contaminant concentrations (µg/L) in open bay produced water discharges in Louisiana (for contaminants reported above detection limits).

	count	minimum	maximum	mean	std dev
METALS					
Antimony	7	11.85	20100	5595.91	8479.477
Arsenic	11	6.9	498.5	74.74	136.76
Cadmium	6	0.93	500	231.19	202.57
Chromium (VI)	6	9.5	200	83.49	70.09
Copper	11	10	710	288.37	197.93
Lead	7	35.36	829000	104263	292839
Mercury	4	0.007	27	7.08	11.26
Nickel	7	57.90	2840	1013.86	1062.08
Selenium	3	11.00	84	63.00	34.79
Silver	5	11.30	400	143.32	160.09
Thallium	4	248.39	3700	1904.74	1535.71
Zinc	12	31.09	6375	1217.10	2102.65
ORGANICS					
Benzene	12	10	9550	1813.23	2690.15
Bis (2-ethylhexyl) phthalate	6	45	80	59.67	12.40
Naphthalene	5	10	118	57.42	41.65
Phenol	13	24	12000	1557.86	3144.72
Toluene	12	16	2800	831.62	944.56
Xylenes	9	7	862	183.30	265.84

Table 5-3. Radium concentrations (pCi/l) in open bay discharges.

	^{226}Ra	^{228}Ra
number	47	47
mean	191.4	250.0
standard deviation	122.4	163.6
minimum	0.0	0.0
maximum	592.0	560.0

Figure 5-4. Relationship between ^{226}Ra and ^{228}Ra concentrations in effluents.



5.2.4 Effluent Toxicity

Toxicity tests are useful tools because they can directly measure potential aquatic effects. This is particularly true in the case of complex effluents, such as produced water, where a broad range of toxicants can be present at low levels.

Toxicity data were available in LDEQ permit files for 58 assumed continuing discharge sites. Data were available for acute toxicity tests (96-hr LC_{50}) on *M. bahia* (a shrimplike mysid crustacean) and *Cyprinodon variegatus* (sheepshead minnow); 7-day chronic growth and survival NOEL tests on the same two species; and fecundity studies on *M. bahia*. The acute LC_{50} data and NOEL growth and survival data are summarized in Tables 5-4 and 5-5.

These data are uncertain because many permits have more than one discharge, and it was often difficult to relate the toxicity data to the correct discharge point. They are also uncertain because concentrations change over time, and a single sample may be of limited value.

Table 5-4. Results (percent effluent) of acute toxicity (LC_{50}) tests, *Mysidopsis bahia* and *Cyprinodon variegatus*.

	Mysidopsis bahia	*Cyprinodon variegatus*
N	41	39
mean	9.5	24.4
median	7.9	18.5
standard deviation	11.0	38.2
minimum	0.2	2.4
maximum	71.2	250

Table 5-5. Results (NOEL, growth and survival, percent effluent) of chronic toxicity tests.

	Mysidopsis bahia		*Cyprindon variegatus*	
	survival	growth	survival	growth
N	43	42	41	39
mean	2.9	4.0	7.1	9.0
median	2.2	3.6	6.9	7.5
standard deviation	2.9	3.5	5.7	6.9
minimum	4×10^{-4}	0.1	0.2	0.2
maximum	11.4	12.1	19.1	25.2

5.3 Transport Modeling

The USEPA surface water transport model CORMIX 2.1 (Cornell Mixing Zone Expert System Model; Doneker and Jirka, 1990) was used to estimate the dilution expected at 50 and 200 feet from open bay discharges. The CORMIX model may be used for the prediction of aqueous toxic or conventional pollutant discharges to surface water bodies. Its major emphasis is on prediction of plume geometry and dilution within an initial mixing zone, but the model also predicts plume behavior at larger distances (Bouchard *et al.*, 1995). The current version allows simulation of submerged or surface, single and multiport discharges. CORMIX has been used by USEPA in rulemaking for produced water discharges.

Table 5-6 summarizes the input parameters used in the analysis. A depth of 8 feet (2.44 m) was chosen to represent the assumed continuing open bay discharges in Louisiana (see Figure 5-3). A range of discharge rates was modeled (Table 5-7) to cover the range of discharge rates for the open bay discharges (see Figure 5-2).

Because of the shallow depth, the model was run using an unstratified scenario with a surface and bottom water density of 1005 kg/m^3. These values were derived from temperature and salinity data published in literature reviewed by USEPA (USEPA, 1995a). A produced water discharge density of 1020 kg/m^3 was derived from USEPA's review of produced water effluent density estimates, and an ambient velocity of 0.05 m/s was used (USEPA, 1995a).

CORMIX forces a submerged single port discharge to be in the bottom 1/3 of the water column. The model was run with the discharge pipe pointing straight up from the lower 1/3 of the water column. This is unrealistic for produced water discharges, because they are normally released on or close to the surface. Our decision to run the model with this discrepancy was based on the assumption that differences in dilution rates resulting from a discharge pointing up at the surface or down toward the bottom in a shallow bay environment would be negligible.

To test this assumption, sensitivity runs using altered input parameters were run to "fool" the model into simulating a more accurate scenario. The model can be adjusted to make the projections more accurate by creating a mirror image using a stratified water column and inverting the ambient densities (Avanti Corporation, 1993). Specifically, the depth was increased from 2.44 m to 3.44 m, the discharge pipe was placed at 2.44 m with the theta angle at 90°, pointing straight up (i.e., a mirror image of effluent being discharged directly onto the surface). To complete this mirror imaging, the effluent had to be changed from a negatively buoyant plume (i.e., surface to bottom) to one with a positive buoyancy. The water column data was modeled as stratified with surface density at 1018 kg/mg^3 and the bottom density at 1020.15 kg/mg^3. The discharge density was then reduced to 970 km/m^3. The resulting scenario was modeling a plume traveling the entire depth of the receiving environment from the bottom to the surface, simulating the same characteristics as a surface discharge of a negatively buoyant effluent. Results of this sensitivity analysis indicated that differences in predicted dilution rates are negligible. The dilution factor for a worst case scenario of 37,500 bbl/day discharge at 200 ft is 13.8 as opposed to 12.0 for the unaltered input parameters.

Table 5-6. CORMIX input parameters.

AMBIENT PARAMETERS	
cross section	unbounded
average depth	2.44 m
depth at discharge	2.44 m
ambient velocity	0.05 m/s
Darcy-Weisbach friction factor	0.0524
Manning's friction factor	0.03
wind velocity	2 m/s
stratification type	unstratified
surface density	1005 kg/m^3
bottom density	1005 kg/m^3
DISCHARGE PARAMETERS	
discharge description	submerged single port
nearest bank	left
distance to bank	1609.76 m
port diameter	0.127m
port cross-section area	0.0126m^2
discharge flow rate	100 - 37,500 bbl/day
discharge port height	0.8 m
vertical discharge angle	90 degrees
horizontal discharge angle	0 degrees
discharge density	1020 kg/m^3
density difference	-15 kg/m^3
buoyant acceleration	-0.1464 m/s^2
discharge concentration	100 percent
surface heat exchange coeff.	0 m/s
coefficient of decay	0 m/s

CORMIX uses a 13 step procedure to determine the flow category of a discharge. CORMIX classified the flow as "NV5" for discharge rates between 7,500 bbl/day and 37,500 bbl/day, and as "NV2" for discharge rates up to 5000 bbl/day. Both of these classifications show that the model treated the discharge as a negatively buoyant discharge in a uniform ambient layer. Class NV2 has an extremely strong negative buoyancy causing upstream spreading and does not have layer or surface interaction. Class NV5 has an interaction and unstable discharge configuration with vertical mixing and recirculation zones. After determining the flow classification CORMIX selects an algorithm that best represents the discharge scenario (Doneker and Jirka, 1990). The NV5 algorithm did not predict dilutions at 50 feet from the discharge.

Results are presented in terms of the expected dilution factor in the plume at 50 and 200 feet (Table 5-7) where :

DF (dilution factor) = Concentration in Effluent / Concentration in Water

These data (Table 5-7) were used to derive empirical relationships between discharge rates and dilution factors (Figure 5-5):

For discharge rates $\leq$ 5000 bbl/d

$DF_{50\,ft} = 10633 * (\text{DISCHARGE RATE})^{-0.867}$ (R=0.997)

$DF_{200\,ft} = 46303 * (\text{DISCHARGE RATE})^{-0.946}$ (R=0.9997)

For discharge rates > 5000 bbl/d

$DF_{200\,ft} = 36061 * (\text{DISCHARGE RATE})^{-0.762}$ (R=0.9997)

In modeling the dilution factors at 200 ft, CORMIX automatically switched from the NV2 to the NV5 algorithm, at release rates greater than 5000 bbl/d. Table 5-7 shows that there is a 100% increase in $DF_{200\,ft}$ in the transition from 5000 bbl/d to 7500 bbl/d. The $DF_{200\,ft}$ derived from hypothetical release rates between >5000 and <7500 bbl/d were not a good fit to the empirical relationship derived from the NV5 algorithm results. An attempt to fit these release rates to the relationship derived from the NV2 algorithm also yielded a poor fit. Only three of the assumed continuing open bay discharges (Appendix B, Table B-2) fell into this transition (5365 bbl/d; 6800 bbl/d, 7368 bbl/d). In the risk analysis, we opted to use $DF_{200\,ft}$ values derived by the NV5 algorithm for these discharges, with the assumption that any overestimates of dilution would be offset by the conservatism of the CORMIX model.

Table 5-7. Estimates of dilution factors in the plume at 50 and 200 feet.

Discharge Rate	Dilution Factor		CORMIX Flow Class
(bbl/d)	50 feet[1]	200 feet	
1	14661	33539	NV2
3	6561.6	76824	NV2
5	3514.6	22471	NV2
10	2385.5	10016	NV2
25	771.30	6294.3	NV2
50	350	3002.7	NV2
100	168.3	1135.5	NV2
200	85.2	435.4	NV2
500	36.0	127.5	NV2
1000	19.7	53.4	NV2
2000	11.4	24.4	NV2
3000	9.4	17.3	NV2
4000	11.2	17.9	NV2
5000	13.0	19.1	NV2
7500	-	41.0	NV5
10000	-	32.3	NV5
12500	-	27.1	NV5
15000	-	23.5	NV5
22500	-	17.3	NV5
37500	-	12.0	NV5

[1]NV5 does not predict a dilution factor at 50 feet.

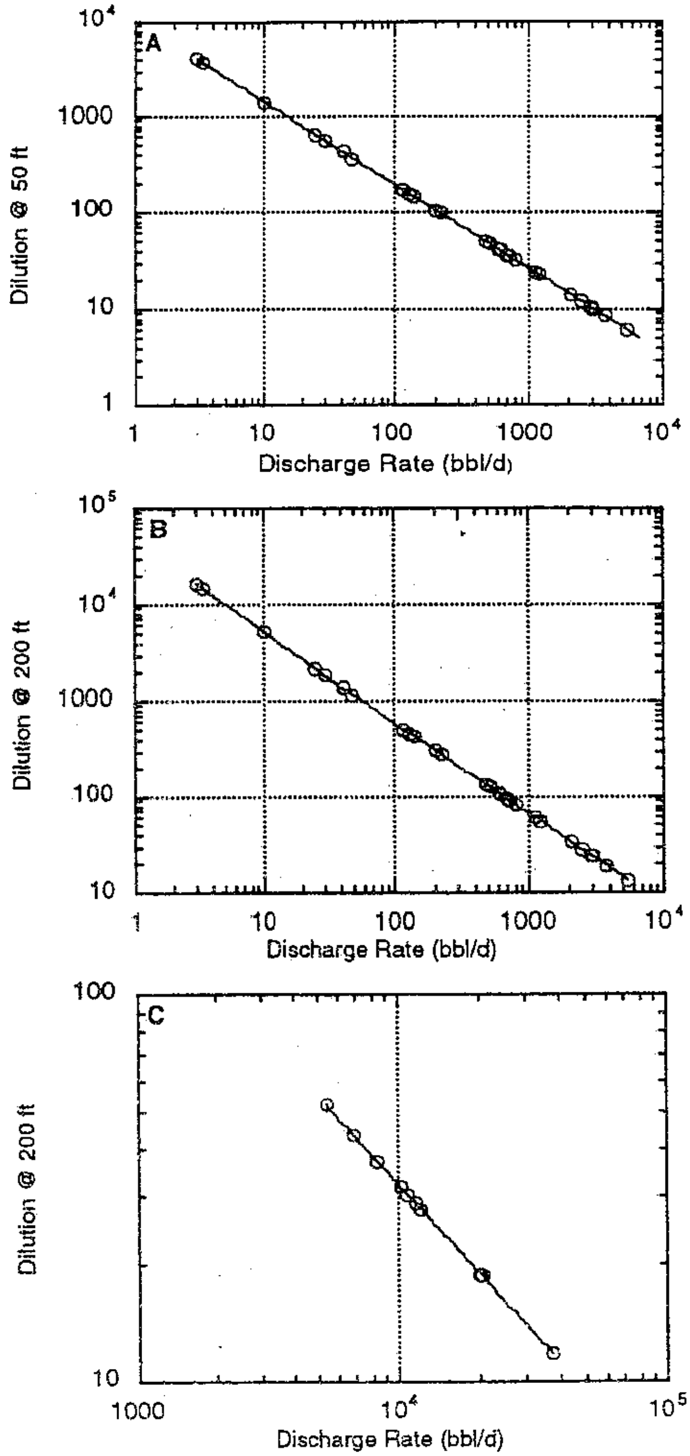
While low discharge rates (1 and 3 bbl/d) yielded good fits to the empirical relationship derived for $DF_{50\ ft}$, they yielded poor fits to relationship derived for $DF_{200\ ft}$. Therefore, $DF_{200\ ft}$ for 1 and 3 bbl/d was calculated using the relationship derived for 5 to 5000 bbl/d.

The empirical relationships were applied to the distribution of discharge rates for the open bay discharges (Table 5-1) to produce a distribution of dilution factors for 50 and 200 feet (Table 5-8). The dilution factor distributions were also used to develop a distribution of percent effluent expected in the water column at 50 and 200 feet (Table 5-8).

Table 5-8. Dilution factors and effluent concentrations (percent effluent) in the plume for open bay discharges.

	50 feet		200 feet			
	≤ 5000 bbl/d		≤ 5000 bbl/d		> 5000 bbl/d	
	Dilution Factor	**Percent Effluent**	**Dilution Factor**	**Percent Effluent**	**Dilution Factor**	**Percent Effluent**
mean	537.7	0.03	198.1	0.01	30.1	0.04
median	74.4	0.02	207.0	0.01	29.1	0.03
standard dev.	999.1	0.04	3935.9	0.02	11.4	0.02
minimum	8.5	0.0002	19.4	0.0001	11.9	0.02
maximum	4102.6	0.12	16378.0	0.05	5.2	0.08
95th percentile	2451.0	0.10	9431.0	0.04	47.7	0.07

Figure 5-5. Relationships between discharge rates and model-derived dilution factors in the plume at 50 and 200 feet from discharges: A and B, NV2 algorithms; C, NV5 algorithm.



6 HUMAN HEALTH RISK ASSESSMENT FOR RADIUM

6.1 Introduction and Approach

Radium may be accumulated by aquatic organisms, and there is a potential human health risk associated with the ingestion of radium in fish and shellfish caught near open bay produced water discharges. Screening and quantitative probabilistic human health risk assessments were done for open-bay radium discharges in Louisiana.

The two data sets used in this risk assessment were:

- measured concentrations of ^{226}Ra and ^{228}Ra in finfish and crustaceans (pCi/g) caught near the discharge at the Delacroix Island and Bay de Chene study sites (pre-termination data; section 4); and
- measured concentrations of ^{226}Ra and ^{228}Ra in 47 continuing open bay discharges (pCi/l, section 5).

6.2 Screening Assessment

6.2.1 Concentrations in Edible Seafood

Biota Near USDOE Open Bay Study Sites

Biota were collected in Spring 1993 from two USDOE study platform locations (Delacroix Island, Bay de Chene) and two reference stations for each platform. Screening assessments were done on radium measured in these biota.

Only one value for each isotope was available for each species sampled from each site at Delacroix Island (Table 6-1). For each isotope in each species, the value of the concentration at the discharge site and the higher of the two reference site values were used in the screening analysis. Multiple samples were taken for each species in the study at Bay de Chene. The highest concentrations of radium detected in each species at each site (Table 6-1) were used in the screening analysis.

Table 6-1. Maximum radium concentrations measured in biota at Delacroix Island and Bay de Chene Study Sites (pCi/g).

	Delacroix Island				Bay de Chene			
	Discharge		Reference		Discharge		Reference	
	^{226}Ra	^{228}Ra	^{226}Ra	^{228}Ra	^{226}Ra	^{228}Ra	^{226}Ra	^{228}Ra
croaker	0.025	0.037	0.018	0.112	0.024	0.094	0.032	0.05
spot	0.005	0.027	0.002	0.076	0.034	0.086	0.029	0.042
sea trout	NS	NS	NS	NS	0.021	0.159	0.016	0.036
blue crab	0.013	0.032	0.025	0.09	0.023	0.059	0.024	0.01
shrimp	NS	NS	NS	NS	0.011	0.026	0.027	0.124

NS = no sample

Fish Near Continuing Discharges

Mean and maximum radium concentrations from the data set for continuing open bay discharges were used to estimate water concentrations in the plume at 200 feet (Table 6-2). A conservative dilution factor of 20 was chosen to estimate worst-case water concentrations. A dilution factor of 20 was chosen to estimate worst-case concentrations because it yields concentrations similar to the worst-case concentrations predicted by the CORMIX model (section 5) at 50 and 200 feet from the discharge. A conservative bioaccumulation factor of 100 (IAEA, 1982) was used to calculate concentrations of radium in edible fish:

$$CF = \left(BAF \times CW \right) \times \frac{1l}{1{,}000g} \qquad (6.1)$$

where:
CF= radium concentration in fish (pCi/g)
BAF = bioaccumulation factor (100)
CW = radium concentration in water (pCi/l)

Estimated concentrations in edible fish for mean and maximum radium discharge concentrations are given in Table 6-2. The estimated concentrations in fish (Table 6-2) are based on a series of conservative models and assumptions and are significantly higher than radium concentrations measured in field studies (e.g. Table 6-1).

6.2.2 Exposure Assessment

The screening analyses used a conservative value of 70 years as the exposure period. A conservative ingestion rate of 132 g/d was used (USEPA 1989a; 95th percentile value). Exposure was calculated for ^{226}Ra and ^{228}Ra separately as:

$$I_{Ra} = I_{fish} \times [Ra]_{fishes} \qquad (6.2)$$

where:
I_{Ra} = radium intake rate (pCi/d)
I_{fish}= intake rate of fish (132 g/d)
$[Ra]_{fishes}$ = concentration of radium in fishes (pCi/g)

Table 6-2. Screening analysis; estimated water and fish concentrations 200 feet from continuing open bay discharges.

	Effluent (pCi/l)	Water (pCi/l)	Fish (pCi/g)
^{226}Ra			
mean	191.4	9.6	1.0
maximum	592	29.6	3.0
^{228}Ra			
mean	250	12.5	1.3
maximum	560	28	2.8

6.2.3 Dose-response Assessment

USEPA (Federal Register, 1991) uses risk factors of 4.4×10^{-6} for ^{226}Ra and 3.8×10^{-6} for ^{228}Ra (individual lifetime fatal cancer risk per pCI/l of drinking water), assuming an intake rate of 2 l/d of drinking water. These risk factors can be converted to units of individual lifetime fatal cancer risk per pCi/d by dividing by 2, resulting in unit risk factors of 2.2×10^{-6} for ^{226}Ra and 1.9×10^{-6} for ^{228}Ra (per pCi/d). These unit risk factors were used in the screening analyses.

6.2.4 Risk Characterization

Individual lifetime fatal cancer risks were calculated separately for ^{226}Ra and ^{228}Ra and then summed. Individual lifetime risk of cancer mortality (ILR) was calculated as:

$$ILR = I_{Ra} \times RF \qquad (6.3)$$

where:
ILR = individual incremental lifetime fatal cancer risk
I_{Ra} = radium intake rate (pCi/d)
RF = risk factor (risk per pCi/d, 70 year exposure period)

6.2.5 Results

Results of the screening risk assessments for radium measured at the Delacroix Island and Bay de Chene study sites, and for the continuing open bay discharges are given in Table 6-3.

Table 6-3. Screening human health risk assessment for Delacroix Island and Bay de Chene study sites, and modeled continuing discharges; individual lifetime fatal cancer risk.

	Delacroix Island		Bay de Chene		Modeled Discharges	
Species	**Discharge**	**Reference**	**Discharge**	**Reference**	**Mean**	**Maximum**
croaker	1.6×10^{-5}	3.3×10^{-5}	3.0×10^{-5}	2.1×10^{-5}	--	--
spot	8.2×10^{-6}	2.0×10^{-5}	3.1×10^{-5}	1.9×10^{-5}	--	--
sea trout	NS	NS	4.6×10^{-5}	1.4×10^{-5}	--	--
blue crab	1.2×10^{-5}	3.0×10^{-5}	2.2×10^{-5}	9.4×10^{-6}	--	--
shrimp	NS	NS	9.7×10^{-6}	3.9×10^{-5}	--	--
fish	--	--	--	--	6.2×10^{-4}	1.6×10^{-3}

NS=no sample

Estimated risks in the screening analysis for the ingestion of radium in fishes exceed 1×10^{-6} in all cases. Note that estimated cancer risks from eating seafood sampled at reference stations at Delacroix Island and Bay de Chene are similar to those for ingestion of seafood caught near the discharges (pre-termination).

For the modeled continuing discharges, maximum predicted risks are greater than 1×10^{-3}. These results do not represent reasonable estimates of risk because of the conservative nature of the screening level assessment, suggesting a need for a more detailed, probabilistic assessment. This quantitative assessment is presented in the following section.

6.3 Probabilistic Assessment

6.3.1 Exposure Assessment

6.3.1.1 Concentrations in Edible Fish

USDOE Open Bay Sites

Preliminary data on concentrations of radium in muscle from fishes sampled at the discharge sites were assumed to conservatively represent the concentrations in edible flesh of all fishes caught by recreational fishermen.

Distributions for radium concentrations in finfish at Delacroix Island and Bay de Chene were derived for the probabilistic human health risk assessment. At Delacroix Island, only one croaker and one spot were sampled. Therefore the concentrations (pCi/g) of ^{226}Ra (0.005, 0.025) and the concentrations of ^{228}Ra 0.027, 0.037) were used to represent the concentration of radium in fish, with equal probabilities for the values from the two species.

For the three species of finfish sampled (croaker, spot and seatrout) at the Bay de Chene discharge, the range of all values of ^{226}Ra in muscle could not be distinguished from a normal distribution, while those for ^{228}Ra fit a lognormal distribution. The combined values for ^{226}Ra concentrations were assumed to be a normal distribution, averaging 0.015 pCi/g (range, 0.004 to 0.034). For ^{228}Ra the combined values were assumed to be a lognormal distribution averaging 0.067 pCi/g (range, 0.018 to 0.159).

Continuing Discharges

Radium concentrations in edible fish were estimated for an assumed continuation of open bay discharges in Louisiana in two steps.

In the first step, the distribution of radium water concentrations in the plume was estimated by modifying the distribution of ^{226}Ra and ^{228}Ra concentrations reported for the open bay discharges (Table 5-3) by a distribution of dilution factors derived for the plume at 200 feet using the CORMIX model (section 5; Table 5-8).

Radium concentrations in fish (in the plume at 200 feet) were then derived applying the bioaccumulation factor method in equation (6.1).

A BAF distribution based on data collected in coastal Louisiana (Meinhold and Hamilton, 1992) was used to estimate radium concentrations in fish. This distribution is lognormal, has a range of 2 to 100, a mean of 30.4 and a standard deviation of 28. Table 6-4 gives the estimated distributions for radium concentrations in fish. These values over-estimate the concentration of radium in fish near open bay platforms because they use concentrations predicted in the plume, not average concentration in the water column.

Table 6-4. Estimated radium concentrations in water and fish in modeled plumes 200 feet from open bay discharges.

	Water Concentration (pCi/l)		Fish Concentration (pCi/g)	
	^{226}Ra	^{228}Ra	^{226}Ra	^{228}Ra
mean	5.6×10^{-1}	6.7×10^{-1}	1.5×10^{-2}	1.9×10^{-2}
median	2.4×10^{-1}	3.0×10^{-1}	5.2×10^{-3}	6.4×10^{-3}
std. dev	9.8×10^{-1}	1.1	3.3×10^{-2}	4.2×10^{-2}
95th percentile	2.1	2.5	6.1×10^{-2}	7.6×10^{-2}

Fish Away From Platforms

For comparison, risks from ingestion of fish caught away from platforms in the Gulf of Mexico were estimated. Radium concentrations in fish not associated with platforms were assumed to be uniformly distributed, with a range of 0 to 0.01 pCi/g (Meinhold *et al.,* 1995).

6.3.1.2 Ingestion Rates

Ingestion rates for recreational fishermen and their families were derived in section 4.3.2. The derived distribution of intake rates was lognormal, had a mean value of 38.4 g/d, a median value of 31.5, a standard deviation of 26.4 and a 95th percentile value of 89.5.

6.3.1.3 Exposure Period

Exposure periods (i.e. number of years fishermen catches and eats fish close to a open bay produced water discharge) may vary from several years to a large part of a lifetime. The probabilistic risk assessment assumed that the exposure period for recreational fishermen ranged from 5 to 65 years, and was described by a triangular distribution with the most frequent value set at 20 years.

6.3.1.4 Calculation of Radium Exposure

Daily ^{226}Ra and ^{228}Ra intake rates during the exposure period were calculated by using the distributions described above, in equation (6.2).

6.3.2 Dose Response Assessment

Current practice in radiation protection is to assume there is a cancer risk associated with even small doses of radiation. Risk factors are derived from epidemiological data and extrapolated down to low doses to describe the cancer risk associated with small exposures. Appendix C summarizes the basic concepts in radiation protection applicable to risk assessment, discusses in detail the USEPA risk factors for radium and derives the distribution for the cancer mortality risk factors used in the probabilistic assessment presented here (Table 6-5).

Table 6-5. Risk factor distribution for ^{226}Ra and ^{228}Ra (lognormal distributions; individual lifetime fatal cancer risk per pCi/day).

	^{226}Ra	^{228}Ra
mean	1.5×10^{-6}	1.0×10^{-6}
standard deviation	9.0×10^{-7}	1.4×10^{-6}
lower 90% confidence limit	9.4×10^{-7}	4.7×10^{-7}
upper 90% confidence limit	2.2×10^{-6}	1.9×10^{-6}

6.3.3 Risk Characterization

This section presents the risk characterization analysis for the ingestion of radium in fishes harvested near offshore produced water outfalls in the Gulf of Mexico. The risk characterization step includes the calculation of individual lifetime fatal cancer risk. The risk factor for the exposure period (5 - 65 years for recreational fishermen) was modified by adding 10 years to account for radium retention (see Appendix C):

$$RF(EP) = \frac{(EP + 10) \times URF_{70}}{70y} \tag{6.4}$$

where:
RF(EP) = risk factor as a function of exposure period EP (lifetime risk per pCi/day)
EP = exposure period (years)
URF_{70} = USEPA unit risk factor for lifetime exposure (lifetime risk per pCi/day)

Individual lifetime fatal cancer risks were calculated as:

$$ILR = I_{Ra} \times RF(EP) \tag{6.5}$$

where:
ILR = individual lifetime fatal cancer risk
I_{Ra} = average daily radium intake during the exposure period (pCi/day)

Individual lifetime risks were calculated separately for ^{226}Ra and ^{228}Ra and then summed.

6.3.4 Results and Discussion

Results of the probabilistic risk assessment for radium in fishes at Delacroix Island and Bay de Chene (pre-termination) are given in Table 6-6. The 95th percentile lifetime fatal cancer risks for both sites were less than 1×10^{-5}.

Results from the modeling analysis of continuing open bay discharges in Louisiana are also presented in Table 6-6. The 95th percentile lifetime fatal cancer risk was 4.3×10^{-6}. Assumed background concentrations of radium in fish yielded a 95th percentile value of 3.2×10^{-8}.

The results from the two study sites are in good agreement with the results of the modeling analysis. These results suggest that ingestion of radium in fish caught near open bay produced water platforms does not present an important risk to human health.

There are a number of uncertainties associated with this analysis, including:

- uncertainty due to limited data describing radium concentrations in animals at USDOE study sites;
- uncertainty in modeling of radium dilution and bioaccumulation for continuing discharges;
- uncertainty in ingestion rate distribution; and
- uncertainty in radium dose-response function.

These uncertainties are considered in the probabilistic risk assessment by describing each of the relevant variables as a distribution in the Monte Carlo analysis. The results based on modeling continuing discharges overestimate risk from radium ingestion because of the conservatism of the CORMIX dilution model (see section 5.3), assumptions used in its application (e.g. all radium remains in solution), and the use of modeled plume concentrations at 200 feet to estimate exposure.

Table 6-6. Probabilistic risk assessment for radium in fishes: individual lifetime fatal cancer risk.

SITE	Individual Lifetime Fatal Cancer Risk				
	mean	median	std. deviation	5th percentile	95th percentile
Delacroix Island	2.1×10^{-6}	1.3×10^{-6}	2.7×10^{-6}	2.4×10^{-7}	6.6×10^{-6}
Bay de Chene	2.0×10^{-6}	1.1×10^{-6}	3.3×10^{-6}	1.1×10^{-7}	6.7×10^{-6}
Continuing Discharges[1]	1.1×10^{-6}	2.5×10^{-7}	3.4×10^{-6}	1.6×10^{-8}	4.3×10^{-6}
Background	8.7×10^{-9}	4.2×10^{-9}	1.4×10^{-8}	4.4×10^{-10}	3.2×10^{-8}

[1] risk is for ingestion of fish, living in the plume 200 feet from the discharges.

7 ECOLOGICAL RISK ASSESSMENT FOR RADIONUCLIDES

7.1 Background and Approach

An aquatic organism may be irradiated externally by radionuclides in water and sediment, and internally by radionuclides taken into the body by ingestion or direct absorption. Most incorporated radionuclides are differentially distributed among the organs and tissues of the organism. Radium, for example, tends to accumulate in bone, skin and exoskeleton.

Exposure to ionizing radiation can result in injury at the molecular, cellular and whole body levels. Most of the available studies of the effects of radiation on aquatic organisms are concerned with the induction of deterministic, somatic effects. These effects include increases in mortality and pathophysiological, developmental and reproductive effects. There is little information available concerning induction of cancer and genetic effects, although a few studies of stochastic genetic effects in organisms are available (Anderson and Harrison, 1986).

Appendix C reviews the terminology and units used in radiation protection, and summarizes the data available that describes the effects of radiation exposure on aquatic animals.

The National Council on Radiation Protection and Measurements recently reviewed the literature on the effects of ionizing radiation on aquatic organisms. NCRP (1991) suggested a reference dose rate to protect aquatic populations of 10 mGy/d. NCRP also suggested a detailed assessment if an initial analysis results in an estimated dose rate above 2.4 mGy/d .

