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PRODUCED WATER DISCHARGES TO THE GULF OF MEXICO: BACKGROUND INFORMATION FOR ECOLOGICAL RISK ASSESSMENTS

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

June 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
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ACRONYMS

AET	Apparent Effects Threshold
AF	Application Factor
API	American Petroleum Institute
AQUIRE	Aquatic Toxicity Information Retrieval Database
BAF	Bioaccumulation Factor
BAT	Best Available Technology Economically Achievable
BCF	Bioconcentration Factor
BEAG	Biomedical and Environmental Assessment Group
BEDS	Biological Effects Database
BGOF	Buccanear Gas and Oil Field
BNL	Brookhaven National Laboratory
CEAM	Center for Exposure Modeling
CORMIX	Cornell Mixing Zone Expert System Model
CSA	Continental Shelf Associates, Inc.
EC_{50}	Median Effective Concentration
EIS	Environmental Impact Statement
ERL	Effects Range Low
ERM	Effects Range Median
FFPI	Fossil Fuel Pollution Index
FWS	Fish and Wildlife Service
IAEA	International Atomic Energy Agency
ICRP	International Commission on Radiological Protection
IFEM	Integrated Fate and Effects Model

LC_{50}	Median Lethal Concentration
LD_{50}	Median Lethal Dose
LDEQ	Louisiana Department of Environmental Quality
LOEC	Lowest Observed Effects Concentration
LOAEL	Lowest Observed Adverse Effect Level
MATC	Maximum Acceptable Toxicant Concentration
MFO	Mixed Function Oxidase
MMS	Minerals Management Service
NEPA	National Environmental Policy Act
NCRP	National Council on Radiation Protection and Measurements
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NOEC	No Observed Effects Concentration
NOEL	No Observed Effects Level
NORM	Naturally Occurring Radioactive Material
NPDES	National Pollution Discharge Elimination System
OEI	Offshore Ecology Investigation
OOC	Offshore Operators Committee
PAC	Polyaromatic Carbon
PAH	Polycyclic Aromatic Hydrocarbon
QF	Quality Factor
RBE	Relative Biological Effectiveness
SEP	Sediment Equilibrium Partitioning
SI	International System of Units (Systeme International)

SWACOM	Standard Water Column Model
TDS	Total Dissolved Solids
USDOC	United States Department of Commerce
USDOE	United States Department of Energy
USDOI	United States Department of Interior
USEPA	United States Environmental Protection Agency

EXECUTIVE SUMMARY

Introduction

Oil and gas production is often accompanied by the production of a saline wastewater, called produced water. In offshore and coastal areas, this wastewater may be discharged to surface water. Produced water may contain a number of contaminants, including oil and grease, organics, heavy metals and radionuclides. Many of these contaminants are toxic to marine organisms at high concentrations.

Environmental impacts associated with discharges of produced water are of concern to regulators at the state and federal levels, the public, environmental interest groups and industry. Most of the current (and projected future) oil and gas platforms in the U.S. are located in the central and western Gulf of Mexico. This area supports economically important commercial and recreational fisheries, as well as unique, socially valued ecosystems, and several endangered and threatened species.

This report reviews ecological risk assessment concepts and methods; describes important biological resources in the Gulf of Mexico of potential concern for produced water impacts; and summarizes data available to estimate exposure and effects of produced water discharges. The emphasis is on data relating to produced water discharges in the central and western Gulf of Mexico, especially in Louisiana. Much of the summarized data and cited literature are relevant to assessments of impacts in other regions. Data describing effects on marine and estuarine fishes, mollusks, crustaceans and benthic invertebrates are emphasized.

This review 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 United States Department of Energy (USDOE). These assessments will provide input to regulators in the development of guidelines and permits, and to industry in the use of appropriate discharge practices.

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 more quantitative risk-based approach to decision-making for environmental protection.

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. 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.

Selection of endpoints is a critical step in an ecological risk assessment. Selection of assessment endpoints includes identifying the valued components of the environment that are at risk, and developing an operational definition of effects. USEPA recommends that this selection consider ecological relevance, policy goals and societal values, and susceptibility to the stressor (USEPA, 1992).

Approaches to ecological risk assessment were reviewed in the context of USEPA's proposed framework. Methods for exposure assessment include application of transport and fate models, and approaches to the estimation of dose and internal exposure. Methods and data for the effects assessment phase include data from acute and chronic toxicity tests at the individual and population level; and methods to extrapolate effects between species and genera and from acute to chronic effects. Methods for risk characterization include comparing exposure and effects values or distributions, and application of population and ecosystem models.

Major uncertainties in ecological risk assessments come from three fundamental sources: the heterogeneity or stochasticity of natural systems, measurement error, and lack of knowledge. Uncertainties in ecological risk assessments that come from a basic lack of knowledge should be described qualitatively. Other major uncertainties, including natural heterogeneity or stochasticity, and parameter error, can be treated analytically.

A commonly used tool in 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 run to interact in a model with samples from other input parameters.

Biological Resources In The Gulf Of Mexico

A description of the important fisheries resources in the Gulf of Mexico is needed to complete the problem formulation phase of an ecological risk assessment and to identify endpoints for a specific analysis. Additional site or region specific data may be needed for a specific analysis, and species with important social value other than commercial or recreational uses should also be considered.

The Gulf of Mexico includes a wide variety of habitats for marine biota, both in the water column and on the seafloor. Important coastal ecosystems associated with the Gulf of Mexico include extensive wetlands and estuaries. Wetlands provide habitat for a great number and diversity of invertebrates, fish, reptiles, birds and mammals, and are important nursery grounds for many species of fish.

The commercial fishery resources of Texas and Louisiana are of national importance, and the Gulf of Mexico provides almost 20 percent of commercial fish landings in the United States (MMS, 1993). Marine recreational fishing in the Gulf of Mexico accounts for an estimated $769 million in sales and employment for over 15,000 people (Sports Fishing Institute, 1987 as cited in MMS, 1993). Recreational fishing takes place from shore or within state waters, as well as offshore from private or charter boats.

Sportfishing in Louisiana and Texas is concentrated around oil and gas structures. Ditton and Auyong (1984) found heavy use of offshore platforms by private recreational fishing boats. Most private recreational boats were bottom fishing, with snapper and seatrout reported most frequently as the major target species. Croaker was reported as a major target species in the Delta Region.

Platform Communities

A description of the communities associated with coastal and offshore platforms in the Gulf of Mexico is needed to complete the problem formulation phase of an ecological risk assessment and describe the environment at risk. Community descriptions are also required to identify assessment endpoints.

Oil and gas platform structures are colonized by microorganisms, algae, and sessile invertebrates that live attached to the structure and form the biofouling mat. These organisms provide food and habitat for many motile invertebrates and small fishes that live in close association with the biofouling mat. There is also a diverse assemblage of demersal and pelagic fishes associated with the platforms, some of which are residents. The composition of the biofouling community and assemblage of demersal and pelagic fishes associated with platforms varies with distance from shore, water depth, latitude and age of the platform (Gallaway and Lewbel, 1982).

Endangered Species And Sensitive Ecosystems

Descriptions of the major ecosystems and biota potentially at risk will support the problem formulation step in an ecological risk assessment. Endangered species and sensitive ecosystems represent unique social values and should be considered in identifying assessment endpoints.

Seven species of baleen whales have been reported in the Gulf of Mexico, but are rare: the northern right whale, blue whale, fin whale, sei whale, Bryde's whale, minke whale and the humpback whale. Five of these species are listed as endangered. Twenty-five species of toothed whales and dolphins have been reported in the Gulf of Mexico. The sperm whale is the only one of the toothed whales and dolphins listed as endangered.

Endangered and threatened species of coastal and marine birds potentially impacted by produced water discharges include the brown pelican, bald eagle, arctic peregrine falcon, piping plover and the whooping crane. One species of fish listed as threatened is potentially affected by produced water discharges -- the Gulf Sturgeon (a subspecies of the Atlantic sturgeon).

Five species of marine turtles occur in the Gulf of Mexico, and all are listed as threatened or endangered (loggerhead turtle, green turtle, leatherback, hawksbill and Kemp's ridley).

Unique and sensitive biological resources of the Gulf of Mexico include coastal wetlands, the pinnacle trend live-bottom features, topographic features inhabited by hard-bottom benthic communities, and deep water chemosynthetic benthic communities.

Chemical/Physical Characterization Of Produced Water

Data describing contaminant concentrations and discharge rates of produced water are needed to formulate the problem, identify potential impacts and describe the source term for an ecological risk assessment. Data presented here are limited and additional data derived from permit files and other sources will be needed in a site or area specific assessment.

Produced waters usually have high total dissolved solids (salinity) and total organic carbon, and are low in dissolved oxygen. Other components of potential concern include heavy metals, dissolved and dispersed petroleum hydrocarbons, various treatment chemicals and radionuclides. Contaminants and contaminant concentrations in produced water vary widely, because the characteristics of the saline water and oil in the formation varies and because treatment methods and efficiencies vary over time and space.

Concentrations of metal discharges in produced waters vary widely. Metals that have been measured in produced waters at concentrations greater than seawater include aluminum, barium, beryllium, cadmium, chromium, copper, iron, lead, manganese, mercury, nickel, silver vanadium and zinc (MMS, 1993).

Produced waters contain petroleum components, with volatile and soluble acid-extractable components present in higher concentrations than the heavier components (PAHs) (Middleditch, 1984).

Produced Water Toxicity

Studies of the toxicity of produced water discharges can be used to assess potential effects on organisms. Problems with using these data include the importance of biocides in causing toxicity, and the change in toxicity that occurs with time and space. Site-specific toxicity tests are preferable to data derived from other sources.

Results of produced water bioassays conducted in the laboratory range from providing evidence of very low toxicities (Middleditch, 1984), to evidence that produced water was highly toxic (Federal Register, 1992). This could be due to differences in the toxicity of the produced water, problems with protocols used in testing, or the presence of biocides in some discharges.

The largest produced water toxicity data base used in permitting applications consists of self-monitoring compliance data required by Louisiana Department of Environmental Quality (LDEQ) discharge permits (Avanti Corporation, 1993). LC_{50}s for mysids ranged from 0.05% to >100% effluent, with a mean 96 hour LC_{50} of 12.1%. LC_{50}s for sheepshead minnow ranged from 1.17% to >100%, with a mean of 27.4%.

Bioaccumulation Of Major Toxic Components Of Produced Waters

Data describing the bioaccumulation of produced water components can be used to formulate the problem, identify contaminants of potential concern and assess exposure. They can also be used to assess effects, although data relating body burdens to effects are limited. The (bioaccumulation factor) BAF approach is commonly used in both human health and ecological risk assessments, but its reliability may be questionable.

Measurements on field-collected specimens are the preferred method for estimating site-specific bioaccumulation, but they may be fiscally prohibitive (Lee, 1992). There is limited information on bioaccumulation of specific contaminants from produced waters in marine and estuarine organisms in the Gulf of Mexico. Because the data are limited, a BAF modeling approach is often used to estimate concentrations of organics, metals and radionuclides in animals.

Only limited data are available for BAFs for organics and metals in saltwater organisms. Bioaccumulation factors in the literature should be reviewed in the context of their relevance and appropriateness for application to a specific organism and specific circumstance. Generic values are often used in screening-models (Strenge and Peterson, 1989) and for organics may be calculated from octanol-water partition coefficients. Bioaccumulation factors for some contaminants in produced water have been estimated by USEPA (Avanti Corporation, 1993) and others are available in the USEPA AQUIRE database (Russom *et al.,* 1991).

Toxicity of Chemical Components of Produced Water

Data on toxicity of produced water chemical components are needed to support an effects assessment. Limited data are available to describe toxicity to marine organisms native to the Gulf of Mexico, and extrapolating from laboratory studies performed on standard test organisms adds uncertainty to an analysis.

A good place to start in developing the toxicity data needed in an analysis are the USEPA water quality criteria developed to protect saltwater animals as well as other values (USEPA, 1986). Additional data are available in the documents that support these criteria, in the open literature, and in electronic data bases.

USEPA maintains a comprehensive AQUatic toxicity Information REtrieval Database (AQUIRE) that is updated quarterly (Russom *et al.*, 1991). The goal of this data base is to enhance ecological and human health risk assessment processes, by providing comprehensive access to up-to-date available information on aquatic pollutants, including: standardized nomenclature and CAS registry numbers for each chemical; test organism identification by scientific and common names; parameters such as life stage; taxonomic information; test conditions and location; exposure duration and type; water chemistry and chemical analyses; adequacy of controls; effects parameters; and references for each entry.

Another important concern in terms of potential impacts from produced water are effects on animals living on or in the sediment. There are limited data available to describe toxicities to these organisms, but sediment quality criteria have been derived for some contaminants (Long et al., 1995). Limited data also are available to suggest relationships between exposure to contaminants in water and sediment, and genotoxic and histopathologic effects in fish.

Several studies have related particular groups of chemicals to increased incidence of histopathological lesions in fish from saline environments. High

PAH concentrations in sediments were associated with hepatic lesions (Johnson *et al.*, 1993), and biomarkers (Goksøyr *et al.*, 1994) in marine fishes.

Radionuclide Effects

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.

NCRP (1991) reviewed several models for estimating dose to aquatic animals based on concentrations in water. Models described include CRITR (Soldat et al., 1974), EXREM III, and BIORAD (Trubey and Kaye, 1973). IAEA (1976) presents a method for estimating the dose to aquatic organisms for radionuclides in water, sediment and accumulated in tissue.

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

Exposure to ionizing radiation can result in injury at the molecular, cellular and whole body levels. Most 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.

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

IAEA (1988) came to similar conclusions, but expressed their reference levels in terms of dose equivalent rather than absorbed dose. 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).

Effects On Benthic Communities

Site-specific assessments of benthic effects from produced water discharges may have limited application for assessments at other sites. However, available studies do provide estimates of threshold levels for effects useful in the development of the problem formulation phase of an ecological risk assessment for produced water discharges.

Effects of platforms and platform discharges on benthic communities can vary from reductions in populations and diversity to increases in populations and diversity. This variation includes an increase in diversity at the expense of preexisting species, or an increase in numbers of particular species while total diversity declines. Some of these results are attributable to the introduction of a new structure (i.e., the platform), as well as the accumulation of contaminants in sediments. In the latter case relatively low levels of contaminants may increase populations (hormesis) of opportunistic species that can either metabolically adapt to the contaminants, or compensate for toxic effects on particular life-stages by increases of more resistant stages in the populations.

A number of studies have shown differences in benthic communities with distance from platforms discharging produced water. This was particularly true for coastal sites, in contrast to offshore areas. Although some studies found correlations between the number of species and individuals and the chemical constituents in the near bottom waters or surficial sediments, the findings were site specific and not consistent across all studies. Some studies found little or no disturbance in the benthic communities. Results from studies which found a correlation would be difficult to use for predicting effects, because it is not clear whether the effect was due to periodic contact with toxic substrates in the produced water, substrate disturbance due to currents eddying around the platform leg and removing the substrate, or some other confounding factor (Harper et al., 1981).

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1 INTRODUCTION

1.1 Problem

Oil and gas production is often accompanied by the production of a saline wastewater, called produced water. In offshore and coastal areas, this wastewater may be discharged to surface water. Produced water may contain a number of contaminants, including oil and grease, organics, heavy metals and radionuclides.

Environmental impacts associated with discharges of produced water are of concern to regulators at the state and federal levels, the public, environmental interest groups and industry. Most of the current (and projected future) oil and gas platforms in the U.S. are located in the central and western Gulf of Mexico (Louisiana and Texas; Figures 1-1, 1-2). This area supports economically important commercial and recreational fisheries, as well as unique, socially valued ecosystems and several endangered and threatened species.

Most of the contaminants discharged in produced water occur naturally in the geologic reservoir along with the oil and gas. Many of the contaminants are toxic to marine organisms at high concentrations. Some produced waters also contain added biocides or other treatment chemicals that may be toxic to aquatic organisms.

In offshore 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, and effects on marine life are likely to be minimal. In shallower, coastal environments, contaminants have been 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, 1989a; Gallaway *et al.,* 1981a).

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 United States Department of Energy (USDOE). These assessments will provide input to regulators in the development of guidelines and permits, and to industry in the development and application of appropriate discharge practices.

Figure 1-1. Offshore oil and gas platforms in the Gulf of Mexico.

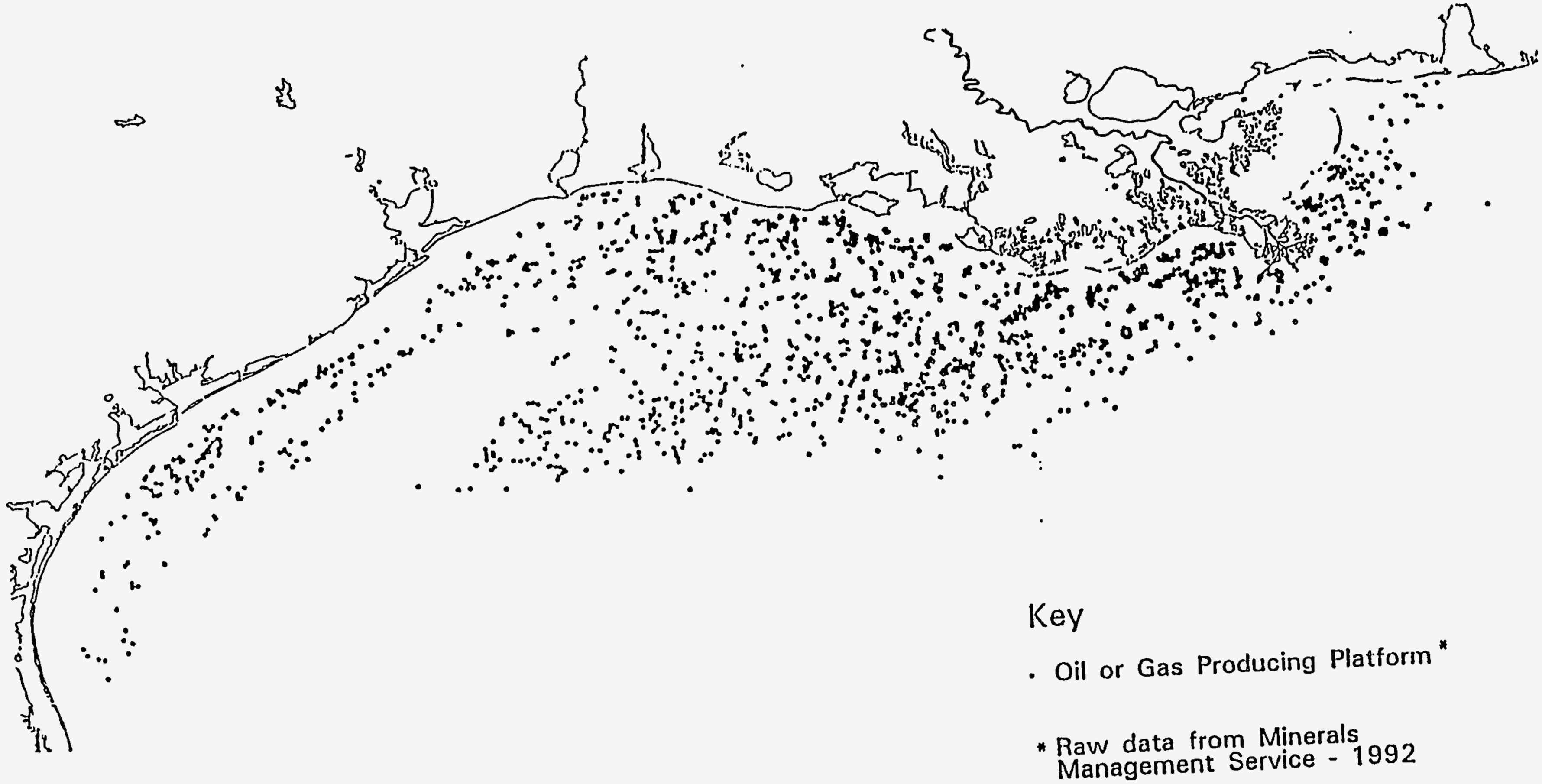


Figure 1-2. Coastal produced water discharges in the Gulf of Mexico.

1.2 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 more quantitative risk-based approach to decision-making for environmental protection.

"Risk assessment can be defined as the process of assigning magnitudes and probabilities to the adverse effects of human activity or natural catastrophes" (Suter, 1993a). Risk management is the process of decision-making concerning risks -- environmental regulation is a form of risk management. 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.

Risk assessment is useful in environmental decision-making because it (Suter, 1993a):

- Provides a quantitative basis for comparing and prioritizing risks;
- Provides a systematic means of improving the understanding of risks;
- Acknowledges the inherent uncertainties in predicting future environmental states;
- Estimates clear and consistent endpoints; and
- Clearly separates the scientific process of estimating the risks (risk assessment) from the process of choosing among alternatives and determining the acceptability of risk (risk management).

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

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.

Human health risks are described as the probability of an adverse health effect (e.g., cancer death or toxic effect) occurring in an individual in an exposed population, or the number of health effects expected in the population (individual and population risk). A major characteristic of a risk analysis is that risks are described in terms of probabilities of effects, and uncertainties are explicitly considered in both the analysis and the expression of its result.

With some modifications and additional uncertainties, this framework can be applied to ecological assessments. 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) has proposed a framework for ecological risk assessment that includes three phases:

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

1.3 This Report

This report presents a summary review of the data available to support both traditional qualitative assessments and more quantitative ecological risk assessments of produced water impacts. The emphasis is on data relating to produced water discharges in the central and western Gulf of Mexico, especially in Louisiana. Much of the summarized data and cited literature is relevant to assessments of impacts in other regions. Data describing effects on marine and estuarine fishes, mollusks, crustaceans and benthic invertebrates are emphasized. Data for macroalgae, zooplankton and phytoplankton, and basic ecological processes are not as readily available or as easily applied in an assessment.

This report does not present an impact analysis or ecological risk assessment. Environmental assessments relating to subsets of produced water discharges in the Gulf of Mexico (e.g., discharges in open Louisiana bays, offshore discharges) are being developed using the data and information presented here, combined with data collected in a field study conducted by the USDOE and results of environmental transport and exposure modeling.

Not all data or models needed in a specific ecological risk assessment are presented here. The intent of this report is to summarize data available for the Gulf of Mexico, review more generic data, describe and document additional data sources, and introduce the concepts and methods of ecological risk assessment.

Section 2 reviews important ecological risk assessment concepts and methods. Sections 3 through 5 describe the important biological resources in the Gulf of Mexico of potential concern for produced water impacts. These resources include fish and shellfish resources, platform communities, endangered and threatened species, and sensitive and unique ecosystems.

Sections 6 through 11 summarize data available to estimate exposure and effects of produced water discharges including:

- Produced water characteristics;
- Toxicity studies of produced waters;
- Bioaccumulation of produced water components;
- Toxicity studies of produced water components and USEPA water quality criteria;
- Sediment chemical toxicity and sediment quality criteria;
- Radionuclide effects; and
- Field studies of effects on benthic communities.

2 ECOLOGICAL RISK ASSESSMENT

2.1 Introduction

Ecological risk assessment is a relatively new field, but regulatory requirements and the desire to improve the bases for environmental decision-making have accelerated its development. Several books summarizing various approaches to risk assessment are available (Suter, 1993b; Bartell *et al.,* 1992; Calbrese and Baldwin, 1993), and USEPA has recently developed a framework for ecological risk assessment (USEPA, 1992).

As used here, "ecological risk assessment" is the study of risks to the natural environment. The objective of ecological risk assessment is to use available toxicological data to estimate the probability of some specific effect on individual organisms, natural populations, communities or ecosystems. An important feature of risk assessment is "the explicit, quantitative consideration of uncertainties in the analysis and the expression of the final estimated effects as a probability" (Bartell *et al.,* 1992). This quantitative result is the goal of an ecological risk assessment, but in practice assessments are often deterministic or qualitative (USEPA, 1992).

2.2 Framework for Ecological Risk Assessment

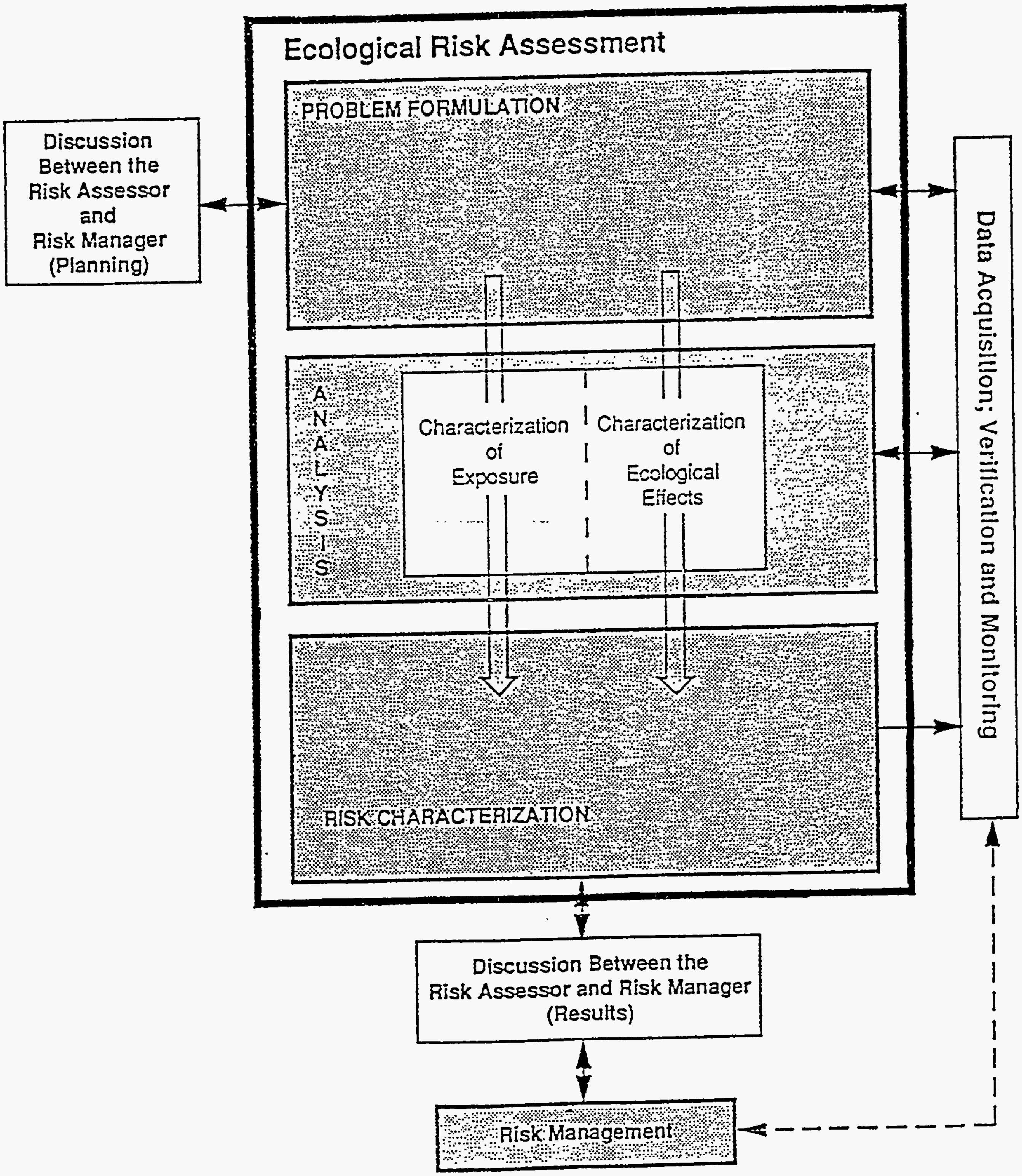
Several similar frameworks for ecological risk assessment have been suggested (Suter, 1993c; USEPA, 1992). All of these frameworks are derived from the human health risk assessment paradigm suggested by the National Research Council (NRC, 1983). USEPA (1992; 1994) developed a framework for ecological risk assessment composed of three phases: problem formulation, analysis, and risk characterization.