IAEA (1988) suggested similar reference dose rates where effects on aquatic biota would be minimal. IAEA (1988) concluded that:

- increased mortality is expected above 10 mSv/hr (240 mSv/d);
- reduced reproductive success may occur between 1 and 10 mSv/hr (24-240 mSv/d);
- some somatic effects which would be eliminated by natural selection could occur between 0.004 and 1 mSv/hr (0.1-24 mSv/d); and
- no adverse effects are expected below background levels of 0.004 mSv/hr (0.1 mSv/d).

IAEA (1988) developed dose conversion factors that relate exposure to an organism to a unit concentration of a radionuclide in the water in which the organism lives (Table 7-1). These dose conversion factors are based on models using assumptions concerning the bioaccumulation factor, K_d, and the sizes and shapes of the animals (IAEA, 1988). These factors are useful for screening purposes.

Table 7-1. IAEA dose conversion factors (mSv/hr per Bq/m^3).

ORGANISM	RADIONUCLIDE				
	^{226}Ra	^{228}Ra	^{210}Pb	^{210}Po	^{228}Th
FISH					
bathypelagic	1.38×10^{-4}	1.62×10^{-7}	4.96×10^{-8}	1.22×10^{-4}	2.21×10^{-4}
benthic	1.45×10^{-4}	3.83×10^{-6}	8.00×10^{-6}	1.22×10^{-4}	1.26×10^{-3}
MOLLUSKS	2.85×10^{-4}	4.41×10^{-6}	8.51×10^{-5}	6.10×10^{-4}	1.60×10^{-3}
CRUSTACEANS					
large, bathypelagic	2.77×10^{-5}	2.82×10^{-8}	2.46×10^{-7}	3.05×10^{-3}	3.68×10^{-4}
large, benthic	3.54×10^{-5}	4.03×10^{-6}	1.82×10^{-5}	3.05×10^{-3}	1.52×10^{-3}
small, bathypelagic	2.76×10^{-5}	1.86×10^{-8}	1.67×10^{-7}	1.83×10^{-3}	3.68×10^{-3}
small, benthic	3.70×10^{-5}	4.76×10^{-6}	6.14×10^{-4}	1.83×10^{-3}	5.12×10^{-3}

The IAEA screening dose-rate factors were used in a conservative screening analysis to identify the potential for ecological effects from radium and other radionuclides discharged in produced water to Louisiana open bays.

The data sets available for the analysis were:

- measured concentrations of ^{226}Ra, ^{228}Ra, ^{210}Pb, ^{210}Po and ^{228}Th in the discharge at Delacroix Island and Bay de Chene Study Sites (section 4).

- measured concentrations of ^{226}Ra and ^{228}Ra in 47 continuing open bay discharges (section 5).

A dilution factor of 20 was applied to the concentrations of radionuclides measured in these effluents. A dilution factor of 20 was chosen to estimate worst-case concentrations because it yields more conservative concentrations than those predicted by the CORMIX model (section 5) at 50 and 200 feet from the discharge. The resulting water concentrations (in the plume at 200 feet from the discharge) were used to estimate the dose to aquatic animals using the IAEA dose conversion factors.

7.2 USDOE Open Bay Sites

Concentrations of radionuclides measured in the effluent at the Delacroix Island and Bay de Chene study sites are given in Table 7-2. A conservative dilution factor of 20 was applied to these concentrations to estimate worst-case radium concentrations 200 feet from open bay discharges (Table 7-2). The IAEA dose conversion factors were applied to these estimated water concentrations, and the total dose to aquatic organisms calculated (Table 7-3). No estimated doses exceeded the IAEA (1988) range of 0.1-24 mSv/d associated with the potential for only minor effects on individual animals.

Table 7-2. Screening-level concentrations of radionuclides predicted for 200 feet at the Delacroix Island and Bay de Chene study sites.

Radionuclide	Delacroix Island		Bay de Chene	
	Discharge (pCi/l)	Water Conc. (pCi/l)	Discharge (pCi/l)	Water Conc. (pCi/l)
^{210}Pb	60.3	3.0	78.0	3.9
^{210}Po	<2.0*	<0.1	<1.1*	<0.06
^{226}Ra	162.5	8.1	218.5	10.9
^{228}Ra	317.5	15.9	264.5	13.2
^{228}Th	15.0	0.8	20.5	1.0

*lower limit of detection values were used in the analyses

Table 7-3. Screening level dose estimates for Delacroix Island and Bay de Chene study sites (mSv/d).

ORGANISM	Delacroix Island	Bay de Chene
FISH		
bathypelagic	1.2	1.5
benthic	2.0	2.6
MOLLUSKS	3.5	4.6
CRUSTACEANS		
large, bathypelagic	0.7	0.8
large, benthic	1.7	2.0
small, bathypelagic	3.0	3.6
small, benthic	5.8	7.2

7.3 Continuing Discharges

Radium concentrations measured in 47 open bay discharges are given in Appendix B, and are summarized in Table 5-3. Mean and maximum concentrations are given in Table 7-4. A conservative dilution factor of 20 was applied to these concentrations to estimate worst-case radium concentrations 200 feet from open bay discharges (Table 7-4). A dilution factor of 20 was chosen to estimate worst-case concentrations because it yields more conservative concentrations than those predicted by the CORMIX model (section 5) at 200 feet from the discharge.

Mean and maximum doses calculated using the IAEA dose rate conversion factors (Table 7-1) are given in Table 7-5. No estimated doses exceeded the IAEA (1988) range of 0.1-24 mSv/d associated with the potential for only minor effects on individual animals.

Table 7-4. Screening-level concentrations of radionuclides predicted for water 200 feet from open bay discharges.

Radionuclide	Discharge		Water Conc. (pCi/l)	
	mean (pCi/l)	maximum (pCi/l)	mean (pCi/l)	maximum (pCi/l)
^{226}Ra	191.4	592.0	9.6	29.6
^{228}Ra	250.0	560.0	12.5	28.0

Table 7-5. Screening level dose estimates for radium in continuing open bay discharges (mSv/d).

ORGANISM	Dose rate (mSv/d)	
	mean	maximum
FISH		
bathypelagic	1.2	3.6
benthic	1.3	3.9
MOLLUSKS	2.5	7.6
CRUSTACEANS		
large, bathypelagic	0.2	0.7
large, benthic	0.4	1.0
small, bathypelagic	0.2	0.7
small, benthic	0.4	1.1

7.4 Discussion

In this simple conservative screening analyses, doses to aquatic animals did not exceed the range associated with only minor effects of individual organisms (IAEA, 1988). No effects are expected to be found in aquatic animals in open bays in Louisiana, because of the conservative screening analysis yielded worst-case estimates of exposure.

8 HUMAN HEALTH RISK ASSESSMENT FOR METALS AND ORGANICS

8.1 Introduction and Approach

A screening human health risk assessment was done (section 8.2) for metals and organic compounds measured in continuing open bay discharges (section 5). This analysis followed the USEPA approach to estimating risks from toxic materials and carcinogens by applying RfD (reference dose) and slope factor values to conservative estimates of chemical intake rates (USEPA, 1989a). Conservative predictions of water concentrations were also compared to USEPA and Louisiana human health surface water criteria.

A second level assessment (section 8.3) using a probabilistic approach was done for contaminants that the initial screening analysis suggested may be of potential concern. A separate probabilistic risk assessment was done for lead (section 8.4).

8.2 Screening Assessment

8.2.1 Concentrations in Water and Fish

Concentrations in the effluent for continuing open bay discharges were described by the data abstracted from LDEQ permit files (Table 5-2). These data overestimate average concentrations because only contaminants detected in the effluent above the reported detection limit are given.

A conservative dilution factor of 20 was chosen to estimate worst-case water contaminants concentrations in the plume 200 feet from the discharge. Most contaminants were assumed to remain in solution. Dissolved fractions of copper, lead and zinc were assumed to be 0.88, 0.38 and 0.59, respectively (USEPA, 1995a).

In this assessment, contaminants were assessed only if they were reported above detection limits in more than two of the LDEQ permit files; and if toxicity data were available in IRIS or other USEPA literature. Worst-case mean and maximum chemical contaminant concentrations in effluents and in water at 200 feet are given in Table 8-1.

Conservative, generic bioaccumulation factors (Strenge and Peterson, 1989); were used to calculate concentrations of contaminants in edible fish (Table 8-1):

$$CF = (\ BAF \times CW\ ) \times \frac{1kg}{1{,}000g} \tag{8.1}$$

where:
CF = contaminant concentration in fish (µg/g)
BAF = bioaccumulation factor (l/kg)
CW = contaminant concentration in water (µg/l)

Estimated concentrations in edible fish for worst-case mean and maximum contaminant discharge concentrations are given in Table 8-1.

Table 8-1. Estimated worst-case mean and maximum contaminant concentrations in the effluent, in the plume 200 feet from the discharge, and in edible fish.

Contaminant	Effluent (µg/l)		Diss.* Fract.	Concentration in Water at 200 feet (µg/l)		BAF** (l/kg)	Concentration in Fish (µg/g)	
	max	mean		max	mean		max	mean
Antimony	20100	5595.9	1	1005	279.8	1	1.0	0.3
Arsenic	498.5	74.8	1	24.925	3.7	1	0.02	0.004
Cadmium	500	231.2	1	25	11.6	200	5	2.3
Chromium (VI)	200	83.5	1	10	4.2	20	0.2	0.1
Copper	710	288.4	0.88	31.2	12.7	50	1.6	0.6
Lead	829000	104263	0.38	15751	1981	100	1575	198.1
Mercury	27	7.1	1	1.35	0.4	2.0E5	270	70.8
Nickel	2840	1013.9	1	142	50.7	100	14.2	5.1
Silver	400	143.3	1	20	7.2	2.3	0.05	0.02
Zinc	6375	1217.1	0.59	188.1	35.9	2.0E3	376.1	71.8
Benzene	9550	1813.2	1	477.5	90.7	24.1	11.5	2.2
Naphthalene	118	57.4	1	5.9	2.9	168	1.0	0.5
Phenol	12000	1557.9	1	600	77.9	7.57	4.5	0.6
Toluene	2800	831.6	1	140	41.6	69.9	9.8	2.9
Xylenes	862	183.3	1	43.1	9.2	177	7.6	1.6

* dissolved fraction (USEPA, 1995a)
**bioaccumulation factors (Strenge and Peterson, 1989)

8.2.2 Risk Factors

Risk factors (slope factors for carcinogens and reference doses (RfD) for toxicants) were obtained from the USEPA IRIS data base (April, 1995) and other sources. Table 8-2 summarizes these values.

Reference Dose

The RfD (chronic reference dose) is "an active estimate (with uncertainty spanning perhaps an order of magnitude or greater) of a daily exposure level for the human population, including sensitive subpopulations, that is likely to be without an appreciable risk of deleterious effects during a lifetime. Chronic RfDs are specifically developed to be protective ..." (USEPA 1989a).

Each RfD includes uncertainty factors (UFs). Depending on the derivation of the RfD, uncertainty factors can inflate the RfD by up to 10,000 times. Therefore, an estimated exposure that exceeds an RfD for a particular contaminant may or may not exceed a threshold for toxicity. RfDs for many of the chemicals commonly found in produced water discharges are highly uncertain, as shown in Table 8-2.

RfDs undergoing review at USEPA are not available in IRIS. At the time of this analysis, current RfD's were not available for copper, mercury, lead and naphthalene, all contaminants with potential toxic effects. Estimates were available for mercury and naphthalene in HEAST (1991). These reference doses are interim values and have not been formally verified by USEPA.

No RfDs are available for lead or copper. Screening level estimates were derived for these contaminants as described below.

Copper:

- current maximum contaminant level goal for drinking water is 1.3 mg/l
- assume based on 2 l/day water intake
- assume 70 kg adult
- RfD = 0.04 mg/kg-day

Lead:

- current data suggest effects at a blood level concentration of 10 μg/dl (Carlisle and Wade, 1992)
- slope of 0.04 μg Pb/dL blood per μg/day in diet (Carlisle and Wade, 1992)
- assume 70 kg adult
- RfD = 3.6×10^{-3} mg/kg-day

Table 8-2. RfDs, uncertainty factors (U), slope factors and human health water quality criteria.

Contaminant	RfD (mg/kg-day)	Confidence	U	Weight of Evidence	Slope Factor risk per mg/kg-day	Human Health Criteria For Fish Ingestion(µg/l)	
						USEPA	LDEQ
Antimony	4.00×10^{-4}	Low	1000	--	--	4.50×10^{4}	--
Arsenic	3.00×10^{-4}	Medium	3	A	5.00×10^{-5}	1.75×10^{-2}	--
Cadmium	1.00×10^{-3}	High	10	B1***	--	--	--
Chromium (VI)	5.00×10^{-3}	Low	500	A***	--	--	--
Copper*	4.00×10^{-2}	--	--	D	--	--	--
Lead*	3.60×10^{-3}	--	--	B2	--	--	--
Mercury**	3.00×10^{-4}	--	--	D	--	1.46×10^{-1}	--
Nickel	2.00×10^{-2}	Medium	300	--	--	1.00×10^{2}	--
Silver	5.00×10^{-3}	Low	3	D	--	--	--
Zinc	3.00×10^{-1}	Medium	3	D	--	--	--
					--	--	--
Benzene	--	--	--	A	2.90×10^{-2}	4.00×10^{1}	12.5
Naphthalene**	4.00×10^{-3}	--	--	D	--	--	--
Phenol	6.00×10^{-1}	Low	100	D	--	--	50
Toluene	2.00×10^{-1}	Medium	1000	D	--	4.24×10^{5}	6.93×10^{4}
Xylenes	2.00	Medium	100	D	--	--	--

* no RfD available in IRIS, screening values derived in text
** no RfD available in IRIS, screening values from HEAST (1991)
*** evidence is for inhalation carcinogenesis only

Hazard Quotients

For noncarcinogenic toxicity risk characterization of individual contaminants, USEPA (1989a) uses a hazard quotient (HQ), "the ratio of a single substance exposure level over predicted a specified time period (e.g. subchronic) to a reference dose for that substance derived from a similar exposure period". In this report the *HQ concept* is extended to utilize any comparable reference standard for human health or ecological risks. Such standards include RfDs and human health water quality criteria. The term HQ is reserved for the ratio derived using the RfD; WHQ (water quality criteria hazard quotient) is the ratio of the predicted water concentration to the USEPA human health water quality criteria for the contaminant.

Slope Factor

A slope factor is "a plausible upper-bound estimate of the probability of a response per unit intake of a chemical over a lifetime. The slope factor is used in risk assessments to estimate an *upper-bound* (italics added) lifetime probability of an individual developing cancer as a result of a lifetime exposure to a level of a particular carcinogen" (USEPA, 1989a). The upper bound is usually the upper 95th percent limit of the slope of a calculated dose-response curve. "In some cases slope factors based on human dose-response data are based on the "best" estimate instead of the upper 95 percent confidence limits" (USEPA, 1989a) Each USEPA slope factor is accompanied by a weight-of evidence classification, a "...system for characterizing the extent to which the available data indicate that an agent is a human carcinogen" (USEPA, 1989a). The weight of evidence classification used by USEPA is as follows:

A Human carcinogen
B1 Probable human carcinogen based on limited human data
B2 Probable human carcinogen based on sufficient evidence in animals only
C Possible human carcinogen
D Not classifiable as to human carcinogenicity
E Evidence of noncarcinogenicity in human beings

8.2.3 Exposure Assumptions

The screening analyses used a conservative value of 70 years as the duration of exposure, to reflect an assumption of a lifetime exposure. A conservative ingestion rate of 132 g/d was used (USEPA 1989a; 95th percentile value), along with an exposure frequency of 365 d/year. An assumed body weight of 70 kg for adults was used in the analysis (USEPA, 1990). Intakes were averaged over a 70 year lifetime.

8.2.4 Exposure Assessment and Risk Characterization

Intake rates for contaminants in finfish caught near coastal open bay platforms were calculated following USEPA methods developed for the assessment of Superfund sites (USEPA, 1989a).

$$I = \frac{\left(CF \times I_{fish} \times F \times EF \times ED\right)}{\left(BW \times AT\right)} \qquad (8.2)$$

where:

I = intake rate (mg/kg-d)
CF = concentration in finfish (mg/kg)
I_{fish} = ingestion rate (0.132 kg/d; USEPA, 1989a)
F = fraction of fish from contaminated source (1.0)
EF = exposure frequency (365 d/year; USEPA, 1989a)
ED = exposure duration (70 years; USEPA, 1989a)
AT = averaging time (70 years x 365 d/year; USEPA, 1989a)
BW = body weight (70 kg; USEPA, 1989a)

The risks associated with the ingestion of contaminants in finfish caught near coastal open bay platforms were calculated following EPA methods developed for assessments at Superfund sites (USEPA, 1989a).

Toxicity

$$HQ = \frac{I}{RfD} \quad (8.3)$$

where:
HQ = hazard quotient
I = intake rate (mg/kg-d)
RfD= reference dose (mg/kg-d)

Hazard quotients greater than one suggest a potential for chronic toxic effects.

Carcinogenicity

$$IR = I \times SF \quad (8.4)$$

where:
IR = individual incremental lifetime fatal cancer risk
I = intake rate (mg/kg-d)
SF = slope factor (risk per mg/kg-d, 70 year exposure period)

8.2.5 Water Quality Criteria

Worst-case mean and maximum predicted water concentrations at 200 feet from the discharge were compared to USEPA and LDEQ water quality criteria for human health (for fish ingestion; Table 8-2). A WHQ [predicted water concentration/water quality criteria] was calculated for each contaminant. Where WHQs are greater than one, this conservative screening analysis predicts that the human health water quality will be exceeded.

8.2.6 Screening Analysis Results

Results of the screening risk assessment for the continuing open bay discharges in Louisiana are given in Tables 8-3 and 8-4.

Arsenic, chromium, copper, silver, naphthalene, toluene and xylenes were eliminated from further consideration. Contaminants with screening hazard quotients greater than one were antimony, cadmium, lead, mercury, nickel, and zinc. Screening cancer risk estimates for benzene exceed 1×10^{-4}. Benzene is the only carcinogen of potential concern. Contaminants that exceeded human health water quality criteria in the screening analysis were: mercury, nickel, benzene and phenol.

Major uncertainties and conservative assumptions in this screening assessment included:

- use of worst-case water concentrations;
- use of average chemical concentrations that excluded zero values;
- use of conservative ingestion rates and exposure periods;
- use of generic bioaccumulation factors; and
- use of uncertain reference doses that either include large safety factors or are not verified by USEPA (lead, mercury, antimony, nickel).

A more realistic and quantitative assessment was performed for contaminants identified in this screening analysis (section 8.3). Because of the concern for lead exposure to children, and the current belief that the dose-response function for lead exposure does not have a threshold, lead was analyzed in a separate probabilistic risk assessment (section 8.4).

8.3 Quantitative Analysis for Antimony, Cadmium, Mercury, Nickel, Zinc, Benzene and Phenol

For chemicals not eliminated by the screening assessments, distributions of concentrations in produced water discharges were developed from permit data (Table 8-5). Values for chemicals that were not detected were assigned one-half the reported detection limit value. Each chemical, except cadmium and copper, was assigned a lognormal distribution, after a log probability plot of the frequency of measured values yielded a linear fit (Layton *et al.*, 1987). Cadmium was assigned a custom distribution that matched the relative frequencies of the values in the available data set.

Table 8-3. Hazard quotients (HQ[1]) and cancer risk estimates (shaded values exceed a HQ of 1.0 or an individual lifetime fatal cancer risk of 1×10^{-4}).

Contaminant	HQ[1]		Individual Lifetime Fatal Cancer Risk	
	maximum	mean	maximum	mean
Antimony	4.7	1.3	--	--
Arsenic	0.2	0.02	2.4×10^{-9}	3.5×10^{-10}
Cadmium	9.4	4.4	--	--
Chromium (VI)	0.07	0.03	--	--
Copper	0.07	0.03	--	--
Lead	825	104	--	--
Mercury	1697.1	445.1	--	--
Nickel	1.3	0.5	--	--
Silver	0.02	0.006	--	--
Zinc	2.4	0.5	--	--
			--	--
Benzene	--	--	6.3×10^{-4}	1.2×10^{-4}
Naphthalene	0.5	0.2	--	--
Phenol	0.01	0.002	--	--
Toluene	0.1	0.03	--	--
Xylenes	0.01	0.002	--	--

[1] HQ = Intake Rate/RfD

Table 8-4. Water Quality Criteria Hazard Quotients (WHQ[1]) at 200 feet (shaded values exceed a ratio of 1.0).

Contaminant	Louisiana Criteria		USEPA Criteria	
	maximum	mean	maximum	mean
Antimony	--	--	0.02	0.006
Arsenic	--	--	--	--
Cadmium	--	--	--	--
Chromium (VI)	--	--	--	--
Copper	--	--	--	--
Lead	--	--	--	--
Mercury	--	--	9.3	2.4
Nickel	--	--	1.4	0.5
Silver	--	--	--	--
Zinc	--	--	--	--
	--	--	--	--
Benzene	38.2	7.3	1.2	2.3
Naphthalene	--	--	--	--
Phenol	12	1.6	--	--
Toluene	0.002	0.0006	3.3×10^{-4}	9.1×10^{-5}
Xylenes	--	--	--	--

[1] WHQ = predicted concentration at 200 feet / water quality criteria for human health

Table 8-5. Distributions of concentrations of contaminant (µg/l) found in discharges from open bay platforms.

Chemical	Distribution	Mean	SD	Minimum	Maximum
Antimony	Lognormal	3192.6	6268.3	11.8	20,100
Cadmium	Custom	217.9	235.6	0.0015	540
Mercury	Lognormal	4.3	11.1	0.0005	41
Nickel	Lognormal	569.1	947.9	20	2,480
Zinc	Lognormal	1465.3	2768.3	2.5	10,800
Benzene	Lognormal	1315.5	1909.6	2.5	6,420
Phenol	Lognormal	1257.3	2743.4	5	12,000

These distributions were used with the relationships derived from results of CORMIX modeling to obtain distributions of the concentrations of each contaminant at 200 feet in the plume (Table 8-6; see section 5-3). Each chemical was assumed to be totally soluble in water except for zinc which was assumed to have a fractional solubility of 0.59.

Table 8-6. Predicted concentrations of contaminants (µg/l) in plumes, at 200 feet from discharges of produced waters.

Chemical	Mean	SD	Minimum	Maximum
Antimony	1.8	2.2	0.02	13.4
Cadmium	0.5	1.3	9.2×10^{-6}	13.2
Mercury	0.01	0.03	3.6×10^{-6}	0.6
Nickel	1.4	3.3	3.9×10^{-3}	52.2
Zinc	0.5	0.4	0.007	4.2
Benzene	3.2	7.6	0.01	117.7
Phenol	3.3	9.7	0.01	170.0

These concentrations were used in a probabilistic analyses of potential human health effects. Exposure of humans was assumed to be from eating 100% of their recreationally caught fish intake as fish caught in a plume, within 200 feet of a discharge.

The distribution of concentrations of a contaminant in fish (CF, Table 8-7) was calculated by applying the distribution for contaminant concentrations in water, and the bioaccumulation factors given in Table 8-1 to equation (8.1).

The distribution of exposure (mg/kg-d) to humans by ingestion of fish caught in the plume was calculated using the parameter distributions and values from Table 8-8 in Equation (8.2).

Table 8-7 Predicted concentrations of chemicals (mg/kg) in finfishes, assumed to live in plumes, within 200 feet of discharges of produced waters.

Chemical	Mean	SD	Minimum	Maximum
Antimony	1.8×10^{-3}	2.2×10^{-3}	2.4×10^{-5}	1.3×10^{-2}
Cadmium	0.11	0.3	1.8×10^{-6}	2.6
Mercury	2.0×10^{-10}	6.5×10^{-10}	7.2×10^{-14}	1.2×10^{-8}
Nickel	0.14	0.05	3.9×10^{-4}	5.2
Zinc	1.5×10^{-4}	1.9×10^{-4}	2.1×10^{-5}	1.3×10^{-3}
Benzene	0.08	0.18	1.2×10^{-4}	2.8
Phenol	2.5×10^{-2}	7.3×10^{-2}	9.4×10^{-5}	1.3

Table 8-8. Parameters used in the exposure calculations.

Parameter	Value or Distribution
CF concentration in fish (mg/kg)	calculated from equation 8.1 and Table 8-7.
I_{fish} fish ingestion rate (kg/d)	lognormal, mean: 38.4; sd: 26.4; range: 3.3-228.6 (section 5)
F fraction of fish from contaminated source	1.0
ED exposure duration (y)	Triangular: most frequent 20; range 5 to 65
EF exposure frequency (d/y)	365 d/y (USEPA, 1989a)
BW body weight (kg)	Age-weighted lognormal: mean 58; SD 14 (McKone and Daniels, 1991).
AT averaging time (d)	ED (y) x 365 (d/y)

The HQ (ratio of the predicted range of exposures to the RfD (Table 8-9) was calculated for each contaminant (with the exception of benzene) and the percent probability of exceeding the RfD was determined.

Table 8-9. Probability that the HQ (from ingestion of fish caught within 200 ft of a produced water discharge) exceeds 1.0.

Contaminant	RfD (mg/kg-d)	Mean HQ	Maximum HQ	%p (HQ) > 1
Antimony	4×10^{-4}	3×10^{-3}	5×10^{-2}	0
Cadmium	1×10^{-3}	0.08	3.0	1.4
Mercury	3×10^{-4}	7×10^{-9}	4×10^{-7}	0
Nickel	2×10^{-2}	5×10^{-3}	0.16	0
Zinc	2×10^{-3}	3×10^{-4}	4×10^{-3}	0
Phenol	6×10^{-1}	3×10^{-5}	1×10^{-3}	0

The results show that intakes of chemical contaminants, by eating fish, pose a negligible toxic hazard to human health, when the contaminants are considered individually. The only chemical that marginally exceeded its oral RfD value was cadmium (Figure 8-1).

For benzene, the slope factor (2.9×10^{-2}) from USEPA's IRIS was multiplied by the predicted range of exposures to yield a distribution of values for incremental individual lifetime risk of cancer mortality: mean, 1.6×10^{-6}; SD, 3.9×10^{-6}; 95th percentile, 7.4×10^{-6} (Figure 8-2). This is within the range considered acceptable by USEPA (1×10^{-6} to 1×10^{-4}; Federal Register, 1991).

These analyses used several conservative assumptions. The first assumption was that all the fish spend all of their time living and feeding within the plume, although they probably spend only a fraction of time within a plume. The predicted concentrations represent values at the midline of the plume 200 feet from the discharge. These values were generated by a model that conservatively underestimates dilution (Smith *et al.*, 1993). It was also assumed that all the fish eaten by a person were captured at the midline of a plume, while people may eat fish from several sources. Although contaminant concentrations in water should increase with decreasing distances from a discharge, bioaccumulation in fish would be offset by expected reduced residence of fish within the smaller plume volumes.

Figure 8-1. Hazard quotient (HQ) for chronic oral exposure to cadmium.

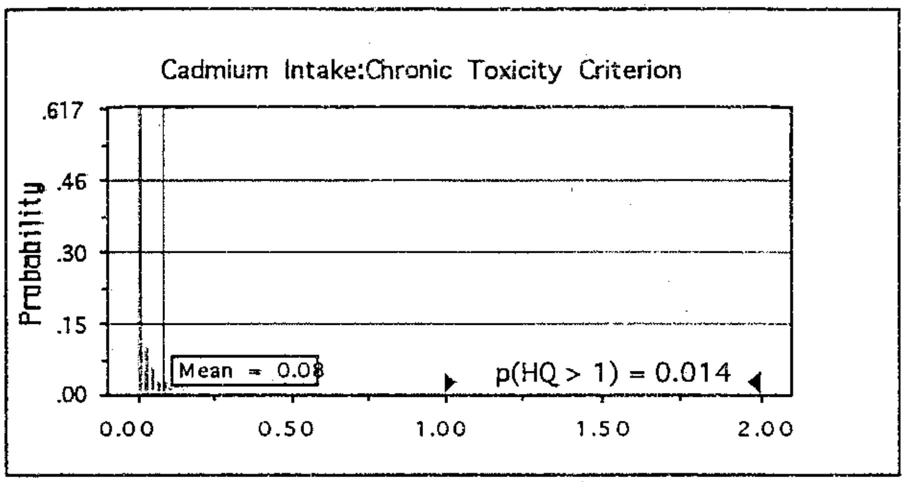
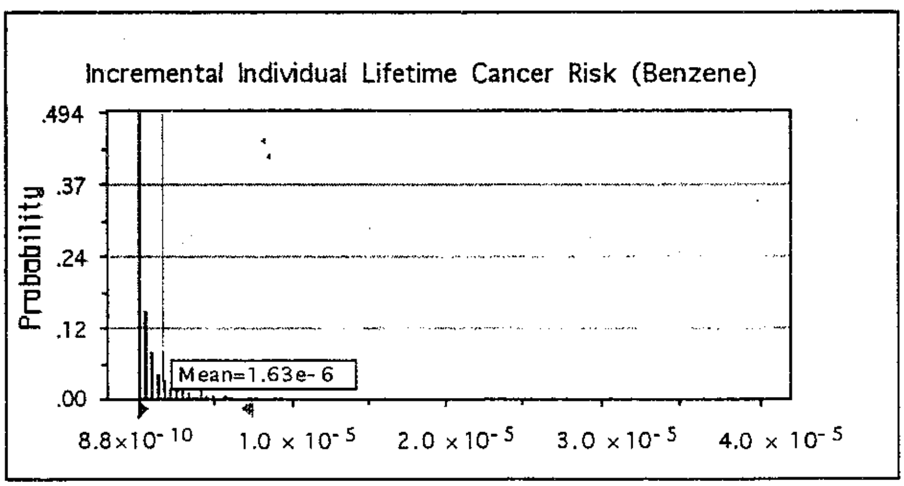


Figure 8-2. Incremental individual lifetime risk of cancer from benzene intake by ingesting fish (right marker on x axis is at the 95th percentile).



8.4 Probabilistic Risk Assessment for Lead

8.4.1 Concentrations in Water

Measured concentrations of lead in open bay produced water discharges, reported in LDEQ permit files, are summarized in Table 8-10. The largest lead concentration reported in permit files (800,000 μg/l) was several orders of magnitude larger than maximum values reported in other studies (Stephenson, 1992; Middleditch, 1984) and was not included in the data set for the risk assessment. Many of the lead concentrations in produced water were reported as "less than (<)" the detection limit. The detection limit for lead ranged from 50 to 125 μg/l. These values were replaced by one-half the value of the reported detection limit.

To estimate ambient water concentrations, the distribution of lead concentrations reported for open bay discharges was modified by the distribution of dilution factors (DFs). Thirty-eight percent of lead was assumed to remain in solution based on calculations performed by LDEQ (USEPA, 1995a). Table 8-10 gives estimated lead concentrations in the water column at 200 feet.