Figure 2-1 outlines the steps in an ecological risk assessment as described by USEPA (1992). These steps are described in more detail below (summarized from USEPA, 1992; Suter, 1993c).

2.2.1 Problem Formulation

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. These steps are not independent of one another, and the decisions and information gathered in each step may influence the others.

Figure 2-1. Framework for ecological risk assessment (from USEPA, 1992).



Planning

This step in the problem formulation phase involves discussion with risk managers to ensure that risk assessments will be relevant to regulatory needs and public concerns. Risk assessment and risk management are separate processes, but interaction with regulators and other stakeholders at this early stage in the process will support the development of relevant assessments.

Stressor Characteristics

Identification of stressor characteristics includes identifying the potential chemical and physical stressors of concern. In this step, source terms are described in terms of concentration or magnitude, duration, frequency, and spatial scale. Source terms may be estimated directly from emissions data, or derived using transport models and monitoring data.

Ecosystem Potentially at Risk

This step involves describing the ecosystem potentially at risk from the identified stressor(s). Properties of the ecosystem to be considered include physical aspects of the environment, and ecosystem structure and function.

Ecological Effects

This step is a preliminary assessment of available data on ecological effects to help focus the assessment on important stressors and ecosystem components.

Endpoint Selection

Selection of endpoints is a critical step in an ecological risk assessment. An assessment endpoint is a formal expression of the environmental values to be protected (Suter and Barnthouse, 1993). Selection of assessment endpoints includes identifying the valued components of the environment that are at risk, and developing an operational definition of effects.

USEPA recommends that this selection consider ecological relevance, policy goals and societal values, and susceptibility to the stressor (USEPA, 1992). Suter and Barnthouse (1993) make similar recommendations and give five criteria for an assessment endpoint: societal relevance, biological relevance, unambiguous operational definition, accessibility to prediction and measurement, and susceptibility to the hazardous agent.

There is a distinction between assessment endpoints and measurement endpoints. Measurement endpoints are an expression of the results of toxicity tests or field monitoring studies. Assessment endpoints refer to effects on populations or ecosystems -- things that usually cannot be measured and must be derived by extrapolation from measurement endpoints. Measurement endpoints should be selected on the basis of how well they represent the assessment endpoint.

Conceptual Model

This is the development of the planned assessment approach. The conceptual model "(1) describes how a given stressor might affect the ecological components in the environment; (2) describes the relationships among the assessment and measurement endpoints, the data required, and the methodologies that will be used to analyze the data and (3) summarizes the steps that will be taken to ensure that laboratory or field data collected for the assessment will be sufficient to achieve the intended objectives" (Barnthouse and Brown, 1994). In the conceptual model, possible exposure scenarios are described, and a series of hypotheses are developed about how the stressor might affect the ecosystem at risk.

2.2.2 Analysis Phase -- Exposure Assessment

In the 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. Much of the effort in an exposure assessment involves using models to estimate concentrations of contaminants in various media. Estimates of exposure or dose to the endpoint organisms or systems are then developed. The rates of exposure derived in the exposure assessment must be provided in units compatible with the dose-response data developed in the effects assessment. Methods and data used in exposure assessment are described in more detail in section 2.3.

2.2.3 Analysis Phase -- Effects Assessment

The effects assessment determines the relationship between exposure to the contaminant and effects on the measurement endpoint. The effects assessment is usually based on the results of toxicity studies. These results are extrapolated to relate the effects on individual organisms to effects on populations, communities and ecosystems. Most effects data for ecological risk assessment are in terms of external exposure in air, soil or water. For mammals and birds, intake and uptake may be used, and internal dose or body burdens estimated. Section 2.4 describes data and statistical extrapolation approaches that may be used in an effects assessment.

2.2.4 Risk Characterization

The risk characterization phase 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. Risk characterization also involves describing the analysis to the risk manager, and discussing the uncertainties associated with the analysis and their implication for the results. Approaches to risk characterization are discussed in section 2.5.

2.2.5 Risk Management

Risk management decisions are made based on the results of the ecological risk assessment. Risk management uses results of the risk assessment to support decisions relating to acceptable risks from environmental discharges. Regulatory agencies involved in standard setting and regulation development are engaged in risk management. The goal should be to minimize risks without undermining other societal values. 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.

2.3 Methods and Data for Exposure Assessment

Exposure assessment comprises two major efforts: an analysis of the transport and fate of the stressor in the environment; and quantification of the exposure of the identified receptor(s) to the contaminant.

2.3.1 Transport and Fate

This part of exposure assessment involves translating a source term into estimates of concentration in environmental media. Models are used to simulate transport, dilution, transformation, degradation and partitioning between media, and to generate predictions of the temporal and spatial dynamics of concentration (Suter, 1993c). The input data to these transport models include release rates, chemical and physical characteristics of the pollutant, and characteristics of the receiving environments.

Mackay and Paterson (1993) and Suter et al. (1994) discuss the application of transport models to ecological risk assessment. Models used in human health risk assessment and in assessments for regulatory purposes are applicable. Compilations of model codes are available from the USEPA Center for Exposure Modeling (CEAM; Bouchard et al., 1995), and other reviews of available codes have been published (Moskowitz et al., 1995).

Two models of specific interest to the modeling of produced water discharges in the Gulf of Mexico include the Offshore Operators Committee Model (OOC) and the USEPA Cornell Mixing Zone Expert System model (CORMIX).

The OOC model was developed by the Offshore Operators Committee (OOC) and Exxon Production Research Company to predict the initial fate of drilling mud and cuttings to the marine environment (Brandsma and Sauer, 1983a, 1983b; O'Reilly *et al.,* 1988; Brandsma *et al.,* 1992). The model was modified to allow prediction of the initial dynamics and passive diffusion of produced waters.

The OOC model simulates the descent of a jet of discharged material through the water column, dynamic collapse as the material spreads out on the bottom or within the water column, and passive diffusion. Currents can be variable in three dimensions, density profiles can change with time, and the model can incorporate variable depths and land boundaries. The OOC model has been validated by comparing model predictions to laboratory and field observations (O'Reilly *et al.,* 1988; Brandsma *et al.,* 1992).

The CORMIX model (Doneker and Jirka, 1990) 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 greater 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.

2.3.2 Quantification of Exposure

Exposure defines the contact between the receptors and the contaminant in the environment. Factors considered in estimating exposure include receptor behavior and bioavailability of the contaminant.

2.3.2.1 Estimation of Dose

The calculation of dose or effective concentration is the most common approach to estimating exposure. Estimated environmental concentrations are combined with assumptions or parameters that describe the receptor's contact with environmental media (Suter *et al*., 1994).

Exposure of aquatic organisms to contaminants in water, and exposure of terrestrial organisms to contaminants in respired air, is usually estimated by assuming that the contaminants are well mixed and that the organism is exposed to a representative concentration. Exposure of terrestrial organisms through ingestion in food, water and soil is estimated by combining ingestion rates with estimates of concentrations in environmental media. USEPA (1993a) tabulated parameters needed for an exposure assessment for several birds and mammals. Methods are available to estimate the radiation dose to animals from both external and internal exposure (IAEA, 1979; NCRP, 1991; see section 10).

2.3.2.2 Internal Exposure

Biomarkers and Body Burdens

Most ecological risk assessments describe exposure in terms of external exposure, or for large terrestrial organisms in terms of intake. In many cases, estimates of internal exposure would be more appropriate. Quantification of internal exposure can be estimated through measurement of biomarkers and body burdens.

Biomarkers measure biochemical or physical changes in an exposed organism. Biomarkers currently have only limited application to ecological risk assessment because of the lack of data relating biomarker measurements to effects (Suter *et al.*, 1994).

The most common approach to estimating internal exposure is through the measurement (or estimation through modeling) of body burdens. This approach works well for exposure to radiation in the environment, because methods are available to estimate dose and related effects (see section 10). For other contaminants the use of body burdens is problematic because there are few data available to relate internal exposure to effects.

Field data, describing concentrations of contaminants in organisms, are usually limited because of the effort and expense involved in sampling biota. A modeling approach is commonly used to estimate the concentration of contaminants in organisms from concentrations in environmental media. The approach used for aquatic organisms is called the bioaccumulation factor approach.

Bioaccumulation Factor

A bioaccumulation factor (BAF) is useful for estimating tissue concentrations, when the alternative of direct tissue measurements is not available (Lee, 1992). Choice of BAF values are dependent on their biological relevance to potential toxicity. USEPA (1989a) defines a bioconcentration factor (BCF) as the ratio of concentration of a contaminant within an organism to the concentration in water. Practically, BCF is usually based on exposure of an aquatic organism in the laboratory to water containing a contaminant. A BAF is the accumulation of a contaminant in an organism from all sources of exposure, including the ambient medium and trophic considerations. In addition to measurements in the field, BAFs are also estimated by calculations that combine BCFs with factors for trophic levels and biomagnification.

BCFs and BAFs are highly uncertain parameters, depending on time of exposure, species, and ambient chemical and physical conditions. Influences on bioaccumulation that are difficult to quantify include intraspecific and interspecies variability, conditioning factors and developmental stages (Franke *et al.,* 1994). Trophic factors add another layer of uncertainty to calculated BAFs.

Although site- and organism-specific factors are desirable, they can be difficult and expensive to obtain. Use of a generic BAF or BCF assumes a steady-state linear relationship between ambient concentrations of a contaminant (C_{amb}) and concentration in an organism (C_{org}), ignoring uncertainty arising from these influences.

$$BAF = \frac{C_{org}}{C_{amb}} \tag{2.1}$$

The linear relationship between C_{org}/C_{amb} and BAF is usually assumed to be independent of the concentration in the environment. In general, this is valid only for relatively small environmental concentrations. Bioaccumulation factors are often calculated as the geometric mean of a set of bioaccumulation factors in similar organisms that were tested in a specific environment. The BAF approach can produce severe errors if this assumption is not correct. These assumptions, allow the use of generic concentration factors in assessment models. The assumptions may not always be justified, and generic factors can only be used in a preliminary assessment as a first order estimate of bioaccumulation.

In the absence of site-specific BAFs, USEPA recommends that BCF values be obtained in the laboratory, with careful consideration of the problems described above (USEPA, 1989a). A Limited number of BAF values are available for organics, metals and radionuclides (section 8.3).

2.4 Methods and Data for Effects Assessment

2.4.1 Toxicity Testing

By using different methods, periods of exposure, life stages and species; bioassays provide toxicological information on a variety of endpoints for different levels of biological organization. Acute tests usually use mortality as an endpoint, while chronic tests include sublethal endpoints, such as growth, development, reproduction and behavior. When combined with other studies they provide opportunities to determine cause and effect relationships.

Most standard testing is done in the laboratory, for exposures in water. Standard tests for contaminants associated with sediment have also been developed. Most toxicity studies are performed on individual animals, but there are also methods available to test effects on populations and ecosystems.

Organism Level Tests: Exposure in Water

Results of toxicity tests on individual animals are usually the basis for the effects assessment. These can be used directly, but more often must be modified to account for differences in response of different species, or to extrapolate to a higher level of organization. Most toxicity tests on individual organisms are performed in the laboratory, although field tests are also possible.

Two types of tests are standard for aquatic organisms: acute and chronic. The standard acute endpoint is the 96-hr or 48-hr median lethal concentration LC_{50} (USEPA, 1982; ASTM, 1991). Dose-response functions are derived from the data collected in these toxicity tests, and the LC_{50} value estimated from the function. Models are used both to calculate the single value LC_{50} endpoint and to describe the dynamics of the dose-response function.

Commonly used functions to analyze acute toxicity data are the S-shaped probit and the logit function. These functions assume no threshold and a dichotomous response (i.e. mortality) (Suter, 1993d). Continuous responses and nondichotomous responses can also be fit to these functions by representing the response as a proportion of the control response. For a more detailed description of methods, see Suter (1993d), Stephan (1977), and Kooijman (1983a, 1983b). In the absence of functions relating concentration to response, linear approximations can be constructed by using the origin and the acute toxicity benchmark (LC_{50}) to define a line for interpolating the expected response in relation to an exposure concentration (Bartell *et al.,* 1992).

Problems with LC_{50} tests include the fact that they do not protect early life stages, would allow mass mortality of late stages, and that in most cases only the 96-hour response is reported (Suter, 1993d).

The standard chronic endpoint has been the maximum acceptable toxic concentration (MATC) or "chronic value" which is the threshold for statistically significant effects on survival, growth or reproduction. The MATC is the geometric mean of the lowest concentration producing a statistically significant effect (LOEC, lowest observed effect concentration) and the highest concentration producing no effect on survival, growth or fecundity (NOEC, no observed effect concentration). Because the MATC is derived using hypothesis testing it is not of great value in estimating ecological effects (Suter, 1993c).

Another standard chronic endpoint is the EC_{50} (median effective concentration) for effects on growth, reproduction and development. The EC_{50} is estimated from a dose-response function fit to data from chronic toxicity tests.

Fish Standard tests for fish are acute lethality tests, life cycle tests and early life stage tests. Conditions may be flow-through, static, or static renewal (periodic renewal of test solution). The standard acute endpoint for fish is the 96-hr median lethal concentration LC_{50} (USEPA, 1982; ASTM, 1991). Chronic toxicity tests for fish include full life cycle tests, partial life cycle tests, early life stage tests, and five to eight day chronic tests involving eggs and or larvae. The standard chronic endpoint has been the MATC or "chronic value", and the EC_{50} is also in common use.

Aquatic Invertebrates The most common salt water test invertebrates are the shrimp *Penaeus duorarum* and the mysid crustacean *Mysidopsis bahia*. These animals are commonly used in 96-hr LC_{50} tests, and 28-d tests of mortality, reproduction and growth. A variety of other aquatic invertebrates are used in 48-h or 96-h lethality tests, including annelids (*Neanthes arenaceodentata*) (Suter, 1993d).

Mammals Few data are available for assessing the toxic effects of contaminants on mammals. The most common test endpoint available for assessing effects on mammals is the acute, oral, median lethal dose (LD_{50}) for laboratory rodents (Suter, 1993d). LD_{50}s for domestic mice (*Mus musculus*) and rats (*Ratus ratus*) are the most common endpoints. No wild animal species is used in routine testing. Testing of birds is based on acute LD_{50}s for adults as well as subacute lethal dietary toxicities for young birds.

Organism Level Tests: Exposure in Sediment

Sediment toxicity tests directly describe the interactive effects of both measured and unmeasured chemicals in field-collected sediment samples. The tests also account for the influence of biotic and abiotic factors in sediments. Tests are performed on whole sediment, suspended sediment, pore or interstitial water, and sediment extracts (*e.g.*, aqueous and organic solvent). Sediment tests can be performed on field samples to assess the toxicity of chemicals bound to the sediments or in the pore water, or sediment samples can be treated with specific chemicals or chemical mixtures.

Nonvertebrates are the preferred organisms for testing sediments, providing logistical advantages for assaying sediment toxicity. Invertebrate infauna are the preferred eukaryotes because they have the most contact with sediments. Unfortunately, the Gulf of Mexico is one of the geographic regions that are not represented by the species routinely used in sediment toxicity tests. Sets of bioassay organisms have been recommended for sediments from aquatic environments of interest (E.V.S. Consultants, 1990):

- Marine waters:
  amphipod, *Rhepoxynius abronius* ;
  bivalve larvae;
  and Microtox® (bacterial luminescence)

- Estuarine/brackish waters:
  amphipods *Hyalella azteca, R. abronius* ;
  and Microtox®

Amphipods, such as *R. abronius* and *H. azteca*, are readily available sediment dwellers widely used for acute lethal bioassays. Amphipod sensitivity to sediment pollution is well characterized. They have been described as the first organisms to disappear from benthic communities in contaminated sediments (E.V.S. Consultants, 1990; Lamberson, DeWitt and Swartz, 1992). Amphipods are a major food source for bottomfish (Franz and Tanacredi, 1992). Testing with these animals requires a standardized interstitial salinity of 25 ppt. *R. abronius* are unsuitable for sediment samples less than 15 ppt salinity, and salinities from 15 to 24 ppt have to be adjusted to 25 ppt. *H. azteca*, a freshwater amphipod, should be used at salinites of 15 ppt or less. Positive and negative controls are required for toxicants, because of varying sensitivity of life stages and field populations, and sensitivity of the organisms to sediment-sample grain size. Field collected test animals have to be accurately identified by a qualified taxonomist. *H. azteca* has an advantage because it can be cultured.

Bioassays with bivalve larvae are sensitive sublethal 48-hr tests that generally use oysters (*Crassostrea gigas*) or mussels (*Mytilus edulus*). The tests measure normal or abnormal development of fertilized ova to free swimming larvae. These bivalves generally do not reside in the types of sediments to be characterized. Therefore the tests are highly sensitive general indicators of toxicity, rather than of ecological significance. Testing is limited by seasonal spawning characteristics of the species, and a lower salinity limit of 10 ppt. Positive and negative controls are required for population-specific and seasonal influences on sensitivity to toxicants.

Microtox® assays measure the degree of inhibition of light emission from the bacterium *Photobacterium phosphoreum* by aqueous contaminants. It is a measure of the metabolic condition of the organism. This bioassay has been adapted successfully from freshwater testing to marine and estuarine/brackish waters. It is a well-documented, simple, sensitive bioassay, useful for toxicity screening rather than determining ecological significance. This assay depends on extraction procedures, and should be used to test sediments for both water-soluble chemicals in aqueous extracts and chemicals extracted by other means. This test can show responses to naturally-occurring chemicals as well as anthropogenic contaminants in sediments, and sediments from areas considered to be uncontaminated can produce positive responses.

Population/Ecosystem Level Tests

Population and ecosystem level testing are not as well standardized or as widely used as the tests for effects on individual organisms. Two major kinds of tests are in use: microcosms and mesocosms.

Microcosms are laboratory systems that physically simulate an ecosystem or subsystem. Microcosms may be assembled from a standard set of species (assembled microcosms) or created from natural populations removed from the environment (excised microcosms). Standard protocols are available for the standard aquatic microcosm (an assembled microcosm; Federal Register, 1987; ASTM, 1991), and for three excised microcosm tests: mixed flask culture, pond microcosm, and site specific aquatic microcosm (Federal Register, 1987; ASTM, 1991; Suter and Bartell, 1993).

Microcosm endpoints include organism-level parameters, abundance of component organisms, community parameters such as number of species and diversity indices, and parameters that describe ecosystem function (Suter and Bartell, 1993).

Mesocosms are outdoor experimental systems that are to some extent enclosed. Mesocosms are more realistic than microcosms, but are more expensive and less standardized. They may include assembled mesocosms, or delimited portions of natural ecosystems. Assembled mesocosms in use include artificial ponds and streams. Delimited mesocosms include plastic bags and plastic cylinders called lymnocorrals used to enclose portions of a natural ecosystem.

2.4.2 Effects Extrapolation

Statistical extrapolation models are used to estimate a toxic effect of interest from a measured effect in another species, life stage or test type. Models have been used to incorporate taxonomic differences, difference in life stage and size, mode of exposure, severity and proportion responding. The following summary is abstracted from Suter (1993d).

Specific Taxonomic Extrapolations

Specific taxonomic extrapolations account for the difference between two species, and allow data from one species to predict responses in another. One approach to specific extrapolation is to use the ratio of the responses of two species to other chemicals for which they have both been tested (Schaefer et al., 1983). Regression analysis has also been used to relate the response of one species to that of another (Sloff et al., 1986; Mayer et al., 1987). This approach has limited application because many of the species of interest in an ecological risk assessment are not standard test species.

Suter *et al.,* (1983) and Suter and Rosen (1988) used the taxonomic relationships between test animals and species of interest to estimate the response for species that have not been tested. Regressions were performed between all pairs of species that occur in a common genus, all pairs of genera within a family, families within orders, etc. Extrapolations are made between taxa having the next higher level in common. This approach is based on the assumption that similarity of response is related to taxonomic similarity. Extrapolations between taxa within the same family can be made with fair certainty, but extrapolations between orders, classes or phyla are highly uncertain (Suter, 1993d).

The response of a whole taxon can be predicted on the basis of the response of a test species using regression analysis. Suter et al., (1987) regressed toxicity data for three standard test species (fathead minnow, bluegill, rainbow trout) against all other species that had been tested for a given chemical. 95% of the time, toxicity for other species fell within ±1.31, ±1.37 and ±1.20 of a log unit of the regression line for fathead minnow, bluegill and rainbow trout respectively. Holocombe et al., (1988) used the same technique and found that most species fell within ±1 order of magnitude of the regression line.

Extrapolations can also be done on the basis of allometric regression for dose scaling, natural history and guild theory information, and biochemical traits (Suter 1993d).

Generic Taxonomic Extrapolations

Generic extrapolations use data from one or several species to estimate the sensitivity distribution of the members of the community at risk. The usual approach is to try to identify sensitive species in a community, with the assumption that endpoints for sensitive species will protect the entire community.

In general, it has been found that arthropods are more sensitive than fish, and that fish are more sensitive than amphibian larvae. Salmonids are the most sensitive fish. Mysid and peneid shrimp have been found to be the most sensitive marine organisms (Suter and Rosen, 1988). No one species or set of species can be assumed to be consistently the most sensitive.

Observed ranges of species sensitivity can be used as correction factors (Calabrese and Baldwin, 1993). This approach assumes that the relative sensitivity of the test species and the most sensitive species in the community is equal to the most and least sensitive species in the data set (Suter, 1993d).

Another approach is to assume that the sensitivity of species follows some probability distribution, and to define the concentration that affects the most sensitive species as the lower Xth percentile (e.g. lower 5th percentile; Suter, 1993d).

Acute to Chronic Extrapolations

In aquatic toxicology, the terms acute and chronic are used to describe both the severity of the effect, and the duration of the exposure; confounding the need to estimate low severity, chronic effects from severe effects. Acute exposures are assumed to be both of shorter duration and result in a more severe effect (mortality) than do chronic exposures.

Many more acute than chronic toxicity tests have been performed for aquatic animals, and chronic endpoints have been estimated from the results of acute toxicity tests.

One approach is the use of an application factor (AF) which is the ratio of the chronic threshold concentration to the acute LC_{50}. USEPA uses the acute to chronic ratio in deriving water quality criteria (USEPA, 1986). Application factors can be derived across all species and chemicals or can be derived for specific chemicals (Suter 1993d).

Another approach is to regress chronic threshold values against acute LC_{50} values. This approach is useful for some chemicals because the AF decreases with an increase in the LC_{50}. Suter et al., (1983, 1986) derived acute/chronic equations for fish.

2.5 Methods for Risk Characterization

Approaches to risk characterization in ecological assessment were reviewed by Suter (1993b) and Wiegart and Bartell (1994). Major approaches to risk characterization include comparing effects and exposure data (single values or distributions) and the use of mechanistic models to estimate effects on populations and ecosystems.

2.5.1 Compare Exposure and Effects Values

Quotient Method

A common approach to ecological risk characterization is the quotient method (USEPA, 1992; Suter 1993c; Calabrese and Baldwin, 1995). The quotient method is a ratio of an exposure concentration to an effect value. Uncertainty or safety factors are used to adjust the effect value. If the quotient is one or more, an adverse effect is considered likely to occur. Quotients are most useful for screening purposes (Wiegart and Bartell, 1994).

Comparing Distributions

This approach compares distributions of exposures and effects. Risk is quantified by the degree of overlap between the two distributions. The use of distributions recognizes the variability in exposure in space and time and the natural variability in response of individuals and populations. Approaches to risk characterization based on comparison of distributions of exposure and response are described in detail in Suter (1993c).

2.5.2 Population and Ecosystem Level Effects

Barnthouse (1993) and Suter and Bartell (1993) reviewed methods and models for assessing population and ecosystem effects. These reviews are summarized here. Few example applications of these approaches are available in the literature.

Population Models and Approaches

Models for assessing risk to wildlife should address endpoints of regulatory relevance, easily incorporate toxicological information, use available population data, and be linkable to models of chemical exposure (Emlen, 1989). Potential endpoints at the population level include alteration of mean population densities or biomass, alteration of the age or size distribution of the population, and probability of extinction.

Approaches to assessment of risks at the population level should consider the potential influence of life history and density dependence on sensitivity to stress from exposure to toxicants.