8.4.2 Concentrations in Fish

8.4.2.1 Fish Near Platforms

A distribution for a lead BAF was developed from published estimates. In a report prepared for USEPA, Avanti Corporation (1993) cited a range of 10 to 100 for bioaccumulation of lead. IAEA (1982) presented a default BAF of 300 for lead in seawater. A triangular distribution for BAF of lead ranging from 10 to 300, with a most likely value of 100 was used in this analysis.

Lead concentrations in fish near produced water discharges (Table 8-10) were estimated using the bioaccumulation factor (BAF) method in equation (8.1).

Table 8-10. Lead concentrations in open bay produced water discharges, and estimated concentrations in water and fish in the plume at 200 feet.

	Effluent[1]	Ambient Water[2]	Fish[3]
mean	546.8	0.53	0.07
sd deviation	934.5	1.14	0.17
minimum	25.0	0.0	0.0
maximum	2,600	12.9	1.95

[1]Measured in effluent: μg/l
[2]Modeled concentrations in water: μg/l
[3]Modeled concentrations in fish: μg/g

8.4.2.2 Fish Away From Platforms

For comparison, concentrations of lead in fish caught in the Gulf of Mexico away from platforms (and associated health risks) were estimated. Distributions of lead in fish not associated with platforms were derived from measured concentrations of lead in whole fishes at two Environmental Monitoring and Assessment Program (EMAP) sites on the coast of Louisiana (USEPA, 1995b). These measurements may under- or overestimate background concentrations because the samples were of whole fish rather than edible fillets. Concentrations in fish caught away from platforms were assumed to be lognormally distributed with an arithmetic mean value of 0.05 µg/g (standard deviation: 0.06; range: 0.01 - 0.28). Although the data used in deriving this distribution have been funded wholly or in part by the USEPA through its EMAP-Estuaries Program, it has not been subjected to Agency review, and therefore does not necessarily reflect the views of the Agency and no official endorsement should be inferred.

8.4.3 Intake

8.4.3.1 Background Intake

Lead is ubiquitous in the environment, and children, in particular, are exposed to lead through a number of pathways. Sources of lead exposure to children include food, drinking water, air, soil and dust. Exposures from specific sources are added to background exposures experienced by children and increase the probability of exceeding blood lead levels of concern identified by USEPA. This analysis assumed age-specific background intakes for children ages one-half to 7 years, as described in USEPA (1994).

8.4.3.2 Recreational Fishing

Lead intake was estimated for children eating fish caught either near platforms, or away from platforms. Distributions of lead intake in recreationally caught fish were calculated as:

$$I_{Pb} = I_{fish} \times [Pb]_{fishes} \tag{8.5}$$

where:
I_{Pb} = lead intake (µg/day)
I_{fish} = intake of fish (g/day) for children of recreational fishermen (section 5)
$[Pb]_{fishes}$ = concentration of lead in fish (µg/g)

Intake estimates were divided into groups (15 groups for fish caught near platforms, 13 groups for fish caught away from platforms) and the midpoint of the intake range for each group was used to represent the intake of lead ingested in recreationally caught fish. Daily lead ingestion rates in food were calculated for each year of life to age 7 by adding the background intake for that age (USEPA, 1994) to the estimated intake from recreationally caught fish. This approach slightly overestimates lead intake in food because recreationally caught fish would actually replace a small amount of lead in fish and meat obtained from other sources.

8.4.4 Dose-Response Assessment

Lead exposure can affect a number of systems, including the brain, hematopoietic system, cardiovascular system and the developing fetus (Derosa *et al.*, 1991). Extensive data are available to link low-level lead exposure of young children to deficits in neurobehavioral-cognitive performance (Rosen, 1995). Federal agencies and advisory groups including USEPA (USEPA, 1986), have defined a level of concern for children as a blood lead level ≥10 µg/dl (Rosen, 1995; USEPA, 1994). USEPA has developed a biokinetic/uptake model for lead (UBK Model; USEPA, 1994) that relates intake in food, air, water and soil to the probability of exceeding a blood lead level of 10 µg/dl (BL>10). This analysis used this probability as the metric for risk from ingestion of lead in fish.

8.4.5 Risk Characterization

The UBK model (USEPA, 1994) was used to estimate the blood lead concentration and the probability of BL>10 for each level of intake of recreationally caught fish. All other UBK model parameters reflected USEPA (1994) estimates of average background intakes.

Blood lead levels were estimated for two age groups: age 1-2 years when they are at their maximum level for a given intake; and averaged over age 0 to 7 years. Figure 8-3 shows the relationship between the intake of lead in recreationally caught fish and the probability of BL>10. For comparison, background intakes of lead are associated with a probability of BL>10 of 1.56% for age 0-7 years and of 4.42% for age 1-2 years.

The total risk (that is, the probability that BL>10 across all predicted intake rates) was calculated as:

$$TP = \sum P\left(I_{Pb}\right) \times P\left(BL > 10 \mid I_{Pb}\right) \qquad (8.6)$$

where:
TP = total probability (%) of exceeding a blood lead level of 10 μg/dl
P (I_{Pb}) = probability (%) of a given lead intake in recreationally caught fish
P(BL>10 | I_{Pb}) = probability (%) of exceeding a blood lead level of 10 μg/dl for a given intake of lead in fish

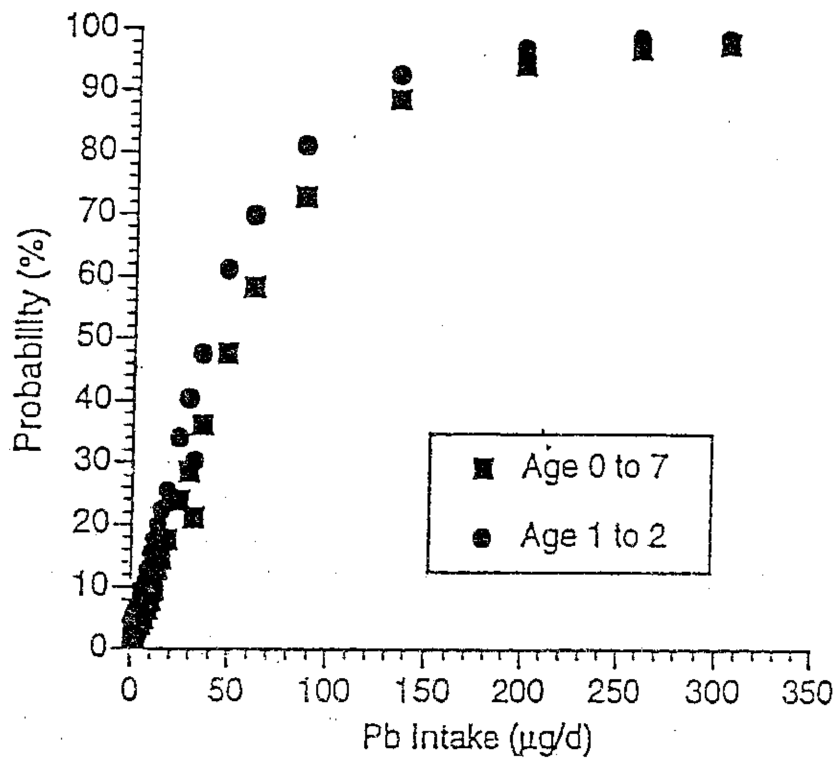
8.4.6 Results

Table 8-11 shows the total probability of BL>10 for fish caught near platforms, fish caught away from platforms and background intakes. Risk from ingestion of fish caught near from platforms only slightly exceeded risks from background intake of lead and was similar to those associated with ingestion of fish caught away from platforms in the Gulf of Mexico. Because of the conservatisms embedded in the analysis (assumptions concerning "less than" effluent concentrations, underestimate of dilution at low discharges rates) the risk from ingestion of lead discharged from open bay discharges in Louisiana appears to be small.

Table 8-11. Total probability (%) of exceeding a blood lead level of 10 μg/dl.

	0-7 years	1-2 years
Fish Near Platforms	2.3	4.8
Fish Away From Platforms	2.0	4.8
Background	1.6	4.4

Figure 8-3. Relationship between intake of lead in recreationally caught fish and probability of exceeding 10 µg/dl blood lead for two age groups.



9 ECOLOGICAL RISK ASSESSMENT FOR METALS, ORGANICS AND TOTAL EFFLUENT

9.1 Introduction and Approach

Three screening analyses were used to identify potential ecological effects and important receptors:

1. *Sediment toxicity* - Sediment metal and PAH concentrations measured at the Delacroix Island and Bay de Chene USDOE study sites (pre-termination data, section 4) were compared to proposed sediment quality criteria.

2. *Potential toxicity of individual contaminants in the water column* - Worst-case predicted water column concentrations of contaminants measured in continuing open bay effluents (LDEQ permit files, section 5) were compared to USEPA and Louisiana water quality criteria.

3. *Total effluent toxicity* - Predicted water column concentrations of effluent were compared to results of acute and chronic toxicity tests performed in the laboratory with standard test organisms (LDEQ permit files, section 5).

9.2 Sediment Toxicity -- USDOE Open Bay Sites

Marine environments containing high levels of (multiple) contaminants may be associated with adverse effects on biota. However, no direct causal relationship has been established between a contaminant and a biological effect in a marine environment. Therefore, sediment quality criteria rely on prudent use of the best information available and empirical data (E.V.S. Consultants, 1990).

A screening ecological risk assessment was performed, using preliminary data that describe concentrations of heavy metals and PAHs in sediment cores taken at sampling stations at the Delacroix Island and Bay de Chene USDOE study sites (Appendix A). These data were compared to sediment quality criteria (Table 9-1) developed for contaminants in marine and estuarine sediments (Long et al., 1995). These criteria are based on specific levels of probability of toxicological effects described in a biological effects database (BEDS) for contaminant concentrations in marine and estuarine sediments. The criteria were recently updated, but remain generally consistent with those previously reported (Long and Morgan, 1990).

BEDS includes a wide variety of adverse biological effects and information derived from all the types of measurements described above. Concentrations in each study included in BEDS were assigned an effects/no effects descriptor, and ascending orders of concentration were assigned percentile values to describe the distributions. The lower tenth percentile level was identified as the Effects Range Low (ERL) value, and the fiftieth percentile was identified the Effects Range Median (ERM) value. Measured sediment values below the ERL value of a contaminant represent a minimal effects range, where effects "would rarely be observed". Concentrations at and above the ERL value, but less than the ERM value, "represent a possible-effects range within which effects would occasionally occur". Concentrations at or above the ERM value "represent a probable effects range within which effects would frequently occur" (Long et al., 1995).

Table 9-1 Proposed sediment quality criteria (from Long et al., 1995).

	Sediment Quality Criteria	
Contaminant	**ERL[1]**	**ERM[2]**
Metals	**(ppm)[3]**	**(ppm)[3]**
Arsenic	8.2	70
Cadmium	1.2	9.6
Chromium	81	370
Copper	34	270
Lead	46.7	218
Mercury	0.15	0.71
Nickel	20.9	51.6
Silver	1.0	3.7
Zinc	150	410
Organics	**(ppb)[3]**	**(ppb)[3]**
Total PCBs	22.7	180
Total PAH	4022	44792
Acenaphthene	16	500
Acenaphthylene	44	640
Anthracene	85.3	1100
Fluorene	19	540
2-Methylnaphthalene	70	670
Naphthalene	160	2100
Phenanthrene	240	1500
Low Molec. Weight-PAH	552	3160
Benzo(a)anthracene	261	1600
Benzo(a)pyrene	430	1600
Chrysene	384	2800
Dibenzo(a,h)anthracene	63.4	260
Fluoranthene	600	5100
Pyrene	665	2600
High Molec. Weight-PAH	1700	9600

[1] ERL: effects range low
[2] ERM: effects range median
[3] dry weight

Table 9-2 shows the results of the screening assessment for metals in sediment, and Tables 9-3, 4 and 5 show the results of PAH analyses. None of the measured concentrations of metals in sediment samples exceeded their respective ERM value. In general, measured sediment concentrations were below the ERL (minimal effects range), with the exception of arsenic and nickel. Each of these metals exceeded its ERL value in samples from at least one reference station, and both discharges. Excess arsenic was detected up to 500 m from the Bay de Chene discharge (Table 9-2). Excess nickel was detected up to 500 m from the Delacroix Island discharge, and up to 1,000 m from the Bay de Chene discharge. There was no clear pattern of concentration with distance from a discharge.

Table 9-2. Measured metal concentrations that exceed ERL sediment criteria (Long *et al.*, 1995), at sampling stations (0 to 5 cm depth) around the Delacroix Islands and Bay de Chene study sites.

	As (ppm)	Ni(ppm)
ERL	8.2	20.9
Delacroix Island[1]		
R1	4.7*	25.1*
R2	3.6*	20.0*
Discharge	10.7	22.7
100NW		23.0
300NW		22.6
100NE		21.7
300NE		21.6
500NE		22.6
Bay de Chene[1]		
R1	8.7*	20.6*
R2	7.5*	21.*
Discharge	11.0*	24.2*
100NW	10.4*	28.2*
300NW	13.9	25.7
500NW	8.7	23.9
100SW		25.3
300SW		22.8
1000SW		22.9
100NE		26.0

[1] sample locations, distance from discharge in meters; R = reference
*mean value

Table 9-3. Sediment samples from the Delacroix Island area that exceed ERL values (Long et al., 1995) for total and individual PAH concentrations.

Contaminant	ERL (ppb dry weight)	Measured (ppb dry weight)	Location	Sediment Depth (cm)
Total PAH	4,022	9,406	Discharge	0 to 5
		8,143	Discharge	20 to 25
		20,065	Discharge	0 to 5
		6,913	Discharge	35 to 40
		9,142	Discharge	0 to 5
		16,401	Discharge	20 to 25
		6,056	100 m NW	0 to 5
Acenaphthene	16	22	Discharge	0 to 5
		130	Discharge	20 to 25
		41	Discharge	35 to 40
		50	Discharge	0 to 5
		64	Discharge	20 to 25
		190	Discharge	35 to 40
		24	Discharge	0 to 5
		280	Discharge	20 to 25
		19	Discharge	35 to 40
		99	100 m NW	0 to 5
		180	300 m NW	0 to 5
		69	500 m NW	0 to 5
		210	100 m NE	0 to 5
		71	300 m NE	0 to 5
		140	500 m NE	0 to 5
Anthracene	85	150	Discharge	20 to 25
		200	100 m NW	0 to 5
Fluorene	19	53	Discharge	0 to 5
		83	Discharge	20 to 25
		100	Discharge	0 to 5
		48	Discharge	20 to 25
		58	Discharge	35 to 40
		50	Discharge	0 to 5
		76	Discharge	20 to 25
Naphthalene	160	160	Discharge	0 to 5
		200	Discharge	0 to 5
		160	Discharge	0 to 5
		260	Reference 1	35 to 40
Benzo(a)anthracene	261	320	Discharge	20 to 25
		350	Discharge	35 to 40
		1,000	Discharge	20 to 25
		350	100 m NW	0 to 5

Table 9-3. (cont.)

Contaminant	ERL (ppb dry weight)	Measured (ppb dry weight)	Location	Sediment Depth (cm)
Benzo(a)pyrene	430	470	Discharge	20 to 25
Chrysene	384	470	Discharge	20 to 25
		1,200	Discharge	20 to 25
Dibenzo(a,h)anthracene	63	67	Discharge	20 to 25
Fluoranthene	600	1,000	Discharge	20 to 25
		620	Discharge	35 to 40
		1,400	Discharge	35 to 40
		3,500	Discharge	20 to 25
		900	100 m NW	0 to 5
Pyrene	665	2,200	Discharge	20 to 25
		880	Discharge	35 to 40

Table 9-4. Sediment samples from the Bay-de Chene area that exceeded ERL values (Long et al., 1995) for total and individual PAH concentrations.

Contaminant	ERL (ppb dry weight)	Measured (ppb dry weight)	Location	Depth (cm)
Total PAH	4022	23723	Discharge	0 to 5
		18003	Discharge	20 to 25
		35369	Discharge	35 to 40
		162152	Discharge	0 to 5
		28980	Discharge	20 to 25
		49963	Discharge	35 to 40
		32179	Discharge	0 to 5
		31482	Discharge	20 to 25
		43359	Discharge	35 to 40
		6336	300 m NE	0 to 5
		5370	100 m NW	0 to 5
		4075	300 m NW	0 to 5
		11577	100 m NE	0 to 5
Acenaphthene	16	180	Discharge	0 to 5
		69	Discharge	20 to 25
		99	Discharge	35 to 40
		210	Discharge	0 to 5
		71	Discharge	20 to 25
		140	Discharge	35 to 40
		250	Discharge	0 to 5
		110	Discharge	20 to 25
		140	Discharge	35 to 40
		48	100 m NE	0 to 5
		20	300 m NE	0 to 5
Anthracene	85.3	250	Discharge	0 to 5
		150	Discharge	20 to 25
		160	Discharge	35 to 40
		1000	Discharge	0 to 5
		300	Discharge	20 to 25
		220	Discharge	35 to 40
		470	Discharge	0 to 5
		210	Discharge	20 to 25
		180	Discharge	35 to 40
		86	100 m NE	0 to 5
Fluorene	19	230	Discharge	0 to 5
		130	Discharge	20 to 25
		240	Discharge	35 to 40
		390	Discharge	0 to 5
		150	Discharge	20 to 25
		350	Discharge	35 to 40
		340	Discharge	0 to 5
		210	Discharge	20 to 25
		320	Discharge	35 to 40
		22	100 m NW	0 to 5
		33	300 m NW	0 to 5
		67	100 m NE	0 to 5

Table 9-4. (cont.)

Contaminant	ERL (ppb dry weight)	Measured (ppb dry weight)	Location	Depth (cm)
Naphthalene	160	160	Discharge	0 to 5
Phenanthrene	240	890	Discharge	0 to 5
		300	Discharge	20 to 25
		600	Discharge	35 to 40
		1800	Discharge	0 to 5
		370	Discharge	20 to 25
		890	Discharge	35 to 40
		1400	Discharge	0 to 5
		490	Discharge	20 to 25
		680	Discharge	35 to 40
		250	100 m NE	0 to 5
		260	300 m NE	0 to 5
Benzo(a)anthracene	261	960	Discharge	0 to 5
		470	Discharge	20 to 25
		330	Discharge	35 to 40
		12000	Discharge	0 to 5
		780	Discharge	20 to 25
		490	Discharge	35 to 40
		1400	Discharge	0 to 5
		760	Discharge	20 to 25
		340	100 m NE	0 to 5
		350	300 m NE	0 to 5
Benzo(a)pyrene	430	850	Discharge	0 to 5
		9000	Discharge	0 to 5
		530	Discharge	20 to 25
		1200	Discharge	0 to 5
		650	Discharge	20 to 25
Chrysene	384	1000	Discharge	0 to 5
		600	Discharge	20 to 25
		470	Discharge	35 to 40
		11000	Discharge	0 to 5
		790	Discharge	20 to 25
		600	Discharge	35 to 40
		1300	Discharge	0 to 5
		820	Discharge	20 to 25
		470	100 m NE	0 to 5
Dibenzo(a,h)anthracene	63.4	150	Discharge	0 to 5
		78	Discharge	20 to 25
		1700	Discharge	0 to 5
		95	Discharge	20 to 25
		83	Discharge	35 to 40
		210	Discharge	0 to 5
		130	Discharge	20 to 25
		70	100 m NE	0 to 5

Table 9-4 (cont.)

Contaminant	ERL (ppb dry weight)	Measured (ppb dry weight)	Location	Depth (cm)
Fluoranthene	600	2100	Discharge	0 to 5
		1000	Discharge	20 to 25
		780	Discharge	35 to 40
		8100	Discharge	0 to 5
		1300	Discharge	20 to 25
		1200	Discharge	35 to 40
		2700	Discharge	0 to 5
		1700	Discharge	20 to 25
		800	Discharge	35 to 40
		910	100 m NE	0 to 5
		650	300 m NE	0 to 5
Pyrene	665	1500	Discharge	0 to 5
		810	Discharge	20 to 25
		6100	Discharge	0 to 5
		940	Discharge	20 to 25
		960	Discharge	35 to 40
		1900	Discharge	0 to 5
		1300	Discharge	20 to 25
		730	100 m NE	0 to 5

With the exception of acenaphthene, individual and total PAH concentrations exceeded ERL criteria at, and 100 m from the discharge at Delacroix Island (Table 9-3). Acenaphthene concentrations exceeded the ERL values at the discharge, 100, 300 and 500 m stations. Neither individual nor total PAH concentrations in sediment samples from Delacroix Island exceeded ERM criteria.

Individual and total PAH concentrations exceeded ERL criteria at the discharge, and 100 m and 300 m from the discharge at Bay de Chene (Table 9-4). Individual and total PAH concentrations in samples from the discharge sediment exceeded ERM criteria (Table 9-5).

Table 9-5. PAH concentrations in marine sediments (dry weight) at Bay de Chene that exceed ERM concentrations.

Contaminant	ERM (ppb)	Measured (ppb)	Location	Sediment Depth (cm)
Total PAH	44,792	162,152	Discharge	0 to 5
		49,963	Discharge	35 to 40
Benzo(a)anthracene	1,600	12,000	Discharge	0 to 5
Benzo(a)pyrene	1,600	9,000	Discharge	0 to 5
Chrysene	2,800	11,000	Discharge	0 to 5
Dibenzo(a,h)anthracene	260	1,700	Discharge	0 to 5
Fluoranthene	5,100	8,100	Discharge	0 to 5
Pyrene	2,600	6,100	Discharge	0 to 5
High Molecular Weight PAH	9,600	47,900	Discharge	0 to 5

The field studies showed depression of numbers of species (amphipod, gastropod, bivalve, and polychaetes) and/or individuals at less than 100 m from the discharges (Mulino *et al.*, 1995; 1996). The pre-termination benthic effects were greater at the Delacroix Island discharge station than at the comparable Bay de Chene station. Mulino *et al.* (1995; 1996) explained this on the basis of hydrology of the environment. Although the Delacroix discharge was approximately half that at Bay de Chene, there was less opportunity for turbulent mixing and dilution of the discharge because the Delacroix environment was semi-enclosed. It was suggested that the Delacroix discharge was more likely to produce a hypersaline nonoxygenated layer on the bottom, as supported by data on the chlorinity of pore water from the 2 sites.

Mulino *et al.* (1996) did a stepwise multiple regression analysis to look for correlations of PAH concentrations at the stations with benthic biota data. Fluoranthene at Bay de Chene was the only PAH, of those exceeding the criteria values in Tables 9-3 and 9-4, that showed a negative correlation with the benthic data. Dibenzothiophene was the only other PAH that exhibited a (negative) correlation with the benthic biota data.

These results cannot be applied to all other open bay discharge sites with much confidence, but the pre-termination discharge rates and depths of the Bay de Chene and Delacroix Island study sites are comparable (discharge rates are on the high end of distribution) to those that are continuing to discharge (see section 5). Screening criteria for the individual chemicals in this case can only indicate potential ecological problems, while field surveys present the effects on biota of the total set of conditions at the time of sampling. Nevertheless, there is good general agreement between the results of the screening assessment with the observations of the field surveys.

9.3 Toxicity of Individual Produced Water Components - Continuing Open Bay Discharges

9.3.1 Screening Analysis

A screening analysis was performed for potential toxic effects from individual contaminants in plumes from continuing open bay discharges. Average and worst-case concentrations of contaminants measured in the discharges (LDEQ permit files) were used to predict water column concentrations. The predicted concentrations were then compared to USEPA and Louisiana water quality criteria.

Concentrations in the discharges were described by data abstracted from LDEQ permit files (section 5). These data contain only values for contaminants detected in the effluent above the reported detection limit, and therefore overestimate average concentrations.

In this preliminary assessment, contaminants were assessed only if: they were reported above detection limits in more than two of the LDEQ permit files; and water quality criteria were available. Mean and maximum chemical contaminant concentrations in the data set for continuing open bay discharges were diluted by a factor of 20 to estimate water concentrations in a plume (Table 9-6). A dilution factor of 20 was chosen to estimate worst-case concentrations because it yields more conservative concentrations than those predicted by the CORMIX model (section 5) at 50 and 200 feet from the discharge. Although most contaminants were assumed to remain totally in solution, dissolved fractions of copper. lead and zinc were assumed to be 0.88, 0.38 and 0.59, respectively (USEPA, 1995a).

Louisiana and USEPA water quality criteria (Table 9-6) were compared to the predicted water concentrations. Ratios were calculated for each contaminant by dividing the concentration predicted in water by the contaminant's acute and chronic water quality criteria. These ratios are here called the Acute Hazard quotient (AHQ) and Chronic Hazard Quotients (CHQ), respectively. Hazard quotients greater than 1 suggest a potential for toxic effects. Results are given in Tables 9-7 and 9-8. Acute criteria were used as standards for LDEQ's mandated toxicity determinations at 50 feet, while chronic criteria were used as standards for LDEQ's mandated determinations at 200 feet.

Table 9-6. Screening concentrations of chemicals at 50 feet (acute) and 200 feet (chronic) from open bay discharges, and water quality criteria.

Contaminant	Predicted Concentration (μg/l)		Acute Water Quality Criteria (μg/l)		Chronic Water Quality Criteria (μg/l)	
	mean	maximum	LDEQ	USEPA	LDEQ	USEPA
Antimony	279.8	1005	–	1500	–	500
Arsenic	3.7	24.9	69	69	36	36
Cadmium	11.6	25.0	45.6	43	10	9.3
Chromium (VI)	4.2	10.0	1100	1100	50	50
Copper	12.7	31.2	4.37	2.9	4.37	–
Lead	1981.0	15751	220	140	8.5	5.6
Mercury	0.4	1.4	2.1	2.1	0.025	0.025
Nickel	50.7	142	75	75	8.3	8.3
Silver	7.2	20.0	–	7.2	–	0.92
Zinc	35.9	188.1	95	95	86	86
Benzene	90.7	477.5	2700	5100	1350	700
Naphthalene	3.0	5.9	–	2300	–	–
Phenol	77.9	600	580	5800	290	–
Toluene	41.6	140	950	6300	475	5000

Worst-case predicted water concentrations exceeded acute water quality criteria for copper, lead, nickel, silver and zinc (Table 9-7). The mean concentration of copper exceeded acute criteria approximately 3 to 4 fold, while the maximum concentration exceeded these criteria by 7 to 11 fold. The mean concentration of lead was approximately one order of magnitude higher than acute criteria, while the maximum concentration was approximately seventy times to slightly more than one order of magnitude higher than acute criteria. Acute criteria values were exceeded two fold by the predicted maximum concentrations of nickel and zinc. Mean and maximum silver concentrations were 2 and 3 times higher than the USEPA acute criterion. Only the maximum concentration of phenol equaled LDEQ's acute criterion value (one order of magnitude lower than that of USEPA).

Table 9-7. Screening-level Acute Hazard Quotients (AHQ): predicted concentrations at 50 feet/ acute water quality criteria (shaded values are those that exceed 1).

Contaminant	AHQ based on LDEQ Water Quality Criteria		AHQ based on USEPA Water Quality Criteria	
	mean	maximum	mean	maximum
Antimony	--	--	0.2	0.7
Arsenic	0.1	0.4	0.1	0.4
Cadmium	0.3	0.6	0.3	0.6
Chromium (VI)	0.004	0.01	0.004	0.01
Copper	2.9	7.1	4.4	10.7
Lead	9.0	71.5	14.1	112.5
Mercury	0.2	0.6	0.2	0.6
Nickel	0.7	1.9	0.7	1.9
Silver	--	--	1.0	2.8
Zinc	0.4	2.0	0.4	2.0
Benzene	0.03	0.2	0.02	0.1
Naphthalene	--	--	0.001	0.002
Phenol	0.1	1.0	0.01	0.1
Toluene	0.04	0.2	0.01	0.02

Table 9-8. Screening-level Chronic Hazard Quotients (CHQ): predicted concentrations at 200 feet / acute water quality criteria (shaded values are those that exceed 1).

Contaminant	CHQ based on LDEQ Water Quality Criteria		CHQ based on USEPA Water Quality Criteria	
	mean	maximum	mean	maximum
Antimony	--	--	0.6	2.0
Arsenic	0.1	0.7	0.1	0.7
Cadmium	1.2	2.6	1.2	2.7
Chromium (VI)	0.1	0.20	0.1	0.2
Copper	2.9	7.1	--	--
Lead	233.0	1853.0	353.7	2512.7
Mercury	14.2	54.0	14.2	54.0
Nickel	6.1	17.1	6.1	17.1
Silver	--	--	7.8	21.7
Zinc	0.4	2.2	0.4	2.2
Benzene	0.1	0.4	0.1	0.7
Naphthalene	--	--	--	--
Phenol	0.3	2.1	--	--
Toluene	0.1	0.3	0.01	0.03

Chronic water quality criteria were exceeded by predicted concentrations of antimony, cadmium, mercury, and the contaminants that exceeded acute toxicity criteria. The maximum concentration of antimony was twice the USEPA's chronic toxicity criterion. Mean and maximum concentrations of cadmium were approximately 1 and 3 times the water quality criteria of both USEPA and LDEQ. LDEQ's acute toxicity criterion is the only available value for copper, and that criterion was exceeded by predicted mean and maximum concentrations by approximately 3 and 7 times. Lead concentrations exceeded chronic criteria values from more than 2 to more than 3 orders of magnitude. USEPA and LDEQ use the same chronic toxicity criteria for each of mercury, nickel, and zinc. Predicted mean and maximum concentrations of mercury respectively exceeded the chronic toxicity criterion by 14 and 54 times. Predicted mean and maximum concentrations of nickel exceeded the chronic toxicity criterion by 6 and 17 times respectively. The predicted maximum concentration of zinc was approximately twice the chronic toxicity criterion. Phenol was the only organic chemical to exceed LDEQ's chronic toxicity criterion; by a factor of two at the predicted maximum concentration.

Because of the conservative nature of this screening analysis, no important effect on aquatic biota can be assumed. Major uncertainties and conservative assumptions in this screening assessment included:

- use of worst-case water concentrations;
- use of average chemical concentrations that excluded zero values; and
- simple comparison to water quality criteria with no reference to specific receptors or end-points of concern in open Louisiana bays.

These analyses serve to eliminate contaminants that do not warrant further time and attention. Arsenic, chromium, benzene, naphthalene and toluene were eliminated from further consideration. Chronic and/or acute chronic water quality criteria were exceeded for antimony, cadmium, copper, lead, mercury, nickel, silver, zinc and phenol.

Contaminants that exceeded chronic water quality criteria (AHQ or CHQ greater than 1) were assessed in a quantitative risk assessment.

9.3.2 Quantitative Risk Assessment

For contaminants not eliminated by the initial screening assessment, permit data (Table 9-9) were used to develop distributions of concentrations in produced water discharges. Contaminants that were not detected were assigned one-half the reported detection limit value. Each contaminant, except cadmium and copper, was assigned a lognormal distribution, after a log probability plot of the frequency of measured values fit a straight line (Layton et al., 1987). Cadmium and copper were assigned custom distributions that matched the relative frequencies of their respective values.