In general, long-lived vertebrates such as large mammals and predatory birds are more sensitive to mortality imposed on adults than are short-lived, highly fecund species (Barnthouse, 1993). Short-lived species are often more vulnerable to short-term stresses that affect critical life stages. Most populations exhibit some form of density dependence (Barnthouse, 1993): when population numbers are high, mortality increases and reproduction decreases; and when numbers are low, mortality decreases and reproduction increases. Populations in which survival or reproduction is strongly related to density should be less vulnerable to stress from exposure to toxicants (or exploitation from man) than populations with a low degree of population dependence.

Two major approaches to population analysis are used in models for ecological risk assessment: quantification of reproductive potential, and age-structure projection matrices. These approaches are summarized by Barnthouse (1993).

Toxicity tests can be linked to population models to estimate population-level effects. One approach uses the reproductive potential index to quantify the effect of exposure to a toxicant on changes in mortality and reproduction. Another approach uses matrix-type life cycle models to estimate changes in yield, abundance and risk of extinction.

Barnthouse *et al.* (1987, 1989, 1990) linked toxicity data to fish population models. Two different approaches to population modeling were used. The first used standard survival and reproduction data to calculate an index of reproductive potential (Barnthouse *et al.,* 1987). These indices are used as relative measures of impact, expressed as a fractional reduction in reproductive potential (Barnthouse, 1993).

This approach does not account for natural environmental variability or density dependence. Barnthouse et al. (1990) developed density dependent stochastic matrix models of two well studied populations: the Gulf of Mexico menhaden population and the Chesapeake Bay striped bass population.

Barnthouse *et al.* (1990) quantified population level effects of chronic exposure by coupling standard toxicity tests data to matrix type population models for the Gulf of Mexico menhaden and Chesapeake Bay striped bass populations.

Ecosystem Models

Ecosystem models attempt to represent mathematically the ecological processes and structure of an ecosystem, and can be used to predict adverse ecological effects in ecosystems. Some existing models were developed to assess the ecological effects of toxic materials, while others were designed as basic research tools. Suter and Bartell (1993) summarized existing aquatic ecosystem models that might be used in ecological risk analysis.

Aquatic ecosystem models of particular interest here include the SWACOM (Standard Water Column Model; O'Neil *et al.,* 1982; O'Neil *et al.,* 1983; Bartell *et al.*, 1988a) and the Integrated Fate and Effects Model (IFEM; O'Neil *et al.,* 1982; Bartell *et al.* (1988b) .

The SWACOM model (O'Neil *et al.,* 1982; O'Neil *et al.,* 1983; Bartell *et al.,* 1988a, Bartell et al., 1992) was developed to extrapolate the results of acute toxicity bioassays to probabilistic estimates of specific toxic effects in aquatic systems. The model describes the temporal biomass production of 10 populations of phytoplankton, 5 populations of herbivorous zooplankton, 3 populations of planktivorous fish and a single population of piscivorous fish. Each population is defined by parameters that determine rates of photosynthesis, respiration, feeding, mortality and optimal conditions for growth. The model uses difference equations to simulate daily changes in biomass concentrations. Toxicity data for species representative of the food web populations and an estimated (time invariant) exposure are needed. This model permits an evaluation of the potential higher-order effects of toxic chemicals on system structure and function.

The IFEM model (O'Neil *et al.,* 1982; Bartell *et al.* 1988b) integrates the physiological processes associated with chemical kinetics, with dynamic estimates of chemical fate. The code currently models PAH's only. The model requires toxicity data and kinetic information, and estimates population-specific toxic effects as a function of body burden. The food web consists of single populations of algae, periphyton, macrophytes, bacteria, zooplankton, benthic insects, larger benthic invertebrates, and detritivorous and omnivorous fishes. The model predicts the time varying concentration of toxicant in each model component and the time varying change in population size that results from sublethal effects.

2.6 Uncertainty

The current application of the National Research Council risk assessment paradigm (NRC, 1983) to the estimation of risk to the natural environment requires explicit description of uncertainties in assumptions, models and parameters and incorporation of these uncertainties in 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 potential costs of decisions made based on conservative assumptions (Burmaster et al., 1990; Paustenbach et al., 1991).

A common way to describe uncertainties in risk assessment is to recognize three fundamental sources of uncertainty: the heterogeneity or stochasticity of natural systems, measurement error, and lack of knowledge. Other approaches to describing uncertainty have also been suggested (Smith and Shugart, 1994).

Uncertainties in ecological risk assessments that come from a basic lack of knowledge should be described qualitatively. Other major uncertainties, including natural heterogeneity or stochasticity, and parameter error can be treated analytically. Statistical methods can be used to derive the variance on a parameter estimate by fitting models to data (Suter, 1993c; Suter and Rosen, 1988).

A commonly used tool in 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 run 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).

Uncertainties occur in all phases and steps of an ecological risk assessment. Table 2-1 (from Smith and Shugart, 1994) summarizes some of the uncertainties in ecological risk assessment and their potential influence.

2.7 Data for Ecological Risk Assessments in the Gulf of Mexico

Much of the basic data and information needed in an ecological risk assessment for produced water discharges in the Gulf of Mexico are summarized in the following sections. Each of the sections describe data or information relevant to the phases of an ecological risk assessment as described above.

Sections 3, 4 and 5 describe important biological resources in the Gulf of Mexico to support the problem formulation phase and guide the identification of assessment endpoints.

Data describing produced water characteristics, including chemical concentrations and discharge rates are provided in section 6 to support the development of source terms and exposure assessments. Bioaccumulation data are also summarized to support exposure assessments (section 8).

Data describing the toxicity of produced water and component chemicals are reviewed to support effects assessments (sections 7,9, 10). Documented biological effects from produced waters are largely limited to impacts on benthic invertebrate communities, and these are summarized to support effects assessments at the community level (Section 11).

Table 2-1. Uncertainties and their importance in ecological risk assessment (from Smith and Shugart, 1994; adapted from Cothern, 1988).

Source of Uncertainty	Importance	Magnitude of Effect
Poor knowledge of System	Without any knowledge of the system, it is not possible to build a useful model.	Many orders of magnitude
Extreme variation, incorrect scales	Great variation in weather, for example, may cause a large change in the importance of the stressor. Modeling large-scale phenomenon using a small-scale model may lead to great uncertainty.	
Wrong model, endpoints, exposure routes	Measuring the wrong endpoint may lead to missed effects. Lack of knowledge of the exposure or model may lead to large errors.	
Surprises	Unexpected effects may occur caused either by important gaps in knowledge or by random effects. Despite low probability of occurrence, effects can have great consequence.	
Data collection practices	Errors in data collection and entry may lead to mistakes in interpreting statistical analyses.	Order(s) of magnitude
Design of laboratory experiments and quality control	Adherence to laboratory standards is necessary to avoid errors induced by lack of care.	
Variability in mesocosms or other ecosystem surrogates	Mesocosm studies have higher variability than laboratory studies and need to be carefully designed.	
Extraneous variables	Physical conditions may have a strong effect on laboratory results.	
Mistakes in statistical analysis	Outliers, wrong statistical model.	

Table 2-1. (cont.)

Source of Uncertainty	Importance	Magnitude of Effect
Interactions	Uncertainty may be introduced by failing to account for interactions among species or combined effects of chemicals or other stressors.	
Parameterization of Computer Model	Parameter estimates are taken from the literature, not from a fit to actual observations.	
Mistakes in computer code of simulation model	Errors in code may lead to gross prediction errors.	
Extrapolations across one species to another species in community or from laboratory to field spatial scale (local to regional)	Using a model developed for a simple endpoint may lead to errors when applied to estimate a more complex endpoint.	
Variability in laboratory test conditions	Variation in test organisms or concentrations of chemicals, for example, may cause under- or over-estimation of effects.	Up to one order of magnitude
Minor mistakes in choice of statistical model	Including variables that are not necessary in the model may lead to increased variance; missing variables may add a bias.	
Statistical design of manipulative studies (choice of stressor levels, randomization, number of experimental units, number of units per treatment)	Proper statistical design is important in laboratory and field studies especially when sample sizes are limiting. Estimates of quantities, such as No Observed Effect Levels, may be greatly affected by sample size and other factors.	Potentially of great importance
Design of field study	Haphazard design of field studies may lead to incorrect decisions regarding effects.	

3 BIOLOGICAL RESOURCES IN THE GULF OF MEXICO

3.1 Major Ecosystems

The Gulf of Mexico includes a wide variety of habitats for marine biota, both in the water column and on the seafloor. Darnell and Phillips (1988) describe the major ecosystems of the Texas-Louisiana Continental shelf and the upper continental slope. Three basic systems with transitional systems in between are recognized: the water column, soft bottom benthos, and hard substrate structure-related systems (Table 3-1).

Important coastal ecosystems associated with the Gulf of Mexico include extensive wetlands and estuaries. Wetlands along the Gulf Coast include fresh, brackish and saline marshes, forested wetlands and small areas of mangroves. Coastal wetlands are characterized by high organic productivity, high detritus production, and efficient nutrient recycling (MMS, 1993). Wetlands provide habitat for a great number and diversity of invertebrates, fish, reptiles, birds and mammals, and are important nursery grounds for many species of fish.

Table 3-1. Major ecological systems of the Texas-Louisiana continental shelf and upper continental slope (modified from Darnell and Phillips, 1988).

Major Systems	Subsystems	Components
Water Column	Coastal waters	Phytoplankton Zooplankton
	Oceanic waters	Neuston Nekton
Soft-bottom Benthic	Continental shelf	Meiofauna Macrofauna
	Upper continental slope	Megafauna Demersal fauna
Hard substrate, structure-related	Natural substrates	Hard bank biota Biofouling mat
	Artificial substrates	Niche fauna Structure-associated fauna
Transitional	Combinations of the above	Mixtures and combinations of the above

3.2 Biota

The larger, free swimming animals (nekton) of the Gulf of Mexico include squid, fish, sea turtles and cetaceans. Both coastal and pelagic assemblages are recognized (Darnell and Phillips, 1988). Table A-1 in Appendix A lists common nektonic species in the Northern Gulf of Mexico. The following summary is abstracted from MMS (1993) and Darnell and Schmidly (1988).

Coastal pelagic species such as mackerels, cobia, bluefish, amberjack and dolphin move seasonally within the Gulf of Mexico (MMS, 1993). King and Spanish mackerel winter in the southeastern Gulf, and spawn and summer in the Northeastern Gulf along the continental shelf.

Oceanic species including yellowfin and bluefin tuna are found mainly beyond the continental shelf during winter and spring, and move into the Atlantic Ocean after spawning. Black marlin, white marlin, sailfish and swordfish spawn in the northeastern Gulf, mostly beyond the continental shelf.

The demersal fauna are those associated with near bottom waters and include shrimp, crabs, and fishes. Tables A-2 and A-3 in Appendix A list demersal species common along the continental shelf and upper continental slope in the northern Gulf of Mexico.

In the continental shelf areas, populations from the inshore shelf zone (7-14m) are dominated seasonally by Atlantic croaker, spot, drum, silver seatrout, southern kingfish and Atlantic threadfin. The middle shelf zone (27-46 m) is dominated by longspine porgies and sciaenids. In the outer shelf zone (64-110m) blackfin searobin, Mexican searobin and shoal flounder are dominant.

As many as 15 species of shrimp are found in the coastal and estuarine areas of the Gulf of Mexico. Brown, white and pink shrimp are the most common. About eight species of portunid crab are found along the Gulf Coast. The blue crab is the only species that represents a substantial fishery.

Natural reefs and banks support large numbers of grouper, snapper, gag, scamp and seabass. Reef fish occur wherever hard bottoms with rocks or crevices are available, and are also associated with platforms. Snappers are estuary independent fish that remain close to underwater features or structures.

Many of the fishes in the Gulf of Mexico are estuary dependent. Estuary-dependent species spawn on the continental shelf, move into the estuaries as eggs, larvae or juveniles, grow and mature in the estuary, and migrate back to the shelf to spawn. Important estuary-dependent species include menhaden, shrimps, crabs, oysters and sciaenids (croaker, red drum, black drum, spotted seatrout).

Benthic Fauna include the infauna (animals that live in the substrate) and epifauna (animals that live on or are attached to the substrate). The most important factor in the distribution of benthic fauna is the type of substrate (MMS, 1993). The vast majority of the Northern Gulf of Mexico consists of soft, muddy bottoms dominated by polychaetes.

Benthic biota may be classified in terms of size (Darnell and Schmidly, 1988):

- microbenthos < 0.062 mm
- meiobenthos 0.062 - 1.0 mm
- macrobenthos 1.0 - 25.4 mm
- megabenthos > 25.4 mm

The microbenthos includes primarily bacteria, protozoa, fungi and blue-green algae. The meiobenthos includes nematodes, kinorhynchs, polychaetes, and harpacticoid copepods. Macrobenthos recorded in the Gulf of Mexico include well over a thousand species (see Table A-4 in Appendix A), and are dominated by polychaetes, crustaceans, and mollusks. The megabenthos includes bivalves, gastropods, and crustaceans. Depth related faunal assemblages have been identified by Defenbaugh (1976), see Table A-5 in Appendix A.

Estuaries in the Gulf of Mexico are often characterized by intertidal reefs constructed by oysters. When submerged, these reefs provide habitat and food for finfishes, crabs, and shrimp.

The communities associated with hard bottom substrate, reefs and banks, and platform structures are very different from soft bottom communities. Hard-bottom communities may include barnacles, oysters, hermatypic (reef-building) and ahermatypic (non-reef building) corals, and reef fish. Demersal and pelagic species with various levels of association to the structures are also found nearby. The communities associated with artificial platform structures are described in more detail in section 4. The biota associated with topographic highs identified as unique and sensitive ecosystems are described in section 5.

3.3 Commercial and Recreational Fisheries

The commercial fishery resources of Texas and Louisiana are of national importance, and the Gulf of Mexico provides almost 20 percent of commercial fish landings in the United States (MMS, 1993). The most important species in terms of quantity in 1991 was menhaden (1.2 billion pounds, $41 million; USDOC, 1992). Shrimp harvest was the most important in terms of value (229 million pounds, $411 million; USDOC, 1992). In 1991, the oyster fishery in the

Gulf of Mexico accounted for 43 percent of the national total (13.7 million pounds meats, $35.5 million) and 29 percent of the national total for the blue crab fishery (65.4 million pounds, $23.5 million).

In Louisiana, menhaden was the most important species in terms of quantity (1.0 billion pounds, $48 million), and shrimp was the highest value shellfish (27.3 million pounds, $36.7 million). In 1991, the following nine species accounted for landings valued at over $1 million: black drum, red mullet roe, shark, snapper, spotted sea trout, bluefin tuna, yellowfin tuna, blue crab, and American oyster (USDOC 1992; MMS, 1993).

In Texas in 1991, shrimp ranked first in both quantity and value (92 million pounds, $17 million). During 1991, the following species accounted for landings values over $500,000: red snapper, black drum, blue crab and American oyster (USDOC, 1992; MMS, 1993).

Many of the commercially important fish species in the Gulf of Mexico are believed to be in decline due to overfishing (USDOC, 1992). Fisheries in danger of collapse include shrimp, red snapper, black drum, shark, tuna, and spiny lobster. Fisheries Management Plans have been implemented to assess and manage commercial species in need of conservation. Fisheries Management Plans have been implemented for the following Gulf species: shrimp, stone crab, spiny lobster, coastal pelagics (king and Spanish mackerel), coral, reef fish (red snapper), swordfish, red drum, sharks, snapper and grouper.

Marine recreational fishing in the Gulf of Mexico accounts for an estimated $769 million in sales and employment for over 15,000 people (Sports Fishing Institute, 1987 as cited in MMS, 1993). Approximately 43% of the fish taken by recreational fishermen in the United States in 1991 were from the Gulf of Mexico (excludes Texas; data from tables in USDOC, 1992). Recreational fishing takes place from shore or within state waters, as well as offshore from private or charter boats.

In nearshore waters, recreational fishing is aimed at estuary-dependent species, particularly members of the drum family (sand and spotted seatrout, croakers, red drum).

Sportfishing in Louisiana and Texas is concentrated around oil and gas structures. In 1984, approximately 37% of all saltwater fishing trips in Louisiana and 28% in Texas were within 200 feet of an oil or gas structure. It has been estimated that over 70% of the fishing trips beyond three miles from shore are to areas near oil and gas structures (Witzig, 1986; Reggio, 1987). In the Gulf of Mexico, Ditton and Auyong (1984) found heavy use of offshore platforms by private recreational fishing boats. Most private recreational boats were bottom fishing, with snapper and seatrout reported most frequently as the major target species. Croaker was reported as a major target species in the Delta Region. The catch near oil and gas platforms was primarily (80%, excluding saltwater catfish) red snapper, sand seatrout, and Atlantic croaker (Witzig, 1986). On non-rig fishing trips, round scad, grunts and snappers made up over 70% of the catch (Witzig, 1986).

Stanley and Wilson (1990) surveyed recreational fishermen and charter boat operators in Louisiana in 1987 and 1988, and found the most common fishing method was offshore bottom fishing at oil and gas structures. The five most frequently caught species (or groups of species) in the 1987 survey were red snapper, spotted seatrout, silver/sand seatrout, other snapper and greater amberjack. The most frequently caught species in 1988 were red snapper, spotted seatrout, other snapper, silver/sand seatrout and gray triggerfish. The major target species for offshore bottom fishing was red snapper, and nearshore fishing near platforms targeted spotted seatrout. Stanley and Wilson also documented catch rates for offshore trolling near platform structures. They found that the catch was not dominated by any one species but included blue runner, dolphin, king mackerel, little tunny and Spanish mackerel. Table 3-2 summarizes the composition of the catch for the survey participants.

Recreational fishermen also take pelagic species not associated with platform structures including tarpons, cobias, dolphins, amberjacks and other jacks, little tunnys and billfishes (Linton, 1988).

3.4 Application to Ecological Risk Assessment for Produced Water Discharges

A description of the important fisheries resources in the Gulf of Mexico is needed to complete the problem formulation phase of an ecological risk assessment and to identify appropriate endpoints for a specific analysis. More site or region specific data may be needed for a specific analysis, and species with important social value other than commercial or recreational should also be considered (section 5).

Table 3-2. Composition of catch around oil and gas platforms off Louisiana (modified from Stanley and Wilson, 1990).

Species/group	1987	1988	Species/group	1987	1988
	%	%		%	%
Atlantic croaker	2.0	0.5	Little tunny	0.8	1.0
Atlantic spadefish	0.1	0.7	Lookdown	0.0	0.2
Bearded brotula	0.0	0.0	Other jacks	0.1	0.1
Black drum	0.4	0.5	Other snapper	4.7	8.1
Blackfin tuna	0.1	0.1	Pinfish	0.3	0.4
Bluefish	2.8	1.8	Puffer	0.0	--
Blue marlin	0.0	0.0	Rainbow runner	0.0	0.0
Blue runner	0.6	1.5	Rays	0.0	0.0
Cobia	1.1	1.1	Red drum	2.9	1.1
Crevalle Jack	0.1	0.2	Red snapper	34.6	41.2
Cubbyu	0.0	--	Sharks	1.5	1.2
Dolphin	0.9	3.7	Sheepshead	0.1	0.1
Florida pompano	0.7	0.1	Shrimp eel	0.0	0.1
Flounder	0.0	0.1	Silver/sand seatrout	8.2	5.8
Gafftopsail catfish	0.2	0.3	Skipjack tuna	1.4	1.5
Great barracuda	0.2	0.1	Spanish mackerel	1.7	1.3
Greater amberjack	4.1	4.3	Spotted seatrout	21.8	9.4
Grey triggerfish	2.7	5.9	Squirrelfish	0.0	0.1
Grouper	2.7	4.7	Tarpon	0.0	0.0
Grunts	0.2	0.8	Tripletail	0.2	0.0
Hake	0.0	0.0	Wahoo	0.1	0.2
Hardhead catfish	1.1	0.3	White spotted soapfish	0.0	--
King mackerel	1.2	1.2	Yellowfin tuna	0.0	0.0
Ladyfish	0.1	0.0			

4 PLATFORM COMMUNITIES

4.1 Introduction

Oil and gas platform structures are colonized by microorganisms, algae, and sessile invertebrates that live attached to the structure and form the biofouling mat. These organisms provide food and habitat for many motile invertebrates and small fishes that live in close association with the biofouling mat. There is also a diverse assemblage of demersal and pelagic fishes associated with the platforms, some of which are residents. The composition of the biofouling community and assemblage of demersal and pelagic fishes associated with platforms varies with distance from shore, water depth, latitude and age of the platform (Gallaway and Lewbel, 1982). Gallaway and Lewbel (1982) and Darnell and Schmidly (1988) summarized the results of several studies of platform communities (Gallaway et al., 1981a, b; Gallaway 1981; Bert and Humm, 1979; Fotheringham 1981; George and Thomas, 1979; Middleditch, 1981). The following summary is from Gallaway and Lewbel (1982) and Darnell and Schmidly (1988).

4.2 Biota Associated With Platform Structures

Biofouling Community

Gallaway and Lewbel (1982) described three faunal groups at platforms in the Northwestern Gulf of Mexico: coastal (below 30 meters depth); offshore (30-60 meters depth); and bluewater (greater than 60 meters depth). The biomass of the biofouling mat generally ranges from about 1-5 kg/m^2 (oceanic waters) to 15 kg/m^2 (nearshore surface waters) (Darnell and Schmidly, 1988).

A large number of algal species are associated with platforms, but most are small or microscopic. The greatest portion of the algae growth is close to the surface. Brown and red algae become more abundant in oceanic waters.

Sponges form a large proportion of the mat of fouling organisms, covering the shells of barnacles and bivalves on the platforms. Sponges are relatively unimportant on bluewater platforms.

On coastal platforms in Louisiana, the dominant biofouling organisms are the stalked barnacles *(Balanus amphitrite niveus = B. reticulatus)* and *B. improvisus*. On coastal platforms in Texas, *B. tintinnabulum* is dominant. Barnacles are relatively unimportant components of the biofouling community at offshore platforms. On bluewater platforms, stalked barnacles are the primary biofoulers (Darnell and Schmidly, 1988).

At least four species of oysters are found on Gulf of Mexico platforms: *Crassostrea virginica*, *Ostrea equestris*, *Isognomon bicolor* and *Hyotissa thomasi.* Platforms in Louisiana do not support large numbers of bivalves at shallow depths, but deeper portions of platforms may support large numbers of oysters. Oysters and other bivalves are common on coastal platforms in Texas waters. On offshore platforms in Louisiana, bivalves replace barnacles as the dominant biomass. Important species include tree oysters (*Isognomon bicolor)* and leafy jewel boxes (*Chama macerophylla*).

Hydroids are patchy but can be extremely abundant on offshore and coastal platforms in both Louisiana and Texas. Hydroids are the dominant species in many near-bottom samples, but are often the most abundant (on a weight basis) near the surface.

Anemones are important components of the biofouling community, especially on coastal platforms in both Texas and Louisiana. They tend to form distinct zones at specific depths, especially toward the bottom.

Bryozoans are patchy in time and space on coastal and offshore platforms in Louisiana, but may be extremely abundant. Bryozoans are common on Texas coastal platforms, but tend to die back in winter months.

Stony corals and octocorals have been found at platforms, but are numerically unimportant. Identified Octocoral species include *Telesto* sp., a low-profile encrusting species, and *Leptogorgia virgulata* (gorgonian sea whip). Several species of small ahermatypic (non reef-building) corals have been found at platforms, including *Astrangia sp.*, *Phyllangia americana*, and *Oculina diffusa.*

Table 4-1 summarizes the dominant organisms for coastal, offshore and blue water biofouling communities. A partial list of the invertebrates and algae associated with platform structures in the Gulf of Mexico, including species that comprise the biofouling community, is given in Appendix A (Table A-6).

Table 4.1. Dominant organisms (by weight) for coastal, offshore and blue water biofouling communities (modified from Darnell and Schmidly, 1988).

Assemblage	Dominant Organisms	
Coastal	Reticulated barnacle	*Balanus reticulatus*
	Bay barnacle	*Balanus improvisus*
	Mediterranean barnacle	*Megabalanus antillensis*
	Virginia oyster	*Crassostrea virginica*
	Horse oyster	*Ostrea equestris*
Transitional (Offshore)	Leafy jewel box	*Chama macerophylla*
	Tree oyster	*Isognomon bicolor*
Blue water	Striped goose barnacle	*Conchoderm virgatum*
	Common goose barnacle	*Lepas anatifera*

Non-Attached Species Living in Close Association With the Biofouling Community

Species that live in close association with the biofouling community include a variety of worms, crustaceans, echinoderms and other small invertebrates and blenny fish. General conclusions about the motile epifaunal invertebrates on platforms in the Northwestern Gulf of Mexico were made by Gallaway and Lewbel (1982):

"1. An extremely diverse assemblage of small and large motile invertebrates utilize the shelter and food provided by sessile members of the community.

2. The most abundant amphipods are tube-dwelling forms such as corophiids, stenothoids, and caprellids which are typically symbiotic with hydroids and other mat organisms.

3. The most commonly reported polychaetes are syllids, a group often associated with hydroids and sponges.

4. Pycnogonids, another epibiotic group, are frequently found on the fouling mat.

5. Nermerteans are common predators on the other epifaunal invertebrates on the platforms.

6. Ophiuroids may be present in very high densities embedded in the fouling mat.

7. Large, conspicuous invertebrates such as lobsters and crabs may be found in low densities around and beneath platforms".

Small fishes, particularly blennies, may reside in old barnacle shells and other niches in the biofouling mat (Darnell and Schmidly, 1988). A partial list of the invertebrates and algae associated with platform structures in the Gulf of Mexico, including non-attached species, is given in Appendix A (Table A-6).

Large Mobile Species

Larger mobile species that have a relatively loose association with the platform structure include shrimp, larger crabs and lobsters, fishes, and sea turtles. Common crustaceans on platforms include pistol shrimp *(Synalpheus* spp.*),* arrow crabs (*Stenorhynchus seticornis*), Xanthid crabs (Xanthidae) and in deeper water spiny lobster (*Panulirus argus*) (Gallaway and Lewbel, 1982). A large number of fish species have been reported around platforms in the Gulf of Mexico (Appendix A, Table A-7).