Table 9-9. Distributions of concentrations of contaminants (µg/l) found in discharges from open bay platforms.

Contaminant	Distribution	Mean	SD	Minimum	Maximum
Antimony	Lognormal	3192.6	6268.3	11.8	20,100
Cadmium	Custom	217.9	235.6	0.0015	540
Copper	Custom	227.8	208	5	710
Lead	Lognormal	546.8	934.4	25	2600
Mercury	Lognormal	4.3	11.1	0.0005	41
Nickel	Lognormal	569.1	947.9	20	2,480
Silver	Lognormal	88.8	118	5	400
Zinc	Lognormal	1465.3	2768.3	2.5	10,800
Phenol	Lognormal	1257.3	2743.4	5	12,000

These distributions were used with the relationships established by the CORMIX algorithms to obtain concentrations of each chemical at 200 feet in the plume (see section 5-3). The concentrations at 200 feet (Table 9-10) accounted for fractional solubility of each chemical in water (Table 8-1): 1 for all chemicals, with the exception of: 0.88 for copper; 0.38 for lead; and 0.59 for zinc. The assessment was performed for 200 ft because this is the chronic mixing zone under LDEQ's regulations, and because of limitations on the ability of CORMIX to generate concentrations at the edge of the acute mixing zone (see section 5).

The distributions of predicted chemical concentrations were then used in probabilistic analyses of potential toxicity to biota. These distributions were compared to the lowest of the LDEQ and USEPA acute and chronic toxicity criteria for marine biota (Table 9-6). The comparisons were expressed as ratios (Table 9-11). None of the predicted chemical concentrations (200 ft) exceeded their respective acute toxicity criteria.

Antimony, phenol, and zinc concentrations did not exceed any of their respective chronic toxicity criteria. With the exception of mercury, none exceeded chronic toxicity criteria by an order of magnitude (Table 9-11). The distributions of ratios were then used to determine the probabilities of exceeding the criteria values (Table 9-12, Figure 9-1).

Table 9-10. Predicted concentrations of contaminants (µg/l) in plumes, at 200 feet from discharges of produced waters.

Contaminant	Mean	SD	Minimum	Maximum
Antimony	1.8	2.2	0.02	13.4
Cadmium	0.5	1.3	9.2×10^{-6}	13.2
Copper	0.5	1.2	1.0×10^{-5}	13.6
Lead	0.5	1.1	2.0 x 10'3	12.9
Mercury	0.01	0.03	3.6×10^{-6}	0.6
Nickel	1.4	3.3	3.9×10^{-3}	52.2
Silver	0.2	0.5	2.8×10^{-4}	7.5
Zinc	0.5	0.4	0.007	4.2
Phenol	3.3	9.7	0.01	170.0

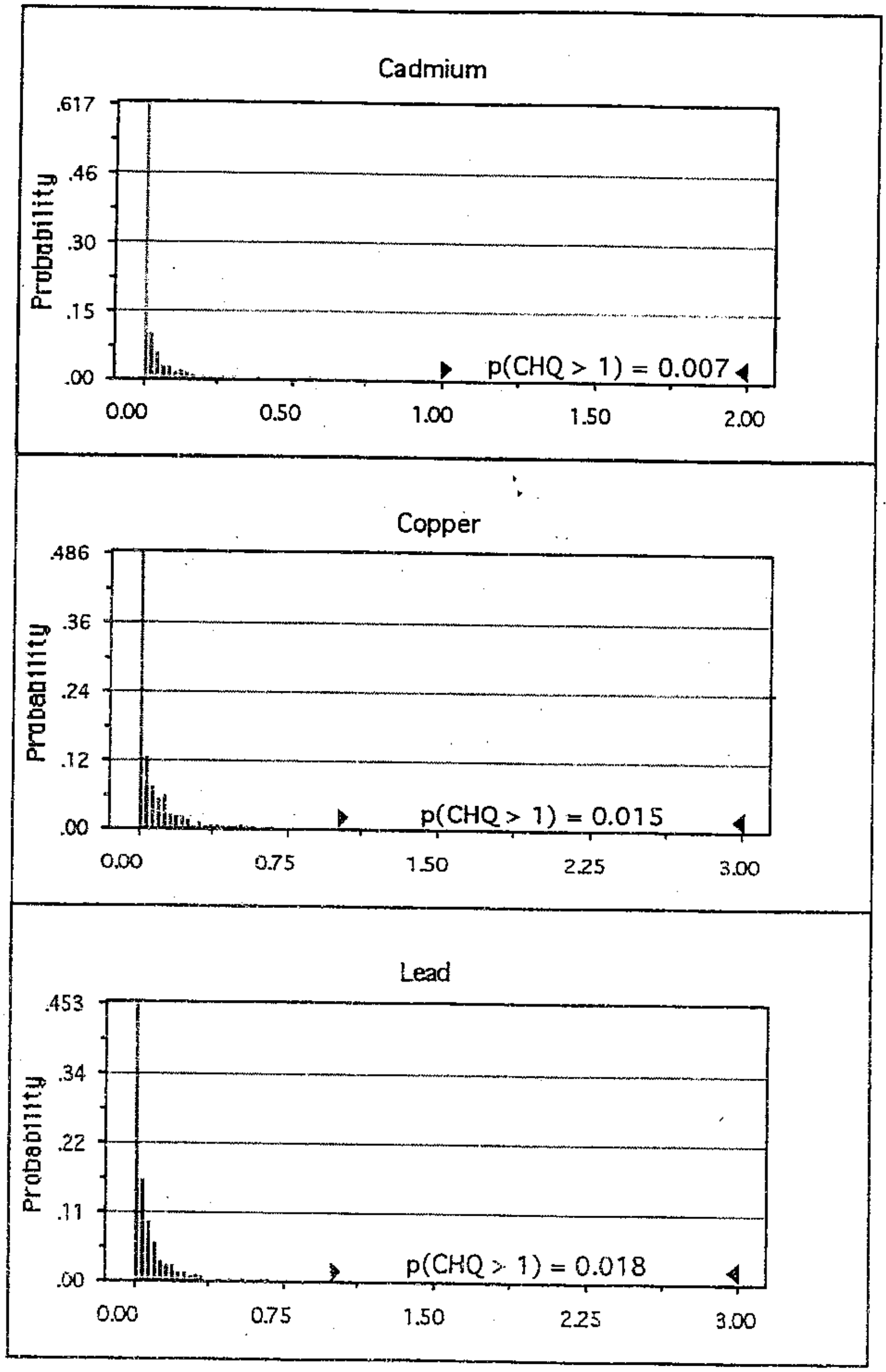
Table 9-11. Chronic Toxicity Hazard Quotients (CHQ) ratios of contaminant concentrations to water quality criteria for chronic toxicity.

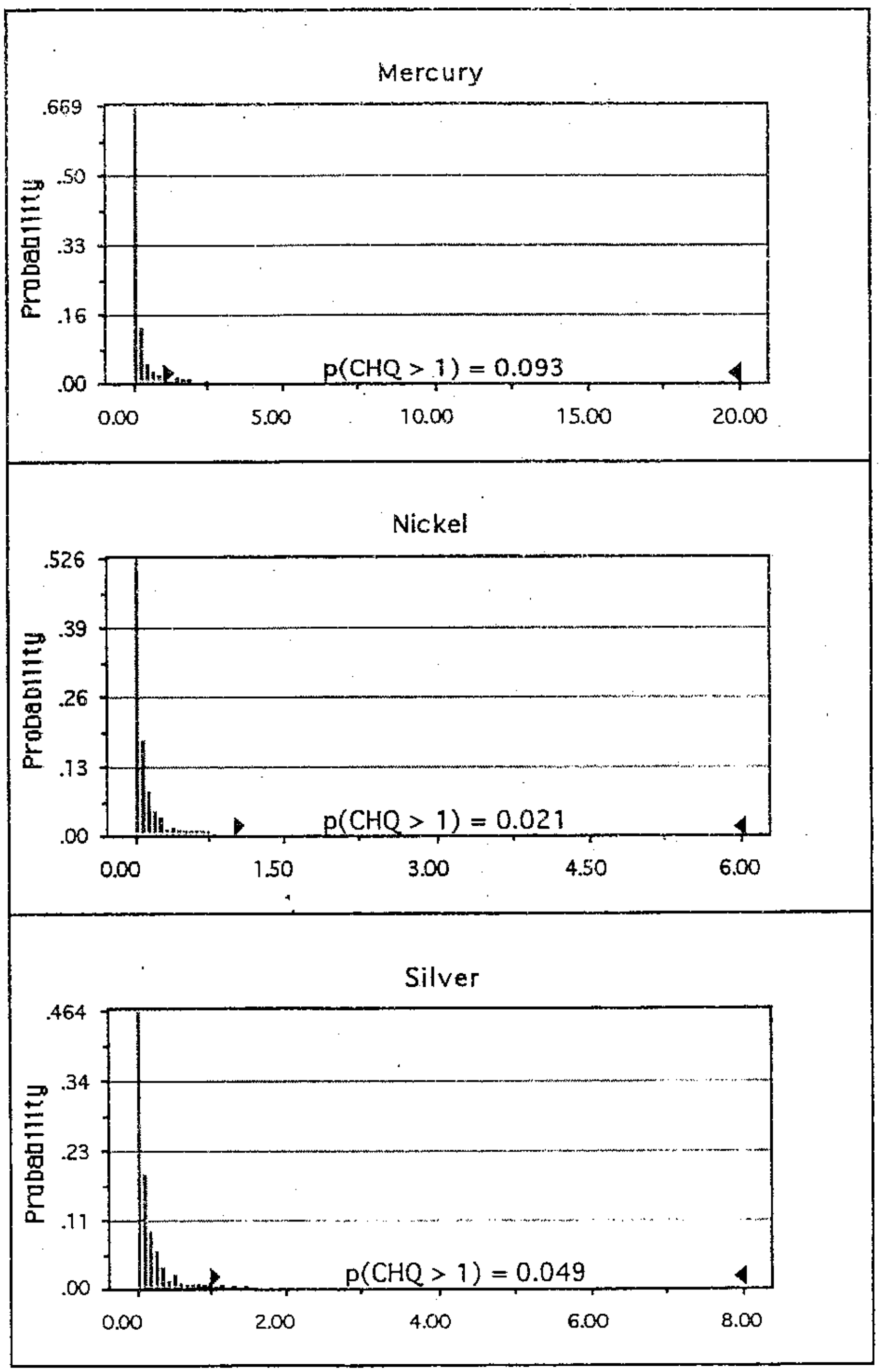
Contaminant	Average CHQ	Median CHQ	SD	Maximum
Antimony	1.3×10^{-4}	4.1×10^{-5}	3.3×10^{-4}	5.8×10^{-3}
Cadmium	0.06	0.01	0.14	1.42
Copper	0.12	0.03	0.27	3.11
Lead	0.12	0.04	0.26	2.93
Mercury	0.40	0.01	1.29	23.20
Nickel	0.16	0.06	0.40	6.29
Silver	0.24	0.10	0.54	8.17
Zinc	5.8×10^{-3}	3.1×10^{-3}	7.3×10^{-3}	4.9×10^{-2}
Phenol	1.1×10^{-2}	3.3×10^{-3}	3.3×10^{-2}	5.9×10^{-1}

Table 9-12. Contaminants with concentrations at 200 feet that were > chronic toxicity criteria for marine organisms, and percent probability of exceeding those criteria.

Contaminant	%p(CHQ) >1
Cadmium	0.7
Copper	1.5
Lead	1.8
Mercury	9.3
Nickel	2.1
Silver	4.9

Figure 9-1. Distribution of chronic hazrd quotients (CHQ) for contaminants that have a CHQ > 1.




Mercury
Probability
.669
.50
.33
.16
.00
p(CHQ > 1) = 0.093
0.00
5.00
10.00
15.00
20.00
Nickel
Probability
.526
.39
.26
.13
.00
p(CHQ > 1) = 0.021
0.00
1.50
3.00
4.50
6.00
Silver
Probability
.464
.34
.23
.11
.00
p(CHQ > 1) = 0.049
0.00
2.00
4.00
6.00
8.00

The predicted values in Table 9-9 represent modeled concentrations of chemicals that would be found at midlines of plumes at 200 ft (~61 m) from discharges of produced water. None of the discharges included in the model yielded concentrations that exceeded acute toxicity criteria at 200 feet. With the exception of mercury, less than five percent of the concentrations of each contaminant, at 200 ft, are expected to result in chronic toxicity to biota. More than 90% of the predicted concentrations of mercury are expected to be below its chronic toxicity criterion.

Physical-chemical properties are not accounted for and it is assumed that components of produced water discharges stay in solution in their plumes, and are freely available to biota. For example, predicted lead concentrations in the water column appear to most greatly exceed acute and chronic toxicity criteria. Under the ordinarily alkaline conditions of briny waters, such as those that might be found in open bays, lead would be expected to form insoluble salts and complexes that tend to precipitate. Thus, metals might not be readily be available to biota in the water. Since these all represent midline values for the plumes, the expectation would be that environmental impacts of the individual chemicals would be limited. However, produced waters are complex mixtures of contaminants that may have a range of interactions from no toxicity to high toxicity. Therefore, the next step was an analysis of actual toxicity testing of diluted whole effluents from produced water discharges (section 9.4).

9.3.3 Relationships between the Screening and Probabilistic Assessments of Individual Components

In the screening assessment, a total of nine individual chemicals exceeded chronic toxicity criteria at 200 ft. Lead, mercury, nickel and silver exceeded one or more of the criteria by at least one order of magnitude. In the quantitative assessment, these chemicals had a 1.8% to 9.2% probability of exceeding at least one chronic criterion. Cadmium and copper exceeded one or more chronic toxicity criteria by less than an order of magnitude in the screening assessment, and respectively had a 0.7% and 1.5% probability of exceeding at least one criterion in the quantitative assessment. In the screening results, antimony, zinc, and phenol exceeded at least one chronic criterion approximately two fold, but did not exceed any criteria in the probabilistic assessment. This suggests that cadmium, copper, lead, mercury, nickel and silver may serve as sentinels for potential toxicity of produced water effluents.

9.4 Toxicity of Whole Effluents -- Continuing Open Bay Discharges

Toxicity tests are useful analytical tools because they can directly measure potential aquatic effects. This is particularly true in the case of complex effluents, such as produced water, where a broad range of toxicants can be present at low levels.

Toxicity test data in LDEQ permit files for assumed continuing discharge sites are summarized in section 5 (Tables 5-4 and 5-5). These data are uncertain because many permits have more than one discharge point, and it was often difficult to correctly match discharge points and toxicity data. These data are also uncertain because both discharge rates and toxicity are likely to change over time.

The estimated distribution of percent effluent expected at 50 ft (~ 15 m; LDEQ acute standard mixing zone) and 200 feet (~ 61 m; LDEQ chronic mixing zone) for the continuing discharges in open bays is given in Table 5-8. For flow rates reported to the LDEQ, previously described relationships between discharge (flow) rates and dilution factors (section 5.3) were used to estimate concentrations of effluents at 50 m and 200 m from discharges (Table 9-13).

Standard laboratory test organisms, a shrimplike crustacean (*Mysidopsis bahia*) and the sheepshead minnow (*Cyprinodon variegatus*), were used in toxicity tests that were reported in LDEQ permits. Predicted water column concentrations of effluents were compared with reported results of acute and chronic toxicity tests on diluted effluent samples. Toxicity test data were expressed in the same way as the predicted water column concentrations: as percent effluent.

Produced water test procedures usually measure mortality responses, with results of acute tests expressed as an effluent median lethal concentration for an exposure duration of 96 hrs (96-hr LC_{50}), or the effluent concentration which results in the mortality of 50% of the test organisms in a 96-hr exposure period. Acute toxicity ratios (AHQ) were calculated between the estimated percent effluent at 50 ft and 200 ft from the discharge and the available corresponding LC_{50} values (*M. bahia*; *C. variegatus*) for each discharge (Tables 9-14, 9-15). Ratios of one or greater indicate potential lethality to each species. Fewer data points were used in this analysis than are reported in Table 5-4 because only discharges with discharge rates less than or equal to 5,000 bbl/d could be used to predict water concentrations at 50 feet (see section 5.3).

At 50 ft, 17% of the modeled effluent concentrations exceeded their respective LC_{50} values *for M. bahia*, and 6% exceeded their respective LC_{50} values for *C. variegatus* (Table 9-14). At 200 ft, 15% of the modeled effluent concentrations exceeded their respective LC_{50} value for *M. bahia* and 3% exceeded their respective LC_{50} value for *C. variegatus* (Table 9-15). These results suggest a potential for lethal effects for some discharges at 50 and at 200 feet.

The data in tables 9-14 and 9-15 suggest either a specific component, or group of components in the effluent from platform 2072 is responsible for the toxicity to *C. variegatus*; or that the fish used were especially sensitive. The AHQ at 200 ft is relatively high, without any comparable toxicity to the usually more sensitive *M. bahia*. This is further supported by the CHQ results.

Table 9-13. Estimated effluent dilutions and concentrations at open bay discharges near the Louisiana coast (BPD = barrels per day) .

Platform Site	Flow Rate BPD	Dilution at 50 ft	% Effluent at 50 ft	Dilution at 200 ft	% Effluent at 200 ft
2856	3	4102.6	0.0244	16378.0	0.0061
3023	3.4	3680.8	0.0272	14549.3	0.0069
2479	10	1444.8	0.0692	5243.6	0.0191
	10	1444.8	0.0692	5243.6	0.0191
2857	10	1444.8	0.0692	5243.6	0.0191
	10	1444.8	0.0692	5243.6	0.0191
3032	25	652.9	0.1532	2203.9	0.0454
	30	557.4	0.1794	1854.7	0.0539
	30	557.4	0.1794	1854.7	0.0539
	41	425.2	0.2352	1380.2	0.0725
1870	49	364.3	0.2745	1166.1	0.0858
	117	171.3	0.5837	511.9	0.1954
2915	130	156.4	0.6395	463.3	0.2158
2816	140	146.6	0.6819	431.9	0.2315
2881	204	105.8	0.9451	302.5	0.3306
	220	99.1	1.0090	281.7	0.3550
	489	49.6	2.0165	132.3	0.7558
2816	510	47.8	2.0914	127.1	0.7865
	600	41.5	2.4078	109.0	0.9172
	614	40.7	2.4564	106.7	0.9374
	701	36.3	2.7554	94.1	1.0626
	729	35.1	2.8505	90.7	1.1027
	802	32.3	3.0964	82.9	1.2069
	1103	24.5	4.0816	61.3	1.6316
	1201	22.8	4.3942	56.5	1.7684
	2065	14.2	7.0294	33.9	2.9528
2084	2484	12.1	8.2502	28.4	3.5167
	2485	12.1	8.2531	28.4	3.5180
2825	2910	10.6	9.4636	24.5	4.0847
2898	3000	10.3	9.7168	23.8	4.2041
	3017	10.2	9.7645	23.7	4.2267
	3720	8.5	11.7086	19.4	5.1529
2523	5364	--	--	52.0	1.9218
2860	6800	--	--	43.4	2.3024
2672	8366	--	--	37.1	2.6962
1901	10123	--	--	32.1	3.1175
2859	10807	--	--	30.5	3.2767
	11500	--	--	29.1	3.4355
3063	11500	--	--	29.1	3.4355
2142	12076	--	--	28.0	3.5658
	20077	--	--	19.0	5.2520
2072	20250	--	--	18.9	5.2865
2504	37113	--	--	11.9	8.3863

Table 9-14. Effluents ≥ LC_{50} at 50 ft from discharges, and ratios of their concentrations[1] to their respective LC_{50} values for each species*.

Platform	Discharge (bbl/d)	Ratio of Effluent Concentration to LC_{50}	
		Mysidopsis bahia[2]	*Cyprinodon variegatus*[3]
2816	140	3.4	--
2084	802	4.4	--
2084	2,484	5.2	1.8
2825	2,910	1.007	--
2898	3,000	1.2	1.6

* only discharges ≤ 5,000 bbl/d
[1]Percent effluent
[2]LC_{50} results available for 30 discharges
[3]LC_{50} results available for 32 discharges

Table 9-15. Effluents ≥ LC_{50} at 200 ft from discharges, and ratios of their concentrations[1] to their respective LC_{50} values for each species.

Platform	Discharge (bbl/d)	Ratio of Effluent Concentration to LC_{50}	
		Mysidposis bahia[2]	*Cyprinodon variegatus*[3]
2816	140	1.2	--
2084	802	1.7	--
2084	2,484	2.2	--
1901	10,123	1.7	--
2859	10,807	3.0	--
2072	20,250	--	2.2
2504	37113	2.9	--

[1]Percent effluent
[1]Percent effluent
[2]LC_{50} results available for 41 discharges
[3]LC_{50} results available for 39 discharges

Chronic toxicity ratios were calculated for the estimated percent effluent at 200 ft and the available corresponding chronic NOEL values for survival and growth inhibition. Ratios greater than one suggest a potential for toxic effects. Results of these ratio tests are shown in Tables 9-16 and 9-17.

At 200 ft, 37% of the modeled effluent concentrations exceed their respective survival NOEL values for *M. bahia*, and 20% exceed their respective survival NOEL value *for C. variegatus* (Table 9-16). At 200 ft, 38% of the modeled effluent concentrations exceeded their respective growth-inhibition NOEL values for *M. bahia*, and 18% exceeded their respective growth-inhibition NOEL values for *C. variegatus* (Table 9-17).

The results suggest a potential for chronic effects within 200 feet of some discharges. All the AHQs and CHQs were determined to be lognormal distributions, as exemplified by the linearity of the plot in Figure 9-2.

Table 9-16. Survival ratios greater than one (percent effluent at 200 feet/ percent effluent NOEL).

Platform	Discharge (bbl/d)	Survival Ratio: percent effluent/NOEL	
		Mysidopsis bahia[1]	*Cyprinodon variegatus*[2]
2816	140	5.8	1.5
2816	614	6.7	--
2084	701	--	1.2
2084	802	6.4	--
2084	1,201	--	1.4
2084	2,484	12.6	--
2881	2,485	3.2	--
2825	2,910	1.7	--
2898	3,000	6.7	2.1
2084	3,017	--	1.5
2084	3,720	1.7	1.7
2523	5,364	1.1	--
1901	10,123	5.0	--
2859	10,807	3.3	--
1901	11,500	6.9	--
3063	11,500	--	1.4
2142	12,076	1.4	--
1901	20,077	5.2	--
2072	20,250	--	4.8
2504	37113	5.1	--

[1]survival test results available for 43 discharges
[2]survival test results available for 41 discharges.

A ratio of one was exceed for AHQ and CHQ values by at least two times greater percent of tests on *M. bahia* than on *C. variegatus*.

These results should be taken only as an indicator of potential toxicity. The percent effluent values exceeded their respective LC_{50} and NOEL values by small amounts. Controlled laboratory conditions of the toxicity tests, and the conservative CORMIX modeling constraints, do not reproduce the variable chemical and physical conditions of the open bay environment. Under natural conditions, effluent components probably vary in the water column. Therefore, it is likely that comparisons of percent effluent (at 50 or 200 feet) with percent effluent acute or chronic toxicity values yielded toxicity ratios that are overestimates.

Table 9-17. Growth-inhibition ratios greater than one (percent effluent at 200 feet/ percent effluent NOEL).

Platform	Discharge (bbl/d)	Growth Inhibition Ratio: percent effluent/NOEL	
		Mysidopsis bahia[1]	*Cyprinodon variegatus*[2]
2816	140	3.3	1.5
2816	614	3.5	--
2084	802	6.4	--
2881	1,103	1.6	--
2084	2,484	1.6	--
2896	3,000	6.7	2.1
2084	3,017	1.5	1.5
2084	3,720	3.4	1.7
2523	5,364	1.1	--
2860	6,800	1.2	--
1901	10,123	5.0	--
2859	10,807	3.3	--
1901	11,500	6.9	--
3063	11,500	--	1.4
2142	12,076	1.4	1.4
1901	20,077	10.5	--
2072	20,250	--	4.8
2504	37113	1.3	--

[1]survival test results available for 42 discharges
[2]survival test results available for 39 discharges.

Since the percent effluent values compared to the NOELs in this analysis represent the concentrations at the midline of the plume at 200 ft from the discharges, an organism would have to live totally in the plume, within 200 ft of the discharges for at least the period of the chronic test to be affected. This is unlikely because the plume is a relatively small fraction of the volume of water within 200 ft of a platform. That volume, in turn, is a small fraction of the body of water in which the discharge occurs. Therefore, major impacts to local populations or to the ecology of the region around open bay discharges are not expected.

The estimates of toxicity to biota are highly uncertain because of the previously described variability in natural conditions versus the controlled conditions of laboratory tests. It is also difficult to sort out the uncertainty associated with the estimation of individual effluent concentrations because of the limitations of the conservative CORMIX model (section 5-3).

Sensitivity analyses were done to see the effects of lowering all effluent concentrations at 50 and 200 ft by 20% (Table 9-18).

Figure 9-2. Logarithmic distribution of ratios between % effluent concentrations and the LOEL % effluent concentrations for inhibition of growth in *Mysidopsis bahia* in produced waters from Louisiana open bay platforms.

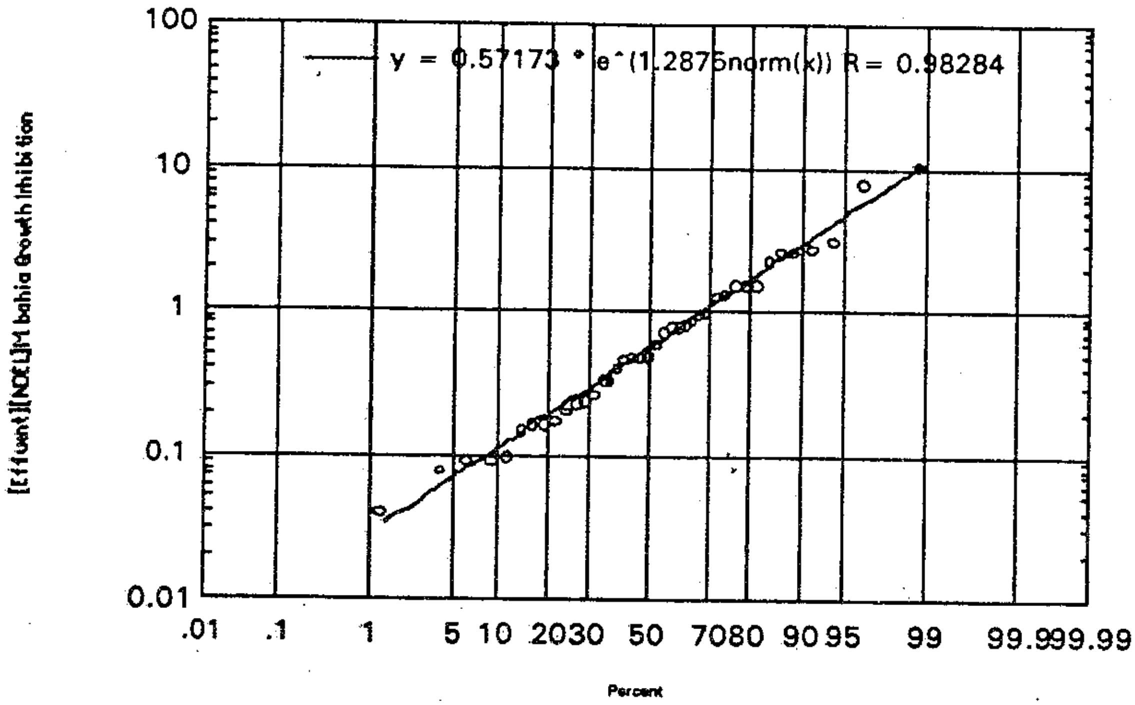


Table 9-18. Sensitivity of results to 20 percent reductions of all effluent concentrations, or when discharges equal or exceed 5000 bbl/d. Values are percentage of effluent concentration values that equal or exceed respective toxicity assay results.

Organism & Analysis	LC_{50} (50 ft)	Survival (200 ft)	Growth Inhibition (200 ft)
Mysidopsis bahia			
[Effluent] .	17	37	38
[Effluent x 0.8]	10	33	33
Cyprinodon variegatus			
[Effluent]	6	20	18
[Effluent x 0.8]	6	17	18

Twenty percent reductions in effluent concentrations produced varying reductions in toxicity parameters (Table 9-18: 1 versus 2; 3 versus 4).

Although the effluent concentration estimates may be uncertain, the findings of potential toxicity up to 200 ft from the discharges agree with field observations of reduced numbers of benthic species and individual animals within 100 m of discharges in open bays off the coast of Louisiana (Mulino *et al.*, 1995)

Regression methods were used to look for linear, exponential, logarithmic or power relationships for the following sets of data:

- between estimated concentrations at 50 and 200 ft from each discharge and the respective LD_{50} values (acute toxicity) for each discharge;
- between LD_{50} values (acute) and their respective NOEL values (chronic);
- between NOEL values for survival and NOEL values for growth inhibition;
- between estimated effluent concentrations at 50 and 200 ft and their respective acute and chronic ratios;
- between acute and chronic ratios at 50 and 200 ft
- relationships between data for *M. bahia* and data for *C. variegatus*.

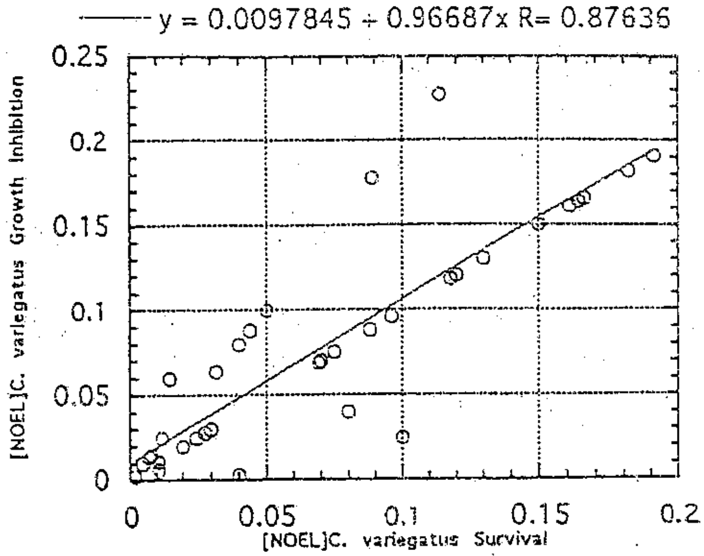
No significant relationships ($R \geq .75$) were detected, except between NOEL values for survival and NOEL values for growth inhibition, as demonstrated in Figure 9-3. The absence of relationships may arise from several sources:

- over-estimates of effluent concentrations for low discharge rates, as described above;
- qualitative and quantitative variability in the toxic components of the effluents;
- interspecific and intraspecific differences in response to toxicity of the effluents.