Some of the resident fish species are trophically dependent on the biofouling mat (e.g., sheepshead, gray triggerfish, butterflyfishes). Other residents, including spadefish, red snappers and groupers are mainly trophically independent of the biofouling mat and are attracted by the structure itself for cover.

The barracuda (*Sphyraena barracuda*), almaco jack (*Seriloa rivoliana*) hammerhead sharks (*Sphryna* spp.), cobia (*Rachycentron canadum*) and bluefish *(Pomatomus saltatrix)* are predator species that feed upon other resident platform species and may have a longer residence time than do the other large predators (Gallaway and Lewbel, 1982). Large predatory species believed to be highly transient include mackerels (Scombridae), jacks (*Caranx* spp.) and the little tunny (*Euthynnus alletteratus*). These species come and go to platforms for periods of hours to days as they follow schools of prey fish such as scads and sardines. Except for bluefish, large predators that feed upon the platform residents are not numerous. Around a given structure bluefish are usually present in schools of up to about 5,000 individuals (Gallaway and Lewbel, 1982). For food, bluefish may depend on pelagic prey species, surrounding soft bottom fish, and crustacean populations as well as platform residents.

Coastal Platforms

Gallaway and Lewbel (1982) summarized the vertical zonation of the assemblage of fishes at coastal Louisiana platforms. "According to Shinn (1974), the vertical zonation of fishes around Louisiana coastal platforms is characterized by spadefish, barracuda, lookdown and sheepshead in the upper part of the water column, red snapper and large groupers typically near the bottom, but often in mid-water, and on the bottom, species such as speckled trout (*Cynoscion nebulosus*), sand trout (*Cynoscion arenarius*) and flounders (*Paralichthys* sp.). To the bottom group we would add the Atlantic croaker (*Micropogon undulatus)*, and note that we have not observed speckled trout at coastal platforms although they may be present, particularly near the beach. Schools of bluefish and some jackfishes like blue runner appear to be quite abundant around coastal platforms at all depths. In the upper and middle part of the water column, Atlantic moonfish are typically abundant at Coastal platforms in Louisiana as are an occasional gray snapper. At coastal platforms more distant from shore, it is not unusual to encounter large schools of baitfish such as round scad, Spanish sardine and scaled sardine (*Harengula pensacolae*) in the upper part of the water column".

Gallaway and Lewebel (1982) summarized the composition of the assemblage of fishes at Texas coastal platforms studied in the Buccaneer Gas and Oil Field (BGOF; Middleditch, 1981). "The composition and vertical zonation of pelagic fishes around the BGOF structures were similar to those observed at Coastal platforms in Louisiana waters. Spadefish (dominant), sheepshead and barracuda were characteristic of the upper column; red snapper and groupers were common to the bottom and often seen at mid-depths; and schools of bluefish, blue runner and baitfish were common. The fish fauna at BGOF structures differed notably from those in Louisiana in that large schools of lookdown were never observed over the four years of investigation, and by the high abundance of the tomtate. The sciaenid fishes listed by Shinn (1974) were not common to the bottom of BGOF structures, but the cubbyu (*Equestus umbrosus*) and, sometimes, the bigeye (*Priacanthus arenatus*) were basically similar to those at Louisiana platforms, including the most common inhabitants, belted sandfish, cocoa damselfish, sergeant major, night sergeant, gray triggerfish and an occasional butterflyfish."

Offshore Platforms

Gallaway and Lewbel (1982) describe the fish assemblages at two offshore Louisiana platforms: one classified as "ecotonal" between coastal and offshore assemblages and one they consider to represent a true offshore assemblage (Gallaway et al., 1981b).

" The ecotonal platform in the offshore zone of Louisiana was located at a distance of 42 km from the shore, in water 35 m deep. Dominant fishes during the summer were bluefish, spadefish, and mixed schools of lookdown and moonfish. Blue runner, amberjack and almaco jack were common. Sheepshead and gray triggerfish were common but not abundant. Large predators were barracuda, cobia, and nurse shark. Reef fish were not abundant, and included cocoa damselfish, cubbyu, whitespotted soapfish, bigeye and bermuda chub. The snapper grouper assemblage was a major component, and included large groups of gray snapper and medium to small schools of red and lane snapper. Spadefish, lookdown, and gray snapper dominated near the surface; spadefish, bluerunner and gray snapper were most abundant at mid-depth; and snapper were most common at 23 m. Large Atlantic croaker were caught by angling at the bottom.

The spadefish was the dominant pelagic species at the Offshore platform, and most of the other species characteristic of the coastal fish assemblages were also well represented and abundant (lookdown, moonfish, blue runner, sheepshead, gray triggerfish). The assemblage differed from the coastal assemblage in the abundance of gray and red snapper and the richness of the tropical species such as cocoa damselfish, blue and French angelfish, sergeant major, brown chromis, filefishes, tangs, flamefish and the creole fish. Also well represented were the almaco jacks, greater amberjack, bar jack, and rainbow runner. Other large predators included barracuda, crevalle jack, cobia and hammerhead shark."

Bluewater Platforms

Gallaway and Lewbel (1982) summarized the fish assemblage at blue water platforms in the Northern Gulf of Mexico. "At Bluewater platforms, the huge pelagic schools of spadefish, lookdowns, and bluefish are absent, seemingly replaced by numerous creole fish and almaco jacks along with the ubiquitous blue runner. The grazing sheepshead is replaced by the gray triggerfish and a host of tropical species. In the upper part of the water column down to 30-m depths, mycteropercid groupers and hinds (e.g., *Epinephelus adscensionis*) are common to abundant. The vertical members of the Bluewater platforms are surrounded by swarms of wrasses (particularly the creole wrasse, *Clepticus parrai*, and Spanish hogfish (*Bodianus rufus*) and other tropical species including damselfishes, angelfishes, tangs, rock beauty (*Holocanthus tricolor*), red spotted hawkfish *(Amblycirrhitus pinos*) and red hogfish (*Decodon puellaris*). The most abundant large predator, at least within safe diving depths (30 m) is the barracuda; hammerhead sharks are also common."

4.3 Conceptual Model

Conceptual models of platform communities have been developed by Gallaway and Margraf (1979), Fucik and Show (1981), Gallaway et al. (1981a) and Gallaway and Lewbel (1982). Darnell and Schmidly (1988) presented a simplified summary model (Figure 4-1).

The fouling mat includes primary producers, attached filter feeders (barnacles, mollusks, sponges, hydroids, bryozoans), and small browsers and detritus feeders trophically dependent on the fouling mat. The fouling mat receives input from sunlight, plankton and nutrients and produces organic detritus which may remain suspended or fall to the bottom. In the surrounding water are the non-attached plankton feeders, mat browsers, detritovores, omnivores and predatory species. Some of these are residents, while others are transient.

Fucik and Show (1981) developed a mathematical model based on the BGOF studies of platforms near Galveston Bay, Texas. This model suggested that the major flow of organic material entered the system through the phytoplankton, zooplankton, plankton feeders, and fouling flora and fauna compartments of the system, and that the other compartments serve to export carbon. Based upon results of simulations of the model, uptake of contaminants in platform discharges appeared limited, in large part, to the fouling community.

4.4 Application to Ecological Risk Assessment for Produced Water Discharges

A description of the communities associated with coastal and offshore platforms in the Gulf of Mexico is needed to complete the problem formulation phase of an ecological risk assessment and describe the environment at risk. Community descriptions are also required to identify assessment endpoints.

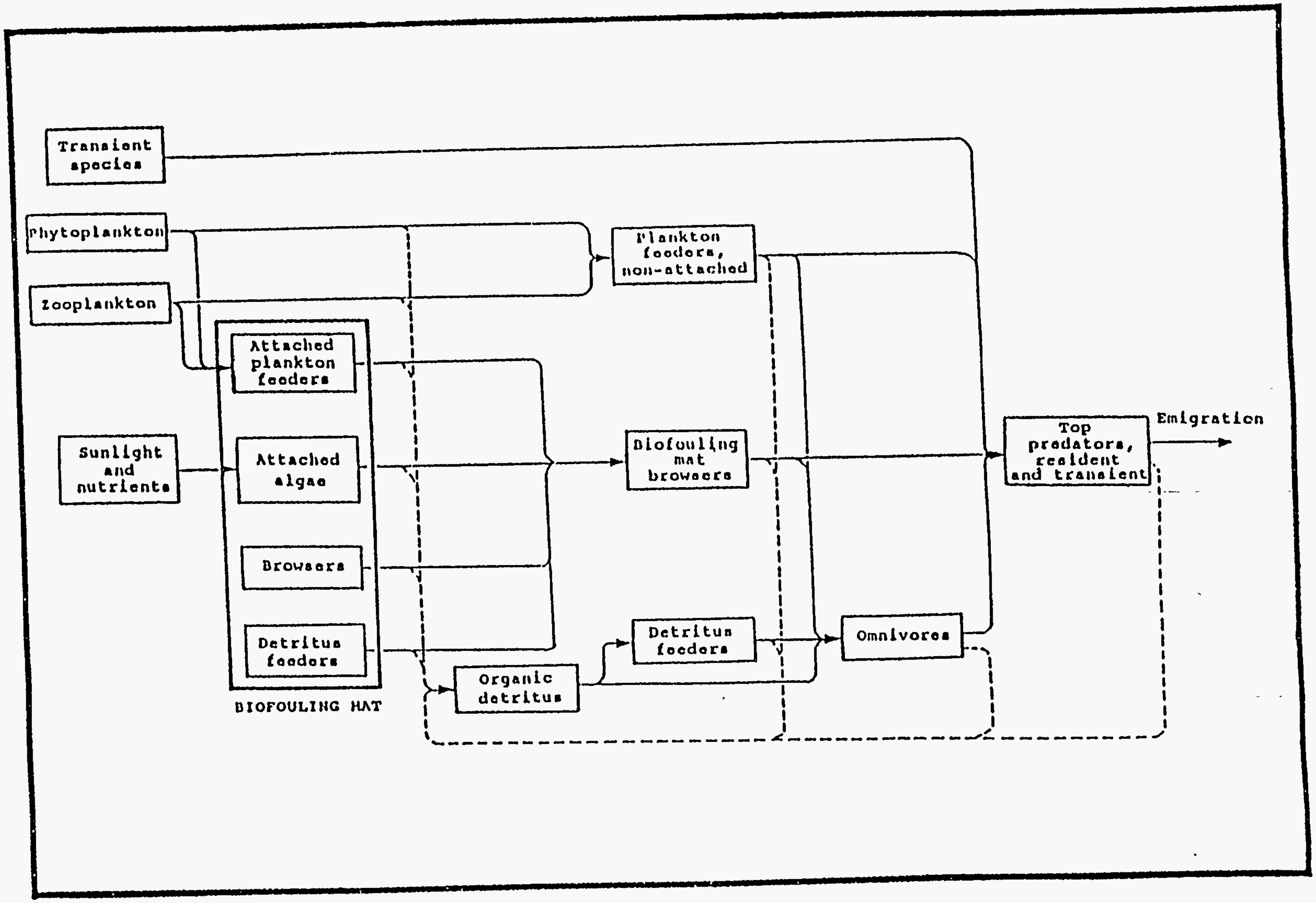


Figure 4-1. Conceptual model of a platform community (from Darnell and Schmidly, 1988).

5 ENDANGERED SPECIES AND SENSITIVE ECOSYSTEMS

5.1 Introduction

Endangered species and sensitive ecosystems in the Gulf of Mexico have special social and sometimes ecological value, and represent potential assessment endpoints in ecological risk assessments. The following summaries are based largely on information presented in the Environmental Impact Statements produced by the Department of Interior, Minerals Management Service (DOI, MMS) to assess the potential impacts of proposed oil and gas lease sales on the Gulf of Mexico (MMS, 1994; 1993; 1990).

5.2 Endangered Species and Marine Mammals

The Endangered Species Act of 1973 (16 U.S.C. 1531-1543) as amended, and the Marine Mammal Protection Act (MMPA) of 1972 (16 U.S.C. 1361 et seq.) give special protection to endangered and threatened species and to marine mammals. These species are unique in that the risk to individual animals may be of concern.

Marine Mammals

Marine mammals that occur in the Northern Gulf of Mexico are listed in Table 5-1. Seven species of baleen whales have been reported in the Gulf of Mexico but are rare: the northern right whale, blue whale, fin whale, sei whale, Bryde's whale, minke whale and the humpback whale. Five of these species are listed as endangered (Table 5-1).

Twenty-five species of toothed whales and dolphins have been reported in the Gulf of Mexico (Table 5-1). The sperm whale is the only one of the toothed whales and dolphins listed as endangered. It is the most abundant large whale in the Gulf of Mexico and tends to occur in deeper water (MMS, 1993).

Common toothed whales and dolphins in the Gulf of Mexico include the grampus, Atlantic bottlenose dolphin, striped dolphin and the Atlantic spotted dolphin (Table 5-1). Bottlenose dolphins are common on the continental shelf and nearshore waters, and Atlantic spotted dolphins frequent mid-shelf to outer-shelf waters. Grampus are frequently sighted along the shelf edge. The striped dolphin frequents deeper waters.

Table 5-1. Marine mammals of the central and western Gulf of Mexico (modified from Schmidly and Scarborough, 1990; as cited in MMS, 1993).

Species	Occurrence	Status
Baleen Whales		
Northern Right Whale (*Eubalaena glacialis*)	R	E
Blue Whale (*Balaenoptera musculus*)	R	E
Fin Whale (*Balaenoptera physalus*)	R	E
Sei Whale (*Balaenoptera borealis*)	R	E
Bryde's Whale (*Balaenoptera edeni*)	R	
Minke Whale (*Balenoptera acutorostrata*)	R	
Humpback Whale (*Megaptera novaeangliae*)	R	E
Toothed Whales and Dolphins		
Sperm Whale (*Physeter macrocephalus*)	C	E
Pygmy Sperm Whale (*Kogia breviceps*)	C	
Dwarf Sperm Whale (*Kogia simus*)	U	
Sowerby's Beaked Whale (*Mesoplodon bidens*)	Ext	
Blainville's Beaked Whale (*Mesoplodon densirostris*)	R	
Antillian Beaked Whale (*Mesoplodon europaeus*)	U	
Goosebeaked Whale (*Ziphius cavirostris*)	U	
Killer Whale (*Orcinus orca*)	R	
False Killer Whale (*Pseudorca crassidens*)	U	
Pygmy Killer Whale (*Feresa attenuata*)	U	
Short-finned Pilot Whale (*Globicephala macrorhynchus*)	C	
Grampus/Risso's Dolphin (*Grampus griseus*)	U	
Melon-headed Whale (*Peponocephala electra*)	R	
Atlantic Bottlenose Dolphin (*Tursiops truncatus*)	C	
Saddleback Dolphin (*Delphinus delphis*)	R	
Rough Toothed Dolphin (*Steno bredanensis*)	R	
Striped Dolphin (*Stenella coeruleoalba*)	C	
Pantropical Spotted Dolphin (*Stenella attenuata*)	R	
Short-snouted Spinner Dolphin (*Stenella chymene*)	U	
Atlantic Spotted Dolphin (*Stenella frontalis*)	C	
Long-snouted Spinned Dolphin (*Stenella longirostris*)	U	
Fraser's Dolphin (*Lagenodelphis hosei*)	R	

C = common, U = uncommon, R = rare, Ext= extralimital record
E = endangered

Endangered and Threatened Species

Endangered and threatened species in the Northern Gulf of Mexico potentially impacted by produced water discharges are listed in Table 5-2 (biological opinions in MMS, 1993). This list does not include all endangered and threatened species in the region but is restricted to: those identified as potentially impacted by the Fish and Wildlife Service (FWS) and the National Marine Fisheries Service (NMFS) in the biological opinions referenced in the Environmental Impact Statements for the central and western Gulf of Mexico (MMS, 1993); all marine turtles that occur in the Gulf of Mexico; and all marine mammals that occur in the Gulf of Mexico. No terrestrial organisms have been included.

Coastal and Marine Birds

Endangered and threatened species of Coastal and Marine Birds potentially impacted by produced water discharges include the brown pelican, bald eagle, arctic peregrine falcon, piping plover and the whooping crane (Table 5-2).

The brown pelican is classified as an endangered species except along the U.S. Atlantic coast, Florida and Alabama. The brown pelican is rarely found away from saltwater, and does not move more than 20 miles out to sea. Feeding is by plunge-diving for fish in coastal waters.

The bald eagle is listed as endangered in most of the conterminous U.S., and as threatened in WA, OR, MN, WI and MI. The bald eagle eats primarily fish, combined with opportunistic capture of small vertebrates.

The arctic peregrine falcon is listed as a threatened species, and is a subspecies of the peregrine falcon of North America. The arctic peregrine falcon nests in tundra areas of North America and Greenland, and migrates south to the Gulf coast, West Indies and Central and South America. The arctic peregrine falcon feeds on a wide variety of birds. Migrant arctic peregrine falcons concentrate on beaches, flats and wetlands along the Gulf coast.

The piping plover is listed as endangered in the Great Lakes watershed and as threatened in the remainder of its range, including the Gulf Coast. Nesting areas include the Central and North Atlantic seaboard of the U.S., Atlantic Canada, the Great Lakes and a portion of the Northern Great Plains. The piping plover winters on the Atlantic Ocean and Gulf Coasts and on Caribbean islands. Texas is an important wintering area, and the Louisiana barrier islands provide favorable habitat.

The whooping crane is listed as an endangered species. The wintering range of the entire reproducing population of the whooping crane is along the Texas coast. There is an experimental population established in southeastern Idaho. Whooping cranes feed on crabs and clams in tidal flats, shallow bays and channels.

Fishes

One species of fish listed as threatened is potentially affected by produced water discharges -- the Gulf Sturgeon (a subspecies of the Atlantic sturgeon). The Gulf Sturgeon is anadromous and is known to occur on most major rivers from the Mississippi River to the Suwannee River and in marine waters of the central and eastern Gulf of Mexico. Adults and immature fish spend eight to nine months in rivers and three to four of the coldest months in estuaries or the Gulf of Mexico.

Marine Turtles

Five species of marine turtles occur in the Gulf of Mexico, and all are listed as threatened or endangered.

The loggerhead turtle is the most abundant species of marine turtle in Gulf of Mexico waters, and is listed as a threatened species. Loggerheads inhabit coastal areas of the continental shelf and are most common in water less than 50 m deep, but are also found in deep water (NMFS, biological opinion in MMS, 1993).

Green turtles are listed as threatened species, except for breeding populations in Florida and on the Pacific coast of Mexico where they are listed as endangered. The green turtle was once a commercial fishery in inshore Texas Bays but the population has not recovered from over-exploitation. Green turtles prefer depths less than 20 m, and the only major feeding grounds for the juvenile and subadult Green Turtle in the Gulf of Mexico is the upper west coast of Florida. Older green turtles are unlikely to reside permanently in most areas of the Gulf of Mexico because of the scarcity of sea grass pastures (NMFS Biological Opinion in MMS, 1993).

Leatherbacks are the largest and most oceanic marine turtle, and feed primarily on jellyfish. They are distributed throughout the Atlantic, Pacific, Caribbean and the Gulf of Mexico. Nesting is concentrated in the tropical latitudes, but there are occurrences in Florida. The leatherback turtle is listed as endangered.

The hawksbill turtle is relatively uncommon in the northern Gulf of Mexico, and is listed as endangered. Hawksbill turtles prefer reefs and shallow coastal waters, and are more common in tropical areas of the Atlantic, Gulf of Mexico and Caribbean.

The Kemp's ridley turtle is the most endangered of the sea turtles. The only major nesting area is on a stretch of beach in Rancho Nuevo, Mexico. Limited nesting occurs on Padre and Mustang Islands in Texas, and the NMFS oversees a hatching and rearing program on Padre Island. The foraging range of the Kemp's ridely turtle is restricted to the Gulf of Mexico. These turtles feed primarily in shallow coastal waters on bottom-living crustaceans.

Marine Mammals

Threatened and Endangered species of marine mammals in the Northern Gulf of Mexico include the northern right whale, blue whale, fin whale, sei whale, humpback whale and the sperm whale (Table 5-2 ; see previous discussion).

Table 5-2. Endangered and threatened species in the northern Gulf of Mexico potentially impacted by produced water discharges (MMS, 1993).

Species	Status
Coastal and Marine Birds	
Brown Pelican (*Pelicanus occidentalis*)	E
Bald Eagle (*Haliaeetus Leucocephalus*)	E
Arctic Peregrine Falcon (*Falco peregrinus tundrius*)	T
Piping Plover (*Charadrius melodus*)	T
Whooping Crane (*Grus americana*)	E
Fishes	
Gulf of Mexico Sturgeon (*Acipenser oxyrhnchus*)	T
Marine Turtles	
Green Turtle (*Chelonia mydas*)	T
Kemp's Ridley Turtyle (*Lepidochelys kempi*)	E
Hawksbill Turtle (*Eretmochelys imbricata*)	E
Loggerhead Sea Turtle (*Caretta caretta*)	T
Leatherback Sea Turtle (*Dermochelys coriacea*)	E
Marine Mammals	
Northern Right Whale (*Eubalaena glacialis*)	E
Blue Whale (*Balaenoptera musculus*)	E
Fin Whale (*Balaenoptera physalus*)	E
Sei Whale (*Balaenoptera borealis*)	E
Humpback Whale (*Megaptera novaeangliae*)	E
Sperm Whale (*Physeter macrocephalus*)	E

E = endangered; T = threatened

5.3 Unique and Sensitive Ecosystems

Unique and sensitive biological resources of the Gulf of Mexico include coastal wetlands, the pinnacle trend live bottom features, topographic features inhabited by hard-bottom benthic communities, and deep water chemosynthetic benthic communities. The Minerals Management Service (MMS) describes these biological resources in detail in the Environmental Impact Statements developed for proposed oil and gas lease sales in the Gulf of Mexico outer continental shelf (OCS) (MMS, 1994; 1993). The following descriptions are abstracted from MMS (1993); see this source for more detail and original data sources.

Wetlands

Wetlands along the Gulf Coast include fresh, brackish and saline marshes, forested wetlands and small areas of mangroves. Coastal wetlands are characterized by high organic productivity, high detritus production, and efficient nutrient recycling (MMS, 1993). Wetlands provide habitat for a great number and diversity of invertebrates, fish, reptiles, birds and mammals, and are important nursery grounds for many species of fish. Figure 5-1 shows the extensive wetlands along the coast of Louisiana.

Live-Bottoms (Pinnacle Trend)

The pinnacle trend is a region of topographic relief in the northwestern Gulf of Mexico between 67 and 110 m in depth that appears to be carbonate reef structures in an intermediate stage between growth and fossilization. Additional features occur outside of the identified pinnacle trend. The pinnacles provide surface area for the growth of sessile invertebrates and attract large numbers of fish. In a study of hard bottom features, Continental Shelf Associates (1992a) found that biological communities were dominated by tropical and subtropical suspension-feeding invertebrates.

Topographic Features

The shelf and shelf edge of the central and western Gulf of Mexico contain topographic features or banks inhabited by hard-bottom benthic invertebrates. These areas are important because they support hard-bottom communities of high diversity and high biomass and large numbers of recreationally and commercially important fish. Topographic features are also unique in that they are small and isolated areas in a region of much lower diversity (MMS, 1993). These topographic features present proper conditions for coral growth. Seven distinct biotic zones have been identified (Rezak et al., 1985) and are described in more detail in MMS (1993). Figure 5-2 shows the location of major topographic features in the Gulf of Mexico. The East and West Flower Garden

Banks are of particular interest due to the extensive development of the hermatypic coral reef community, and have been designated a National Marine Sanctuary.

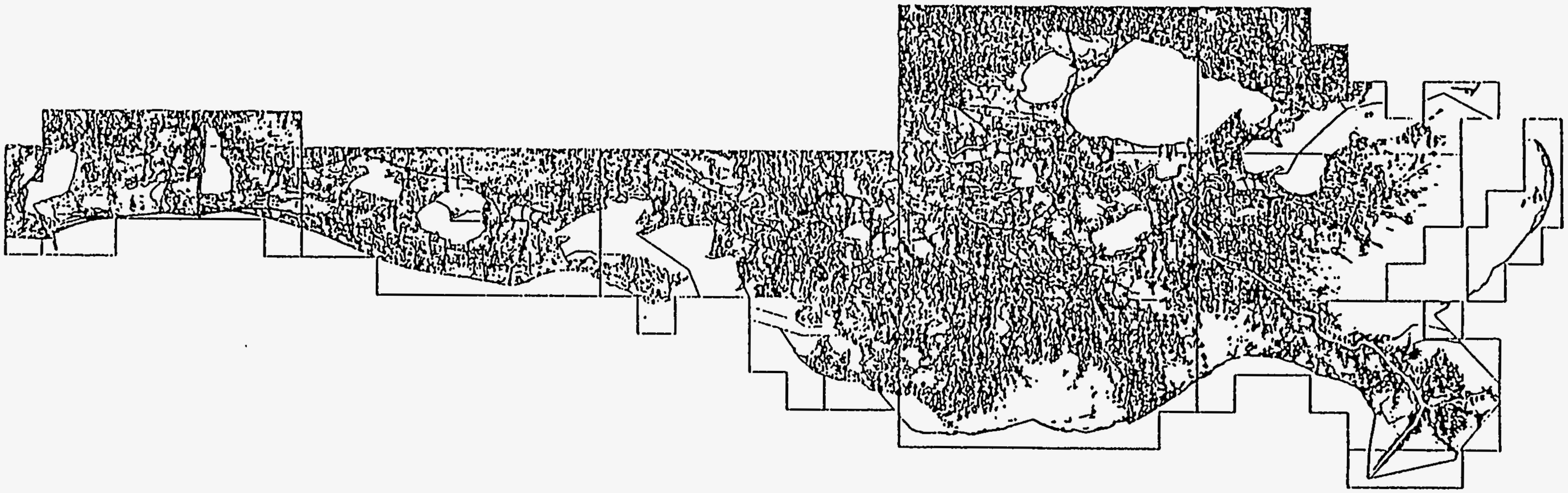
Deepwater Chemosynthetic Benthic Communities

Chemosynthetic clams, mussels and tube worms have been discovered in deep waters of the Gulf of Mexico (MMS, 1993; Figure 5-3). These chemosynthetic communities are associated with hydrocarbon and H_2S seep areas at water depths greater than 400 m. The communities are characterized by bacterial mats, dense beds of tube worms, clams and mussels, numerous small gastropods and galatheid crabs (MMS, 1993). The worms, clams and mussels contain autotrophic bacterial symbionts (Brooks et al., 1987).