9.5 Discussion

Comparison of the results of the analyses of toxicology testing of whole effluents, and the results of the analyses of individual components of produced waters, suggest that individual component analyses are not enough to explain the toxicity of produced water effluents in the water column. These analyses, the screening study of sediment components, and the field observations on benthic animals indicate that there is a potential for detrimental effects on open bay biota within LDEQ's chronic mixing zone (200 feet from the discharge). Permanent damage to regional populations of organism and ecosystems are not expected, because mixing zones represent relatively small volumes, in bodies of water with greater energy than previously studied coastal waters (*e.g.*, canals; Boesch and Rabalais, 1989; St. Pe', 1990).

Figure 9-3. NOEL (as percent effluent) for growth inhibition as a power function of the NOEL for survival of *Cyprinodon variegatus* exposed to produced waters from Louisiana open bay platforms.



10 SUMMARY AND CONCLUSIONS

A tiered risk assessment approach was used for human health and ecological risks. Screening-level assessments identified potentially important contaminants and eliminated others from further consideration. Based on the results of these preliminary analyses, additional probabilistic risk assessments were done for the human health and ecological risks of contaminants that were identified as being of potential concern.

10.1 Human Health Risk Assessment for Radium

Screening and probabilistic human health risk assessments were done for open bay radium discharges in Louisiana. In the conservative screening analysis, estimated risks for ingestion of radium in fishes exceeded 1×10^{-6} in all cases. These results are from a conservative, screening level assessment, and do not represent best estimates of risk associated with radium discharged by open bay platforms. They do, however, suggest the need for a more detailed, probabilistic assessment.

A probabilistic risk assessment was done using distributions of: radium concentrations in fish; rates of ingestion of fish by recreational fishermen and their families; and risk factors. The 95th percentile individual lifetime fatal cancer risks for both DOE study sites (Delacroix Island and Bay de Chene) were less than 1×10^{-5}. The 95th percentile individual lifetime fatal cancer risk for continuing open bay discharges was 4.3×10^{-6}, in good agreement with the DOE study site results.

These results suggest that the ingestion of radium in fish near open bay produced water platforms does not present an important risk to human health.

10.2 Ecological Risk Assessment for Radionuclides

In a simple screening analysis, none of the predicted doses to aquatic animals from radionuclides in produced water discharges exceeded the IAEA range associated with only potential minor effects on individual animals. Because of the conservative nature of this initial analysis, it can be concluded that no effects on aquatic animals from radionuclides discharged in produced water to open bays in Louisiana are expected.

10.3 Human Health Risk Assessment for Chemical Contaminants

A screening human health risk assessment was done for metals and organic compounds measured in continuing open bay discharges. This analysis followed the USEPA approach to estimating risks from toxic materials and carcinogens by applying RfD (reference dose) and slope factor values to estimates of chemical intake rates (USEPA, 1989). Predicted water concentrations were also compared to USEPA and Louisiana human health water quality criteria.

Arsenic, chromium, copper, silver, naphthalene, toluene and xylenes were eliminated from further consideration. Contaminants of potential concern identified in this screening step included antimony, benzene, cadmium, lead, mercury, nickel, zinc and phenol.

A more realistic and quantitative assessment was performed for contaminants identified in this screening analysis. The results show that intakes of contaminants discharged to open bays in produced water pose a negligible hazard to human health.

The potentially toxic contaminants examined (antimony, cadmium, mercury, nickel, zinc and phenol; lead was analyzed separately) all had low risks of toxic effects. The only contaminant that marginally exceeded its oral RfD value was cadmium.

Because of the concern for lead exposure to children, and the current belief that the dose-response function for lead exposure does not have a threshold, lead was analyzed in a separate probabilistic risk assessment. Risk from ingestion of lead in fish caught near platforms only slightly exceeded risks from background intake of lead and was similar to risks from ingestion of lead in fish caught in the Gulf of Mexico but not near platforms.

For benzene, the predicted distribution of values for incremental individual lifetime risk of carcinogenic mortality had a mean value of 1.6×10^{-6} and a 95th percentile value of 7.4×10^{-6}. This is within the range considered acceptable by USEPA (1×10^{-6} to 1×10^{-4}; Federal Register, 1991).

10.4 Ecological Risk Assessment for Chemical Contaminants and Total Effluent

Three ecological risk assessments were performed: a screening assessment of chemical toxicity to benthic biota; an assessment of potential toxicity of individual produced water components to fish and crustaceans in the water column; and an assessment of whole effluent toxicity to fish and crustaceans.

Screening Assessment Of Sediment Toxicity

Sediment metal and PAH concentrations measured at USDOE study sites (data collected before termination of discharges) were compared to proposed sediment quality criteria (ERM: Effects Range Median; ERL: Effects Range Low; Long *et al.*, 1995).

None of the measured concentrations of metals in sediment samples exceeded their respective ERM values. In general, measured sediment concentrations were below the ERL, with the exception of arsenic and nickel. Each of these metals exceeded its ERL value in samples from at least one reference site, and each discharge site. There was no clear pattern of concentration with distance from a discharge.

With the exception of acenaphthene, individual and total PAH concentrations exceeded ERL criteria at, and 100 m from the discharge at Delacroix Island. Acenaphthene concentrations exceeded the ERL values at the discharge, 100, 300 and 500 m sample sites. Neither individual nor total PAH concentrations in sediment samples from Delacroix Island exceeded ERM criteria.

Individual and total PAH concentrations exceeded ERL criteria at the discharge site, and 100 m and 300 m from the discharge at Bay de Chene. Individual and total PAH concentrations in samples from the discharge site exceeded ERM criteria.

In preliminary results of the benthos sampling performed at the USDOE study sites Mulino *et al* (1995; 1996) depressed numbers of individuals and numbers of species were found only at distances less than 100 m from the discharges. Although comparisons of PAH concentrations to criteria were generally consistent with the results of benthos observations, they could not explain differences between the benthic biota at the two study sites. Mulino *et al.*, (1996) attributed the more severe impacts at Delacroix Island (smaller discharge) to hydrologic influences on salinity and oxygen content of the water.

These results are preliminary, and cannot be applied to all other open bay discharge sites with much confidence, but the discharge rates and depths of the Bay de Chene and Delacroix Island study sites are comparable (discharge rates are on high end of distribution) to those that are continuing to discharge.

Assessment Of Potential Toxicity Of Individual Contaminants In The Water Column

Worst-case predicted water column concentrations of contaminants measured in continuing open bay effluents (LDEQ permit files) were compared to USEPA and Louisiana water quality criteria.

Worst-case predicted water concentrations exceeded acute water quality standards for copper, lead, nickel, silver and zinc. Chronic water quality criteria were exceeded for antimony, cadmium, copper, lead, mercury, nickel, silver, zinc and phenol. Contaminants eliminated from further consideration included arsenic, chromium, benzene, naphthalene and toluene.

For contaminants not eliminated by the initial screening assessment, a quantitative risk assessment was done. Distributions of predicted chemical concentrations were compared to acute and chronic toxicity criteria for marine biota.

None of the predicted chemical concentrations (200 ft) exceeded their respective acute toxicity criteria. Antimony, phenol, and zinc concentrations did not exceed any of their respective chronic toxicity criteria. Less than five percent of the concentrations of cadmium, copper, lead, nickel and silver, at 200 ft, are expected to result in chronic toxicity to biota. More than 90% of the predicted concentrations of mercury are expected to be below its chronic toxicity criterion. Since these all represent midline values for the plumes, the expectation would be that environmental impacts of the individual chemicals would be limited.

Assessment Of Effluent Toxicity

Standard laboratory test organisms, an amphipod (*Mysidopsis bahia*) and the sheepshead minnow (*Cyprinodon variegatus*), were used in toxicity tests that were reported in LDEQ permits. Predicted water column concentrations of effluents were compared with reported results of acute and chronic toxicity tests on diluted effluent samples. For the results of each type of toxicity test, data were expressed in the same way as the predicted water column concentrations: as percent effluent.

For discharges reported to the LDEQ, modeled relationships between discharge (flow) rates and dilution factors were used to estimate concentrations of effluents at 50 m and 200 m from discharges.

Acute toxicity ratios (AHQ) were calculated between the estimated percent effluent at 50 ft and 200 ft from the discharge and the available corresponding LC_{50} values for each platform. Ratios of one or greater indicate potential lethality to each species.

At 50 ft, 17% of the modeled effluent concentrations exceeded their respective LC_{50} values *for M. bahia*, and 6% exceeded their respective LC_{50} values for *C. variegatus*. At 200 ft, 15% of the modeled effluent concentrations exceeded their respective LC_{50} value for *M. bahia* and 3% exceeded their respective LC_{50} value for *C. variegatus*. The results suggest a potential for lethal effects for some discharges at 50 and at 200 feet.

Chronic toxicity ratios were calculated for the estimated percent effluent at 200 ft and the available corresponding chronic NOEL values for survival and growth inhibition. Ratios greater than one suggest a potential for toxic effects.

At 200 ft, 37% of the modeled effluent concentrations exceed their respective survival NOEL values for *M. bahia*, and 20% exceed their respective survival NOEL value *for C. variegatus. At* 200 ft, 38% of the modeled effluent concentrations exceeded their respective growth-inhibition NOEL values for *M. bahia*, and 18% exceeded their respective growth-inhibition NOEL values for *C. variegatus*. Approximately two times more of the predicted effluent concentrations exceeded chronic NOEL values (both survival and growth-inhibition) for *M. bahia* than for *C. variegatus*.

The results suggest a potential for chronic effects within 200 feet of some discharges. These results should be taken only as an indicator of potential toxicity. The percent effluent values exceeded their respective NOEL values by small amounts.

Since the percent effluent values compared to the NOEL in this analysis represent the concentrations at the midline of the plume at 200 ft from the discharges, an organism would have to live totally in the plume, within 200 ft of the discharges for at least the period of the chronic test to be affected. This is unlikely because the plume is a relatively small fraction of the volume of water within 200 ft of a platform. That volume, in turn, is a small fraction of the body of water in which the discharge occurs. Therefore, major effects to local populations or to the ecology of the region around open bay discharges is not expected.

10.5 Conclusions

The tiered approach to risk assessment is a cost-effective way to provide information needed to make risk management decisions. This screening assessment for human health and ecological risks from open bay produced water discharges in Louisiana eliminated a number of contaminants from further consideration. More quantitative assessments were performed on contaminants of potential concern.

Human health risks from radium in produced water appear to be small. Ecological risks from radium and other radionuclides in produced water also appear to be small.

Intakes of chemical contaminants in fish caught near open bay produced water discharges are expected to posed a negligible toxic hazard or carcinogenic risk to people.

Potential impacts to benthic biota and fish and crustaceans in the water column are possible for some discharges within the 200 ft mixing zone. Permanent damage to populations of organisms and ecosystems are not expected, because mixing zones represent relatively small volumes and animals are not expected to remain continuously in the plume.

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APPENDIX A

USDOE OPEN BAY SITES: PRELIMINARY DATA

Table A-1. Preliminary radium data in tissue collected at Delacroix Island and Bay De Chene.

Site	Survey	Station	Organism	Tissue Type*	Number of Specimens in Composite	^{226}Ra [LLD**] (pCi/g)	^{228}Ra [LLD] (pCi/g)
Bay de Chene	Spring 1993	Discharge	Croaker	Whole	11	0.021 [0.004]	0.038 [0.012]
					11	0.014 [0.004]	0.094 [0.013]
					11	0.024 [0.004]	0.067 [0.012]
					11	0.008 [0.004]	0.040 [0.012]
					11	0.004 [0.004]	0.029 [0.012]
Bay de Chene	Spring 1993	Discharge	Spot	Whole	15	0.034 [0.007]	0.073 [0.014]
					15	0.023 [0.003]	0.086 [0.009]
					15	0.024 [0.003]	0.018 [0.007]
					15	0.026 [0.003]	0.048 [0.009]
					15	0.019 [0.003]	0.026 [0.009]
Bay de Chene	Spring 1993	Discharge	Seatrout	Whole	8	0.021 [0.004]	0.057 [0.012]
					8	0.016 [0.004]	0.159 [0.011]
					8	0.016 [0.006]	0.121 [0.014]
					8	0.004 [0.003]	0.037 [0.009]
					8	0.004 [0.003]	0.105 [0.009]
Bay de Chene	Spring 1993	Discharge	Blue Crab	Edible	2	0.023 [0.003]	0.056 [0.009]
					2	0.009 [0.003]	0.058 [0.009]
					2	0.020 [0.003]	0.041 [0.008]
					2	0.017 [0.003]	0.059 [0.008]
Bay de Chene	Spring 1993	Discharge	Shrimp	Edible	73	0.007 [0.004]	0.026 [0.010]
					73	0.006 [0.004]	BDL† [0.010]
					73	0.006 [0.004]	BDL [0.010]
					73	0.007 [0.004]	BDL [0.010]
					73	0.011 [0.004]	BDL [0.016]
Bay de Chene	Spring 1993	Reference 1	Croaker	Whole	14	0.027 [0.003]	BDL [0.015]
					14	0.009 [0.003]	BDL [0.015]
					14	0.011 [0.003]	BDL [0.015]
					14	0.010 [0.004]	0.046 [0.019]
					14	0.024 [0.003]	BDL [0.018]

Site	Survey	Station	Organism	Tissue Type*	Number of Specimens in Composite	^{226}Ra [LLD**] (pCi/g)	^{226}Ra [LLD] (pCi/g)
Bay de Chene	Spring 1993	Reference 1	Spot	Whole	8	0.029 [0.003]	BDL [0.018]
					8	0.020 [0.003]	BDL [0.019]
					8	0.024 [0.003]	BDL [0.018]
					8	0.022 [0.003]	BDL [0.018]
					8	0.024 [0.003]	BDL [0.019]
Bay de Chene	Spring 1993	Reference 1	Seatrout	Whole	4	0.016 [0.004]	0.020 [0.007]
Bay de Chene	Spring 1993	Reference 1	Shrimp	Edible	28	0.027 [0.004]	BDL [0.021]
					28	0.013 [0.004]	BDL [0.021]
					28	0.005 [0.003]	BDL [0.018]
					28	0.014 [0.003]	BDL [0.019]
Bay de Chene	Spring 1993	Reference 1	Blue Crab	Edible	20	0.012 [0.003]	BDL [0.019]
Bay de Chene	Spring 1993	Reference 2	Croaker	Whole	13	0.031 [0.004]	BDL [0.010]
					13	0.032 [0.004]	BDL [0.011]
					13	0.024 [0.004]	BDL [0.011]
					13	0.014 [0.004]	BDL [0.010]
					13	0.032 [0.004]	BDL [0.012]
Bay de Chene	Spring 1993	Reference 2	Spot	Whole	10	0.021 [0.004]	BDL [0.013]
					10	0.023 [0.004]	BDL [0.012]
					10	0.008 [0.004]	BDL [0.013]
					10	0.022 [0.004]	BDL [0.012]
					10	BDL [0.004]	0.042 [0.010]
Bay de Chene	Spring 1993	Reference 2	Seatrout	Whole	9	BDL [0.004]	0.036 [0.010]
					9	0.007 [0.004]	0.032 [0.010]
					9	0.012 [0.004]	BDL [0.010]
Bay de Chene	Spring 1993	Reference 2	Shrimp	Edible	111	0.021 [0.003]	BDL [0.017]
					111	0.013 [0.003]	BDL [0.018]
					111	0.006 [0.003]	BDL [0.018]
					111	0.010 [0.003]	BDL [0.018]
					111	0.010 [0.004]	0.124 [0.01]

Site	Survey	Station	Organism	Tissue Type*	Number of Specimens in Composite	^{226}Ra [LLD**] (pCi/g)	^{228}Ra [LLD] (pCi/g)
Bay de Chene	Spring 1993	Reference 2	Blue Crab	Edible	4	0.007 [0.004]	BDL [0.010]
					4	0.024 [0.003]	BDL [0.018]
					4	0.023 [0.003]	BDL [0.018]
					4	0.024 [0.003]	BDL [0.018]
Delacroix Island	Spring 1993	Discharge	Croaker	Edible	16	0.025 [0.004]	0.037 [0.007]
			Spot	Edible	8	0.005 [0.003]	0.027 [0.006]
			Blue Crab	Edible	12	0.013 [0.003]	0.032 [0.006]
Delacroix Island	Spring 1993	Reference 1	Croaker	Edible	16	0.005 [0.004]	0.112 [0.007]
			Spot	Edible	4	BDL [0.004]	0.076 [0.007]
			Blue Crab	Edible	19	0.025 [0.004]	0.090 [0.006]
Delacroix Island	Spring 1993	Reference 2	Croaker	Edible	29	0.018 [0.003]	0.039 [0.007]
			Spot	Edible	6	BDL [0.003]	0.017 [0.006]
			Blue Crab	Edible	13	0.023 [0.004]	0.013 [0.008]
Delacroix Island	Spring 1994	Discharge	Croaker	Edible	56	0.019 [0.003]	0.0159 [0.007]
			Spot	Edible	11	BDL [0.004]	0.036 [0.008]
			Blue Crab	Edible	23	0.007 [0.004]	0.046 [0.008]
Delacroix Island	Spring 1994	Reference 1	Croaker	Edible	16	0.028 [0.022]	0.266 [0.045]
			Spot	Edible	4	BDL [0.004]	0.025 [0.008]
			Blue Crab	Edible	22	0.007 [0.003]	BDL [0.008]
Delacroix Island	Spring 1994	Reference 2	Croaker	Edible	14	0.063 [0.018]	BDL [0.042]
			Spot	Edible	5	BDL [0.003]	0.107 [0.008]
			Blue Crab	Edible	20	0.012 [0.003]	0.041 [0.008]

* Whole = whole specimen analyzed; edible = edible tissue analyzed.
** LLD = Lower limit of detection
† BDL = Below detection limit

Table A-2. Codes used to identify organic compounds in sediment.

Analyte	Code	Analyte	Code
Naphthalene	CON	Benzo[e]pyrene	BEP
C_1-Naphthalene	CIN	Perylene	PER
C_2-Naphthalene	C2N	Indeno[1,2,3c,d]pyrene	IND
C_3-Naphthalene	C3N	Dibenzo[a,h]anthracene	DAH
C_4-Naphthalene	C4N	Benzo[g,h,i]perylene	BGP
Acenaphthylene	ACEY		
Acenaphthene	ACE		
Biphenyl	BIP		
Fluorene	COF		
C_1-Fluorene	CIF		
C_2-Fluorene	C2F		
C_3-Fluorene	C3F		
Dibenzothiophene	COD		
C_1-Dibenzothiophene	C1D		
C_2-Dibenzothiophene	C2D		
C_3-Dibenzothiophene	C3D		
Phenanthrene	COP		
Anthracene	COA		
C_1-Phenanthrene/Anthracene	C_1P/A		
C_2-Phenanthrene/Anthracene	C_2P/A		
C_3-Phenanthrene/Anthracene	C_3P/A		
C_4-Phenanthrene/Anthracene	C_4P/A		
Fluoranthene	Flant		
Pyrene	Pyr		
C1-Fluoranthene/Pyrene	C_1F/P		
C_2-Fluoranthene/Pyrene	C_2F/P		
C_3-Fluoranthenec/Pyrene	C_3F/P		
Chrysene	COC		
C_1-Chrysene	C1C		
C_2-Chrysene	C2C		
C_3-Chrysene	C3C		
C_4-Chrysene	C4C		
Benzo[a]anthracene	BAA		
Benzo[b]fluoranthene	BBF		
Benzo[k]fluoranthene	BKE		
Benzo[a]pyrene	BAP		

Delacroix Island Sediment PAH

Site	Depth (cm)	C0N ng/g	2-C1N ng/g	1-C1N ng/g	2,6-C2N ng/g	2,3,5-C3N ng/g	C1N ng/g	C2N ng/g	C3N ng/g	C4N ng/g
Discharge	0 to 5	160	290	170	260	160	350	960	1300	1200
Discharge	20 to 25	17	32	25	85	45	41	210	370	320
Discharge	35 to 40	4.1	0	0	0	1.2	0	0	14	13
Discharge	0 to 5	200	530	310	530	200	640	2200	3300	2600
Discharge	20 to 25	12	6.3	2.9	9.5	11	7.5	53	95	110
Discharge	35 to 40	9.8	5.9	2.6	6.9	7	7	57	100	96
Discharge	0 to 5	160	290	180	210	140	350	960	1300	1100
Discharge	20 to 25	16	8.5	8.8	0	5.3	14	34	87	92
Discharge	35 to 40	0	0	0	0	0	0	0	0	0
Reference 1	0 to 5	3.2	0	0	0	0	0	0	0	0
Reference 1	20 to 25	11	0	0	0	0	0	0	0	0
Reference 1	35 to 40	260	6.4	5.5	7.5	1.3	8.6	17	22	21
Reference 1	0 to 5	6.2	0	0	0	0	0	0	0	0
Reference 1	20 to 25	2.1	0	0	0	0	0	0	0	0
Reference 1	35 to 40	0	0	0	0	0	0	0	0	0
Reference 1	0 to 5	4.9	0	0	0	0	0	0	0	0
Reference 1	20 to 25	4.7	0	0	0	0	0	0	0	0
Reference 1	35 to 40	12	0	0	0	0	0	0	0	0
Reference 2	0 to 5	3.6	2	2	0	0	3.1	0	0	0
Reference 2	20 to 25	8	0	0	0	0	0	0	0	0
Reference 2	35 to 40	7.2	3	1.8	0.8	0	3.6	7.5	8	11
Reference 2	0 to 5	3.2	2.9	2	1.7	0.84	4	8.9	7.4	12
Reference 2	20 to 25	10	0	0	0	0	0	0	0	0
Reference 2	35 to 40	7.4	5	3.6	4.4	1.7	7.2	17	26	35
Reference 2	0 to 5	2.9	1.7	1.1	1.8	0	2.1	5.2	6.7	0
Reference 2	20 to 25	4.9	1.6	1.4	1.1	0	2.5	4.3	4.2	9.1
Reference 2	35 to 40	27	7.8	4.7	2.3	0	8.7	10	11	12
1000 South	0 to 5	1.9	1.8	0.95	1.9	0.55	2.5	4.3	4.8	6.7
500 South	0 to 5	8.6	2.8	1.7	0.99	0	3.8	6.6	5.7	6.6
300 South	0 to 5	2	1.8	1.5	1.4	0.64	2.7	6.3	7.7	13
100 South	0 to 5	3.7	2.6	1.9	1	0.7	3.7	6.3	6.5	8.6
100 NW	0 to 5	8.6	9.5	6.9	14	6.5	11	41	54	53
300 NW	0 to 5	3.5	2.6	2.1	1.5	0.71	3.8	7.1	7.2	11
500NW	0 to 5	2	1.6	1.2	1.3	0.35	2.3	5.1	6.7	7.5
1000 NW	0 to 5	2.2	1.8	1.4	1.6	0.59	2.8	6.6	7.3	9.3
100 NE	0 to 5	8.5	11	6.4	8.3	2.4	12	26	32	49
300 NE	0 to 5	4.8	4	2.1	3.8	2.7	4.2	12	31	63
500 NE	0 to 5	3.8	3.9	2.8	6.3	1.3	4.4	11	16	21

Table A-3. PAHs in sediment collected at Delacroix Island and Bay De Chene.

Delacroix Island Sediment PAH

Site	Depth (cm)	ACEY ng/g	ACE ng/g	BIP ng/g	COF ng/g	C1F ng/g	C2F ng/g	C3F ng/g	C0A ng/g	C0P ng/g
Discharge	0 to 5	0	22	38	53	150	420	520	22	110
Discharge	20 to 25	0	130	8.1	83	64	140	170	94	160
Discharge	35 to 40	0	41	0	11	12	17	24	12	12
Discharge	0 to 5	0	50	67	100	320	910	1100	46	220
Discharge	20 to 25	0	64	5.2	48	25	49	64	23	82
Discharge	35 to 40	4.4	190	5.2	58	54	64	74	73	110
Discharge	0 to 5	0	24	37	50	150	390	460	21	97
Discharge	20 to 25	11	280	7.6	76	89	110	130	150	130
Discharge	35 to 40	0	19	0	0	0	0	0	6.4	1.8
Reference 1	0 to 5	0	0	0	2.7	0	13	14	0	6.8
Reference 1	20 to 25	0	0	0	1.9	0	0	0	2.2	5.9
Reference 1	35 to 40	0	0	9.4	7.9	30	77	63	7.4	15
Reference 1	0 to 5	0	0	0	0	0	0	0	2	4.7
Reference 1	20 to 25	0	0	0	2.9	0	0	0	0	8
Reference 1	35 to 40	0	0	0	0	0	0	0	0	3.9
Reference 1	0 to 5	0	0	0	0	0	0	0	0	4.3
Reference 1	20 to 25	0	0	0	3	4.4	0	0	0	6.7
Reference 1	35 to 40	0	0	0	4.6	0	0	0	0	12
Reference 2	0 to 5	0	0	0	0	0	0	0	0	4.2
Reference 2	20 to 25	0	0	0	3.1	0	0	0	3.1	14
Reference 2	35 to 40	2.1	2	4.7	6.4	5	6.3	5.1	7.2	27
Reference 2	0 to 5	0	0	0	2	3.4	7.9	11	2.3	5.7
Reference 2	20 to 25	2.1	2	3.8	7.9	9	21	25	8	24
Reference 2	35 to 40	0	3.9	3.9	9.1	11	36	33	11	36
Reference 2	0 to 5	0	0	1.1	2.1	2.6	6.2	11	1.8	4.4
Reference 2	20 to 25	9.1	0	2.3	4.2	4.2	10	12	3.9	15
Reference 2	35 to 40	12	6	10	10	17	8.7	13	13	62
1000 South	0 to 5	6.7	0	1.4	1.4	1.6	3.1	8.2	1.5	3.9
500 South	0 to 5	2.5	2.5	4.2	6.6	4.4	5	11	6.3	26
300 South	0 to 5	0	2.6	1.2	3.4	4.3	10	14	5.6	11
100 South	0 to 5	0	6.9	1.5	8.5	4.1	5.5	7.5	19	50
100 NW	0 to 5	17	13	4.1	78	48	53	47	200	410
300 NW	0 to 5	0	0	1.3	1.8	2.4	5.8	7.4	1.4	5.4
500NW	0 to 5	0	0	0	1.6	2.3	6.3	8.3	1.8	4.7
1000 NW	0 to 5	0	0	0	2.7	3.9	7	8.3	6.3	14
100 NE	0 to 5	0	0	5	7.4	12	31	55	9.7	21
300 NE	0 to 5	0.42	2.9	2.3	4.6	13	37	73	4	12
500 NE	0 to 5	0.42	0.92	2.3	3.4	5	12	24	3.6	11

Delacroix Island Sediment PAH

Site	Depth (cm)	1C1P ng/g	C1P/A ng/g	C2P/A ng/g	C3P/A ng/g	C4P/A ng/g	FLANT ng/g	PYR ng/g	C1F/P ng/g	C2F/P ng/g
Discharge	0 to 5	88	400	670	470	230	110	81	120	110
Discharge	20 to 25	41	190	270	180	340	1000	650	580	170
Discharge	35 to 40	7.2	22	49	33	99	620	380	230	56
Discharge	0 to 5	170	810	1400	950	510	240	170	270	240
Discharge	20 to 25	18	69	96	68	68	270	170	110	38
Discharge	35 to 40	45	130	190	99	260	1400	880	580	170
Discharge	0 to 5	86	370	630	450	250	150	100	130	98
Discharge	20 to 25	72	260	340	190	690	3500	2200	1700	440
Discharge	35 to 40	2	7.7	6.3	6.6	6	47	23	9.7	4.4
Reference 1	0 to 5	4.4	15	25	23	13	23	18	13	8.3
Reference 1	20 to 25	2	6.3	9.6	18	20	20	17	12	8
Reference 1	35 to 40	6.7	30	50	42	74	32	17	26	0
Reference 1	0 to 5	3.6	12	14	17	9.9	19	14	11	9.4
Reference 1	20 to 25	3.7	11	13	17	15	26	17	13	8.3
Reference 1	35 to 40	0	0	0	0	0	5.2	5.1	0	0
Reference 1	0 to 5	2.9	9.8	15	21	17	19	14	9.7	7.7
Reference 1	20 to 25	3	7	5.6	11	9.8	23	16	12	6
Reference 1	35 to 40	5.4	13	5.8	0	0	24	18	13	0
Reference 2	0 to 5	1.5	6.9	12	16	12	13	9.7	8.3	6.8
Reference 2	20 to 25	5.1	12	16	16	14	45	33	24	13
Reference 2	35 to 40	4.5	16	13	7	6.1	62	46	32	8.5
Reference 2	0 to 5	2.4	9.4	15	13	9.6	25	20	12	7.2
Reference 2	20 to 25	4.3	21	20	12	14	69	57	33	11
Reference 2	35 to 40	8.9	36	53	35	42	160	120	67	25
Reference 2	0 to 5	2	9.2	14	12	7.9	19	15	9.2	6.2
Reference 2	20 to 25	2.9	10	14	8.5	6.2	42	34	21	6.4
Reference 2	35 to 40	7.1	25	15	9.4	7.3	94	74	47	12
1000 South	0 to 5	1.7	7	12	10	10	22	19	12	8.8
500 South	0 to 5	3.4	13	8.3	6.1	6.4	56	43	24	8.9
300 South	0 to 5	3.5	15	22	17	25	64	50	38	24
100 South	0 to 5	5.5	22	18	11	27	110	88	51	18
100 NW	0 to 5	50	250	150	70	190	900	570	460	130
300 NW	0 to 5	2.2	7.3	11	7.9	9.7	15	12	12	7
500NW	0 to 5	1.9	7.6	14	11	8.6	23	18	11	7.7
1000 NW	0 to 5	4.5	19	19	11	17	53	35	31	13
100 NE	0 to 5	8.7	32	65	62	58	110	99	67	36
300 NE	0 to 5	7.7	24	64	64	47	47	40	38	21
500 NE	0 to 5	5.6	18	33	25	19	43	36	24	12

Delacroix Island Sediment PAH

Site	Depth	C3F/P	C0D	C1D	C2D	C3D	BAA	C0C	C1C	C2C
	(cm)	ng/g	ng/g	ng/g	ng/g	ng/g	ng/g	ng/g	ng/g	ng/g
Discharge	0 to 5	98	15	76	170	180	21	34	38	52
Discharge	20 to 25	69	29	41	73	74	320	470	160	67
Discharge	35 to 40	19	6.2	9.3	16	7.9	130	130	52	16
Discharge	0 to 5	210	29	150	350	370	36	75	110	110
Discharge	20 to 25	26	15	16	25	31	33	44	23	13
Discharge	35 to 40	64	26	37	48	33	350	350	150	50
Discharge	0 to 5	91	14	76	160	180	33	41	45	51
Discharge	20 to 25	150	35	52	83	44	1000	1200	380	120
Discharge	35 to 40	0	0.94	2.1	0	0	2.4	4.2	0	0
Reference 1	0 to 5	5.6	1.5	6	13	13	3.4	6.9	7.2	5.8
Reference 1	20 to 25	6.3	1.6	0	0	0	2.7	5.3	3.4	1.5
Reference 1	35 to 40	0	3.1	27	50	47	2.5	4.2	0	0
Reference 1	0 to 5	3.8	1.3	3.4	7	5.7	2.7	5.5	6	3.9
Reference 1	20 to 25	0	1.6	0	0	0	3.1	4.9	0	0
Reference 1	35 to 40	0	0	0	0	0	0	3.6	0	0
Reference 1	0 to 5	7.2	0.82	3.6	8	9	2.9	5.4	6.1	4.1
Reference 1	20 to 25	0	1	0	0	0	2.9	4.7	4.5	2.8
Reference 1	35 to 40	0	0	0	0	0	0	5.7	0	0
Reference 2	0 to 5	0	0	0	0	0	2.8	6.8	5	3.3
Reference 2	20 to 25	8.9	2.6	0	0	0	0	10	0	0
Reference 2	35 to 40	3	1.9	0	0	0	7.3	15	3.5	2.7
Reference 2	0 to 5	5.6	1.3	3.5	7.9	9.6	4.8	7.6	6.1	4.1
Reference 2	20 to 25	0	3	3.5	7.2	8.7	8.9	14	9.8	5.7
Reference 2	35 to 40	15	4.7	12	27	25	40	82	28	15
Reference 2	0 to 5	2.6	1.2	3.3	7.6	7.3	3.8	5.3	5.5	4.6
Reference 2	20 to 25	1.8	1.6	2.1	3.4	3.3	6.5	11	8.5	4.3
Reference 2	35 to 40	6.3	2.9	2.5	0	0	12	21	12	4.4
1000 South	0 to 5	5	0.93	2.5	6.3	7.2	5.2	7.9	7.4	5.3
500 South	0 to 5	2.9	1.8	1.9	3.8	3.3	8.9	19	7.6	2.2
300 South	0 to 5	11	1.7	3.9	9.8	11	29	43	20	11
100 South	0 to 5	8.8	3.9	4.1	6.8	6.3	60	57	30	10
100 NW	0 to 5	33	27	21	33	28	350	340	140	49
300 NW	0 to 5	2.7	0.99	2.5	4.9	4.7	4	6.2	5.4	4.1
500NW	0 to 5	4.1	0.89	2.8	6.5	7	6	7.9	7.4	5
1000 NW	0 to 5	5.5	1.7	4	6.9	5.9	24	23	13	5.9
100 NE	0 to 5	28	4.9	13	30	37	31	61	30	19
300 NE	0 to 5	19	2.7	10	27	32	12	14	21	12
500 NE	0 to 5	9.9	2.3	6	14	15	12	15	14	7.9

Delacroix Island Sediment PAH

Site	Depth (cm)	C3C ng/g	C4C ng/g	BBF ng/g	BKF ng/g	BEP ng/g	BAP ng/g	PER ng/g	IND ng/g	DAH ng/g
Discharge	0 to 5	58	20	25	6	12	9	110	7.7	0
Discharge	20 to 25	55	25	390	140	160	210	190	120	35
Discharge	35 to 40	10	3.8	100	36	40	48	120	24	8.4
Discharge	0 to 5	110	95	48	17	24	23	190	15	4.1
Discharge	20 to 25	8.5	0	41	13	20	16	140	14	2.3
Discharge	35 to 40	37	7.8	320	92	120	170	180	91	27
Discharge	0 to 5	55	24	32	9	15	13	100	9	0
Discharge	20 to 25	48	81	800	270	320	470	220	230	67
Discharge	35 to 40	0	0	4.7	1.2	1.8	1.3	110	1.3	0
Reference 1	0 to 5	5.4	0	11	2.1	5.4	2.6	52	3.5	0
Reference 1	20 to 25	0	0	7.9	2.4	3.9	2.7	55	3.2	0
Reference 1	35 to 40	0	0	0	0	2.2	0	75	2.1	0
Reference 1	0 to 5	3.2	0	8.1	2.1	4.3	1.6	54	3.2	0
Reference 1	20 to 25	0	0	7.3	2	3.9	2.5	89	3.1	0
Reference 1	35 to 40	0	0	3.3	0.76	1.9	0	140	0	0
Reference 1	0 to 5	0	0	8.5	2.4	4.2	2.7	52	2.9	0
Reference 1	20 to 25	0	0	7.2	2.2	3.7	2.3	60	3.4	0
Reference 1	35 to 40	0	0	5.7	1.7	2.7	1.2	130	0	0
Reference 2	0 to 5	0	0	6.3	1.9	3.3	1.7	41	2.8	0
Reference 2	20 to 25	0	0	9.2	2	4.2	2	45	3.2	0
Reference 2	35 to 40	0	0	15	3.7	7.9	3.9	84	5.3	0
Reference 2	0 to 5	1.8	4.4	12	2.7	6	3.3	63	4	0
Reference 2	20 to 25	0	0	18	4.4	8.6	4.6	60	5.7	1.1
Reference 2	35 to 40	9.8	5.1	60	19	28	30	110	22	5.1
Reference 2	0 to 5	2.9	0	8.4	2.3	4.1	2.4	52	32	0.65
Reference 2	20 to 25	0	0	15	3.1	7.9	4.1	61	5.2	1.5
Reference 2	35 to 40	0	0	24	4.7	11	5.7	71	6.1	0
1000 South	0 to 5	4.9	5.1	14	3.5	7.1	4.1	80	4.6	1
500 South	0 to 5	1.4	2.9	17	5	9	5.9	40	5.3	1.3
300 South	0 to 5	9.8	8.4	52	16	25	25	85	16	4.3
100 South	0 to 5	7	6	77	25	39	55	70	32	8.3
100 NW	0 to 5	40	43	280	110	130	210	140	110	36
300 NW	0 to 5	3.8	3.1	11	2.1	5.3	3	200	3.5	0
500NW	0 to 5	4.4	4.1	14	4	6.7	4.4	78	4.5	1
1000 NW	0 to 5	4.8	2.8	27	9.1	13	15	58	8.1	2.9
100 NE	0 to 5	20	13	52	17	27	22	140	16	4.5
300 NE	0 to 5	11	7.2	23	6.2	11	8.6	190	8.3	1.9
500 NE	0 to 5	5.7	6	24	6.3	13	8.7	140	9.2	2

Delacroix Island Sediment PAH

Site	Depth (cm)	BGP ng/g	Total PAH ng/g
Discharge	0 to 5	10	9405.7
Discharge	20 to 25	100	8143.1
Discharge	35 to 40	22	2456.1
Discharge	0 to 5	16	20065.1
Discharge	20 to 25	16	2071.2
Discharge	35 to 40	81	6912.6
Discharge	0 to 5	11	9142
Discharge	20 to 25	190	16401.2
Discharge	35 to 40	2.3	272.14
Reference 1	0 to 5	5	331.8
Reference 1	20 to 25	5.6	235.4
Reference 1	35 to 40	2.4	1052.2
Reference 1	0 to 5	5.4	244
Reference 1	20 to 25	4.6	259
Reference 1	35 to 40	1.8	165.56
Reference 1	0 to 5	5.9	251.02
Reference 1	20 to 25	5.6	212.5
Reference 1	35 to 40	5.7	260.5
Reference 2	0 to 5	5.4	181.4
Reference 2	20 to 25	7	300.4
Reference 2	35 to 40	6.5	463.5
Reference 2	0 to 5	5.7	342.24
Reference 2	20 to 25	8	525.3
Reference 2	35 to 40	23	1358.8
Reference 2	0 to 5	3.8	294.95
Reference 2	20 to 25	9.2	384.3
Reference 2	35 to 40	10	719.6
1000 South	0 to 5	6.3	334.03
500 South	0 to 5	7.3	420.89
300 South	0 to 5	17	746.54
100 South	0 to 5	33	1027.7
100 NW	0 to 5	91	6055.6
300 NW	0 to 5	5.5	421.9
500NW	0 to 5	5.8	330.34
1000 NW	0 to 5	8.6	517.49
100 NE	0 to 5	21	1420.8
300 NE	0 to 5	12	1058.42
500 NE	0 to 5	12	671.74

Bay de Chene Sediment PAH

Site	Depth (cm)	C0N ng/g	2-C1N ng/g	1-C1N ng/g	2,6-C2N ng/g	2,3,5-C3N ng/g	C1N ng/g	C2N ng/g	C3N ng/g	C4N ng/g
Discharge	0 to 5	56	61	40	77	83	74	340	820	1000
Discharge	20 to 25	46	82	47	75	52	92	360	980	1200
Discharge	35 to 40	61	510	480	670	320	680	2600	3800	3400
Discharge	0 to 5	110	110	68	320	320	130	1400	3600	4200
Discharge	20 to 25	46	92	51	160	110	99	600	1700	2100
Discharge	35 to 40	56	890	810	950	540	1200	4200	5500	4600
Discharge	0 to 5	160	140	120	110	110	180	520	1100	1400
Discharge	20 to 25	57	120	74	160	100	140	740	2000	2400
Discharge	35 to 40	70	670	670	940	490	920	3800	5100	4300
Reference 1	0 to 5	3.5	0	0	0	0	0	0	0	0
Reference 1	20 to 25	6.8	2.6	1.7	0	0	3.1	8.2	6.3	0
Reference 1	35 to 40	2.5	1.5	0	0	0	1.6	5	0	0
Reference 1	0 to 5	6.6	2.6	2.2	0	0	4	7.2	4.4	5.7
Reference 1	20 to 25	6.4	3.1	2	0	0	3.8	7.5	0	0
Reference 1	35 to 40	3.1	0	0	0	0	0	0	0	0
Reference 1	0 to 5	2.8	0	0	0	0	0	0	0	0
Reference 1	20 to 25	6.7	2.1	0	0	0	3.5	8.6	0	0
Reference 1	35 to 40	4	2.6	1.3	0	0	3.2	0	0	0
Reference 2	0 to 5	5	4	0	2.2	0	5.1	9.1	8.1	10
Reference 2	20 to 25	4.7	1.9	0.95	1.4	1.2	2.9	4.8	5.3	5
Reference 2	35 to 40	3.7	2.1	1.5	0	0	2	6.1	6.7	7.2
Reference 2	0 to 5	3.4	2.8	1.5	1.2	0.53	4.4	9.1	9.9	12
Reference 2	20 to 25	3.6	1.4	1	0.52	0.54	1.4	4	4.2	3.2
Reference 2	35 to 40	3.8	1.4	0.97	0.63	0	1.7	4.5	3.2	3.6
Reference 2	0 to 5	5	3.4	1.9	1.04	0.95	3.5	8	7.7	9.6
Reference 2	20 to 25	5	2.7	1.3	1.4	1.1	2.3	6.2	9.3	11
Reference 2	35 to 40	3.7	1.5	0.93	5.9	0.59	1.6	3.3	2.9	1.9
300 ft NE	0 to 5	11	12	6.6	5.6	2.9	12	23	25	34
500 ft NE	0 to 5	5.2	4.9	2.1	2.6	1.3	4.4	9.9	12	19
1000 ft NE	0 to 5	2.7	2.1	0.96	0.59	0.45	1.8	3.9	4.3	4.6
1000 ft SE	0 to 5	2.9	2.5	1.4	1.4	0.76	2.5	5.3	6.2	7.9
500 ft SE	0 to 5	1.6	1.4	1.2	0.79	0.59	1.5	3.9	4.1	0.53
100 ft SW	0 to 5	14	16	6.7	6.8	3.9	14	27	43	58
300 ft SW	0 to 5	3.8	2.6	1.2	1.3	0.57	2.4	6.5	7.4	9.4
500 ft SW	0 to 5	2.2	1.7	1.2	0.74	0.58	1.7	4.3	4.5	5.3
1000 ft SW	0 to 5	4.3	2.9	1.6	2	0.79	3.5	7.9	9.3	13
100 ft NW	0 to 5	32	38	16	23	13	38	74	130	240
300 ft NW	0 to 5	21	20	9.1	16	9	20	56	100	120
500 ft NW	0 to 5	5.8	5	2.2	3.3	1.8	4.9	11	16	25
1000 ft NW	0 to 5	3.6	3.6	1.9	2.1	1.4	4.2	7.9	11	16
100 ft NE	0 to 5	38	32	17	30	43	35	130	400	520

Bay de Chene Sediment PAH

Site	Depth (cm)	ACEY ng/g	ACE ng/g	BIP ng/g	COF ng/g	C1F ng/g	C2F ng/g	C3F ng/g	C0A ng/g	C0P ng/g
Discharge	0 to 5	6.1	180	37	230	280	530	730	250	890
Discharge	20 to 25	6.4	69	20	130	320	650	870	150	300
Discharge	35 to 40	5.5	99	21	240	780	1600	2000	160	600
Discharge	0 to 5	47	210	47	390	960	2000	3100	1000	1800
Discharge	20 to 25	5.2	71	20	150	480	1000	1400	300	370
Discharge	35 to 40	0	140	28	350	1100	2000	2700	220	890
Discharge	0 to 5	40	250	59	340	380	680	900	470	1400
Discharge	20 to 25	8.1	110	25	210	580	1200	1700	210	490
Discharge	35 to 40	5.7	140	24	320	1000	1900	2500	180	680
Reference 1	0 to 5	0	0	0	1.3	1.3	4.3	5	1.3	3.7
Reference 1	20 to 25	0	0	0	0	0	0	0	0	7.7
Reference 1	35 to 40	0	0	0.98	1.3	2.1	0	0	1.3	3.6
Reference 1	0 to 5	5.7	0	2.6	3.3	1.8	7.4	9.8	2.7	11
Reference 1	20 to 25	0	2.1	2	2.8	0	0	0	1.5	7.4
Reference 1	35 to 40	0	0	1.4	1.1	0	0	0	1.2	3.8
Reference 1	0 to 5	0	0	1.2	1.8	3.7	14	19	2.4	6.7
Reference 1	20 to 25	0	0	2	2.5	0	0	0	1.4	8
Reference 1	35 to 40	0	0	1.5	1.6	0	0	0	1.8	5.9
Reference 2	0 to 5	0	0	2.6	5.6	6.7	17	21	5.6	15
Reference 2	20 to 25	0	1.4	2	4.3	5	6.3	4.4	3.8	14
Reference 2	35 to 40	0	1.9	1.9	6	5	5.5	0	4	13
Reference 2	0 to 5	0	0	2	3.5	5.5	14	13	4.3	12
Reference 2	20 to 25	0.68	0.92	2	3.5	3.4	4.9	4.8	2.3	8.4
Reference 2	35 to 40	0.56	1.7	1.8	5.3	3.4	2.9	2.8	2.3	11
Reference 2	0 to 5	0.68	0.83	2.1	4.5	6.3	16	18	4.1	12
Reference 2	20 to 25	0.59	1	2.3	4.5	7.4	13	14	3.7	11
Reference 2	35 to 40	0.48	1.4	1.6	4.6	3.1	3.2	0	2.3	10
300 ft NE	0 to 5	2.8	20	5.2	28	25	33	62	78	260
500 ft NE	0 to 5	0.97	3.4	3	8.2	9.1	20	31	9.6	20
1000 ft NE	0 to 5	0.43	0.65	1.2	1.8	2.3	5.2	8.3	1.7	6
1000 ft SE	0 to 5	0.46	0.63	1.6	2.8	4.2	10	15	2.7	8.8
500 ft SE	0 to 5	0.21	0.46	0.89	1.3	2.1	4.5	8	1.2	4.2
100 ft SW	0 to 5	0.99	6.6	6.6	13	38	88	130	13	32
300 ft SW	0 to 5	0.99	1.6	1.6	3	4.4	10	16	3	9
500 ft SW	0 to 5	0.44	0.99	0.99	1.9	2.4	5.8	9.2	2.1	7.4
1000 ft SW	0 to 5	0.86	2.1	2.1	3.7	6.4	19	25	3.8	12
100 ft NW	0 to 5	2.1	6.2	14	22	150	340	470	30	49
300 ft NW	0 to 5	11	16	10	33	45	88	120	26	150
500 ft NW	0 to 5	0.89	1.8	3	6.7	11	25	37	6.2	17
1000 ft NW	0 to 5	0.63	5.5	1.5	8.4	8.3	16	23	32	56
100 ft NE	0 to 5	5.7	48	18	67	120	260	340	86	250

Bay de Chene Sediment PAH

Site	Depth (cm)	1C1P ng/g	C1P/A ng/g	C2P/A ng/g	C3P/A ng/g	C4P/A ng/g	FLANT ng/g	PYR ng/g	C1F/P ng/g	C2F/P ng/g
Discharge	0 to 5	160	570	1000	810	760	2100	1500	930	480
Discharge	20 to 25	140	460	1100	980	790	1000	810	640	440
Discharge	35 to 40	410	1400	2600	2000	1300	780	650	670	650
Discharge	0 to 5	600	2500	5100	3800	8300	8100	6100	11000	5800
Discharge	20 to 25	240	770	1800	1700	1400	1300	940	1000	980
Discharge	35 to 40	550	2000	3500	2700	1600	1200	960	910	850
Discharge	0 to 5	250	880	1300	1000	1100	2700	1900	1300	690
Discharge	20 to 25	290	920	2100	1700	1400	1700	1300	1000	730
Discharge	35 to 40	510	1700	3200	2500	1500	800	660	680	790
Reference 1	0 to 5	1.9	6.1	9.9	12	8.7	9.3	9.5	6.7	5.7
Reference 1	20 to 25	1.3	4.2	3.1	0	0	7.5	8.7	7	3
Reference 1	35 to 40	0.96	4.3	4.7	0	0	5.9	6.2	8.4	3.8
Reference 1	0 to 5	3.3	9.8	14	12	10	17	19	14	7.5
Reference 1	20 to 25	1.8	5.5	7.1	0	0	7.9	8.9	11	3.3
Reference 1	35 to 40	1.3	3.5	0	0	0	6.5	7.2	7.9	5.9
Reference 1	0 to 5	2.6	9.8	19	18	15	16	21	16	11
Reference 1	20 to 25	1.7	5.9	3.7	2.2	1.7	7.1	7.9	8.3	0
Reference 1	35 to 40	1.4	5.1	5.2	0	0	7	7.2	7.5	5.4
Reference 2	0 to 5	4.6	18	29	24	18	41	35	19	12
Reference 2	20 to 25	4.1	11	15	7	7.8	29	21	12	8.1
Reference 2	35 to 40	4.5	11	9.1	5.3	9.8	23	16	9.1	4.4
Reference 2	0 to 5	3.7	15	26	23	21	36	31	18	13
Reference 2	20 to 25	3	6.3	6.7	4.5	5.8	17	12	6.9	4.1
Reference 2	35 to 40	4	7.4	5.1	2.7	5	14	9.8	7.1	3.5
Reference 2	0 to 5	4.4	16	26	21	16	35	31	17	12
Reference 2	20 to 25	5.3	17	21	15	11	32	25	14	7.7
Reference 2	35 to 40	3.6	6.6	4.5	3.2	4.9	12	9.2	6.4	3.8
300 ft NE	0 to 5	26	100	92	55	160	650	630	360	250
500 ft NE	0 to 5	8.9	29	52	51	38	140	100	48	29
1000 ft NE	0 to 5	2	6.6	11	11	8.1	13	16	9.7	8.4
1000 ft SE	0 to 5	3.5	13	23	24	15	27	26	16	13
500 ft SE	0 to 5	1.8	6	11	12	11	12	14	12	21
100 ft SW	0 to 5	18	52	130	160	110	130	110	86	87
300 ft SW	0 to 5	3.8	13	25	28	18	30	34	21	16
500 ft SW	0 to 5	1.8	7.1	11	12	9.8	18	22	13	9.3
1000 ft SW	0 to 5	4.7	18	36	39	21	42	37	21	18
100 ft NW	0 to 5	33	99	340	470	320	210	190	170	150
300 ft NW	0 to 5	34	120	180	170	140	440	330	190	83
500 ft NW	0 to 5	8.4	28	55	56	39	62	53	34	26
1000 ft NW	0 to 5	15	52	55	40	72	460	360	150	57
100 ft NE	0 to 5	81	280	500	450	350	910	730	500	440

Bay de Chene Sediment PAH

Site	Depth (cm)	C3F/P ng/g	C0D ng/g	C1D ng/g	C2D ng/g	C3D ng/g	BAA ng/g	C0C ng/g	C1C ng/g	C2C ng/g
Discharge	0 to 5	380	99	180	370	380	960	1000	430	320
Discharge	20 to 25	400	66	200	430	460	470	600	300	290
Discharge	35 to 40	660	69	500	950	960	330	470	290	370
Discharge	0 to 5	3000	210	690	1500	1400	12000	11000	6400	2800
Discharge	20 to 25	970	71	310	660	790	780	790	700	800
Discharge	35 to 40	750	96	670	1300	1200	490	600	430	560
Discharge	0 to 5	510	150	250	460	470	1400	1300	610	450
Discharge	20 to 25	500	88	390	800	800	760	820	530	480
Discharge	35 to 40	790	87	630	1200	1200	250	300	310	460
Reference 1	0 to 5	6.8	0.91	1.8	5.3	6.8	2.5	3.7	3.5	2.7
Reference 1	20 to 25	0	0	0	0	0	1.7	3.4	0	0
Reference 1	35 to 40	4.4	0	0	0	0	1.9	3.4	5.9	3.4
Reference 1	0 to 5	4.9	1.1	4	7.2	9.7	4	6	5.1	5.3
Reference 1	20 to 25	0	0	0	0	0	1.8	3.8	0	0
Reference 1	35 to 40	0	0	0	0	0	1.7	3.3	0	0
Reference 1	0 to 5	11	1.5	3.7	10	15	3.9	5.8	6	7.1
Reference 1	20 to 25	0	0	0	0	0	1.7	3.1	0	0
Reference 1	35 to 40	0	0	0	0	0	1.3	2.9	0	0
Reference 2	0 to 5	7	2.6	8.5	19	17	5.6	11	7.8	8.1
Reference 2	20 to 25	3.7	2.3	4.3	5.1	5.2	3.3	6.1	4	3
Reference 2	35 to 40	3.5	1.7	2.4	1.5	0	3.8	6	4.4	2.6
Reference 2	0 to 5	8.4	2.1	5.5	14	15	8.1	12	9.1	7.3
Reference 2	20 to 25	2.1	1.2	1.8	2.5	2.4	1.9	4	3.1	2.7
Reference 2	35 to 40	2	1	0	0	0	1.6	3.4	2.8	2.2
Reference 2	0 to 5	9.2	2.3	5.6	12	13	5	9	9	8.3
Reference 2	20 to 25	6.1	3	5.3	8.9	7.9	4.3	7.5	5.8	5.6
Reference 2	35 to 40	1.8	0.87	0	1.1	0	1.7	2.9	1.9	1.5
300 ft NE	0 to 5	170	16	16	30	45	350	310	200	130
500 ft NE	0 to 5	24	4.5	10	24	28	23	26	23	20
1000 ft NE	0 to 5	6.5	1.1	2.5	5.3	6.4	3.2	4.9	5.6	5
1000 ft SE	0 to 5	11	1.6	4.4	10	13	5.4	8.6	7.9	9.1
500 ft SE	0 to 5	27	1	2.1	5.1	6.3	3.7	5.2	9	16
100 ft SW	0 to 5	68	9.1	26	68	78	40	55	42	48
300 ft SW	0 to 5	14	1.8	4.8	12	14	8.6	10	9.1	11
500 ft SW	0 to 5	8.2	1	2.1	5.3	6.2	6.6	7.3	6.1	6.1
1000 ft SW	0 to 5	16	2.3	6.2	17	19	1.4	12	11	13
100 ft NW	0 to 5	140	24	83	210	230	58	110	79	110
300 ft NW	0 to 5	72	19	34	72	76	100	190	84	57
500 ft NW	0 to 5	25	4.1	10	24	28	13	23	17	20
1000 ft NW	0 to 5	33	7.2	9.8	18	19	190	150	66	31
100 ft NE	0 to 5	370	39	87	180	190	340	470	290	260

Bay de Chene Sediment PAH

Site	Depth (cm)	C3C ng/g	C4C ng/g	BBF ng/g	BKF ng/g	BEP ng/g	BAP ng/g	PER ng/g	IND ng/g	DAH ng/g
Discharge	0 to 5	220	130	1400	420	640	850	440	730	150
Discharge	20 to 25	230	140	670	220	290	390	370	320	78
Discharge	35 to 40	360	190	400	120	170	210	480	170	43
Discharge	0 to 5	2100	640	14000	5200	6000	9000	2900	6100	1700
Discharge	20 to 25	600	330	880	290	390	530	460	360	95
Discharge	35 to 40	380	200	670	240	300	360	600	320	83
Discharge	0 to 5	280	160	1700	580	750	1200	580	900	210
Discharge	20 to 25	430	180	1100	320	490	650	560	550	130
Discharge	35 to 40	410	190	310	100	140	150	520	120	32
Reference 1	0 to 5	4.8	4.6	5.4	1.6	2.8	1.6	36	2.4	0
Reference 1	20 to 25	0	0	6.7	0	3.1	0	480	0	0
Reference 1	35 to 40	0	0	6.6	1	3.1	1.1	510	2.7	0
Reference 1	0 to 5	0	0	7.9	2.5	4.2	2.7	140	3.5	1
Reference 1	20 to 25	0	0	5.2	1.2	2.5	1.1	580	1.9	0
Reference 1	35 to 40	0	0	4.8	1.3	2.6	1.2	430	1.7	0
Reference 1	0 to 5	7.2	4.3	7.8	2.6	4.5	2.7	56	3.2	0.94
Reference 1	20 to 25	0	0	4.5	0	2.2	1.1	390	1.7	0
Reference 1	35 to 40	0	0	4.3	1.6	2.1	0.99	510	1.6	0
Reference 2	0 to 5	2.1	0	12	4.7	6.3	4.7	86	3.5	0.94
Reference 2	20 to 25	0	0	6.2	2.2	3.6	2.4	130	2.4	0
Reference 2	35 to 40	0	0	7.6	2.3	3.8	2.6	160	3.3	0
Reference 2	0 to 5	4.5	0	16	4.9	7.4	5.1	95	6.6	1.3
Reference 2	20 to 25	1.6	1	4.4	0.95	2.3	1.2	83	1.3	0.25
Reference 2	35 to 40	0	0	4	0.88	2.1	0.82	150	0.89	0.16
Reference 2	0 to 5	6.7	3.5	10	2.7	5.6	3.2	60	2.7	0.58
Reference 2	20 to 25	3.9	2.2	9	2.4	4.7	2.7	88	3	0.46
Reference 2	35 to 40	0	0	3.5	0.84	1.7	0.69	110	0.77	0
300 ft NE	0 to 5	110	56	470	170	260	340	170	220	54
500 ft NE	0 to 5	15	7.8	27	8.2	15	11	77	7.2	1.9
1000 ft NE	0 to 5	5.2	3.4	7.9	1.9	4.3	2	52	2.5	0.55
1000 ft SE	0 to 5	8.1	4.8	10	3	6	4.2	41	3.1	0.68
500 ft SE	0 to 5	17	7.3	8.5	2.2	7.4	5.2	44	3.8	1
100 ft SW	0 to 5	41	25	51	17	27	24	100	15	3.7
300 ft SW	0 to 5	6.7	5.1	13	4.2	7.4	5.1	44	3.7	1
500 ft SW	0 to 5	4.5	2.9	10	3	6	5.6	27	4.6	0.89
1000 ft SW	0 to 5	9	5.1	13	3.5	7.4	4.6	57	4.2	1.1
100 ft NW	0 to 5	110	59	93	30	50	44	190	33	8.2
300 ft NW	0 to 5	59	22	190	67	98	89	130	65	16
500 ft NW	0 to 5	16	7.1	22	5.8	12	9.1	76	8.1	1.9
1000 ft NW	0 to 5	19	10	160	58	75	110	75	53	12
100 ft NE	0 to 5	220	100	540	170	350	360	220	260	70

Bay de Chene Sediment PAH

Site	Depth (cm)	BGP ng/g	Total PAH ng/g
Discharge	0 to 5	630	23723.1
Discharge	20 to 25	270	18003.4
Discharge	35 to 40	140	35368.5
Discharge	0 to 5	4400	162152
Discharge	20 to 25	290	28980.2
Discharge	35 to 40	270	49963
Discharge	0 to 5	740	32179
Discharge	20 to 25	440	31482.1
Discharge	35 to 40	110	43358.7
Reference 1	0 to 5	4.7	188.11
Reference 1	20 to 25	4.6	570.7
Reference 1	35 to 40	4.3	601.94
Reference 1	0 to 5	5.2	397.9
Reference 1	20 to 25	3.3	684.9
Reference 1	35 to 40	3.1	492.6
Reference 1	0 to 5	4.3	337.54
Reference 1	20 to 25	2.7	480.3
Reference 1	35 to 40	2.4	587.89
Reference 2	0 to 5	5.5	529.94
Reference 2	20 to 25	3.2	371.35
Reference 2	35 to 40	4.1	368.4
Reference 2	0 to 5	7.7	515.83
Reference 2	20 to 25	1.9	236.66
Reference 2	35 to 40	1.5	283.51
Reference 2	0 to 5	4.1	456.48
Reference 2	20 to 25	7.1	423.65
Reference 2	35 to 40	1.9	234.37
300 ft NE	0 to 5	250	6336.1
500 ft NE	0 to 5	10	1025.17
1000 ft NE	0 to 5	3.7	258.73
1000 ft SE	0 to 5	5.2	394.63
500 ft SE	0 to 5	8	319.07
100 ft SW	0 to 5	19	2157.39
300 ft SW	0 to 5	5.5	454.56
500 ft SW	0 to 5	6	276.83
1000 ft SW	0 to 5	5.5	565.25
100 ft NW	0 to 5	39	5369.5
300 ft NW	0 to 5	78	4075.1
500 ft NW	0 to 5	10	877.09
1000 ft NW	0 to 5	56	2616.03
100 ft NE	0 to 5	380	11576.7