5.4 Application to Ecological Risk Assessment for Produced Water Discharges

Descriptions of the major ecosystems and biota potentially at risk will support the problem formulation step in an ecological risk assessment, including the identification of assessment endpoints in a specific analysis. Endangered species and sensitive ecosystems represent unique social values and should be considered in identifying assessment endpoints

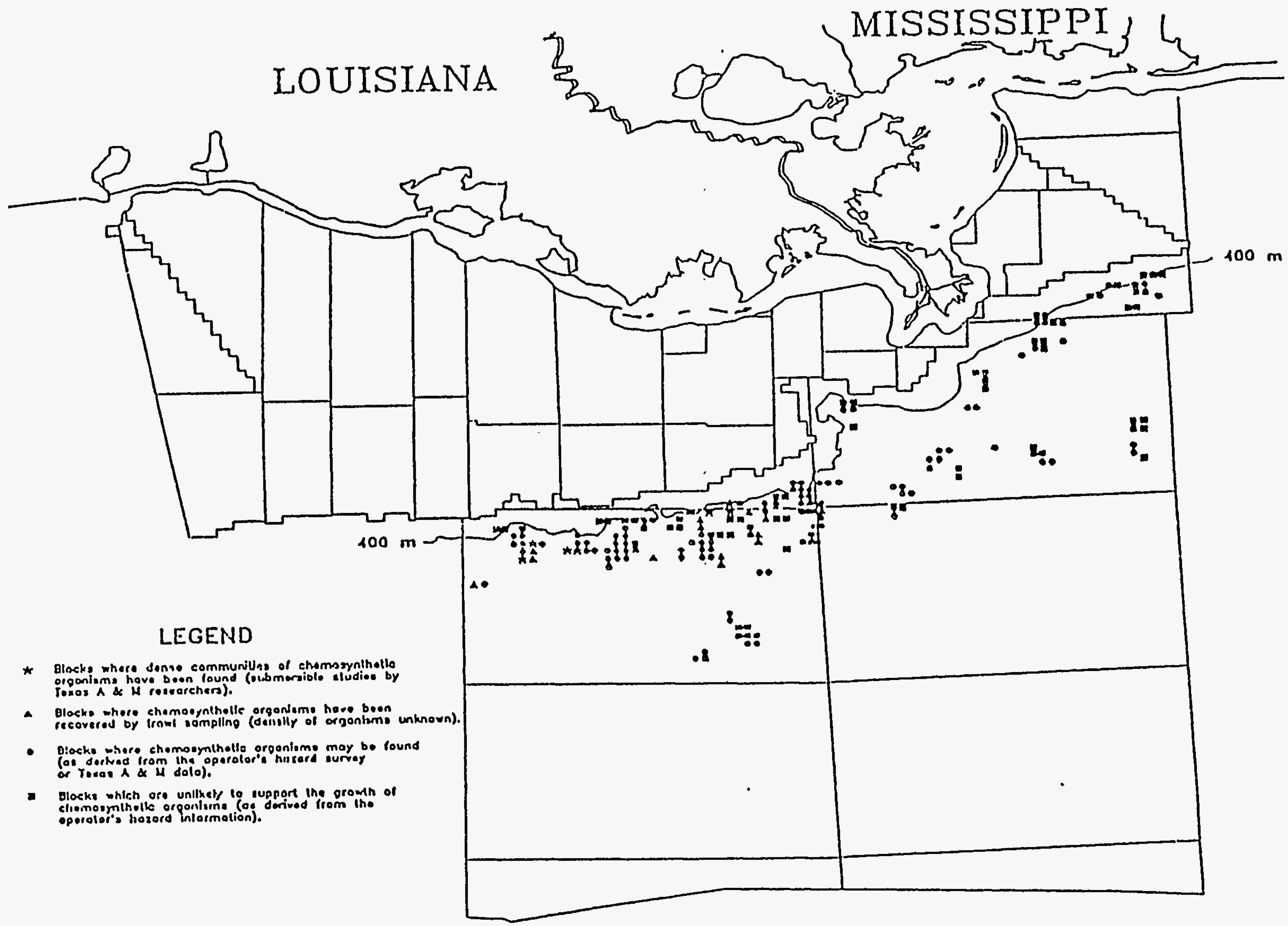
Figure 5-1. Coastal Louisiana wetlands.



* Data From United States Fish and Wildlife Service

Figure 5-2. Topographic features in the Gulf of Mexico (from MMS, 1993).

Figure 5-3. Chemosynthetic benthic communities in the Gulf of Mexico (from MMS, 1993).



6 CHEMICAL/PHYSICAL CHARACTERIZATION OF PRODUCED WATER

6.1 Introduction

Produced waters usually have high total dissolved solids (salinity) and total organic carbon, and are low in dissolved oxygen. Other components of potential concern include metals, dissolved and dispersed petroleum hydrocarbons, various treatment chemicals, and radionuclides.

Detected contaminants and contaminant concentrations in produced water vary widely, because the characteristics of the saline water and oil in the formation varies and because treatment methods and treatment efficiencies vary over time and space. The following sections summarize ranges of contaminant concentrations in produced waters and produced water discharge rates.

6.2 Conventional Pollutants, Heavy Metals and Organics

Produced waters are brine solutions with an ionic composition similar to, but usually more concentrated that of seawater (Neff et al., 1987). The concentration of total dissolved solids (TDS) ranges from a few parts per thousand (ppt) to 300 ppt (MMS, 1993). Most produced waters have higher TDS than seawater (35 ppt).

Metals that have been measured in produced waters at concentrations greater than seawater include aluminum, barium, beryllium, cadmium, chromium, copper, iron, lead, manganese, mercury, nickel, silver vanadium and zinc (MMS, 1993). Concentrations of metal discharges in produced waters vary widely.

Produced waters contain petroleum components, with volatile and soluble acid-extractable components present in higher concentrations than the heavier components (PAHs) (Middleditch, 1984; St. Pé, 1990).

Treatment chemicals are added to treat or prevent operational problems. Production treating chemicals include: scale and corrosion inhibitors, biocides, emulsion breakers, water treating chemicals (coagulants, floculants and reverse emulsion breakers), antifoams and paraffin/asphaltene treating chemicals (Stephenson, 1991). Biocides used at the Buccaneer Gas and Oil Field (Rose and Ward, 1981; Zein-Eldin and Keney, 1979) were K-31 pentanectialdehyde and KC-14 alkyldimethyl benzyl chloride. There were replaced by a formulation of acrolein (2-propenal) of about 90 to 94% purity. Acrolein is a highly volatile, toxic, and reactive substance. Residual acrolein in produced water is scavenged by treatment with sodium bisulfite prior to discharge.

A number of investigators have summarized ranges of major contaminants in produced water effluents (Stephenson, 1992; Neff *et al.,* 1989; Middleditch, 1981). Table 6-1 summarizes some of the more recent data collected in the Gulf of Mexico for heavy metals and organics. Concentrations of all major components vary widely.

6.3 Radionuclides

Naturally occurring radioactive materials (NORM) present in produced water discharges include ^{226}Ra and ^{228}Ra, radon, ^{210}Pb, ^{210}Po, and probably other members of the ^{238}U and thorium decay series. Radionuclide concentrations in produced water are influenced by differences in solubility in the formation, and the nuclides are not in secular equilibrium with their parent compounds.

No comprehensive sampling program has been conducted to document the concentrations and amounts of radium being discharged to the Gulf of Mexico. There are, however, data available for a limited number of produced water discharges (Table 6-2). Few data are available for radionuclides other than radium.

6.4 Discharge Rates

Discharge rates of produced waters are important because the total amount of contaminant discharged to the environment per unit time (i.e. discharge rate times the concentration of contaminant in the discharge) is of more interest in terms of potential environmental effects than is simply the concentration in the effluent. Over the life of a well, the volume of water produced often increases as the volume of oil or gas decreases. In many of the older fields in the coastal waters of Louisiana and Texas, production may be 95% water and 5 percent oil or gas (Neff et al., 1989). Produced water discharge rates for oil wells in the Gulf of Mexico range from 0.5 bbl/d to more than 187,000 bbl/d (Table 6-3).

6.5 Application to Ecological Risk Assessment for Produced Water Discharges

Data describing the contaminant concentrations and discharge rates of produced water are needed to formulate the problem, identify potential impacts and describe the source term for an ecological risk assessment. Data presented here are limited and additional data derived from permit files and other sources may be needed in a site or area specific assessment.

Table 6-1. Ranges of metals and organic contaminant concentrations in produced waters (μg/l).

Contaminant	USEPA (1993d)[1]	SAIC (1991)[2]	St. Pé (1990)[3]
Organics			
benzene	2,978.69	52.48-9008.41	500 - 1,200
ethylbenzene	323.62	23.80-1009.27	9.4-29
chlorobenzene	19.47	47.50	
toluene	1,897.11	80.83-5305.44	150 - 580
total xylenes	695.03	12.51-439.37	58 - 210
di-n-butylphthalate	16.08	10.0	
2-butanone	1,028.96	275.00-1254.80	
phenol	1,538.28	10.00-379.83	240 - 890
p-chloro-m-creosol	25.24	10.00-364.17	
steranes	77.50		
triterpanes	78.00		
creosols			370 - 850
2,4 dimethylphenol	317.13	10.00-146.70	
PAHs			19 - 47
anthracene	18.51	10.00	ND - 0.56
benzo (a) pyrene	11.61	10.00	--
chrysene			<0.18 - 1.9
naphthalene	243.58	18.60-71.70	18 - 41
fluoranthene			<0.17 - 0.56
fluorene			0.33 - 2.8
phenanthrene			0.49 - 9.2
pyrene			<0.17 - 0.67
Metals			
aluminum	78.01	35.00-122.98	<0.44
arsenic	114.19	17.00-308.56	<0.11 - 87
barium	55,563.80	49.67-65853.79	1,521 - 4,644
boron	25,740.25	6850.96-38230.13	
cadmium	22.62	4.00	<0.06
chromium			<0.01 - 0.41
copper	444.66	6.00-135.50	<0.05 - 58
iron	4,915.87	672.29-8260.75	
lead	195.09	50.00	<0.08
mercury			<1.7
manganese	115.87	90.68-230.90	
nickel	1,705.46	30.00	<0.22
titanium	7.00	3.66-12.06	
vanadium			1.2 - 60
zinc	1,190.13	23.23-72.32	7.3 - 25

[1] Table IX-15 in USEPA (1993d), baseline effluent characteristics for priority pollutants achievable by BPT technology based on several industry and EPA databases.
[2] Three facility study, data abstracted from USEPA (1993d)
[3] 4 produced water discharges to coastal environments in Louisiana.

Table 6-2. Radionuclides in produced water discharges in the Gulf of Mexico (pCi/l).

Reference	^{226}Ra	^{228}Ra	$^{226}Ra+^{228}Ra$	^{210}Pb
Stephenson and Supernaw (1990)[1]	4 - 584	18 - 586	--	--
USEPA (1993b)[2]	0 - 450	0 - 560	--	--
Rabalais *et al.*, (1991)[3]	--	--	152 - 1156	0.11 - 12.7
St. Pé ' (1990)[4]	--	355 - 567	--	--
Continental Shelf Associates (1991)[5]	111 - 252	244 - 383	--	--
Continental Shelf Associates (1992b)[6]	111 - 225	240 - 520	--	<10.6
Steimle & Associates (1992)[7]	50.1 - 190	48.2 - 198	--	--
Neff *et. al.*, (1989)[8]	--	--	605 - 1215	--

1 42 Offshore platforms in Louisiana
2 83 outfalls in Louisiana territorial seas
3 7 discharges of OCS produced waters into Louisiana coastal environments
4 4 produced water discharges to coastal sites in Louisiana
5 3 coastal discharges in Louisiana
6 2 offshore platforms in the northern Gulf of Mexico
7 1 offshore, 2 inshore outfalls in Louisiana
8 4 platforms, 1 shallow bay, 3 offshore in Louisiana

Table 6-3. Produced water discharge rates for offshore and coastal discharges in the Gulf of Mexico.

Source	Discharge Rate (bbl/day)		
	mean	std. deviation	range
OOC[1]	6034	5852	8 - 22880
LDEQ[2]	8278	24418	0.5 - 187420
COMBINED[3]	7518	20175	0.5 - 187420

1 Offshore Operators Committee (42 offshore outfalls; Stephenson and Supernaw, 1990)
2 Louisiana Dept. of Environmental Quality (83 outfalls, coastal and offshore within state waters; USEPA, 1991)
3 Combined two data sets

7 PRODUCED WATER TOXICITY

7.1 Introduction

Produced water contains chemical and radioactive contaminants, and has properties that could cause harmful effects in marine organisms and ecosystems. These include elevated salinity, altered ion ratios, low dissolved oxygen, petroleum hydrocarbons and other organics, and heavy metals. Since most produced waters have dissolved solids concentrations higher than that of sea water (30-39 ppt), discharge of such waters to the ocean results in a localized elevation in salinity. Also, the concentration ratios of several major ions (particularly calcium and magnesium) in produced water may be markedly different than those of sea water, leading to adverse reactions in organisms in the receiving waters (Neff et al., 1989). However, mixing and dilution of produced water discharged into deep water offshore regions is rapid, and concentrations of contaminants may be undetectable more than a few meters from the discharge point. Conversely, coastal discharges into poorly flushed brackish or saline systems show a significant possibility for toxicity (St. Pé, 1990).

Toxicity tests are useful analytical tools because they can directly measure potential aquatic effects compared to chemical analyses which are difficult to extrapolate. This is particularly true in the case of complex effluents, such as produced water, where a broad range of toxicants can be present in low levels. Produced water test procedures usually use mortality as the measured response with results of acute tests expressed as an effluent median lethal concentration for an exposure duration of 96 hrs (96-hr LC_{50} equals the effluent concentration which results in the mortality of 50% of the test organisms in a 96-hr exposure period). This section summarizes available toxicity test data for produced water obtained in laboratory and field settings.

7.2 Laboratory Bioassays

Results of produced water bioassays conducted in the laboratory range from providing evidence of very low toxicity (Middleditch, 1984), to showing that produced water was highly toxic (Federal Register, 1992). This could be due to differences in the toxicity of the produced water, problems with protocols used in testing, or the presence of biocides in some discharges.

In evaluating these studies several potential flaws and uncertainties should be kept in mind. One problem concerns the maintenance of appropriate oxygen levels in the test medium. Adequate oxygen levels are usually obtained by aeration, but this can lead to losses of the lighter (more toxic) chemicals. Another significant problem with laboratory bioassays is that the composition of produced waters is almost inevitably altered during transportation and storage.

Bioassay media are prepared by volumetric additions of produced water to receiving water or filtered sea water according to standard protocols. Exposure concentrations may be measured in terms of nominal volume of produced water added per volume of dilution water. Storage samples should be monitored for production of hydrogen sulfide, generated by reduction of sulfur and sulfate by sulfate-reducing bacteria. This may affect the sample's toxicity.

7.2.1 NPDES Permitting Data

The largest produced water toxicity data base used in permitting applications consists of self-monitoring compliance data required by the Louisiana Department of Environmental Quality (LDEQ) discharge permits (Federal Register, 1992; Avanti Corporation, 1993; Table 7-1).

LC_{50}s for mysids (*Mysidopsis bahia*) ranged from 0.05% to >100% effluent, with a mean 96-hour LC_{50} of 12.1%. LC_{50}s for sheepshead minnow (*Cyprinodon variegatus*) ranged from 1.17% to >100%, with a mean of 27.4%.

Table 7-1. Toxicity data, Louisiana DEQ NPDES permits (modified from Avanti Corporation, 1993).

	Mysidopsis bahia				*Cyprinodon variegatus*		
	96-hr LC_{50}	Survival	Growth	Fecundity	96-hr LC_{50}	Survival	Growth
Number of Outfalls	241	226	221	150	239	221	218
Mean	12.1	4.51	5.92	6.44	27.4	8.04	8.23
Median	8.20	2.16	2.08	3.00	17.9	2.50	4.90
Minimum	0.05	0.04	0.06	0.13	1.17	0.14	0.15
Maximum	100	100	100	100	100	100	100
95th percentile	1.31	0.19	0.34	0.29	2.69	0.50	0.56

7.2.2 The Buccaneer Gas and Oil Field Study (BGOF)

The BGOF study has been summarized by Middleditch (1984) and Neff (1987). The BGOF, southeast of Galveston Texas, was 59.3 km^2 (22.9 sq. statute miles) in area. The field contained two production platforms, two quarters platforms, and well jackets surrounded by 13 satellite structures. Middleditch (1984) summarized the observations of two complex series of laboratory bioassays of produced water samples from the BGOF (Zein-Eldin and Keney, 1979; Rose and Ward, 1981). Apparently contradictory results were explained on the basis of presence or absence of biocides in the produced water effluents (Middleditch, 1984).

Biocides

Biocides were added to product streams to protect pipes and storage vessels by reducing formation of corrosive sulfates by sulfur oxidizing bacteria. During the first series of the BGOF study bioassays, gluteraldehyde (K-31) and alkyl-dimethylbenzyl chloride(KC-14), and a surfactant were used. To increase biocidal activity, these agents were replaced by acrolein, a very reactive toxin that is also highly volatile. Before discharge, produced water was treated for residual acrolein by using a scavenger, sodium bisulfite. Biocide treatment was not an apparent factor in the second series of bioassay tests because of removal of acrolein by sodium bisulfite. Acrolein was not detected in produced water sampled during the second series of the BGOF study (Middleditch, 1984).

Bioassay results

Acute toxicity (96 hr LC_{50}) tests were used in each of the two series of bioassays. The first series of bioassays (Zein-Eldin and Keney, 1979) used juvenile brown shrimp (*Penaeus aztecus*) and juvenile white shrimp (*P. setiferous*), under static conditions.

Samples of produced water for the first series were collected from June 1977 through February 1978. Acute toxicities in the first series were high (low LC_{50} values; 1,750 to 6,500 ppm of produced water diluted in sea water) when K-31 and KC-14 biocides were added to production streams, and low (high LC_{50} values may be >10,000 ppm) in the absence of biocide treatment. Although oxygen demand increased during the four days of static testing, there were no significant differences between control and experimental groups.

Other experiments in the first series used acute toxicity testing, after chronic exposure to either synthetic control feeds or experimental feeds (soaked in produced water). After preconditioning with either control or experimental diets, half of each dietary group was subjected to 96 hr exposure to 5,000 ppm of produced water in seawater, while the other half of each group was exposed to sea water only. Prior exposure to produced-water apparently increased the responses of juvenile shrimp to produced water, as indicated by :

1. lower LC_{50} values (higher mortality) in animals preconditioned with feed soaked in produced water; and
2. increased oxygen demand in acutely-tested groups of animals preconditioned with feed soaked in produced water.

The second series of bioassays (Rose and Ward, 1981) used larval and adult brown and white shrimp, barnacles (*Balanus tintinnabulum*), and crested blennies (*Hypleurochilus geminatus*), under static and nonstatic conditions. Produced water was sampled from summer 1978 through spring 1979. Results of testing at four different temperatures (18, 22, 24 and 25°C) indicated that 96 hr LC_{50} values were similar for adult crustaceans, shrimp and barnacles. Larval brown shrimp were the most sensitive test animals, and blennys were most resistant to the toxic effects of produced water.

LC_{50} values were generally lower at higher temperatures. Barnacles and blennys were more sensitive to unaerated than aerated produced water, a possible effect of ammonium bisulfite treatment, or differences in oxygen demand of produced water. Neff (1987) pointed out that aeration has a rapid stripping effect on volatile low molecular weight aromatic hydrocarbons, such as those found in produced water from the BGOF, and that these hydrocarbons may have contributed to the results for unaerated produced water.

7.2.3 Offshore Operators Committee (1992)

Produced-water toxicity data from offshore wells were submitted to USEPA by the Offshore Operators Committee (Federal Register, 1992). Seven-day chronic survival data from one company had a mean survival no effects concentration (NOEC) for mysids of 0.86% effluent (minimum 0.32%; maximum 1.86%) and a mean survival NOEC for sheepshead minnows of 1.0% effluent (minimum 0.26%; maximum 2.7%). Seven-day chronic survival data from another company showed a mean NOEC for mysids of 0.95% effluent (minimum <0.1%; maximum 5%).

7.2.4 St. Pé (1990)

St. Pé (1990) studied the toxicity of produced water discharged at 4 coastal sites in Louisiana. One effluent and one sediment sample were analyzed from each of the four study sites; Lirette site; Delta Farms site; Bully Camp site, and the, Lake Washington site. The effluent samples were analyzed using *C. variegatus* and *M. bahia* in toxicity tests.

<u>Test procedures</u> Acute toxicity tests were performed on one effluent sample, from each of the four study sites. A standard USEPA method was used (USEPA, 1985). LC_{50} (24-hr, 48-hr, and 96-hr) values with 95% confidence levels were calculated by the Binomial Method. Both salinity adjusted and unadjusted tests were run. The unadjusted data are emphasized by St Pé (1990) and are discussed below.

Effluent toxicity Results are summarized in Table 7-2. All tested samples were toxic to *M. bahia.* Salinity readings ranged from 115.2 in Delta Farms to 148.4 in Lake Washington. Salinity readings and toxicity levels did not follow a pattern in these samples; Lake Washington had the highest salinity reading and second highest 96-hr LC_{50} value (5.23%), Delta Farms had the lowest salinity reading and the second lowest 96-hr LC_{50} value (3.54%).

Although high salinity can result in toxicity to mysids, it did not appear to be the major cause of toxicity. The salinity range of the 96-hr LC_{50} values was 23.37 ppt to 31.45 ppt. These values are in the range at which mysids are successfully cultured. St. Pé (1990) concluded that toxicity of the effluents was due to a component of the effluent other than salinity.

Sheepshead minnows exhibited acute toxicity from all samples (Table 7-2). The salinity range of the 96-hr LC_{50} effluent concentrations was 26.82 ppt to 58.06 ppt. Again, the salinity and the toxicity levels did not follow similar patterns: Delta Farms had the lowest salinity reading and the highest LC_{50} value. Although high salinity can be toxic to *C. variegatus*, it does not appear to be the major cause.

Table 7-2. Acute toxicity (LC_{50} value) for four coastal Louisiana produced water effluents (St Pé, 1990).

	24-hour		**48-hour**		**96-hour**	
Test species	**mean**	**range**	**mean**	**range**	**mean**	**range**
Mysidopsis bahia	6.42%	3.81-7.74%	4.12%	3.64-6.37%	4.30%	2.64-5.77%
Cyprinodon variegatus	24.68%	13.62-35.36%	21.23%	7.16-35.36%	20.13%	7.16-33.78%

7.2.5 Enviro-lab (Federal Register, 1992)

Enviro-lab, Inc. conducted acute and chronic toxicity tests on produced water from West Delta Block 52 facility (Federal Register, 1992), Plaquemines Parish, Louisiana. The 96-hr acute lethality LC_{50} tests for *M. bahia* were 5.8% to 15.8% effluent and for *C. variegatus* were 1.5% to 8.1% effluent. Enviro-labs 7-day chronic tests indicated *M. bahia* survival, growth and fecundity NOEC to be respectively, 2.875%, 1.437% and 2.875% effluent. NOEC for *C. variegatus* survival was 1.437% , and the effluent and growth values were <1.437% effluent.

7.3 Field Studies

The few available *in situ* bioassays generally showed that toxicity is low, although there were indications that presence of biocides in the effluents increases toxicity. Field toxicity studies were done at the BGOF (Workman and Jones, 1979).

Workman and Jones (1979) conducted a three season study between the months of June 1977 and March 1978 to determine the effects of produced water discharges on fishes and macro-crustaceans of the BGOF.

In a fall field bioassay experiment (October 1977), 20 crested blennies and 15 seaweed blennies were caged beneath the discharge of a production platform, and 26 crested blennies and 16 seaweed blennies were held in a cage under a control well jacket. The exposure time was 45 hours for the production platform and 43 hours for the well jacket. No biocides were added to the effluent during this sampling period. The results were negative, there were no mortalities.

The experiment was repeated in February 1978, under normal conditions with biocides added to the effluent (Table 7-3). A combination of 40 individuals of the two species were placed in a cage under another platform and 39 blennies were place under a control well jacket, for 48 hrs. During this experiment, a few unidentified blennies were missing from both cages. At the end of the experiment 1 blenny was missing from the platform, and 5 blennies were missing from the control well-jacket (Table 7-3).

Lethal effects on these small reef fishes were exhibited only when biocide was added to the formation water. These effects appeared to be limited to the area immediately under the discharge. Habitat limitations created by the discharge also had an effect on the size and composition of the reef fishes inhabiting the discharge leg of one production platform.

Results of the field bioassay experiment indicate that the produced water discharge by itself had no apparent effect on species of blennies commonly found on the oil structures. Toxic effects strong enough to kill the blennies were observed when biocide was added to the produced water discharge.

Table 7-3. Field bioassay in February 1978 (adapted from Workman and Jones, 1979).

Exposure		Treatment 40 individuals		Control 39 individuals	
		Dead	Alive	Dead	Alive
28 hrs 15 min	*Crested Blennies*	1	18	N/O	N/O
	Seaweed Blennies	3	17	N/O	N/O
48 hrs	*Crested Blennies*	3	16	3	19
	Seaweed Blennies	20	0	0	12

N/O = not observed

7.4 Toxicity Studies on Sediments Near Outfalls

E.V.S. Consultants (1990) did a sediment quality triad study on a platform in the Gulf of Mexico, near Matagorda Island, Texas. Chemical enrichment of the sediment was highest at five stations within 25 m from the platform. Analyses of benthic infauna found subtle differences, that were not easy to distinguish, between these five stations. In contrast to the laboratory evidence of potential toxicity, field-collected data provided evidence of faunal enrichment (more taxa, higher total abundance and abundance of specific taxa) within a 25 m radius of the platform, attributable to either the introduction of the platform structure or a hormetic effect of the contaminant mixture in the sediment. It was suggested that the field observations were due to offsetting factors in the environment, such as principal habitat, grain size effects, and adaptation.

7.5 Summary

Based on available laboratory toxicity testing and field studies, several authors (Middleditch, 1984; Neff, 1987) have indicated that produced water toxicity is generally low, and in large part attributable to the presence of biocides. Studies of individual discharges have found some with relatively high toxicities (Avanti Corporation, 1993; Table 7-1). Under ambient marine conditions, toxicity of produced water in the water column, near the discharges, should not be hazardous to marine animals, especially if these creatures are mobile.

7.6 Application to Ecological Risk Assessment for Produced Water Discharges

Studies of the toxicity of produced water discharges can be used to assess potential effects on organisms, and the geographic location of potential toxicity in sediments. Problems with using these data include the importance of biocides in causing toxicity, and the change in toxicity that occurs with time and space. Site-specific toxicity tests are preferable to data derived from other sources.

8 BIOACCUMULATION OF MAJOR TOXIC COMPONENTS OF PRODUCED WATERS

8.1 Introduction

The amount of contaminant that accumulates in an organism, or its organs, is used as an indicator of dose. This accumulation is expressed as bioconcentration or bioaccumulation. There is confusion in the use of these terms. USEPA (1989b) defined bioaccumulation as the uptake and retention of a contaminant by an organism. Bioconcentration is a specific case of bioaccumulation wherein the concentration of a contaminant in an organism's tissue exceeds its concentration in the medium around the organism. Bioaccumulation is also used as a description of an overall phenomenon in which bioconcentration is a contributing parameter (Franke et al., 1994).

Although it is an indicator of bioavailability, bioaccumulation is of limited use for assessing ecological risks in the real world because it is difficult to assign a detrimental effect to accumulation of a single agent, when the agent is a component of complex mixtures that organisms typically encounter in the environment.

Bioaccumulation of a contaminant is the net result of two basic processes during a specific period of exposure:

1. uptake of a contaminant into an organism; and
2. removal of a contaminant from the organism.

Direct uptake may occur by transfer of a contaminant from external contact with a medium containing the contaminant (e.g., water column or sediment). Direct uptake can also occur by exposure of internal surfaces (gills, alimentary canal) to a contaminant bearing medium. Indirect uptake may be the result of a food-chain transfer, where the contaminant is released in the body as a result of metabolic processes. Metabolism may also convert a relatively harmless xenobiotic agent into a harmful product that is retained in the body. Therefore bioaccumulation involves accumulation in organs and tissues.

Removal may be a simple process of direct elimination of an unaltered contaminant from the body to the environment. Indirect removal can take place by metabolism of contaminants, including either sequestering in structures (scales, exoskeleton) or transfer of metabolites from the body to the environment.

The most direct estimates of bioaccumulation are achieved by measurements of biotic and media specimens simultaneously obtained in the field. Although measurements on field-collected specimens are a preferred method for estimating site-specific bioaccumulation, they may be fiscally prohibitive (Lee, 1992). There is limited information on bioaccumulation of specific contaminants from produced waters in marine and estuarine organisms in the Gulf of Mexico. Section 8.2 summarizes the data that describe concentrations of organics, metals and radionuclides in biota near produced water discharges. Because the data are limited, a modeling approach to estimating concentrations of organics, metals and radionuclides in animals is often used. Section 8.3 summarizes the limited bioaccumulation factors (BAFs) available for organics, metals and radionuclides.

8.2 Field Data

"Measuring tissue residues in field-collected sediment-dwelling organisms is the most straightforward method of assessing bioaccumulation" (Lee, 1992). Aquatic areas that contain high concentrations of contaminants are associated with detrimental effects to biota (Lamberson *et al.*, 1992). This is especially true for animals that are exposed to contaminated sediments, either directly or as a result of trophic activity.

In most field studies detrimental effects occur in a varying milieu of complex mixtures of chemicals, both carbon based and inorganic, and complex physical conditions. Therefore, any attribution of effects to a particular type of contaminant is highly uncertain. Besides expense, obtaining sufficient tissue mass to analyze a full suite of contaminants is a major problem in field sampling, especially when perturbation has already reduced the available biomass (Lee, 1992). An alternative is to place appropriate biota in the field, in adequate numbers and for an adequate duration, to reach equilibrium with their environment. The subsections below describe studies on bioaccumulation of specific contaminants in native animals, and in animals deployed in the field, that are pertinent to investigations on produced waters.

8.2.1 Organic and Hydrocarbon Compounds

Bioavailability of hydrocarbon compounds is directly related to solubility in aqueous media. For sediments this refers to desorption into interstitial water, with a direct relationship to sediment grain size, and an inverse relationship to total organic content of the sediment (Capuzzo, 1987). Capuzzo reviewed bioaccumulation of petroleum related hydrocarbons (*many of which are associated with produced waters*) and reported that the process occurs in every investigated phylogenetic group. One of the major uncertainties in measurements of hydrocarbon compounds is determination of petrogenic and pyrogenic contributions to the suite of contaminants found at a site (Boesch and Rabalais, 1989a; St. Pé, 1990). A fossil fuel pollution index (FFPI - Boehm and Farrington, 1984) is used to indicate the relative percentage of total PAHs that are derived from petroleum sources, where concentrations of alkylated naphthalene, phenanthrene and dibenzothiophene are high relative to the parent compounds.

Benthic crustaceans accumulate organic and hydrocarbon compounds through contact with sediments and water above the sediments, and from food. Depuration after long-term accumulation may be rapid, but considerable fractions of the uptake may persist for years in clean water (Capuzzo, 1987). In addition to species, contaminant-mixture, and site-specific variations, there are important considerations of differential uptake, concentration and retention in organs.

Bivalve mollusks are among the most frequently investigated biota in contaminated habitats, largely because of their association with sediments and the water just above the sediments. Variability in bioaccumulation depends on species-specific physiological characteristics and environmental properties. PAHs remain relatively untransformed in mollusks because of the nature of their mixed function oxidase (MFO) systems (St. Pé, 1990). Uptake of PAHs, and elimination within 24 hours after exposure to clean sea water, are relatively rapid.

Neff *et al.* (1976) studied turnover of four PAH compounds associated with produced waters, and suggested that rates of uptake and retention are probably related to octanol/water partition coefficients. Highly water-soluble naphthalene (a light molecular-weight fraction compound) was quickly taken up, and was released rapidly upon depuration of the clam, *Rangia cuneata*. Phenanthrene accumulation was the fastest, and release the slowest among the compounds. Chrysene and benzo(a)pyrene, low solubility compounds, had the slowest rates of accumulation, and the slowest rates of release. Several studies on various bivalve species have confirmed these observations, although species and site-specific variations were observed (Capuzzo, 1987).

Oysters (*Crassostrea virginica*) and mussels have been used extensively in coastal and estuarine monitoring studies, including placement of indigenous animals in the field. Measurements were made by Boesch and Rabalais (1989a) on: *C. virginica* taken from two inshore locations associated with produced waters (Bayou Rigaud, and East Timbalier Island) and a reference site (LUMCON Port Fourchon Laboratory); and ribbed mussels (*Geukensia demissa*) taken from an inshore location associated with produced waters (Pass Fourchon) and a reference site (LUMCON Marine Center, Cocodrie). Results were reported for Total PAH and Total Saturated Hydrocarbons (Table 8-1), and metals (see section 8.2.2). FFPI values (>0.5) indicate that the test site contaminants were petrogenic in origin, while the reference site contaminants were pyrogenic in origin (those at LUMCON Port Fourchon Laboratory possibly originating from heavy vessel traffic).

Table 8-1. Fossil fuel pollution index (FFPI), and concentrations (ppb) of total polycyclic aromatic hydrocarbons (PAH) and total soluble hydrocarbons (SH) in native bivalves at inshore produced water sites and reference sites* (Boesch and Rabalais, 1989a).

Organism Location	FFPI	Total PAH (ppb)	Total SH (ppb)
Crassostrea virginica			
Bayou Rigaud	0.87, 0.88	2200, 3400	330×10^3
East Timbalier	0.81	3400	550×10^3
LUMCON Port Fourchon*	0.24-0.48	240-2000	$(68\text{-}190) \times 10^3$
Geukensia demissa			
Pass Fourchon	0.72, 0.77	630, 880	$(120\text{-}180) \times 10^3$
LUMCON Cocodrie*	0.00	15	33×10^3

* reference sites

In a bioaccumulation study, oysters were deployed at two oil platform or oil-related locations (Bayou Rigaud and Pass Fourchon) and two reference sites (Bayou Tartellon and Bay Champagne) in coastal waters of Louisiana (Rabalais *et al.*, 1991; Tables 8-2, 8-3). Six sites at each of the oil-related locations represented different spatial-dilutions (distance) of contaminants from produced water discharges. Mortality and lesser weight gains of oysters in the contaminated areas may be related to contamination by produced waters and duration of exposure. Measurements were made of environmental conditions, and contaminant concentrations in sediments and oysters at each site, in a 14-day and a 27-day exposure period. These periods may have been insufficient to bring the animals into equilibrium with ambient conditions, but were necessitated by predation and problems with fouling-induced mortality. A previous investigation (Boesch and Rabalais, 1989a) detected total PAH concentrations in oysters native to Bayou Rigaud, an order of magnitude greater than the concentrations shown in Tables 8-2, 8-3. Total saturated hydrocarbon concentrations were more than two orders of magnitude greater. Low values after 27 days of deployment were attributed to depletion of lipid content by spawning (Boesch and Rabalais, 1989a).

St. Pé (1990) deployed oysters in three oil producing areas and one reference site in low-energy coastal waters of Louisiana (Table 8-4). The values for PAH concentrations in the study by St. Pé (1990) were expressed on a wet weight basis, in contrast to the use of concentrations in lipid by Rabalais *et al.* (1991). When the values are normalized for lipid content of oysters (5% - St. Pé, 1990), the concentrations detected after 27 days exposure by Rabalais *et al.* (1991) are generally higher than those reported by St. Pé, (1990) after 30 days exposure (Table 8-5). Possible explanations might be differences in ambient concentrations, biological status (health) of the test organisms, and physical and chemical conditions of the test areas.

Table 8-2. Measured concentrations (ppm, lipid) of hydrocarbons and PAHs in oysters; controls and 14-day exposures at reference sites and oil production areas (data from Rabalais *et al.*, 1991).

Contaminant	Control	Reference (Two Sites)	Pass Fourchon[1]	Bayou Rigaud[1]
Total Saturated Hydrocarbons	680	660, 640	890-3100	760-2300
Total PAH	56	38, 39	170-900	120-330
Anthracene	trace	trace, ND	ND -1.5	ND -trace
Chrysene	7.5	0.50, trace	0.80-13	trace-0.76
Naphthalene	0.40	0.50, 0.40	0.20-0.90	trace-0.34
Fluoranthene	1.8	1.6, 1.2	2.5-19	1.1-2.0
Fluorene	0.40	0.30, trace	0.70-2.4	ND -0.60
Phenanthrene	0.80	0.40, 0.40	1.2-8.0	0.71-1.8
Pyrene	1.9	1.2, 0.90	2.1-14	0.76-1.7

[1]Range for six sample sites

Table 8-3. Measured concentrations (ppm, lipid) of hydrocarbons and PAHs in oysters; controls and 27-day exposures at reference sites and oil production areas (data from Rabalais *et al.*, 1991).

Contaminant	Control	Reference	Pass Fourchon[1]	Bayou Rigaud[1]
Total saturated Hydrocarbons	530	1100, 500	1100-2000	980-2000
Total PAH	6.8	57,24	110-480	36-170
Anthracene	ND	ND , ND	ND -0.47	ND
Chrysene	trace	0.78, trace	0.92-5.2	trace-0.57
Naphthalene	trace	trace, trace	trace-0.50	trace-0.29
Fluoranthene	0.55	2.1, 0.84	1.9-7.4	0.75-0.98
Fluorene	ND	ND , ND	0.32-1.3	ND -trace
Phenanthrene	0.20	trace, trace	0.41-2.3	trace-0.94
Pyrene	0.46	1.5, 0.63	1.7-6.0	0.48-0.83

[1]Range for six sample sites

Table 8-4. Measured concentrations (ppb, wet weight) of PAHs and volatile hydrocarbons in oysters; 30-day exposures at reference site and oil production areas (data from St. Pé, *et al.*, 1990).

Contaminant	(Reference) Caillou Lake	Lirette	Bully Camp	Lake Washington
Total PAH	trace	22	130	280
Anthracene	ND	ND	ND	ND
Chrysene	ND	ND	trace	72
Naphthalene	ND	ND	ND	ND
Fluoranthene	trace	trace	41	37
Fluorene	ND	ND	ND	ND
Phenanthrene	ND	trace	28	66
Pyrene	trace	22	61	46
Total Volatiles	ND	372	17	3
Benzene	ND	66	7	ND
Toluene	ND	220	9	1
Ethyl Benzene	ND	86	1	1

Table 8-5. Comparison of ranges of concentrations of PAH (ppb, wet weight)[1] from two investigations of produced waters, by deployment of oysters, *Crassostrea virginica*, in Louisiana coastal waters.

Contaminant	(Rabalais *et al.*, 1991)[2]			(St. Pé *et al.*, 1990)[3]	
	Control	Reference	Test Sites	Reference	Test Sites
Total PAH	340	2850, 1215	1800-2400	trace	22-280
Anthracene	ND	ND	ND.-20	ND	ND
Chrysene	trace	390, trace	trace-260	ND	trace-72
Naphthalene	trace	trace	trace-25	ND	ND
Fluoranthene	27.5	105, 42	38-370	trace	trace-41
Fluorene	ND	ND	ND -65	ND	ND
Phenanthrene	10	trace	trace-120	ND	trace-66
Pyrene	23	75, 32	24-300	trace	22-61

[1]Data of Rabalais *et al.* (1991) multiplied by 20×10^3.
[2]Two reference and 12 test sites.
[3]One reference and 3 test sites.

Presley, Boothe and Brooks (1988) summarize a number of field studies performed for platforms in the Gulf of Mexico. High molecular weight hydrocarbons have been found in association with production platforms and coastal produced water discharge sites.

At the Buccanear Gas and Oil Field petroleum hydrocarbons were found in concentrations as high as 4 ppm in the barnacle *Balanus tintinnabulum* collected from the platform, and as high as 16 ppm in barnacles collected from an adjacent flame stack (Middleditch, 1984). Table 8-6 summarizes concentrations of alkanes in biota collected near the platform.

Table 8-6. Mean concentrations of alkanes in biota sampled at the Buccanear Oil and Gas Field (data from Middleditch, 1984).

Species	mean alkane concentration (ppm)		
	muscle	liver	whole
crested blenny (*Hypleurochilus geminatus*)			6.8
sheepshead (*Archosargus probatocephalus*)	4.6	6.1	
spadefish (*Chaetodipterus faber*)	0.6	2.0	
red snapper (*Lutjanus campechanus*)	1.1	1.3	
brown shrimp (*Penaeus aztecus*)			2.5 (maximum)

8.2.2 Metals

Tables 8-7, and 8-8 demonstrate concentrations of metals found in oysters, *C. virginica*, deployed to study produced water contamination of estuarine waters on the coast of Louisiana (Rabalais, *et al.*, 1991), as described in the previous section. Data from a previous study (Boesch and Rabalais, 1989a) suggest problems with the use of deployed animals, as opposed to measurements taken from native animals. Rabalais *et al.* (1991) suggested that deployment for 14 days and 21 days was insufficient for the oysters to reach equilibrium with ambient conditions. The data also suggest that transplantation *per se* may produce uncharacterized stresses that interfere with the animals ability to reach equilibrium, especially when comparisons of reference and control values are done for the two durations of exposure.

Table 8-7. Measured concentrations (ppm, dry weight) of metals in oysters; controls and 14-day exposures at reference sites and oil production areas (data from Rabalais *et al.*, 1991).

Contaminant	Control	Reference	Pass Fourchon[1]	Bayou Rigaud[1]
Aluminum	2600	4100, 3900	3800-5700	4000-6600
Arsenic	54	190,170	170-220	160-500
Barium	39	53, 400	20-180	6.1-61
Cadmium	1.9	2.3, 10	1.6-2.6	2.1-3.3
Chromium	3.8	6.4, 5.9	5.8-8.4	7.0-14
Copper	47	59, 79	31-93	33-71
Lead	0.66	0.15,0.84	0.32-0.63	0.36-0.54
Mercury	ND	0.49, 0.19	ND-0.5	ND-0.77
Nickel	3.9	4.5, 4.3	2.8-5.4	2.8-16
Vanadium	16	85, 85	84-130	80-300
Zinc	1500	2100, 3200	1200-3400	1200-3000

[1]Range for six sample sites

Table 8-8. Measured concentrations (ppm, lipid) of metals in oysters; controls and 27-day exposures at reference sites and oil production areas (data from Rabalais *et al.*, 1991).

Contaminant	Control	Reference	Pass Fourchon	Bayou Rigaud
Aluminum	ND	ND, ND	ND-1.3 x 10^5	ND-1.1 x 10^5
Arsenic	1900	2600, 1900	ND-1000	ND-400
Barium	170	190, 250	ND-160	ND-170
Cadmium	32	110, 69	1.9-69	9.9-77
Chromium	180	270, 190	ND-100	ND-130
Copper	950	1500, 1300	660-1800	810-2800
Lead	35	44, 37	ND-25	ND
Mercury	260	370, 270	ND-120	ND-1300
Nickel	450	560, 540	ND-230	ND-320
Vanadium	520	770, 640	ND-310	ND-220
Zinc	170	390, 320	44-290	93-540

[1]Range for six sample sites

Presley, Boothe and Brooks (1988) summarized results of field studies (both baseline studies and platform contamination studies) that measured trace metal concentrations in biota in the Gulf of Mexico and concluded that the data suggested no significant trace metal contamination.

Table 8-9 is an example of the range of metal concentrations found in three southeastern teleostean fish species (whole fish, liver and fillets) that are not associated with oil and gas production discharges.

Table 8-9. Concentration (ppm, dry weight) ranges[1,2] in three species of finfish (whole fish, liver and fillets), captured in four South Carolina estuaries (abstracted from Matthews, 1994).

Metal	Red Drum *Sciaenops ocellatus*	Flounder *Paralichthys lethostigma*	Seatrout *Cynoscion nebulosus*
Arsenic	ND - 9.56	ND - 22.5	ND - 13.3
Cadmium	ND - 0.42	ND - 0.78	ND - 0.50
Chromium	ND - 1.68	ND - 2.28	ND - 3.46
Copper	ND - 52.9	ND - 22.2	ND - 19.0
Lead	ND - 9.68	ND - 5.78	ND - 3.65
Mercury	ND - 96	ND - 743	ND - 687
Nickel	ND - 2.85	ND - 12.6	ND - 8.21

[1]Mercury in ppb
[2]ND = <0.90 ppb (Hg), <0.010 ppm (As and Cd), <0.040 ppm (Cu), <0.050 ppm (Pb), 0.060 ppm (Cr and Ni)

8.2.3 Radionuclides

Field studies of radium in organisms near produced water discharges include measurements of radium concentrations in: native animals collected near offshore platforms, native animals collected near coastal platforms, and oysters (*Crassostrea virginica*) deployed near platforms in field experiments.

8.2.3.1 Offshore Platforms: Native Organisms

Continental Shelf Associates (1992b)

Samples of biota from near two offshore platforms (at Eugene Island and Ship Shoal) were analyzed for radium. Samples were taken at 0, 30, 100, 300 and 2000 meters from the platform. Fishes were caught by hook and line fishing from the platform, and stone crabs and barnacles were taken from the legs of the platforms. The edible parts of fish and crustaceans were analyzed separately from the inedible bone, skin and exoskeleton. The edible parts of mollusks were analyzed for radium.

Table 8-10 gives the measured radium concentrations in the edible portions of organisms sampled at the Eugene Island platform. Table 8-11 gives the measured concentrations of radium in organisms at Ship Shoal. Barnacles and stone crabs were also taken from the platforms and analyzed for radium. More detailed data and information are given in Continental Shelf Associates (1992b).

Table 8-10. Radium concentrations measured in organisms near the Eugene Island platform.

Distance (meters)	FISH		CRABS	
	^{226}Ra (pCi/g)	^{228}Ra (pCi/g)	^{226}Ra (pCi/g)	^{228}Ra (pCi/g)
0[1,2]	ND	ND	<0.42	ND
50[1,2]	<0.05	ND	ND	<1.64
100[1,2]	<0.31	<1.45	<0.51	4.12
300[1,2]	ND	ND	ND	ND
2000[1,2]	<0.03	<0.37	ND	<3.54
platform[3]	ND	<0.77		
platform[4]	ND	<0.69		

1 fillet, *Arius felis*
2 edible parts, spider crab, *Libina emarginata*
3 fillet, sheepshead, caught by hook and line from the platform
4 fillet, red snapper, caught by hook and line from the platform
ND: not detected; "<" : detected at a concentration less than the quantitation limit

Table 8-11. Radium concentrations measured in organisms sampled near the Ship Shoal platform.

Distance (meters)	FISH		CRABS	
	^{226}Ra (pCi/g)	^{228}Ra (pCi/g)	^{226}Ra (pCi/g)	^{228}Ra (pCi/g)
0[1,2]	ND	ND	<0.05	<1.41
50[1,2]	<0.51	ND	0.16	3.51
100[1,2]	ND	0.93	0.07	0.60
300[1,2]	<0.54	1.78	0.18	1.96
2000[1,2]	ND	ND	<0.45	ND
platform[3]	ND	0.07		
platform[4]	ND	ND		

1 fillet, *Arius felis*
2 edible parts, spider crab, *Libina emarginata*
3 fillet, red snapper, caught by hook and line from the platform
4 fillet, bluefish, caught by hook and line from the platform
ND: not detected; "<" : detected at a concentration less than the quantitation limit

Steimle & Associates (1992)

This study included water, sediment and organism samples taken on and near an offshore platform (South Timbalier). Organisms were taken from the platform legs at depths of 10 to 40 meters below the water surface. Table 8-12 gives the concentrations of radium measured in fishes, crabs and mollusks taken on or near the South Timbalier platform. More detailed data and information are presented in Steimle & Associates (1992), including concentrations measured in barnacles.

8.2.3.2 Coastal Platforms: Native Organisms and Deployed Oysters

Continental Shelf Associates (1991): Native Organisms

Data collected included ^{226}Ra and ^{228}Ra concentrations in water, sediment, fishes, shellfish and crustaceans at three coastal produced water outfalls and three background stations in Louisiana (Continental Shelf Associates, 1991). The three outfalls were located in canals along the Louisiana coast: English Bay, Golden Meadow and Avery Island.

Table 8-12. Radium concentrations measured in organisms sampled from the legs of the South Timbalier platform.

FISHES[1]		CRUSTACEANS[2]		MOLLUSKS	
^{226}Ra (pCi/g)	^{228}Ra (pCi/g)	^{226}Ra (pCi/g)	^{228}Ra (pCi/g)	^{226}Ra (pCi/g)	^{228}Ra (pCi/g)
ND	ND	0.1	ND	0.1 [3]	<0.8 [3]
ND	2.7	0.1	ND	0.1 [4]	1.9 [4]
ND	2.1	0.1	<0.2	0.2 [5]	<0.8 [5]
ND	ND	ND	<0.2		
ND	ND	ND	ND		
ND	ND	0.2	1.5		
		ND	ND		
		0.1	<0.8		
		0.1	<0.9		
		1.3	<0.9		
		ND	2.0		

1 whole crested blenny, *Hypleurochilus geminatus*
2 whole stone crab, *Menippe mercenaria*
3 edible part, bicolor purse oyster, *Isogomon bicolor*
4 edible part, transverse ark clam, *Anadara transversa*
5 edible part, scissor datemussel, *Lithophaga aristata*
ND: not detected
"<" : detected at a concentration less than the quantitation limit

Radium concentrations were measured in water and sediment at stations located 25 and 50 feet from the outfalls, in three directions (the outfalls were located along one side of a canal). Biological samples were taken within 50 feet of the discharge point. All mollusk samples were oyster (*Crassostrea virginica*), and all fish samples were seatrout (*Cynoscion* sp.). Crustaceans sampled at Golden Meadow were crabs (*Callinectes* sp.) and at English Bay and Avery Island 3 were shrimp (*Penaeus* sp.). Oysters were removed from the shell before analysis. Fish and crustacean samples were whole-body samples, including bone, skin and exoskeleton. Table 8-13 gives the concentration data for ^{226}Ra and ^{228}Ra at the three sites and the six background reference stations.

Steimle & Associates (1992): Native Organisms

Steimle & Associates (1992) measured the concentration of ^{226}Ra, ^{228}Ra and ^{210}Pb in a study of two coastal produced water discharges. The Golden Meadow site is located on a canal, and the Quarantine Bay site is an open bay discharge. Tissue samples were collected using commercial crab traps at reference stations and at stations 50 meters from the discharge. Biota sampled at Golden Meadow site were all blue crab (*Callinectes sapidus*, soft and hard tissue analyzed separately). At the Quarantine Bay site, samples of the blue crab (*Callinectes sapidus*) and hardhead catfish (*Arius felis*) were analyzed for radionuclides (soft and hard tissue analyzed separately). Tables 8-14 and 8-15 summarize the data for these two sites.

Rabalais *et al.*, (1991): Deployed Oysters

Rabalais *et al.* (1991) measured accumulated total radium (^{226}Ra + ^{228}Ra) in oysters (*Crassostrea virginica*) deployed for two time periods (April and May 1990) near outer continental shelf (OCS) discharges located in coastal Louisiana canals (Pass Fourchon, Bayou Rigaud). No radium was detected in control animals (< 0.2,2.2) pCi/g. At Pass Fourchon, total radium exceeded detection limits at stations located at the discharge point (4.3,2.2; 2.1,2.2 pCi/l) and 200 m from the discharge (1.1,2.2; <0.2,2.2 pCi/l). No radium was detected in oysters located further from the discharges. At Bayou Rigaud radium above the detection limit was measured only at one station located within about 200 meters from one of the two discharge points during the April deployment.