Table A-4. Metals in sediment collected at Delacroix Island and Bay De Chene

	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O	P	Q
1	Site/Sample		Al	As	Ba	Ca	Cd	Cr	Cu	Fe	Hg	Mn	Mo	Ni	Pb	V	Zn
2	Bay de Chene		(%)	(ppm)	(ppm)	(%)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)
3	Discharge A		5.81	11.0	1370	0.83	0.43	55.30	21.8	2.98	0.172	534	1.2	24.6	35.6	91.0	112.0
4	B		5.24	10.6	466	0.97	0.34	50.60	21.3	2.82	0.423	474	1.2	22.3	19.5	91.7	102.0
5	B		5.05	9.9	1790	0.78	0.29	49.80	19.5	2.73	-	454	1.1	21.9	22.8	75.9	101.0
6	B		5.00	9.7	162	0.82	0.36	42.00	19.0	2.66	-	457	1.1	21.6	12.1	76.8	99.3
7		average B	5.10	10.07	806.00	0.86	0.33	47.47	19.93	2.74	0.42	461.67	1.13	21.93	18.13	81.47	100.77
8	C		5.77	11.9	673	0.90	0.46	56.80	27.5	3.13	0.187	561	1.3	26.0	29.8	93.7	152.0
9	Discharge	Mean (3 #s)	5.56	10.99	949.67	0.86	0.41	53.19	23.08	2.95	0.26	518.89	1.21	24.18	27.84	88.72	121.59
10		SD	0.40	0.92	370.04	0.04	0.07	5.01	3.94	0.20	0.14	51.36	0.08	2.07	8.90	6.43	26.93
11	Reference 1A		4.09	8.6	1090	15.30	0.25	40.30	13.6	2.04	0.047	354	1.8	16.0	16.7	66.8	56.4
12	R1B		5.97	9.1	934	5.39	0.33	56.40	19.1	2.84	0.041	313	2.3	23.6	19.9	100.0	84.0
13	R1C		5.58	8.3	1100	1.95	0.35	52.30	18.3	2.78	0.052	370	2.0	22.2	23.7	85.1	74.5
14	R1	Mean	5.21	8.67	1041.33	7.55	0.31	49.67	17.00	2.55	0.05	345.67	2.03	20.60	20.10	83.97	71.63
15		SD	0.99	0.40	93.09	6.93	0.05	8.37	2.97	0.45	0.01	29.40	0.25	4.04	3.50	16.63	14.02
16	Reference 2A		4.95	5.7	753	1.13	0.23	42.70	15.3	2.29	0.049	454	1.1	19.6	18.9	75.4	68.3
17	R2B		5.47	7.7	850	1.13	0.32	49.90	16.1	2.51	0.046	421	1.6	20.7	21.5	73.1	76.5
18	R2C		6.07	9.0	621	1.03	0.36	53.10	19.9	2.91	0.047	479	1.7	24.3	22.4	95.6	87.0
19	R2	Mean	5.50	7.47	741.33	1.10	0.30	48.57	17.10	2.57	0.05	451.33	1.47	21.53	20.93	81.37	77.27
20		SD	0.56	1.66	114.94	0.06	0.07	5.33	2.46	0.31	0.00	29.09	0.32	2.46	1.82	12.38	9.37
21	100NW		6.35	10.3	1370	0.70	0.40	58.70	23.9	3.25	0.072	427	1.2	27.7	27.2	99.9	109.0
22	100NW		6.38	11.3	990	0.75	0.41	62.70	24.9	3.36	-	440	1.2	28.7	27.3	115.0	115.0
23	100NW		6.44	9.7	570	0.67	0.41	59.50	24.2	3.30	-	433	1.2	28.1	25.6	109.0	109.0
24	100NW	Mean	6.39	10.43	976.67	0.71	0.41	60.30	24.33	3.30	0.07	433.33	1.20	28.17	26.70	107.97	111.00
25		SD	0.05	0.81	400.17	0.04	0.01	2.12	0.51	0.06		6.51	0.00	0.50	0.95	7.60	3.46
26	300NW		5.22	13.9	691	0.66	0.46	48.50	23.5	2.98	0.083	505	1.3	25.7	24.9	86.5	93.0
27	500NW		6.21	8.7	1330	0.99	0.34	54.10	18.6	2.77	0.052	357	1.4	23.9	23.1	86.4	88.4
28	1000NW		5.64	7.7	771	1.22	0.27	48.00	15.3	2.24	0.042	312	1.4	18.6	26.3	77.9	70.8
29	100SW		6.45	7.3	1320	0.76	0.39	58.90	20.9	2.96	1.400	380	1.1	25.3	24.6	101.0	102.0
30	300SW		5.82	8.0	1310	1.80	0.29	53.70	18.5	2.79	0.098	386	2.0	22.8	22.2	84.0	82.3
31	500SW		5.38	6.8	1350	1.24	0.28	45.30	13.2	2.16	0.088	291	1.6	19.9	19.6	68.6	93.7
32	1000SW		6.06	5.9	1100	1.08	0.33	53.30	17.2	2.74	0.040	344	1.7	22.9	22.1	93.2	84.3
33	100NE		5.89	11.4	1090	0.87	0.55	53.40	26.6	3.10	0.065	524	1.4	26.0	35.3	89.3	158.0
34	100NE		5.84	5.3	1460	1.05	0.42	63.80	28.8	3.08	0.096	519	1.2	31.7	28.3	89.5	121.0
35	100NE		5.48	5.8	1360	1.05	0.52	54.60	30.1	3.13	0.133	518	1.6	28.8	32.4	86.8	159.0
36	100NE	Mean	5.74	7.50	1303.33	0.99	0.50	57.27	28.50	3.10	0.10	520.33	1.40	28.83	32.00	88.53	146.00
37		SD	0.22368	3.38674	191.3984	0.10392	0.06807	5.68976	1.76918	0.02517	0.03404	3.21455	0.2	2.85015	3.5171	1.50444	21.6564
38	300NE		5.74	7.3	1560	1.68	0.46	55.50	21.5	2.96	0.500	460	2.2	22.2	33.3	83.2	105.0
39	300NE		5.56	4.6	1670	2.59	0.32	59.20	21.2	2.57	0.072	372	4.0	24.7	25.4	89.3	77.0
40	300NE		5.51	6.2	65.1	1.53	0.32	54.90	20.8	3.07	0.064	376	3.7	28.4	19.4	91.7	81.9
41	300NE	Mean	5.60	6.03	1098.37	1.93	0.37	56.53	21.17	2.87	0.21	402.67	3.30	25.10	26.03	88.07	87.97
42		SD	0.12	1.36	896.52	0.57	0.08	2.33	0.35	0.26	0.25	49.69	0.96	3.12	6.97	4.38	14.95
43																	
44																	

	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O	P	Q
45	Site/Sample		Al	As	Ba	Ca	Cd	Cr	Cu	Fe	Hg	Mn	Mo	Ni	Pb	V	Zn
46	Bay de Chene		(%)	(ppm)	(ppm)	(%)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)
47	500NE		5.42	6.9	947	0.54	0.26	49.70	17.4	2.55	0.034	285	1.1	21.1	20.1	87.3	84.4
48	500NE		6.73	5.6	1260	0.86	0.35	66.60	23.5	3.28	0.061	448	1.2	31.8	23.2	102.0	98.6
49	500NE		6.83	6.6	1120	0.63	0.34	64.20	23.5	3.37	0.058	591	1.0	32.1	24.3	103.0	103.0
50	500NE	Mean	6.33	6.37	1109.00	0.68	0.32	60.17	21.47	3.07	0.05	441.33	1.10	28.33	22.53	97.43	95.33
51		SD	0.79	0.68	156.79	0.17	0.05	9.14	3.52	0.45	0.01	153.11	0.10	6.27	2.18	8.79	9.72
52	1000NE		6.55	8.0	979	1.17	0.34	59.00	20.6	3.09	0.045	329	1.7	26.1	22.4	105.0	90.3
53	1000NE	b	5.94	6.4	1140	1.47	0.25	56.10	20.0	2.87	0.044	321	1.4	31.0	18.9	85.6	73.2
54	1000NE	b	5.94	4.7	325	2.14	0.21	55.60	18.2	2.69	0.042	317	1.3	27.1	16.7	82.2	68.7
55	1000NE	b	5.61	4.7	1060	2.36	0.23	54.20	17.9	2.59	0.042	309	1.2	25.6	18.5	82.1	67.8
56		Average b	5.83	5.27	841.67	1.99	0.23	55.30	18.70	2.72	0.04	315.67	1.30	27.90	18.03	83.30	69.90
57	1000NE		6.77	6.4	1000	1.15	0.23	66.70	22.3	3.12	0.047	283	1.3	30.2	20.1	104.0	86.7
58	1000NE	Mean (3 #s)	6.38	6.56	940.22	1.44	0.27	60.33	20.53	2.98	0.04	309.22	1.43	28.07	20.18	97.43	82.30
59		SD	0.49	1.37	86.00	0.48	0.06	5.82	1.80	0.22	0.00	23.67	0.23	2.06	2.18	12.25	10.89
60	500SE		3.97	6.1	832	12.30	0.20	32.20	10.8	1.73	0.029	387	0.9	14.0	14.0	53.4	51.8
61	1000SE		5.55	7.8	1120	1.50	0.39	49.00	15.5	2.43	0.400	440	1.1	20.3	20.5	77.4	77.5
62	Site/Sample		Al	As	Ba	Ca	Cd	Cr	Cu	Fe	Hg	Mn	Mo	Ni	Pb	V	Zn
63	DELACROIX I.		(%)	(ppm)	(ppm)	(%)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)
64	Discharge A		6.04	11.2	350	0.88	0.25	56.10	23.4	3.49	0.063	1160	1.4	34.4	24.6	101.0	101.0
65	B		5.80	10.7	1380	0.83	0.24	54.10	22.3	3.26	0.067	948	1.4	32.4	27.4	94.2	98.9
66	C		5.86	10.2	358	0.82	0.26	56.00	22.7	3.30	0.075	1196	1.3	1.3	21.2	101.0	97.2
67	Discharge	Mean	5.90	10.70	696.00	0.84	0.25	55.40	22.80	3.35	0.07	1101.33	1.37	22.70	24.40	98.73	99.03
68		SD	0.12	0.50	592.37	0.03	0.01	1.13	0.56	0.12	0.01	134.00	0.06	18.56	3.10	3.93	1.90
69	Reference 1A		5.62	4.7	994	1.09	0.20	54.00	17.3	2.66	0.045	535	1.3	25.8	23.8	85.2	83.4
70	R1A		-	-	-	-	-	-	-	-	0.048	-	-	-	-	-	-
71	R1A		-	-	-	-	-	-	-	-	0.047	-	-	-	-	-	-
72		R1A mean									0.047						
73	R1B		5.50	5.0	1071	1.09	0.17	47.30	14.6	2.33	0.041	606	0.9	23.9	22.1	71.6	73.6
74	R1C		5.56	4.5	1010	0.90	0.19	50.60	17.1	2.57	0.046	653	1.0	25.6	23.9	76.7	75.1
75	R1	Mean	5.56	4.73	1025.00	1.03	0.19	50.63	16.33	2.52	0.045	598.00	1.07	25.10	23.27	77.83	77.37
76		SD	0.06	0.25	40.63	0.11	0.02	3.35	1.50	0.17	0.003	59.41	0.21	1.04	1.01	6.87	5.28
77	Site/Sample		Al	As	Ba	Ca	Cd	Cr	Cu	Fe	Hg	Mn	Mo	Ni	Pb	V	Zn
78	DELACROIX I.		(%)	(ppm)	(ppm)	(%)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)
79	R2A		4.70	3.2	1340	1.10	0.04	36.20	9.9	1.84	0.030	515	0.7	19.0	18.2	58.7	55.8
80	R2B		4.85	4.2	1430	1.05	0.11	45.70	11.6	1.95	0.033	443	0.9	20.8	19.3	59.4	57.4
81	R2B		4.93	4.6	1410	1.02	0.11	41.00	12.4	1.94	-	441	1.0	21.1	18.6	57.6	58.4
82	R2B		4.92	4.2	1290	1.01	0.19	42.30	11.7	1.92	-	447	1.0	21.1	19.3	55.4	59.2
83	R2B	average	4.90	4.33	1376.67	1.03	0.14	43.00	11.90	1.94	0.033	443.67	0.97	21.00	19.07	57.47	58.33
84	R2C		4.79	3.2	1330	1.03	0.08	38.40	9.7	1.77	0.030	416	0.9	20.0	18.0	51.6	55.0
85	R2	Mean (3#s)	4.80	3.58	1348.89	1.05	0.09	39.20	10.50	1.85	0.031	458.22	0.86	20.00	18.42	55.92	56.38
86		SD	0.10	0.65	24.57	0.04	0.05	3.47	1.22	0.08	0.002	51.08	0.14	1.00	0.57	3.79	1.74
87	100GS		4.79	3.2	283	3.96	0.17	41.20	11.0	1.80	0.028	430	0.8	17.7	12.7	56.6	57.9
88																	

	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O	P	Q
89	Site/Sample		Al	As	Ba	Ca	Cd	Cr	Cu	Fe	Hg	Mn	Mo	Ni	Pb	V	Zn
90	DELACROIX I.		(%)	(ppm)	(ppm)	(%)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)	(ppm)
91	500S		4.84	4.0	1220	1.04	0.07	36.00	9.8	1.70	0.032	340	0.7	17.6	16.1	55.9	51.8
92	500S		-	-	-	-	-	-	-	-	0.030	-	-	-	-	-	-
93	500S		-	-	-	-	-	-	-	-	0.031	-	-	-	-	-	-
94	300S		4.73	3.5	933	1.13	0.07	36.50	9.2	1.54	0.029	274	0.7	15.3	13.8	42.4	52.4
95	100S		4.46	3.7	1170	1.18	0.07	35.60	8.7	1.46	0.024	378	0.6	15.6	14.8	35.6	47.5
96	1000NW		3.90	3.8	2850	6.79	0.05	29.50	8.8	1.43	0.230	406	1.0	14.6	13.7	42.3	46.5
97	1000NW		2.79	5.1	73.7	3.84	0.17	28.90	12.7	2.59	0.038	396	2.2	16.7	4.1	44.3	45.2
98	1000NW		3.23	5.4	299	3.20	0.10	29.90	10.7	1.09	0.027	295	1.6	14.1	6.1	46.3	30.0
99	1000NW	Mean	3.31	4.77	1074.23	4.61	0.11	29.43	10.73	1.70	0.10	365.67	1.60	15.13	7.97	44.30	40.57
100		SD	0.56	0.85	1541.98	1.91	0.06	0.50	1.95	0.79	0.11	61.40	0.60	1.38	5.06	2.00	9.17
101	500NW		5.00	3.1	1330	1.05	0.07	40.30	10.5	1.78	0.028	325	0.8	19.5	16.9	50.5	57.4
102	500NW		4.97	4.1	482	1.03	0.11	35.90	11.2	1.78	-	316	0.8	18.8	13.4	41.7	55.2
103	500NW		5.02	3.8	349	1.06	0.15	40.90	10.9	1.80	-	329	0.7	18.6	11.7	45.5	57.6
104	500NW		5.55	5.2	686	1.03	0.30	47.90	13.9	2.29	0.043	489	0.9	26.3	16.1	73.5	65.0
105	500NW		5.14	3.9	1760	1.10	0.24	44.10	11.5	1.84	0.032	382	0.8	22.6	16.0	60.5	54.9
106	500NW	Mean	5.14	4.02	921.40	1.05	0.17	41.82	11.60	1.90	0.03	368.20	0.80	21.16	14.82	54.34	58.02
107		SD	0.24	0.76	601.32	0.03	0.09	4.48	1.34	0.22	0.01	72.30	0.07	3.29	2.19	12.82	4.09
108	300NW		5.66	5.6	834	4.39	0.25	51.90	17.3	2.19	0.030	370	0.9	22.6	17.1	81.5	72.0
109	300NW	b	5.55	5.3	1270	1.45	0.33	49.40	15.7	2.13	0.032	378	0.9	26.5	18.1	74.1	61.0
110	300NW	b	5.63	5.2	1130	1.46	0.27	50.60	15.5	2.15	0.031	375	0.8	24.4	15.7	72.9	65.9
111	300NW	b	5.72	7.2	1160	1.37	0.31	52.10	15.1	2.20	0.030	379	0.9	27.0	16.3	75.7	64.4
112		Average b	5.63	5.90	1186.67	1.43	0.30	50.70	15.43	2.16	0.03	377.33	0.87	25.97	16.70	74.23	63.77
113	300NW		5.19	7.5	1220	1.34	0.30	47.00	12.4	2.10	0.035	620	0.8	25.6	16.1	65.9	57.5
114	300NW	Mean	5.49	6.33	1080.22	2.39	0.28	49.87	15.04	2.15	0.03	455.78	0.86	24.72	16.63	73.88	64.42
115		SD	0.26	1.02	213.89	1.74	0.03	2.55	2.47	0.05	0.00	142.27	0.05	1.85	0.50	7.81	7.27
116	100NW		5.24	6.4	1180	0.97	0.20	43.70	15.6	2.34	0.048	956	0.8	23.0	21.0	61.0	69.3
117	100NW		5.43	7.7	1420	1.04	0.29	48.50	14.0	2.21	0.040	704	0.7	27.0	17.4	68.8	62.5
118	100NW		5.33	4.6	1290	1.24	0.92	41.80	10.9	1.90	0.045	588	0.5	21.2	15.1	54.0	53.5
119	100NW	Mean	5.33	6.23	1296.67	1.08	0.47	44.67	13.50	2.15	0.04	749.33	0.67	23.73	17.83	61.27	61.77
120		SD	0.10	1.56	120.14	0.14	0.39	3.45	2.39	0.23	0.00	188.14	0.15	2.97	2.97	7.40	7.93
121	500NE		5.47	4.7	1050	0.96	0.10	47.00	14.7	2.32	0.037	549	0.8	22.6	20.8	75.1	73.8
122	300NE		5.52	3.9	615	0.93	0.17	44.70	16.3	2.30	0.038	470	1.0	21.6	13.3	68.5	70.8
123	100NE		5.70	4.6	278	0.91	0.23	50.30	17.2	2.44	0.043	380	1.1	21.7	14.2	75.2	77.2

APPENDIX B

CHARACTERIZATION OF CONTINUING OPEN BAY DISCHARGES

Table B-1. Open Bay Discharge Permits Identified by LDEQ (ordered alphabetically by operator, permits may be for more than one discharge, permits in bold removed from further consideration).

Permit Number	Company	Field	Comment[1]
2901	Aviva	Breton Sound 31	I, C
2134	Callon Offshore Pet.	Chandeleur Sound 25	I, C
1934	Callon Offshore Pet.	Main Pass 35	I, C
2860	Callon Offshore Pet.	Black Bay	I, C
2859	Callon Offshore Pet.	East Black Bay	I, C
2142	Callon Offshore Pet.	North Black Bay	I, C
2672	Callon Offshore Pet.	Southeast Black Bay	I, C
1901	Callon Offshore Pet.	West Black Bay	I, C
3023	Clovelly (LL&E)	Chandeleur Sound 51	I, C
2952	Columbia Materials	Breton Sound 20	I, C
4206	Devon	Breton Sound 30	NI
3014	Energy Dev. Corp.	Main Pass 49	I, C
2827	Energy Dev. Corp.	Breton Sound 1	I, C?
2747	**Exxon**	**Lake Raccourci**	**I, N**
2732	**Exxon**	**Lake Sand**	**I, N**
3320	Greenhill Petroleum	Timbalier Bay	I, C
2072	Gulfland (Grasso)	Main Pass 35	I, C
2995	Hubco Exploration	Saturday Island	I, C
3002	Hubco Exploration	SE Saturday Island	I, C
2704	Hunt Petroleum	Caillou Island	I, C
2809	**Kerr-McGee**	**Breton Sound 36**	**I, N**
2810	Kerr-McGee	Breton Sound 32	I, C
2618	Kerr-McGee	Breton Sound 20	I, C

3063	Laurel Operating	West Black Bay	I, C
3072	LL&E (Nerco)	East Lake Sand	I, C
1856	Pennzoil	Quarantine Bay	I, C
1902	**Pennzoil (Amoco)**	**Redfish Point**	**I, N**
2856	Pogo	Breton Sound 2	I, C
2857	Pogo	Breton Sound 23	I, C
2479	Qunitana	Timbalier Bay	I, C
1898	**Samedan**	**Breton Sound 17**	**I, N**
1870	Scana	Chandeleur Sound 51	I, C
2072	Slam Resources	Main Pass 35	NI
2915	Snyder Oil	Chandeuler Sound 71	NI
2084	Texaco	Caillou Island	I, C?
2816	Texaco	Lake Barre	I, C?
2881	Texaco	Lake Pelto	I, C?
2504	Texaco	West Cote Blanche Bay	I, C?
2523	Texaco	Cote Blanche Island	I, C?
3030	Texaco	Queen Bess Island	I, C?
2825	Texaco	Rabbit Island	I, C?
1866	Texoil	Main Pass 4	NI
3032	Texoil	Chandeleur Sound 71	I, C
2273	Torch Operating	Chandeleur Sound 52	I, C
2915	Torch Operating	Chandeleur Sound 71	I, C
2898	Unocal	Caillou Island	I, C

[1] Results of interview, I = interviewed, NI = not interviewed, C= will continue to discharge if allowed, C?= not sure about continuing to discharge, N= plan to reinject or P&A and will not continue to discharge.

Table B-2. Location, receiving water body, depth, discharge rates and other data available for assumed continuing open bay discharges in Louisiana (ordered by receiving water body).

Permit No.	Latitude	Longitude	Receiving Water Body	Average Depth (ft)	Discharge Rate (bbl/day)	Data [1]	Comments
2825	29 26' 53"	91 36' 12"	Atchafalaya Bay		2,910	T x N	
3002	29 24' 35.061	89 54' 21.470"	Barataria Bay	8	2,017	x C x	
1901	29 35' 51"	89 32' 25"	Black Bay	7	10,123	T C N	
1901	29 35' 12"	89 32' 13"	Black Bay	7	20,077	T C N	
1901	29 35' 40"	89 34' 10"	Black Bay	8	11,500	T C N	
2672	29 32' 48.918"	89 29' 10.609"	Black Bay	7	8,366	T x N	
2860	29 34' 9.7"	89 30' 45"	Black Bay		6,800	T C N	
3063	29 35' 40"	89 34' 10	Black Bay	8	11,500	T x x	
2072	29 27' 3.403"	89 24' 11.464	Breton Sound	8	17,500	x C N	
2618	29 34' 41.4"	89 07' 00"	Breton Sound		22,500	x x N	
2856			Breton Sound		3	T C x	
2857	29 35' 31.251"	89 01' 53.993"	Breton Sound		10	T x x	
2857	29 35' 6.121"	89 00' 4.795"	Breton Sound		10	T x x	
2901			Breton Sound		200	x x x	Permit not Available
2901			Breton Sound		876	x x x	Permit Not Available
2952	29 37' 4.813"	89 4' 12.891"	Breton Sound	18	223	x x N	
1870	29 46' 32"	89 15' 09"	Chandeleur Sound		49	T C N	
2273	29 45' 08.65"	89 12' 29.31"	Chandeleur Sound			x C x	
2915	29 42' 16"	89 24' 23"	Chandeleur Sound	6	130	T C N	
3023	29 46' 21"	89 16' 52"	Chandeleur sound	10	3.4	T C x	
3032	29 42' 15.824"	89 24' 23.062"	Chandeleur sound	10	25	T C N	
3032	29 41' 46.466"	89 23' 48.018"	Chandeleur sound	10	25	T C N	
2859	29 33' 45.179"	89 26' 27.147"	E. Black Bay		10,807	T C x	
2816	29 12' 50"	90 29" 20"	Jacko Bay		600	T x x	
2816	29 12' 10"	90 28' 10"	Jacko Bay		220	T x x	
2816	29 12' 50"	90 28' 00"	Jacko Bay		614	T x x	
2816	29 12' 00"	90 28' 50"	Jacko Bay		117	T x x	
2816	29 13" 00"	90 30" 50"	Jacko Bay		int.	T x x	
2816	29 19' 50"	90 30' 10"	Jacko Bay		30	T x x	
2816	29 12' 00"	90 29' 50"	Jacko Bay		int	T x x	
2816	29 13' 00"	90-28' 40"	Lake Barre		510	T x x	
2881	29 06' 20"	90 39' 10"	Lake Pelto		729	T x x	
2881	29 05' 20"	90 38' 30"	Lake Pelto		1,103	T x x	

2881	29 06' 10"	90 38' 40"	Lake Pelto		489	T x x	
2881	29 05' 00"	90 39' 50"	Lake Pelto		2,485	T x x	
1866	29-41' 31.2"	89-22' 0.2"	Main Pass			T x N	
2072			Main Pass		20,250	x C N	
2134	29 46' 26"	89 17' 27"	Main Pass			x x x	Permit not available
2134	29 49' 35"	89 19' 58"	Main Pass			x x x	Permit not available
3014			Main Pass		0	x x x	Permit not available
2142	29 38' 12.03"	89 33' 33.64"	North Black Bay		12,076	T C N	
1856	29 25' 09"	89 30' 49"	Quarantine Bay	10	15,000	T C x	
2995	29 10' 43.943"	90 46' 30.170"	Salt Bay	8		x x x	
2881	29 05' 20"	90 40' 50"	Terrebonne Bay		204	T x N	
2084	29 06' 50"	90 29' 00"	Terrebonne Bay	10	2,484	T x N	
2084	29 05' 30"	90 30' 40"	Terrebonne Bay	10	3,017	T x N	
2084	29 07' 10"	90 30' 10"	Terrebonne Bay	10	3,720	T x N	
2084	29 07' 20"	29 31' 10"	Terrebonne Bay			T x N	
2084	29 06' 00"	90 25' 50"	Terrebonne Bay	10	41	T x N	
2084	29 04' 00"	90 28' 40"	Terrebonne Bay	10	701	T x N	
2704	29 05' 28.293	90 32' 17.027"	Terrebonne Bay	8	524	x x N	
2816	29 11' 20"	90 29' 00"	Terrebonne Bay		30	T x x	
2816	29 22' 30"	90 30' 50"	Terrebonne Bay		140	T x x	
2898	29 04' 25"	90 24' 20"	Terrebonne Bay	4	3,000	T C N	
2898	29 07' 50"	90 29' 50"	Terrebonne Bay			T C N	
2898	29 06' 00"	90 28' 40"	Terrebonne Bay	10	617	T C N	
2479			Timbailier Bay		10	T C N	
2816	29 12' 00"	90 26" 50"	Timbailier Bay		10	T x x	
2898	29 04" 20"	90 25' 30"	Timbailier Bay			T C N	
3320	29 05' 29"	90 18' 30"	Timbailier Bay		4,744	x x N	
3320	29 04' 12"	90 18' 30"	Timbailier Bay		3,873	T C N	
3320	29 04' 33"	90 17' 10"	Timbailier Bay		4,914	x x N	
3320	29 04' 37'	90 19' 2"	Timbailier Bay		7,368	x x N	
3320	29 04' 17'	90 19' 25"	Timbailier Bay		1,680	x x N	
2084	29 06' 20"	90 27' 30"	Timbalier Bay	10	1,201	T x N	
2084	29 07' 00"	90 26' 40"	Timbalier Bay			T x N	
2084	29 06' 10"	90 26 50"	Timbalier Bay			T x N	

2084	29 05' 20"	90 27' 00"	Timbalier Bay			T x N	
2084	29 05' 22"	90 25' 56"	Timbalier Bay			T x N	
2084	29 07' 00"	90 32' 40"	Timbalier Bay	10	802	T x N	
2084	29 05' 20"	90 27' 00"	Timbalier Bay	10	2,126	T x N	
2084	29 08' 00"	90 27' 40"	Timbalier Bay	10	2,065	T x N	
2084	29 06' 50"	90 27' 50"	Timbalier Bay	10	586	T x N	
2084	29 06' 19"	90 27' 58"	Timbalier Bay			T x N	
2504	29 41' 04"	91 47' 59"	West Cote Blanche Bay	10	37,113	T x N	
2523	29 43' 10"	91 42' 00"	West Cote Blanche Bay	7	5,364	T C N	
2523	29 43' 48"	91 41' 35"	West Cote Blanche Bay			T C N	
1934					14,443	x x x	Permit not available
2827					1	x x x	
2915						x x x	Permit not available
3072						x x x	No Data in Permit File
4206						x x x	Permit Not Available

[1] Available Data T= toxicity data; C= chemical data; N= NORM data

Figure C-1. Assumed active discharges in open Louisiana bays.

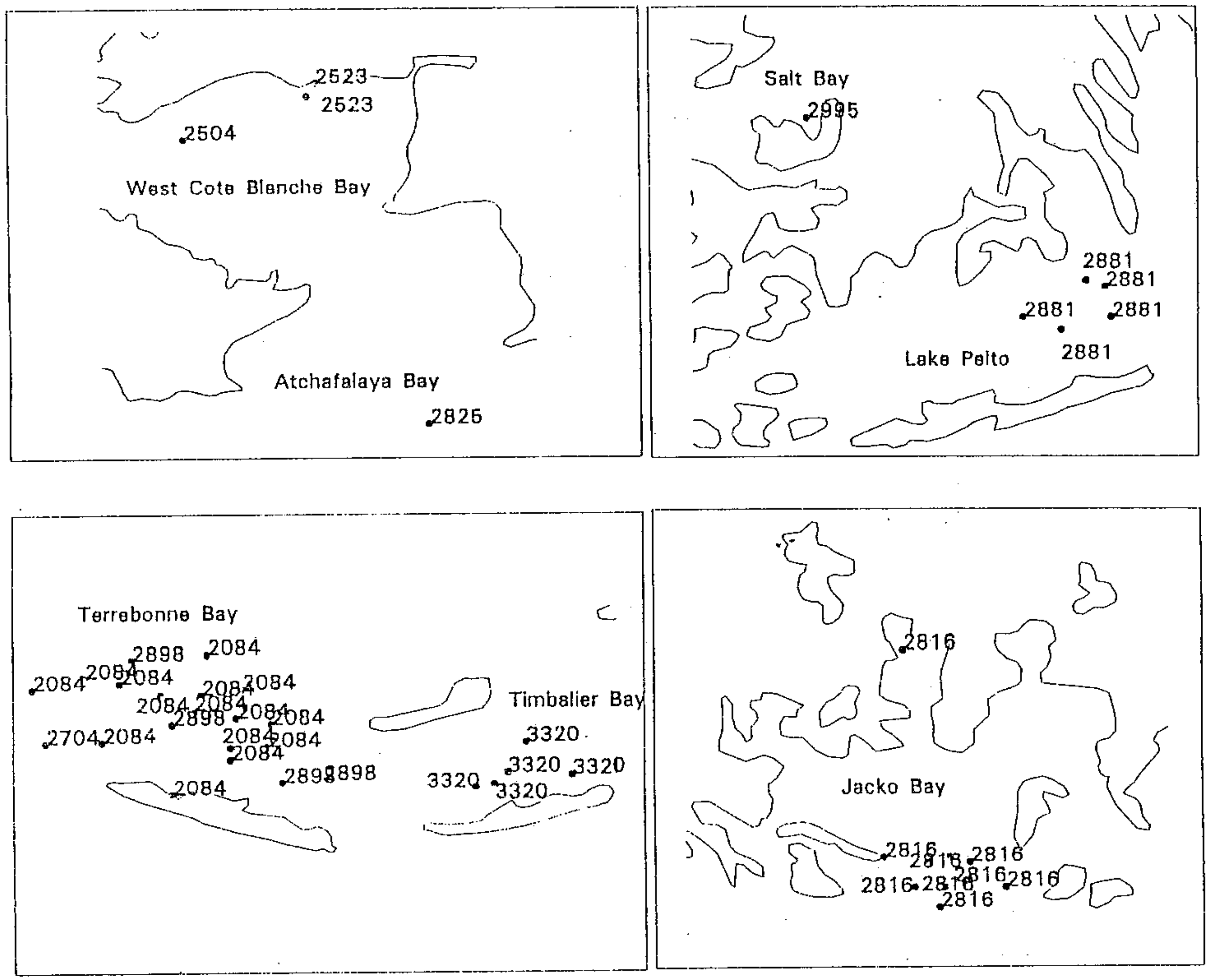
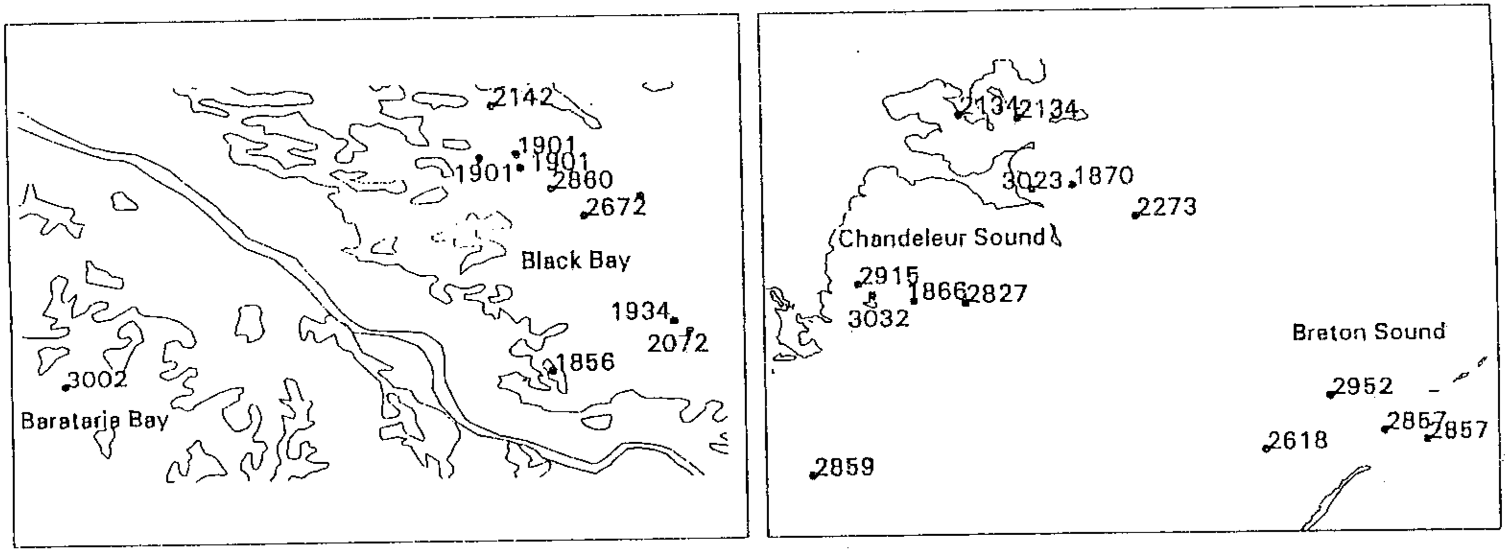


Figure C-1.(cont.) Assumed active discharges in open Louisiana bays.