St Pé (1990): Deployed Oysters

St. Pé (1990) deployed oysters (*Crassostrea virginica*) at three sites in a study of produced water impacts in low-energy coastal environments in Louisiana. Oysters placed at two of the study sites, and the control site did not accumulate ^{226}Ra above detection limits (0.09 - 0.17 pCi/g). At the Lirette Field site, deployed oysters had a ^{226}Ra concentration of 3.1 pCi/g.

Table 8-13. Radium concentrations measured in organisms sampled in study of coastal produced water discharges (Continental Shelf Associates, 1991).

STATION	FISHES[1]		MOLLUSKS[2]		CRUSTACEANS[3]	
	^{226}Ra (pCi/g)	^{228}Ra (pCi/g)	^{226}Ra (pCi/g)	^{228}Ra (pCi/g)	^{226}Ra (pCi/g)	^{228}Ra (pCi/g)
English Bay	0.014	0.005	0.007	0.003	0.07	0.094
Golden Meadow	0	0.022	0.008	0.011	0.07	0.025
Avery Island	0.041	0.012	NS	NS	0.125	0.243
Ref 1a	NS	NS	0.001	0.036	0.014	0.004
Ref 1b	0.005	0.021	0.004	0.011	0.024	0.025
Ref 2a	0.002	0.03	0.004	0.003	NS	NS
Ref 2b	0.031	0.013	0.001	0.01	0.07	0.033
Ref 3a	0.018	0.058	NS	NS	0.016	0.058
Ref 3b	0.03	0.017	NS	NS	0.077	0.045

1 seatrout (*Cynoscion* sp.)
2 oyster (*Crassostrea virginica)*
3 at English Bay and Avery Island, shrimp (*Penaeus* sp.); at Golden Meadow crab (*Callinectes* sp.)
NS = no sample

Table 8-14. Radionuclides in blue crab at Golden Meadow.

STATION	^{226}Ra (pCi/g)	^{228}Ra (pCi/g)	^{210}Pb(pCi/g)
Ref ST	<0.02	ND	<0.06
Ref HT	2.08	0.9	<0.22
50N ST	<0.03	ND	<0.11
50N HT	1.26	1.1	<0.24
50S ST	0.03	ND	ND
50S HT	0.81	0.3	<0.19

ST = soft tissue of blue crab (*Callinectes sapidus*)
HT = hard tissue (exoskeleton) of blue crab
ND = not detected

Table 8-15. Radionuclides in blue crab and hardhead catfish at Quarantine Bay.

STATION	Blue Crab[1]			Hardhead Catfish [2]		
	^{226}Ra (pCi/g)	^{228}Ra (pCi/g)	^{210}Pb (pCi/g)	^{226}Ra (pCi/g)	^{228}Ra (pCi/g)	^{210}Pb (pCi/g)
50NW ST	ND	ND	ND	<0.07	ND	ND
50NW HT	0.49	<0.7	ND	0.09	<3.0	ND
50SE ST	NS	NS	NS	ND	ND	ND
50SE HT	NS	NS	NS	0.07	<0.7	<0.5
1000NW ST	ND	<0.8	ND	ND	ND	ND
1000NW HT	0.3	<0.8	ND	ND	ND	ND
1000NE ST	ND	ND	ND	ND	ND	ND
1000NE HT	0.3	<0.7	ND	ND	ND	ND

1 *Callinectes sapidus*
2 *Arius felis*
ST= soft tissue
HT = hard tissue, bone and exoskeleton

8.3 Bioaccumulation Factors

8.3.1 Bioaccumulation Factors for Polyaromatic Carbon Compounds

Hellou *et al.* (personal communication) recently investigated bioaccumulation of polyaromatic carbon compounds (PAC) in muscle tissue from winter flounder exposed to various concentrations of crude oil in sediment during a four month period. Several levels of bioaccumulation appeared to occur independent of concentration in sediment over wide ranges. The more soluble PAH displayed higher BAF values. For example substituted naphthalenes presented the following order of BAF values from highest to lowest:

- monosaturated methyl and ethyl naphthalenes;
- dimethyl naphthalenes;
- trimethyl naphthalenes;
- C-1 tricyclic naphthalenes.

Hellou et al. (1994) observed that petroleum-derived PAC bioaccumulation begins at sediment concentrations of total PAC greater than 0.27 mg/g (dry weight), while elimination of conjugates begins at sediment concentrations greater than 10 mg/g (dry weight; Hellou and Upshall, 1994). In flounder muscle, the mean concentrations of specific PAH compounds were always greater than the median concentrations of those compounds.

Hellou (1995), Hellou and Upshall (1994, 1995) and Hellou *et al.* (1994) detected only low molecular weight PAH in flounder muscle. PAH compounds with larger numbers of rings were probably biodegraded to smaller compounds. The time for a large PAH to reach equilibrium can exceed the lifetime of the fish. A fugacity model was used to explain BAF values obtained for smaller water-soluble compounds (Hellou et al., 1995). These compounds reach equilibrium levels soonest in exposed fish, whereas the values for less soluble compounds reflect non-attainment of equilibrium, coupled with other contributory mechanisms.

The BCF for chrysene, a high molecular weight PAH was calculated to be 120 to 280. In contrast, Hellou *et al.* (1995) cited studies that reported higher BCF values for PAH in mussels and crustaceans (organisms with low capability to transform PAHs), than for fishes.

Hellou *et al.* (1995) reached the following conclusions. Near equilibrium between flounder muscle and sediment (dry weight/dry weight), regardless of K_{ow} and the concentrations of PAC in the sediment, BAF_{pac} is approximately 5.5. For the more soluble compounds, such as naphthalenes, the range of BAF values is 0.03 to 12. For large parental PAH (fluorene, pyrene, chrysene, and high molecular weight PAH) and less soluble PAC, which fail to reach equilibrium, the range is 0.1 to 0.01. The major factor in the low BAF for larger PAC is slow uptake, possibly combined with biodegradation of the material taken up and slow release from sediments.

8.3.2 Bioaccumulation Factors for Inorganic and Organic Chemical Toxicants

Only limited data are available for BAFs for organic compounds and metals in saltwater organisms. BCFs available in the literature should be reviewed in the context of their relevance and appropriateness for application to a specific organism and specific circumstance. Generic values are often used in screening-models (Table 8-16) (Stenge and Peterson, 1989; Napier *et al.,* 1980; Strenge *et al.,* 1986), and for organics may be calculated from octanol-partition coefficients. USEPA estimated accumulation factors for selected trace metals and petroleum components in produced waters (Table 8-17; Avanti Corporation, 1993). BCFs for some contaminants are available in the USEPA AQUIRE database (Russom *et al.,* 1991). Table 8-18 gives maximum BCFs for three taxa of interest from the AQUIRE database.

8.3.3 Bioaccumulation Factors for Radium

8.3.3.1 Generic Bioaccumulation Factors

Generic or average values for bioaccumulation factors (sometimes referred to as concentration factors) were suggested based on surveys of published data (Thompson et al. 1972; Cherry and Shannon 1974; IAEA 1982; IAEA 1985). Generic bioaccumulation factors are meant for use in radiological assessment models for estimating the dose to man from a number of pathways. Commonly used models contain default bioaccumulation factors for a number of radionuclides and groups of organisms. The generic factors suggested by IAEA (1982) are used by many authors and models and are given in Table 8-19. IAEA (1985) revised these estimates upward for mollusks and fish and based the recommended bioaccumulation factors on the highest values reported in the literature (Table 8-19).

Meinhold *et al.* (1993) described the generic BAFs in IAEA, 1982 (100 for marine fish, mollusks and crustaceans) as consistent with published ranges of BAFs. However, the published values were based on samples taken from water with relatively low levels of ^{226}Ra (0.00052 - 0.518 Bq/l; 0.014 - 14.0 pCi/l).

The bioaccumulation factors commonly used in dose assessment studies (IAEA, 1982, 1985) are appropriate for screening-level assessments, but may overestimate the concentrations in edible portions of marine organisms. The factors suggested by IAEA tend to be based on the highest values reported in the literature, and appear to represent conservative values for radium.

Table 8-16. Generic bioaccumulation factors (from Strenge and Peterson, 1989).

Contaminant	Aquatic Bioaccumulation Factors (l/kg)	
	Finfish	**Shellfish**
Metals		
aluminum	10	63
arsenic	1	40
barium	4	0.2
cadmium	200	2000
chromium	20	2000
copper	50	400
lead	100	100
mercury	200000	200000
nickel	100	100
silver	2.30	770
vanadium	10	3000
zinc	2000	1000
Volatile Hydrocarbons		
benzene	24.1	3.9
ethylbenzene	146	27.1
toluene	69.9	12.3
xylene	177	33.4
Acid Extractable Organics		
phenol	7.57	1.12
acrolein	344	0.726
PAHs		
anthracene	1420	315
benzo (a) pyrene	23800	6570
chrysene	10800	2810
napthalene	168	31.6
fluoranthene	3120	736
fluorene	713	196
phenanthrene	1440	321
pyrene	2800	1630

Table 8-17. Estimated bioccumulation factors for selected trace metals and petroleum components in produced waters (modified from Avanti Corporation, 1993).

Contaminant	Bioaccumulation Factor (BCF x 10^{-4})
Metals	
antimony	0.004-2
arsenic	0.03-2
beryllium	0.01
cadmium	0.01-10
chromium	0.001-0.1
copper	0.01-1
lead	0.001-0.01
mercury	0.1-10
nickel	0.001-0.1
selenium	0.01
silver	0.01-0.1
thallium	0.001-10
zinc	0.01-10
Organics	
benzene	0.0045
toluene	0.0052
napthalene	0.035
anthracene	0.035
phenanthrene	0.22
benzo (a) pyrene	0.1
acenaphthalene	0.12

Table 8-18. Maximum BCFs from USEPA AQUIRE database for exposure in salt water.

Contaminant	Crustacea		Mollusca		Osteichthys	
	BCF	Days/ Life Stage	BCF	Days/ Life Stage	BCF	Days/ Life Stage
Metals						
Cadmium			5000	140/male 6-7 cm	10.5	14/30g
Chromium			20	NR/NR		
Lead					12	14/200g
Mercury			20	NR/NR		
Silver			6750	14/NR		
Zinc	300	9/NR	1200	9/NR	200	9/NR
PAHs						
Benzo[a]pyrene			242	14/NR		
Naphthalene	325	8/NR	62	3/10mo	1158	4/3.35g
Fluoranthene	310	4/NR	5920	4/NR		
Phenanthrene	400	4/NR	1280	4/NR		
Pyrene	500	4/NR	4430	4/NR		
Volatile HCs						
Benzene					84502	
Acid Extractables						
Phenol	8.75	4/NR				

NR = not recorded

Table 8-19. IAEA default bioaccumulation factors for radium.

Organisms	IAEA (1982)	IAEA (1985)
Fish	100	500
Mollusks	100	500
Crustaceans	100	500

8.3.3.2 Bioaccumulation Factors for the Gulf of Mexico

Meinhold and Hamilton (1991) derived radium concentration factors for fish, mollusks and crustaceans from data collected by Continental Shelf Associates (1991) in a study of coastal produced water discharges (see section 8.2.3.2).

BAFs calculated from this data set cover a wide range (Table 8-20). Bioaccumulation factors ranged from 3 to 100 for whole fishes, from 2 to 240 for the soft parts of mollusks and from 4 to 170 for whole crustaceans.

Based on the published data and an independent analysis of data collected in the 1991 Continental Shelf Associates study, it can be concluded that BAFs for radium in whole fishes, soft parts of mollusks, and whole crustaceans are affected by the concentration of radium in the water. The BAFs in Table 8-20 calculated for the concentrations in water at reference sites (^{226}Ra, 0.2 to 0.7 pCi/l; ^{228}Ra, 0.0 to 10.3 pCi/l) in the Continental Shelf Associates (1991) study were smaller, in general, than the IAEA BAFs (100) for salt water fish, mollusks and crustaceans. The BAFs calculated from the 1991 Continental Shelf Associates data suggest that the generic IAEA BAFs may be over-estimates at the relatively high concentrations encountered in produced water discharges.

A BAF for radium may be expected to be a function of the element as a chemical entity. Meinhold *et al.* (1991) estimated BAFs for ^{226}Ra and ^{228}Ra separately. There was disparity between the BAFs for the two isotopes at each site. This may have contributed to the overestimates because measurements were made by radiodetection. Measurements for ^{228}Ra were less sensitive (detection limits were higher), and the two isotopes are not in equilibrium with each other in produced water.

8.4 Application to Ecological Risk Assessment for Produced Water Discharges

Data describing the bioaccumulation of produced water components can be used to formulate the problem, identify contaminants of potential concern and assess exposure. They can also be used to assess effects, although data relating body burdens to effects are limited. The BAF approach is commonly used in both human health and ecological risk assessments, but its reliability may be questionable.

Table 8-20. Calculated radium bioaccumulation factors for organisms sampled in the Continental Shelf Associates (1991) study.

Station	Conc. at 50 feet[1] (pCi/l)	Bioaccumulation factors					
		Fish		Mollusks		Crustaceans	
	$^{226}Ra/^{228}Ra$	^{226}Ra	^{228}Ra	^{226}Ra	^{228}Ra	^{226}Ra	^{228}Ra
		(pCi/g)	(pCi/g)	(pCi/g)	(pCi/g)	(pCi/g)	(pCi/g)
English Bay	0.43/0.93	32.6	5.4	16.3	3.2	169.8	101.1
Golden Meadow	0.52/0.22	NC	100	15.4	50	130	113.6
Avery Island	0.92/4.0	45.6	3.0	NDA	NDA	138.9	60.8
Ref 1a	0.25/0.15	NDA	NDA	4.0	240	56.0	26.7
Ref 1b	0.2/0	25.0	NC	20.0	NC	120.0	NC
Ref 2a	0.7/0	2.9	NC	5.7	NC	NDA	NDA
Ref 2b	0.5/0	62.0	NC	2.0	NC	138.0	NC
Ref 3a	0.7/3.3	25.7	17.6	NDA	NDA	22.9	17.6
Ref 3b	0.7/10.3	42.9	1.7	NDA	NDA	110.0	4.4

NDA: no data

NC: bioaccumulation factor not calculated because water or animal concentration was reported in Continental Shelf Associates (1991) as 0

[1] average of measured concentration at three stations, each 50 feet from discharge

9 TOXICITY OF MAJOR INDIVIDUAL CHEMICAL COMPONENTS

9.1 Introduction

The toxicity of produced water effluents is the most important information to assess risks to aquatic animals that live close to produced water discharge points (i.e. on the platforms as part of the fouling community or within several meters). If this information is available, assessing risks from individual components in the water is not necessary, and could miss combined effects of individual contaminants. The discharge changes character with increasing distance from the source, and various components react and partition differently in the environment. At some point, however, an assessment must rely on data that describe the effects of individual chemical components of produced water discharges. The same types of data are needed in an assessment of the risks to animals exposed to contaminants in sediment

A large amount of laboratory data are available to describe the toxicity of individual contaminants on various saltwater organisms. The task in an ecological risk assessment is to identify those data relevant to the identified endpoint(s) of concern, and to make some judgment about the validity and comparability of laboratory toxicity tests.

A good place to start in developing the toxicity data needed in an analysis are the USEPA water quality criteria developed to protect saltwater animals as well as other values (USEPA, 1986; see section 9.2.1). Additional data are available in the documents that support these criteria, in the open literature, and in electronic data bases such as the USEPA ACQUIRE data base (section 9.2.2). Another important concern in terms of potential impacts from produced water are effects on animals living on or in the sediment. There are limited data available to describe toxicities to these organisms, but sediment quality criteria have been derived for some contaminants and these are reviewed in section 9.3. Limited data are also available to suggest a relationship between exposure to contaminants in water and sediment and genotoxic and histopathologic effects in fish (section 9.4).

9.2 Toxicity of Chemicals in Water

9.2.1 USEPA Water Quality Criteria

The Clean Water Act requires the USEPA to publish and update ambient water quality criteria. "These criteria are to accurately reflect the latest scientific knowledge (a) on the kind and extent of all identifiable effects on health and welfare including, but not limited to, plankton, fish, shellfish, wildlife, plant life, shoreline, beaches, aesthetics, and recreation which may be expected from the presence of pollutants in any body of water including ground water; (b) on the concentration and dispersal of pollutants or their byproducts, through biological, physical and chemical processes; and (c) on the effects of pollutants on biological community diversity, productivity and stability, including information on the factors affecting rates of eutrophication and organic and inorganic sedimentation for varying types of receiving waters" (USEPA, 1986).

Water Quality Criteria Guidelines

Procedures for developing criteria are presented in (USEPA , 1986). Current water quality criteria for important produced water components were developed using the 1980 and 1986 guidelines. These two sets of guidelines are summarized below.

1985 Guidelines (USEPA, 1986, Appendix A)

The 1985 guidelines define acceptable tests, numbers of species and families, acute-chronic ratios, and bioaccumulation factors needed to derive a water quality criterion. "The guidelines indicate that if enough acceptable data on acute toxicity to aquatic animals are available, they are used to estimate the highest 1-hour average concentration that should not result in unacceptable effects on aquatic organisms and their uses. If justified, this concentration is made a function of a water quality characteristic such as pH, hardness or alkalinity. Similarly, data on the chronic toxicity of the material to aquatic animals are used to estimate the highest 4-day average concentration that should not cause unacceptable toxicity during a long-term period" (USEPA, 1986, Appendix A).

The "Final Acute Value" is an estimate of the concentration corresponding to a cumulative probability of 0.05 in the acute toxicity values for the genera with which acceptable acute toxicity tests have been conducted. If a commercially or recreationally important species has a "Mean Acute Value" lower than this "Final Acute Value", then that value replaces the "Final Acute Value" to protect this species. The "Final Chronic Value" may be calculated in the same manner as the "Final Acute Value", or by dividing the "Final Acute Value" by the "Final Acute-Chronic Ratio". Acute-chronic ratios relate the acute and chronic toxicities of a contaminant to aquatic organisms. A "Final Plant Value" is derived to describe the toxicity of the material to plants. A "Final Residue Value" is derived based on bioconcentration or bioaccumulation factors and FDA action levels (to protect humans) and acceptable dietary intakes for wildlife.

The Criterion is derived from these values as follows:

- The criterion maximum concentration (acute criterion, the highest 1-hour average concentration that should not result in unacceptable effects on aquatic organisms), is equal to 1/2 the "Final Acute Value".

- The criterion continuous concentration (chronic criterion, the highest 4-day average concentration that should not cause unacceptable toxicity during a long-term period) is equal to the lowest of the "Final Chronic Value", "Final Plant Value" and "Final Residue Value", unless other data show that a lower value should be used (e.g., behavioral studies, flavor impairment, delayed or cumulative toxicity).

1980 Guidelines (USEPA, 1986, Appendix B)

"Minimum data requirements are identified in four areas; acute toxicity to animals (eight data points), chronic toxicity to animals (three data points), toxicity to plants, and residues". "A "Final Acute Value" is derived from data on acute toxicity to fish and invertebrates". "By taking into account the number and relative sensitivities of the tested species, the "Final Acute Value" (quotes added) is designed to protect most, but not necessarily all, of the tested and untested species" (USEPA, 1986, Appendix B). A "Final chronic Value" can be calculated directly from chronic toxicity test data, or an acute-chronic ratio derived and applied to the "Final Acute Value". "A "Final Plant Value" is obtained by selecting the lowest plant toxicity value" (USEPA, 1986, Appendix B). A "Final Residue Value" is derived based on bioconcentration factors and FDA action levels (to protect humans) and acceptable dietary intakes for wildlife.

The Criterion is derived from these values as follows:

- The maximum concentration (acute criterion) is equal to the "Final Acute Value".

- The 24-hour average value (chronic criterion) is equal to the lowest of the "Final Chronic Value", "Final Plant Value" and "Final Residue Value".

Specific Water Quality Criteria

Current and proposed water quality criteria for important components of produced water are listed in Table 9-1. The following summaries are modified from USEPA (1985; except where noted) and reflect the data available at the time each criterion was developed (1985, 1980, 1976). Summaries of criteria developed under the 1985 guidelines are more detailed than those developed under earlier guidelines. More details are available in the criteria documents available for each contaminant.

TABLE 9-1. USEPA (USEPA, 1985 and proposed) and LDEQ Water Quality Criteria.

Contaminant	USEPA		LDEQ[1]	
	saltwater acute (μg/l)	saltwater chronic (μg/l)	saltwater acute (μg/l)	saltwater chronic (μg/l)
METALS				
arsenic (III)	69.	36.	69.	36.
cadmium	43.	9.3	45.62	10.0
chromium (VI)	1,100	50	1,100	50
copper	2.9	--	4.37	4.37
lead	140	5.6	220.	8.5
mercury	2.1	0.025	2.1	0.025
silver	7.2	0.92	--	--
zinc	95.	86.	95.	86.
ORGANICS				
benzene	5,100.[2]	700.[2]	2,700.	1,370.
ethylbenzene	430[2]	--	8,760	4,380
toluene	6,300.[2]	5,000.[2]	950	475
phenol	5,800.[2]	--	580.	290.
acrolein	55[1]	--	--	--
PAHS				
PAHs (generic)	300.[2]	--	--	--
fluoranthene	40.[2]	16.[2]	--	--
napthalene	2,350.[2]	--	--	--
phenanthrene	7.7[p]	4.6[p]	--	--

1 USEPA (1995)
2 no criterion, value is LOAEL
p proposed criterion
-- no criterion

Metals

<u>Arsenic</u> Twelve species of saltwater animals have acute values for inorganic arsenic (III) from 232 µg/l to 16,030 µg/l and the single acute-chronic ratio is 1.945. The only values available for inorganic arsenic (V) are for two invertebrates and are between 2,000 and 3,000 µg/l. Very few data are available concerning the toxicity of any form of arsenic other than inorganic arsenic (III) to saltwater aquatic life. Inorganic arsenic (V) is acutely toxic to saltwater animals at concentrations as low as 2,319 µg/l. The saltwater acute criterion for arsenic (III) is 69 µg/l, and the chronic criterion is 36 µg/l. There are not enough data to derive saltwater criteria for arsenic (V), but an acute LOAEL of 2,319 µg/l and a chronic LOAEL of 5 µg/l are given (summary chart, USEPA, 1986).

<u>Cadmium</u> Saltwater acute values for cadmium in five fish species ranged from 577 µg/l in larval Atlantic silverside to 14,000 µg/l for juvenile mummichog. Acute values for 30 species of invertebrates ranged from 15.5 µg/l for a mysid to 135,000 µg/l for an oligochaete worm. Acute toxicity generally increases with an increase in salinity. Two life-cycle tests with *Mysidopsis bahia* under different test conditions resulted in similar chronic values of 8.2 µg/l and 7.1 µg/l, with acute-chronic ratios of 1.9 and 15. A life-cycle test with *Mysidopsis bigelowi* also resulted in a chronic value of 7.1 µg/l and an acute-chronic ratio of 15. Acute values appear to reflect effects of salinity and temperature, but chronic values do not. The saltwater acute criteria for cadmium is 43 µg/l, the saltwater chronic criteria is 9.3 µg/l.

<u>Chromium (VI)</u> The acute toxicity of chromium (VI) to 23 saltwater vertebrate and invertebrate species ranges from 2,000 µg/l for a polychaete worm and a mysid to 105,000 µg/l for the mud snail. Chronic values for a polychaete range from <13 to 36.74 µg/l, and the chronic value for a mysid was 132 µg/l. The acute-chronic ratios range from 15.38 to <238.5. The saltwater acute criteria for cadmium (VI) is 1,100 µg/l, and the chronic criteria is 50 µg/l. Data suggest that the acute toxicity of chromium (VI) is salinity dependent, and the 1-hour average concentration (i.e. the chronic criteria) might be underprotective at low salinities.

<u>Copper</u> Acute sensitivities range from 5.8 µg/l for the blue mussel to 600 µg/l for the green crab. In a chronic life-cycle test with a mysid, adverse effects were observed at 77 µg/l but not at 38 µg/l, resulting in an acute-chronic ratio of 3.346. In long-term exposures, the bay scallop was killed at 5 µg/l. The saltwater acute criterion for copper is 2.9 µg/l. No chronic criterion was derived.

<u>Lead</u> Acute values are available for 13 saltwater animal species and range from 315 µg/l for the mummichog to 27,000 µg/l for the soft shell clam. In a chronic

toxicity test conducted with a mysid, unacceptable effects were observed at 37 μg/l but not at 17 μg/l; acute-chronic ratio for this species is 124.8. The acute saltwater criterion for lead is 140 μg/l, and the chronic criterion is 5.6 μg/l (USEPA IRIS database, 1996).

Mercury Data on the acute toxicity of mercuric chloride are available for 29 genera of saltwater animals. Acute values range from 3.5 μg/l for a mysid to 1,678 μg/l for winter flounder. Fishes tend to be more resistant and mollusks and crustaceans more sensitive to the effects of mercury (II). Results of a life-cycle test with the mysid show significant effects on the time of first spawn and productivity at 1.6 μg/l; acute-chronic ratio was 3.1. The acute saltwater criterion for mercury is 2.1 μg/l, and the chronic criterion is 0.025 μg/l.

Silver For saltwater life the acute crtierion is 7.2 μg/l. The chronic criterion is 0.92 μg/l (USEPA IRIS database, 1996).

Zinc For saltwater life the acute crtierion is 95 μg/l. The chronic criterion is 86 μg/l (USEPA IRIS database, 1996).

Organics

Benzene The available data indicate that acute toxicity occurs at concentrations as low as 5,100 μg/l, and would occur at lower concentrations in more sensitive species. No definitive data are available concerning the chronic toxicity of benzene, but adverse effects occur at concentrations as low as 700 μg/l. No criteria are given, but the LOAEL for acute toxicity is given as 5,100 μg/l, and the LOAEL for chronic toxicity is given as 700 μg/l (summary chart, USEPA 1986).

Ethylbenzene The available data for ethylbenzene indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 430 μg/l. No data are available concerning chronic toxicity. There are not enough data to justify water quality criteria, but an acute toxicity LOAEL of 430 μg/l is given (summary table, USEPA, 1986).

Toluene Available data for toluene indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 6,300 μg/l (LOAEL). Chronic toxicity occurs at concentrations as low as 5,000 μg/l (LOAEL). Acute and chronic toxicity would occur at lower concentrations in more sensitive species. Data are not adequate for a water quality criterion.