Permit No.	1856	1856	1856	1856	1856	1870	1901	1901	1901	2072	2142	2523	2623	2618	2856	2859	2860	2898	2916	3002	3023	3032
	sample #1	sample #2	sample #	sample #	average																	
ANTIMONY													134		0.3			20.1		12.5	15.43	11.852
ARSENIC	0.013				0.013	0.011	6.9				10		117	30		44	79	0.011		88.489		27.055
BARIUM														62000								
BENZENE	19600	7270	4910	6420	9550		1400	150	770	950	19600	4910	1760	96		396	10	4400		410		85
CADMIUM													1.6		0.01		0.11	0.5		<.003	0.46	301.515
CHROMIUM (IV)							140						26.9					0.2		17.626	0.05	53.664
COPPER	0.39				0.39		130				710		350	10	0.02		0.16	0.38		244.643	0.27	460.8
LEAD													56		829		1.1	0.3		83.788	2.28	35.355
MERCURY															0.004	0.0034				0.0068	0.001	<0.001
NAPHTHALENE	1180				1180				41	82		51	100				0.01					
NICKEL											310		261				2.1	1.5		57.895	2.84	113.043
PHENANTHRENE	20				20																	
PHENOL	631				631	70	360	180	340	350	91	2200	1530		12000		0.15	3200		1400	24	
SELENIUM																0.077				80		10.986
SILVER													125					0.4		11.304	0.24	14.291
TOLUENE	3890	1660	2280	2980	2702.5		2800	180	580	830	3890	799	759	110		223	0.035	1500	16			
XYLENES	1090	621	737	1000	862		110	10		95	1090	390				67	0.028	220	44		7	
ZINC	0.17				0.17	0.1	110				120		1850	50	0.38		0.26	4.1		31.094	0.24	76.173
Bis (2-ethylhexyl) phth	66				66	60	80				57					45	0.05					
Flourene	11				11																	
2,4-Dimethylphenol	106				106				14	48		8040			990			510		77		
Acetone										390					2200							
2-Methylnaphthalene										53							0.01					
2-Methylphenol							20	10		66							0.02	1600				
4-Methylphenol							150			300							0.14	1700			60	
Benzoic acid										420												
Iron														16000								
2-Butanone															2100			930			230	
Beryllium																		0.05		<.1		0.635
Thallium																		2.6		248.387	2.73	345.313
Methylene Chloride																	0.006			36		
Acetone																	0.039				98	
1,2 Dichloropropane																						55
Carbon Tetrachloride																						45
2-Hexanone																					30	55
2-Chlorophenol																						18
1,4-Dichlorobenzene																						11
1,2-Dichlorobenzene																						12
N-Nitroso-di-n-propylamine																						11
1,2,4-Trichlorobenzene																						13
Acenapthene																						17
Fluorene																						11
4-Methyl-2-Pentanone																					17	
Diethylphthalate															580							
4-Nitrophenol												49										

Table B-3. Chemical contaminant concentrations in open bay produced water discharges in Louisiana (ordered by receiving water body).

Table B-4. Radium concentrations in open bay produced water discharges in Louisiana (ordered by receiving water body).

Permit No.		^{226}Ra (pCi/l)	^{228}Ra (pCi/l)
1866		23	2
1870		73.8	109.0
1901		296	367
1902		178	245
2072		240	273
2084	Tank Bat 2	181	282
2084	Tank Bat 4	65.2	69.2
2084	Tank Bat 6	308	368
2084	Tank Bat 7	87.1	61.4
2084	Tank Bat 8	156	91.4
2084	Tank Bat 9	273	424
2084	Tank Bat 10	172	295
2084	Tank Bat 11	114	171
2084	Tank Bat 14	117	208
2084	Tank Bat 15	247	291
2084	Tank Bat 17	146	283
2084	Tank Bat 18	50.2	56.3
2084	Tank Bat 19	272	353
2084	Tank Bat 20	380	558
2084	Tank Bat 21	311	483
2084	Tank Bat 22	89.2	125
2084	Tank Bat 23	68	47.1
2084	Tank Bat 24	131	225.0
2142		277.0	341.0
2479		3.9	2
2504		108	149
2523	TB#3	207	326
2523	TB#1	129	206
2618		201	289
2672		277.0	341.01
2704		127	400

2825		436	501
2860		120	41.9
2881	TB#2	52.7	109
2881	TB#3	194.0	307
2881	TB#4	173	472
2881	TB#5	290	545.0
2881	TB#1	224	389
2898		0.0	0
2915		290	60
2915		34	66
2952		52	15
3032		592	188
3320		198	265
3320		290	272
3320		284	402
3320		303	361
3320		333	560

APPENDIX C

RADIONUCLIDE EFFECTS

C.1 Quantities and Units

Traditional units in radiation dose measurements (i.e. Ci, rad, rem) are being replaced by the International System (SI) of units (Bq, Gy, Sv). The names and units (traditional and SI) for activity, absorbed dose and dose equivalent are given in Table C-1. Prefixes commonly applied to these units are given in Table C-2.

Table C-1. Radiological names and units.

Quantity	Traditional		SI		Conversion
	Name	Unit	Name	Unit	
activity	curie (Ci)	3.7×10^{10} dis/sec	becquerel (Bq)	1 dis/sec	1 Bq = 2.7×10^{-11} Ci
absorbed dose	rad (rad)	100 erg/gm	gray (Gy)	1 J/kg	1 Gy = 100 rad
equivalent dose	rem (rem)	100 erg/gm	sievert (Sv)	1 J/kg	1 Sv = 100 rem

Table C-2. Prefixes used in radiation protection.

pico (p)	10^{-12}
nano (n)	10^{-9}
micro (μ)	10^{-6}
milli (m)	10^{-3}
kilo (k)	10^{3}
mega (M)	10^{6}
giga (G)	10^{9}
tera (T)	10^{12}

Radioactivity is quantified in terms of the number of spontaneous energy emitting transformations per unit time -- a quantity known as activity. An example of a transformation is the decay of a radium 226 nucleus into a radon 222 nucleus, an alpha particle and gamma rays. The unit of activity has historically been the curie (Ci). One curie is equal to 3.7×10^{10} disintegrations per second. In the SI system, the basic unit of activity has been redefined as one disintegration per second, known as the becquerel (Bq). One curie is equal to 3.7 x 10^{10} Bq.

The biological effects of exposure to a radionuclide are related to the absorbed dose and dose rate. The absorbed dose is a measure of the energy imparted to matter. An absorbed dose of 100 erg/gram is called 1 rad. In the SI system of units, the unit of absorbed dose is the Gray (Gy, 1 Joule/kilogram). An absorbed dose of 1 rad is equal to 0.01 Gy (1 Gy = 100 rads).

The probability of stochastic effects (i.e. cancer and genetic effects) depends not only on the absorbed dose, but also on the type and energy of the radiation causing the dose and on the organs or tissues irradiated. Factors have been developed by the International Commission on Radiological Protection (ICRP, 1991) to account for these relationships in humans.

Radiation weighting factors are used to account for the differences in relative biological effectiveness (RBE) of different radiations. In the past these differences were accounted for by use of quality factors. The radiation weighting factor for gamma radiation (γ) and beta (β) particles has been assigned a value of 1. The weighting factor for alpha (α) particles is set to 20. The absorbed dose modified by the weighting factor is called the equivalent dose and is expressed in units of Joules per kilogram with the name Sievert (Sv) given to 1 Joule/kg. The traditional unit is the rem (see Table C-1). One Sievert is equal to 100 rem.

Tissue weighting factors are used to account for differences in the sensitivity to cancer induction of different human tissues and organs. A tissue weighting factor represents the relative contribution of that organ or tissue to the total effects resulting from uniform irradiation of the whole body. These factors are given in ICRP (1991). The equivalent dose weighted by these tissue weighting factors is referred to as the effective dose. For a uniform, whole body exposure, the equivalent and effective doses have the same value, and are both expressed in units of Sieverts (Sv).

The limited data for the relative biological effectiveness of various radiation types in man indicate that the RBE can be expected to be similar for aquatic organisms, (Woodhead, 1984), because the soft tissues of man and other organisms are generally similar in terms of water content and basic cell structure (IAEA, 1988). IAEA (1988) suggested that it is reasonable to apply the same quality factors (now radiation weighting factors) derived for humans to doses received by aquatic organisms. There are no parallel tissue weighting factors for aquatic organisms, and the usual approach to estimating doses to aquatic animals to assume that the dose is averaged over the whole body of the organism. NCRP (1991) suggests this approach is reasonable, as long as the average whole body exposure is representative of the dose to the gonads.

NCRP also suggests that it may be useful to estimate the dose to the most highly exposed tissue (NCRP, 1991).

C.2 Human Health Effects From Radium Ingestion

C.2.1 Carcinogenicity of Radium

The health effects of radium can be attributed to the radioactive emissions of the radium isotopes and their daughters. The alpha, beta and gamma radiation released by the decay of radium and its daughters cause ionization of cellular components which may result in the mutation or death of affected cells.

Most of the information concerning the health effects of radium come from studies of two groups of people: radium dial painters who ingested radium paint and patients who were injected with radium-224 for treatment of spinal arthritis and tuberculosis of the bone (NAS, 1988). The primary data come from studies of radium dial painters (Rowland et al., 1978, 1983). Radium body burdens were measured in the dial painters and were used to calculate lifetime intake.

In these studies, ingestion of ^{226}Ra resulted in bone cancers (osteosarcomas) and cancers of the linings of the cranial sinuses (head carcinomas). Ingestion of ^{228}Ra resulted in bone cancers. The dose-response function for bone cancer induced by ingestion of ^{226}Ra or ^{228}Ra is purely quadratic, with no excess cancers at lower doses. From a practical point of view, the dose-response function exhibits a threshold at a dose to the skeleton that is well above the worst environmental exposures that have been documented.

The data for head carcinomas can fit either a linear or quadratic function. These carcinomas are attributed to radon-222, a daughter of ^{226}Ra. No excess head carcinomas are associated with ^{228}Ra. The half-life of its daughter product, radon-224, is too short to allow migration to and accumulation in cranial sinuses.

C.2.2 USEPA Risk Factors for Radium

Current practice in radiation protection is to assume there is a cancer risk associated with even small doses of radiation. Risk factors are derived from epidemiological data and extrapolated down to low doses to describe the cancer risk associated with small exposures.

The Science Advisory Board (SAB) has recommended that the USEPA use the epidemiological evidence for bone and head cancers in radium dial painters to derive risk factors for radium (SAB, 1991). The evidence for radium-induction of other soft-tissue cancers is equivocal (Stebbings *et al.*, 1984).

USEPA derived radium risk factors using the RADRISK model, based on effective dose equivalents given in ICRP (1977), modified to account for the specific metabolic behaviors of radioactive daughters (USEPA, 1991). RADRISK incorporates a toxicokinetic model based upon alkaline earth intake, retention and excretion. RADRISK is a linear, no-threshold model that uses the sum of weighted organ doses to arrive at a single dose coefficient used to predict either the risk of getting a cancer or the risk of dying from cancer. RADRISK incorporates a life-table analysis to adjust for age- and sex specific mortality from competing risks.

RADRISK uses a gut uptake factor (f_1) of 0.2, the value recommended by the ICRP (1979). This value is based on data for adult humans who ingested radium in water or incorporated into food (ICRP, 1973; Stehney and Lucas, 1956). Weighting factors in RADRISK were modified from those of the ICRP (USEPA, 1991) to calculate the risks for all cancers (fatal and non-fatal). "Ingested radium is estimated to distribute about 85% to bone and 15% to soft tissue. (UNSCEAR, 1972)" (USEPA, 1991).

The RADRISK model results were adjusted for the over-prediction of leukemias and lack of prediction of head carcinomas (Federal Register, 1991), but the RADRISK model still produces a majority (about two-thirds) of the overall risk estimate for soft tissues, where either no evidence or marginal evidence exists for radium induced cancers. For example, increases in breast cancer and multiple myelomas correlate better with duration of employment, a surrogate for external dose of gamma radiation, than with radium intake (Stebbings et al., 1984). According to the USEPA, the ratio of all cancer risks to the risks for bone and cranial cancers may be overestimated by a factor of between two and five (Federal Register, 1991).

The analysis performed by the USEPA (Federal Register, 1991; USEPA 1991) assumes a linear dose-response relationship for bone sarcoma, although the best fit for bone sarcoma in the radium dial painters is quadratic (USEPA, 1991). If the true relationship is quadratic, the USEPA risk factors will be overestimates. There may also exist a practical threshold for bone sarcoma (USEPA, 1991). Additional uncertainties and assumptions in the USEPA analysis are described in USEPA (1991).

Using RADRISK, the United States Environmental Protection Agency (USEPA) estimated the risk factor associated with the ingestion of ^{226}Ra in drinking water to be 4.4×10^{-6} lifetime risk per pCi/l, and the risk factor for ^{228}Ra to be 3.8×10^{-6} lifetime risk per pCi/l (assuming lifetime exposure) (Federal Register, 1991; USEPA, 1991). These risk factors are based on an assumed water intake of 2 l/day. Unit risk factors (individual lifetime fatal cancer risk per pCi/day) can be derived from these values by dividing the risk factors by two. The USEPA risk factors are then equivalent to 2.2×10^{-6} lifetime risk per pCi/day for ^{226}Ra and 1.9

x 10^{-6} lifetime risk per pCi/day for ^{228}Ra (assuming lifetime exposure) (Table C-3).

C.2.3 Risk Factor Distribution

A risk factor distribution for ^{226}Ra and ^{228}Ra was derived by assuming that the USEPA values represent the upper 90% confidence limit of a lognormal distribution. The lower 90% confidence limit was based on the risk factors for the radium induced cancers in humans for which there is epidemiologic evidence (bone and head carcinomas for ^{226}Ra and bone sarcoma for ^{228}Ra). The methods of Layton et al. (1987) were used to establish lognormal distributions with the arithmetic means and standard deviations given in Table C-4.

Table C-3. USEPA risk factors for ^{226}Ra and ^{228}Ra*.

TYPE	USEPA RISK FACTORS		USEPA UNIT RISK FACTORS	
	^{226}Ra	^{228}Ra	^{226}Ra	^{228}Ra
	risk per pCi/l	risk per pCi/l	risk per pCi/d	risk per pCi/d
Bone Sarcoma	9.4 x 10^{-7}	9.4 x 10^{-7}	4.7 x 10^{-7}	4.7 x 10^{-7}
Head Carcinoma	9.4 x 10^{-7}	0	4.7 x 10^{-7}	0
Leukemia, high LET	2.1 x 10^{-7}	2.6 x 10^{-7}	1.1 x 10^{-7}	1.3 x 10^{-7}
Leukemia, low LET	9.6 x 10^{-8}	2.6 x 10^{-7}	4.8 x 10^{-8}	1.3 x 10^{-7}
All Other	2.3 x 10^{-6}	2.3 x 10^{-6}	1.2 x 10^{-6}	1.2 x 10^{-6}
Total	4.4 x 10^{-6}	3.8 x 10^{-6}	2.2 x 10^{-6}	1.9 x 10^{-6}

* individual lifetime cancer risk, assuming lifetime exposure, from USEPA (1991); divide USEPA risk factors (risk per pCi/l) by two to get risk per pCi/day.

Table C-4. Risk factor distribution for Ra-226 and Ra-228 (lognormal distributions, risk per pCi/day).

Parameter	^{226}Ra	^{228}Ra
Arithmetic Mean	1.5 x 10^{-6}	1.0 x 10^{-6}
Standard Deviation	9.0 x 10^{-7}	1.4 x 10^{-6}
Lower 90% Confidence Limit	9.4 x 10^{-7}	4.7 x 10^{-7}
Upper 90% Confidence Limit	2.2 x 10^{-6}	1.9 x 10^{-6}

Radium is retained in bone and delivers a dose over the remaining lifespan of the exposed individual. The risk factors calculated by the USEPA model RADRISK take account of the total dose accumulated by tissues after intake (called the committed effective dose equivalent), and assume a lifetime exposure.

Retention is the amount of a substance remaining in a tissue or organ at some time after uptake. Within 10 years after an initial intake of radium, most of the radium in the body has been eliminated (*Norris et al.*, 1955). This observation suggests a way to adjust the USEPA lifetime risk factors (and the distributions of risk factors) for exposure periods less than a lifetime. If ten years (to account for the radium left in the body, and delivering a dose after intake and uptake have stopped) is added to the expected exposure period, the maximum risk factor for the expected exposure period can be calculated:

$$RF(EP) = \frac{(EP + 10\ ) \times URF_{70}}{70y}$$

where:
RF(EP) = risk factor as a function of exposure period EP (lifetime risk per pCi/day)
EP = exposure period (years)
URF(70) = USEPA unit risk factor for lifetime exposure (lifetime risk per pCi/day)

This modified risk factor was used in the probabilistic risk assessment for radium described in this report. This method will slightly overestimate the committed dose, but the estimate is less conservative than assuming a seventy year exposure when such an assumption is not realistic.

C.3 Effects on Aquatic Organisms

Exposure to ionizing radiation can result in injury at the molecular, cellular and whole body levels. Most of the available studies of the effects of radiation on aquatic organisms are concerned with the induction of deterministic, somatic effects. These effects include increases in mortality and pathophysiological, developmental and reproductive effects. There is little information available concerning induction of cancer and genetic effects, although a few studies of stochastic genetic effects in organisms are available (Anderson and Harrison, 1986).

Reproductive and early developmental systems of vertebrates are the most sensitive to radiation, and invertebrates appear to be relatively resistant (NCRP, 1991).

Most studies of the effects of radiation on aquatic organisms were performed in the laboratory, with effects determined on individual animals. A few studies of the effects of radiation on natural populations have been performed. The most important consideration on assessing the effects of radionuclides discharged in produced water is the influence radiation exposure has on reproductive success in populations, and consequences in populations and ecosystems. If exposures are limited to protect fertility and fecundity of the population as a whole, it is unlikely that other effects in individuals will be important to the population (NCRP, 1991).

IAEA (1976) and Templeton (1980) examined the possible effects of chronic, low level radiation on recruitment, fecundity and mortality by considering the known regulatory mechanisms of natural populations. Recruitment for highly fecund species is not directly related to standing stock size and the mortality rate operating on eggs and larvae varies from year to year. Survival of eggs and larvae depend to a large degree on the availability of food, and a large number of eggs are produced at each spawning (Templeton, 1980). Density dependent mortality reduces fish larvae populations to the level that can be supported by the available food. If mortality is enhanced by low levels of radiation, recruitment to the stocks of highly fecund fish is not likely to be affected, unless the stocks are already at risk due to over-exploitation or other environmental stresses (IAEA, 1976; IAEA, 1988; NCRP, 1991).

For species with low fecundity (e.g., sharks and marine mammals), recruitment is closely related to parent stock size. It is not possible to predict the effects on recruitment for these species, although effects could be more significant than for highly fecund species. However, at low dose rates, it is reasonable to assume that effects will be small compared to fishing and other pressures (IAEA, 1976). For species with special social value (endangered and threatened species, marine mammals) effects on individuals may be of importance.

Effects at the ecosystem level have been demonstrated only for the large doses received at Eniwetok and Bikini atolls in the Pacific Proving Grounds (Templeton, et al., 1971).

The National Council on Radiation Protection and Measurements recently reviewed the literature on the effects of ionizing radiation on aquatic organisms, and suggested reference levels that would protect aquatic populations (NCRP, 1991). Major conclusions of this review included:

- Experimental studies in the laboratory have shown detectable effects on fecundity down to 10 mGy/d.

- Effects not necessarily deleterious at the population level have been detected at dose rates between 1 and 10 mGy/d. Deleterious effects on natural populations were observed at dose rates ≥ 10 mGy/d. Clearly deleterious effects which would be detected at the population level appear in the range of 10-100 mGy/d.

- Lowest dose rate causing no effect in natural populations: 0.5 mGy/d; lowest dose rate causing no effect in laboratory: 10 mGy/d.

NCRP (1991) suggests a reference dose rate to protect aquatic populations of 10 mGy/d. NCRP also suggests a detailed assessment if an initial analysis results in estimated dose rate above 2.4 mGy/d.

IAEA (1988) suggested similar reference dose rates where effects on aquatic biota would be minimal. IAEA (1988) concluded that:

- increased mortality is expected above 10 mSv/hr (240 mSv/d);
- reduced reproductive success may occur between 1 and 10 mSv/hr (24-240 mSv/d);
- some somatic effects which would be eliminated by natural selection could occur between 0.004 and 1 mSv/hr (0.1-24 mSv/d); and
- no adverse effects are expected below background levels of 0.004 mSv/hr (0.1 mSv/d).

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10.5 Conclusions

The tiered approach to risk assessment is a cost-effective way to provide information needed to make risk management decisions. This screening assessment for human health and ecological risks from open bay produced water discharges in Louisiana eliminated a number of contaminants from further consideration. More quantitative assessments were performed on contaminants of potential concern.

Human health risks from radium in produced water appear to be small. Ecological risks from radium and other radionuclides in produced water also appear to be small.

Intakes of chemical contaminants in fish caught near open bay produced water discharges are expected to posed a negligible toxic hazard or carcinogenic risk to people.

Potential impacts to benthic biota and fish and crustaceans in the water column are possible for some discharges within the 200 ft mixing zone. Permanent damage to populations of organisms and ecosystems are not expected, because mixing zones represent relatively small volumes and animals are not expected to remain continuously in the plume.

ISSUE PAPER

Gulf Of Mexico Coastal Produced Water Discharges
April 1996

Issue

In its proposed coastal effluent limitations guidelines (ELGs), the U.S. Environmental Protection Agency (EPA) proposed zero discharge of produced water to coastal waters of the Gulf of Mexico. Zero discharge is appropriate for shallow or poorly mixed coastal waters, but may be overly restrictive for discharges to open bays or well-mixed deltaic passes. If the coastal ELGs are adopted with the zero discharge requirement in place, any state or regional flexibility will be removed.

Background

EPA proposed ELGs for the coastal subcategory of the oil and gas extraction industry on February 17, 1995 (60 FR 9428). EPA proposed allowing discharges of produced water to Cook Inlet at the levels allowed by the offshore ELGs, but proposed zero discharge of produced water from Gulf of Mexico coastal facilities. EPA's rationale was: "Zero discharge is technologically available because injection of produced water is currently ongoing in much of the coastal subcategory at the present time, and adequate geological formations exist to accept produced water." [EPA Coastal Development Document, p. XIV-11].

In developing ELGs, a type of technology-based limits, EPA must consider the availability and cost of treatment and disposal technology, but does not need to consider water quality impacts [Clean Water Act, §304(b)(2)(B)]. A permit writer must consider both technology-based limits and water quality-based limits when developing a permit, but in developing the ELGs, EPA needs only evaluate technology. In the case of coastal Gulf of Mexico, EPA need not evaluate whether produced water discharges would impair water quality. EPA only needs to examine if a treatment or disposal technology is economically available to coastal operators. Apparently, EPA believes that underground injection is both available and is cost-effective for Gulf of Mexico coastal dischargers, regardless of whether they are in shallow or deep water.

Other Regulatory Issues

Louisiana Regulations - The proposed coastal ELGs are complicated by two related regulatory issues. First, the Louisiana Department of Environmental Quality (LADEQ) promulgated regulations in 1991 that required a phase out of coastal produced water discharges by January 1, 1995. Based on some preliminary U.S. Department of Energy (DOE) comments, on December 16, 1994, LADEQ extended the deadline for discontinuing produced water discharges for certain open bay locations to January 1997.

It is important to note that the LADEQ regulations provide two mechanisms to avoid zero discharge. In lieu of ceasing discharge, operators could treat the produced water to meet very restrictive effluent limits and continue to discharge. As another alternative, dischargers in open bays could show, on a case-by-case basis, that their discharges should be exempted from these regulations on the basis of a DOE study on the environmental impacts of open bay discharges of produced water. The DOE study is now completed.

EPA Region VI Coastal General Permit - The second regulatory issue affecting the proposed coastal ELGs is the EPA Region VI general permit for produced water and produced sand discharges, issued January 9, 1995. The general permit requires zero discharge of produced water into Louisiana and Texas coastal waters, except for discharges derived from offshore wells into the main deltaic passes of the Mississippi River or to the Atchafalaya River below Morgan City. The general permit would not allow discharges from open bays.

However, the general permit is not the only regulatory option available to coastal operators in Louisiana and Texas. The general permit is a mechanism of convenience for Region VI permit writers and many coastal operators. Since many facilities discharge the same waste streams to similar bodies of waters, a general permit provides a more efficient regulatory mechanism. However, operators are not required to be covered under a general permit; they are entitled to apply for an individual National Pollutant Discharge Elimination System (NPDES) permit. Under an individual NPDES permit, a permit writer can make a case-by-case assessment of the appropriate technology-based and water quality-based limits for produced water discharges. EPA and delegated states issue thousands of NPDES permits each year. As was true for the LADEQ regulations, there is a readily available mechanism to seek relief from the zero discharge requirement.

Implications for Gulf of Mexico Coastal Dischargers

The proposed coastal ELGs do not offer the same sort of readily available relief found in the LADEQ regulations and the general permit process. If the final coastal ELGs contain the same zero discharge requirement, all Gulf of Mexico coastal discharges would be required to cease discharging produced water, regardless of their location. This requirement would supersede both the LADEQ regulations and the general permit, rendering their relief mechanisms useless.

The only mechanism that could potentially offer relief from a final ELG is the fundamentally different factors variance (FDFV) allowed by §301(n) of the Clean Water Act. Unfortunately, this is not a very practical alternative. In over 20 years of the FDFV program, EPA received about 250 requests for FDFVs, but only 7 were granted. Some were denied, but many others remain undecided in EPA regional or headquarters offices. FDFVs must be applied for by each company individually; no group applications are allowed.

Potential Mitigating Actions

EPA plans to publish final coastal ELGs in October 1996. Realistically, EPA will finish ELG development by August 1996 and spend the next two months getting internal and external approvals. Any input that would potentially change the proposed position must be provided to EPA by August 1996.

The most valuable input that could provide relief from an across-the-board zero discharge requirement is data demonstrating that facilities located farther from shore, in open bays, or in main deltaic passes cannot cost-effectively inject produced water to the extent that more typical coastal dischargers can. This would give EPA rationale to treat those types of discharges differently and would fall within the clear confines of the ELG-development process.

If such data is not available, state officials could meet with senior EPA officials seeking to retain a greater degree of state flexibility. Under the current Administration, EPA is much more amenable to working with state agencies and operating in a flexible manner. Given sufficient prodding from states, EPA may be willing to allow certain types of coastal dischargers, whose produced water discharges would not cause water quality impacts, to discharge at the levels allowed by the offshore ELGs.

If no positive action is taken by states or the industry, the final coastal ELGs may preclude any produced water discharges to coastal Gulf of Mexico waters. This could result in the loss of potential oil and gas resources, jobs, and revenues to state and federal treasuries.

Comparison of Coastal Produced Water Discharge Requirements under Three Regulatory Approaches

	EPA Proposed Coastal ELGs	LADEQ Regulations	Region VI General Permit
Baseline condition for produced water discharge	zero discharge for Gulf of Mexico	zero discharge by 1997	zero discharge
Exceptions	none	dischargers can choose to meet very strict effluent limits	discharges derived from offshore wells into the main deltaic passes of the Mississippi River or to the Atchafalaya River below Morgan City are not subject to the zero discharge requirement
Mechanisms for gaining relief from zero discharge	operators have 180 days following adoption of ELGs to apply for a fundamentally different factors variance, which is very difficult to get; realistically, no opportunity for relief exists	open bay dischargers can make case-by-case demonstration that environmental impacts from the produced water discharge are acceptable	dischargers can apply for an individual NPDES permit that could possibly allow produced water discharge
Comments	ELGs do not consider effects of the discharge on the environment; they only evaluate the cost and availability of treatment or disposal technology; EPA is therefore unconcerned about where the discharges are located as long as cost-effective treatment or disposal technology exists	following release of DOE study on impact of coastal produced water discharges, some open bay dischargers may be able to make a demonstration of minimal impact if final coastal ELGs require zero discharge for produced water, LADEQ relief mechanism is overruled	if final coastal ELGs require zero discharge for produced water, individual NPDES permits could not provide relief