Phenol Available data indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 5,800 μg/l (LOAEL) and would occur at lower concentrations in more sensitive species. Data are not adequate for a water quality criterion. No data are available concerning chronic toxicity.

Acrolein Available data for acrolein indicate that acute toxicity to saltwater life occurs at concentrations as low as 55 µg/l, and would occur at lower concentrations among more sensitive species than those tested. No data are available concerning chronic toxicity. Data are not adequate for a criterion, value given (55 µg/l is LOAEL).

Polycyclic Aromatic Hydrocarbons (PAH)

PAH (polycyclic or polynuclear aromatic hydrocarbons) are a class of diverse compounds formed during the incomplete combustion of organic material (USEPA, 1991). Priority pollutant PAHs found in the environment include (USEPA, 1991):

- acenaphthtylene
- anthracene
- benz(a)anthracene
- benzo(a)pyrene
- benzo(b)fluoranthene
- benzo(k)fluoranthene
- benzo((g,h,i)perylene
- chrysene
- dibenz(a,h)anthracene
- fluoranthene
- fluorene
- indeno(1,2,3-cd)pyrene
- napthalene
- phenanthrene
- pyrene

PAH (generic) Fluoranthene, napthalene and phenanthrene have their own criteria, proposed criteria or LOAELs. For other PAHS, available data indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 300 µg/l (LOAEL) and would occur at lower concentrations among more sensitive species. Data are not adequate for a water quality criterion. No data are available concerning chronic toxicity.

Fluoranthene Data for fluoranthene indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 40 µg/l and chronic toxicity occurs as low as 16 µg/l and would occur at lower concentrations among species that are more sensitive than those tested. Values are LOAELs, not water quality criteria.

Napthalene The available data indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 2,350 µg/l (LOAEL) and would occur at lower concentrations among more sensitive species. Data are not adequate for a water quality criterion. No data are available concerning chronic toxicity.

Phenanthrene Phenanthrene has a proposed saltwater acute criterion of 7.7 µg/l, and a proposed chronic criterion of 4.6 µg/l.

9.2.2 Available Toxicity Data

USEPA maintains a comprehensive AQUatic toxicity REtrieval Database (AQUIRE) that is updated quarterly (Russom *et al.*, 1991). The goal of this data base is to enhance ecological and human health risk assessment processes, by providing comprehensive access to up-to-date available information on aquatic pollutants, including: standardized nomenclature and CAS registry numbers for each chemical; test organism identification by scientific and common names; parameters such as life stage; taxonomic information; test conditions and location; exposure duration and type; water chemistry and chemical analyses; adequacy of controls; effects parameters; and references for each entry. Table 9-2 illustrates a derivation from retrieved information on toxicity (LC_{50}) of minimal concentrations of contaminants that may be associated with release of produced waters to the Gulf of Mexico.

9.3 Sediment Chemical Toxicity

9.3.1 Background

Sediments are heterogeneous mixtures of abiotic and biotic materials, of natural and anthropogenic origins (Power and Chapman, 1992). Sediment contents reflect movement of chemicals from the water column to the sediments and vice versa (net deposition), from overlying sediments to deeper layers, and transformations of the chemicals within the sediments. A sediment is both a sink and a source for toxic materials in the marine environment (Power and Chapman, 1992), and is an integrated measure of the changes in its chemical contents at the time of sampling.

Chemical contaminants can be found in sediment pore water and/or bound to the solids that form the substrate. Because of their intimate contact with sediments, benthic and interstitial organisms are directly exposed to contaminants by way of ingestion of sediments, or direct transfer at exposed surfaces (e.g. respiratory organs). The key to toxicity of the chemical components is bioavailability of the chemicals.

"Sediment toxicity tests provide a rapid integrated measure of the effects of a substrate and the chemicals found in it on a toxicologic representative of benthic fauna" (Lamberson, DeWitt and Swartz, 1992). These tests directly describe the interactive effects of both measured and unmeasured chemicals in field-collected sediment samples. The tests also account for the influence of biotic and abiotic factors in sediments.

Table 9-2. Minimum Concentrations of Contaminants yielding LC_{50}s in selected taxa of aquatic organisms in salt water. [1]

Contaminant	Taxa								
	Crustacea	Units	DC[2]	Mollusca	Units	DC[2]	Osteichthys	Units	DC[2]
Metals									
Aluminum									
Arsenic	1740	µg/l	1						
Beryllium							>500000	µg/l	2
Cadmium	224	µg/l	1	80	µg/l	2	19	µg/l	2
Chromium	2030	µg/l	1	11.5	µg/l	2			
Copper	55	µg/l	1?	200	µg/l	2	24	µg/l	2
	3.00E-07	µg/l	2						
Lead	3130	µg/l	1						
Mercury	10	µg/l	2				300	µg/l	2
Nickel	508	µg/l	1						
Silver	249	µg/l	1				58000	µg/l	2
Vanadium									
Zinc	499	µg/l	1				30	µg/l	2
PAHs									
Naphthalene	850	µg/l	2	57000	µg/l	2	1200	µg/l	2
Fluoranthene							>560000	µg/l	2
Volatile HCs									
Benzene	20	µl/l	2	190000	µg/l	2	5.8	µl/l	2
							10.9	µl/l	1
Ethylbenzene	0.49	µl/l	2				4.3	µl/l	2
Xylene	7400	µg/l	2	190000	µg/l	2	89000	µg/l	2
Acid Extractables									
Phenol	89500	µg/l	1	53500	µg/l	2	9500	µg/l	1
	5800	µg/l	2						

[1]Derived from USEPA's AQUIRE data February 15, 1995 (*The information does not include salts of metals, or derivatives of parent compounds, and has not been verified from the references supplied by* AQUIRE).

[2]Data Completeness (USEPA rating): 1 = data complete; 2 = data may be adequate, but is missing an element, such as control data; 3 and 4 = data are less than adequate, and are not used.

By using different methods and periods of exposure, and life stages of more than one biological species, bioassays provide toxicological information on a variety of endpoints for different levels of biological organization. Acute tests usually use mortality as an endpoint, while chronic tests include sublethal endpoints, such as growth, development, reproduction and behavior. When combined with other studies they provide opportunities to determine cause and effect relationships. Tests are performed on whole sediment, suspended sediment, pore or interstitial water, and sediment extracts (*e.g.*, aqueous and organic solvent). Unfortunately, the Gulf of Mexico is one of the geographic regions that are not represented by the species routinely used in sediment toxicity tests. Therefore toxicity bioassays are only indicators of potential toxicity, rather than predictors of ecological changes in the Gulf of Mexico.

9.3.2 Methods for Developing Sediment Quality Criteria

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

The two major approaches for developing sediment quality criteria are chemical-by-chemical criteria, and combinations of chemical-by-chemical criteria with chemical mixture criteria. Methods for developing these criteria were evaluated and described (E.V.S. Consultants, 1990; Long and Morgan, 1990; Long *et al.*, 1995):

- Background Sediment Chemistry
- Water Quality Criteria
- Sediment/water equilibrium partitioning (SEP)
- Sediment Bioassay
- Screening level concentration
- Apparent effects threshold (AET)
- Sediment Quality Triad (triad)

This section briefly describes the reviews of the favored choices of methods to be used for the two approaches.

SEP is favored for chemical-by-chemical criteria, and is the basis for USEPA's criteria (Long and Morgan, 1995; USEPA, 1992). This method is not used for complex mixtures. SEP combines a theoretical combination of equilibrium partitioning, as used in the water quality criteria method, with a correction for the effects of organic carbon and, in some cases, acid volatile sulfides. The only field measurements needed are analyses of contaminant concentrations in sediments and inorganic content in those sediments. It takes advantage of methods for comparing sediment interstitial water concentrations to established acute and chronic criteria for the water column. This is also its drawback, because it can only relate to chemicals with established water quality criteria. SEP can only be used for nonpolar organic chemicals. Ionic components are excluded because of the organic carbon factor. In the absence of partitioning data for a specific pollutant, octanol/water partition coefficients are used as estimates. SEP does not account for the interactive effects of chemical mixtures or the effect of a sediment's characteristics (*e.g.*, pH, grain size) on toxicity of a particular chemical.

AET and triad procedures are recommended when chemical mixture criteria are involved.

AET is a field dependent procedure, using concentrations in sediments and one or more indicators of bioavailability/bioeffects, to estimate where significant effects will always be obtained by exceeding a particular level of a contaminant (E.V.S. Consultants, 1990). Thus, the method associates levels of contaminants with effects, but does not establish a causal relationship. Its site-specific capability is one additional advantage. Criteria have already been established for marine waters, and sediment bioassays have been performed with *Rhepoxynies abronius* (amphipod); *Crassostrea gigas* (oyster), and Microtox. Depressions of benthic polychaete, mollusc, and amphipod populations were also used to derive AET values for sediments.

Three measures are used in the triad procedure:

- Chemistry to determine sediment contamination;
- Bioassays to determine sediment toxicity; and
- *in situ* biological changes, including changes in infaunal communities and bottomfish histopathology.

The Triad procedure is useful for broader areas than the site specific application of AET (E.V.S. Consultants, 1990). It's major drawbacks are: further development of criteria are needed; large data bases are required; and methods for bioassays of sediments in the field are not yet standardized. Currently, criteria are only available for Pb, PAHs, and PCBs. Unmeasured covarying contaminants may influence toxicity of the measured components.

Effects of platforms on benthic communities can vary from reductions in diversity and populations of particular species to increases in diversity and populations of particular species (see section 11). This variation includes increases in diversity at the expense of preexisting species, or increases in numbers of particular species while total diversity declines. Some of these results are attributable to the introduction of a new structure (*i.e.*, the platform), as well as the accumulation of contaminants in sediments. In the latter case relatively low levels of contaminants may increase populations (hormesis) of opportunistic species that can either metabolically adapt to the contaminants, or compensate for toxic effects on particular life-stages by increases of more resistant stages in the populations.

E.V.S. Consultants (1990) did a sediment quality triad study on a platform in the Gulf of Mexico, in the vicinity of Matagorda Island, Texas. Chemical enrichment of the sediment was highest at five stations close to the platform (within 25 m), as was potential toxicity. Bioassays of chemical extracts showed that hydrocarbon contamination, not metal contamination, was associated with this toxicity. Analyses of benthic infauna found subtle differences that were not easy to distinguish between these five stations and stations further away. In contrast to the evidence of potential toxicity, field-collected data provided evidence of faunal enrichment within a 25 m radius of the platform, attributable to either the introduction of the platform structure or a hormetic effect of the contaminant mixture in the sediment.

9.3.3 Suggested Sediment Quality Criteria

Toxicity determination of sediment contamination has the same problems as assessment of any complex mixture. Currently the SEP approach for individual contaminants is the most widely available procedure for evaluating toxicity of sediments. Adequately-developed AET and triad approaches, will be useful for confirming the validity of an additive SEP approach to toxicity of multiple contaminants. A major stumbling block to application of the AET/triad approaches is the validation and standardization of endpoints for toxicity evaluation.

Sediment criteria (Table 9-3), based on specific levels of probability of toxicological effects that could be related to compilations of a biological effects database (BEDS) for contaminant concentrations in marine and estuarine sediments, were recently updated, but remain generally consistent with those previously reported (Long *et al.*, 1995).

Table 9-3. 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

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, 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" (Long *et al.*, 1995). Concentrations at or above the ERM value "represent a probable effects range within which effects would frequently occur" (Long *et* *al.*, 1995). In addition to the guideline values, the paper (Long *et al.*, 1995) describes the percent incidence (ratios of data entries) for <ERL, ERL-ERM interval, and >ERM, for 28 chemicals and chemical groups.

9.4 Histopathological and Genotoxic Effects in Fish

Several studies have related particular groups of chemicals to increased incidence of histopathological lesions in fish from saline environments. High PAH concentrations in sediments were associated with hepatic lesions (Johnson *et al.*, 1993; Myers *et al.*, 1991), and biomarkers (Goksøyr *et al.*, 1994) in marine fishes.

Liver neoplasms have only been found in adult fish in field studies (Moore and Stegeman, 1994). Laboratory studies with PAHs have only been able to induce liver tumors by prolonged exposures of fish embryos, larvae and juveniles less than one year of age, while exposures of adults have not been successful. Moore and Stegeman (1994) suggested that preadult stages are more sensitive to tumor induction because of relatively rapid rates of cell replication, and that the field observations are the result of repeated exposures to genotoxic compounds of preadult and adult stages, possibly followed by exposures to epigenetic agents.

Halogenated hydrocarbons and PAHs are found in high levels in harbors on all coasts. Areas such as Boston Harbor and Puget Sound are associated with high levels of neoplasms, especially in the livers, of bottom feeding fish such as the winter flounder, *Pleuronectes americanus* (Johnson *et al.*, 1993, 1994; Moore and Stegeman, 1994; Myers *et al.*, 1991). Associations were made for PAH and chlordane concentrations in sediments with concentrations in stomach contents of several fish species.

Mixed function oxidase systems in fishes are more developed than the enzyme systems in mollusks, and greater transformation of PAHs are to be expected in fishes than in mollusks. PAHs were studied in feral eels and sediment samples, taken from six sites with different levels of water pollution (van der Oost *et al.*, 1994). There were marked differences between parent PAH profiles in the sediments (mostly four-ring compounds) and the profiles in muscle tissue from the eels (mostly two- and three-ring compounds). Further investigations indicated that PAH-DNA adduct levels in liver cells (^{32}P postlabelling technique) were the best markers for elevated exposures to PAH contamination. These observations on PAH-DNA adduct levels are supported by similar observations from mesocosms studies on flounder, *Platichthys flesus* (Baan *et al.*, 1994) and laboratory studies by Hellou and coworkers on winter flounder (*Pleuronectes americanus).*

Baumann and Harshbarger (1995) attributed a significant decline in neoplasms in catfish to a decrease in measured PAH concentrations in sediments in the Black River, Ohio (1982-1987). These declines were associated with the closing of a coking plant in 1983.

E.V.S. Consultants (1990) recommended that histopathology in bottom fish should be dropped from the AET/triad approaches for several reasons: 1) fish are too mobile to be appropriate for evaluating site-specific effects; 2) it will be difficult to establish toxicity indices among geographic areas because of differences in fish assemblages, coupled with interspecific variability in sensitivity to toxic agents; 3) intraspecific differences in life-stage sensitivity to toxic materials; and 4) uncertainty of cause-effect relationships between chemical exposures and histopathological lesions.

9.5 Application to Ecological Risk Assessment for Produced Water Discharges

Data on the toxicity of produced water chemical components are needed to support an effects assessment. USEPA Water Quality Criteria and suggested sediment criteria can be used in a screening assessment to identify potentially important contaminants and exposure pathways. Limited data are available to describe toxicity to marine organisms native to the Gulf of Mexico, and extrapolating from laboratory studies performed on a standard test organisms adds uncertainty to an assessment.

10 RADIONUCLIDE EFFECTS

10.1 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 10-1. Prefixes commonly applied to these units are given in Table 10-2.

Table 10-1. 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 10-2. 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 3.7×10^{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 10-1). 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).

10.2 Dosimetry

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.

10.2.1 Model Codes and Dose Conversion Factors

NCRP (1991) reviewed several models useful for estimating dose to aquatic animals. Models described include CRITR (Soldat et al., 1974), EXREM III, and BIORAD (Trubey and Kaye, 1973).

CRITR (Soldat et al., 1974) allows calculation of radionuclide concentrations in water, sediment and two groups of organisms from an effluent discharged to surface waters. Concentration factors are used to estimate the equilibrium body burden of a radionuclide. Radionuclides are assumed to be uniformly distributed throughout the body. It is assumed that all of the energy emitted from α and β particles inside the body is absorbed within the body. The model uses a set of values derived for muscle-tissue spheres of different radii to account for the effective absorbed energy per dose from γ rays. Dose rates are calculated based on assumed occupancy factors, and dose rates from water are tabulated for the geometry of the human body and assumed to apply to the organisms studied. Dose from radionuclides in the sediment is also considered.

EXREM III and BIORAD (Trubey and Kaye, 1973) use a concentration factor approach to estimate the concentration of a radionuclide in organisms from the concentration in water. Dose rates from radionuclides in water are derived from the mean dose rate in an effectively infinite uniformly contaminated source. The dose rate from radionuclides accumulated by the organism are based on the effective absorbed energy per disintegration for radionuclides in a sphere of 30 cm (except for muskrat: 10 cm). No exposure from the sediment is considered.

IAEA (1988) developed dose conversion factors that relate the radiation exposure to an organism to a unit concentration of the radionuclide in the water in which the organism lives (Table 10-3). These dose conversion factors are based on models using assumptions concerning the bioaccumulation factor, K_d, and the sizes and shapes of the animals (see IAEA, 1988). These factors may be useful for screening purposes.

Table 10-3. IAEA dose conversion factors (mSv/hr per Bq/m^3).

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

10.2.2 IAEA (1976) Method

IAEA (1976) presents a method for estimating the dose to aquatic organisms for radionuclides in water, sediment and accumulated in tissue. The method outlined here is similar to many of the assumptions and models used in the previously discussed codes, but affords greater flexibility in using site specific data and assumptions.

Dose Rates From Incorporated Radionuclides

Mollusks, crustaceans and fish are large in relation to the range of α and β particles. It is assumed that no significant portion of the total energy emitted by incorporated radionuclides in the form of α and β particles is dissipated in the surrounding water. The dose rate from α and β particles closely approach the dose rate in an infinite volume, uniformly contaminated with the radionuclide.

$$D_{(\infty)} = 2.13\ E\ C\ \mu\text{rad/hour} \qquad (10.1)$$

where:
$D_{(\infty)}$ is the dose rate (α or β) in an infinite volume,
E is the average energy (α or β) MeV per disintegration, and
C is the activity of the radionuclide in the organism (pCi/gram).

To estimate the dose from γ rays the dimensions and geometry of the organism are needed. The average dose rate from internal gamma radiation is given by:

$$D_\gamma = \Gamma\, C\, \rho\, g \times 10^{-3}\ \mu\text{rad/hour} \tag{10.2}$$

where:

Γ is the specific γ ray constant in $cm^2 \cdot rad/hr \cdot mCi^{-1}$
C is the specific activity in the organism (pCi/gram)
ρ is the density of the organism
g is the mean geometrical factor in cm (Loevinger *et al.,* 1956)

IAEA (1976) uses specific dimensions and geometrical factors, but other values can also be used in estimating dose. The idealized dimensions and geometrical factors used in IAEA (1976) are as follows:

Mollusks: flat cylinder, 1 cm high, 4 cm in diameter. g = 10 cm
Crustaceans: cylinder, 15 cm long, 6 cm in diameter. g = 25 cm
Fish: cylinder, 50 cm long and 10 cm diameter. g = 41 cm

The tissue density is assumed to be 1 and the activity is assumed to be uniformly distributed throughout the volume. This assumption may result in an underestimate of the dose to some tissue, since radium tends to accumulate in bone and shell.

Dose Rates From Radionuclides in Water

For mollusks, crustaceans and fish IAEA (1976) assumes that external α and radiation from the sea water γ ray-dose is taken to be $D_{(\infty)}$.

$$D_{(\infty)} = 2.13\ E\ C\ \mu\text{rad/hour} \tag{10.3}$$

Dose Rates From Radionuclides in Sediment

The γ and β radiation dose rate above sediments has been taken to be approximately equal to 0.5 $D_{(\infty)}$. The dose from α radiation is assumed to be negligible.

10.3 Effects on Individuals and Populations

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 in assessing the effects of radionuclides discharged in produced water is the effect 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) came to similar conclusions, but expressed their reference levels in terms of dose equivalent rather than absorbed dose. 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).

10.4 Application to Ecological Risk Assessment for Produced Water Discharges

Estimates of external and external dose and doses associated with individual and population effects are needed in an effects assessment for radionuclides discharged in produced water. Generic dose-factors can be used in a screening assessment to identify potential important contaminants and exposure pathways. There is uncertainty in the doses to marine organisms that may be expect to result in effects on individuals or populations, but estimates are available.

11 EFFECTS ON BENTHIC COMMUNITIES

11.1 Introduction

Effects of platforms and platform discharges on benthic communities can vary from reductions in diversity and populations of particular species to increases in diversity and populations. This variation includes an increase in diversity at the expense of preexisting species, or an increase in numbers of particular species while total diversity declines. Some of these results are attributable to the introduction of a new structure (i.e., the platform), as well as the accumulation of contaminants in sediments. In the latter case relatively low levels of contaminants may increase populations (hormesis) of opportunistic species that can either metabolically adapt to the contaminants, or compensate for toxic effects on particular life-stages by increases of more resistant stages in the populations. Sparse densities around the platforms can be caused by any combination of the following three scenarios (Harper *et. al.,* 1981):

1. a toxic substance from the platforms is periodically killing organisms relatively quickly after settling, causing a low abundance and a high turnover rate;

2. the harder substrate in the vicinity of the platforms is unsuitable for habitation for many of the benthos; and

3. the fish and larger invertebrates that congregate around the platform prey heavily on the benthic fauna.

Benthic community impacts were associated with levels of produced water chemical constituents in the sediments which generally decreased with distance from the discharge point. The threshold concentration of contaminants at or above which the benthic community was affected varied among study areas. These differences are a function of the loading to the environment, the sedimentary characteristics of the environment and the type of benthic community (Rabalais *et al.,* 1991). This section summarizes studies of the effects of produced water on benthic communities. Conclusions and findings for each study are described, and where available, threshold levels are presented. In most cases it is not clear if the effect identified was due to toxic exposure or from confounding factors (*e.g.,* substrate removed from the platform, background levels, etc.). Other controlling factors specific to the site must also be considered. For example, sediment grain size is an important parameter because fine-grained sediments have a greater capacity for binding organic and metal pollutants than do coarse sediments, and is an important factor in the structure of benthic communities. These site specific assessments may be of limited use for predictive analysis.

11.2 Coastal Studies

Boesch and Rabalais (1989a)

Boesch and Rabalais (1989a) studied coastal canal environments that received produced water discharges and found that the benthic habitats were disturbed even without the effects of produced water contaminants. The studied environments are channels; in which fine sediments accumulate; which are periodically dredged; and in which vessel traffic disturbs the bottom. Consequently, the benthic fauna is of low diversity and is composed of opportunistic species, including the polychaetes, *Stresblospoi benedicti*, *Mediomastus ambiseta* and *Capitella capitata* and the bivalve, *Mulinia lateralis*. At locations closest to the discharge, where bottom sediments were heavily contaminated, the macrobenthic fauna was essentially eliminated. Low densities of organisms and few species were found under conditions of moderate hydrocarbon contamination of sediments. Although some relatively uncontaminated sediments had a sparse benthic fauna, high PAH levels in sediments were always associated with a depauperate benthos.

The authors selected three coastal study sites for a general chemical and biological assessment: Bayou Riguad, behind Grand Isle; Pass Fourchon; and the bay side of East Timbalier Island. These sites were chosen because of large produced water discharges and because they represented different hydrological conditions (Table 11-1). These general surveys showed evidence of biological effects in terms of: reduced density and diversity of macrobenthic organisms in contaminated sediments; and the accumulation of petroleum hydrocarbon in the tissues of filter feeding mollusks proximate to the discharge sites.

Table 11-1. General characteristics of study areas (adapted from Boesch and Rabalais, 1989a).

Site	Volume (bbl/day)	Receiving Environment	Flow Regime
Bayou Rigaud	150,000	Into terminus of deep slip (7 m) off Bayou Rigaud (4.5 m) and via holding tank and aeration system to drainage ditch (4 m) leading to Bayou Rigaud (5.5 m)	Tidally energetic, swift bottom currents
Pass Fourchon	45,000	Into dead-end dredged channel (2.5 m) near occluded end of Pass Fourchon (3 m)	Less energetic tidal flow, dead-end closed
East Timbalier Island	112,930	Into dredged canals (1.5-2 m) near open bay	Dredged canals with reduced flow; open bay, shallow, well-mixed from waves and tidal currents

Rabalais *et al.*, (1991)

In an expansion of the previous study (Boesch and Rabalais, 1989a) Rabalais *et al.*, (1991) sampled Pass Fourchon, Bayou Rigaud, and East Timbalier Island in more detail and four additional sites (Emeline Pass, Eugene Island, Romere Pass and Empire Waterway). Within each of the study areas, a series of stations was designed for a general assessment of the fate and effects of produced water discharges in the receiving environment.

Pass Fourchon

The benthic infauna of the Pass Fourchon study area was composed primarily of polychaetes, with *Streblospio benedicti* being dominant. Another polychaete opportunist, *Mediomastus ambiseta,* was common in the May 1989 collections. The bivalve, *Mulinia lateralis,* was numerous during February 1990 and was common during the other sample dates. The benthic fauna included additional polychaete species, a few bivalves and gastropods, nemertean worms, oligochaetes and pericaridean crustaceans.

In most cases, benthic infauna was absent or substantially reduced at 400, 500, 600, and 800 m from the discharge point. Infauna was reduced in the deeper parts of the channel where the effluent density plume followed the bottom contours, but was present at higher topographic locations not impacted by the effluent. The benthic fauna at distances greater than 800 m was not impacted by the discharge.

Rabalais *et al.,* (1991) concluded that there were obvious relationships between the number of species and individuals collected at a station to the amount of chemical constituents in near bottom waters or surficial sediments. Where the chemical contaminants exceeded a threshold level, the numbers of benthic fauna were either severely depressed or absent. Table 11-2 summarizes threshold concentrations of contaminants in the sediments and bottom waters above which benthic fauna was depressed.

Table 11-2. Threshold concentrations of chemical constituents in near-bottom waters and surficial sediments at or above which benthic fauna was depressed at Pass Fourchon (adapted from Rabalais et al., 1991)

Chemical Constituent	Threshold (for species)	Threshold (for Individuals)
Near-Bottom Water	**(µg/l)**	**(µg/l)**
Volatiles	25	5.8
Alkylated PAH	0.9	0.9
Total PAH	1.6	1.6
Total Saturated HC	38	38
Surficial Sediments	**(µg/g)**	**(µg/g)**
Alkylated PAH	5,900	5,900
Total PAH	6,200	6,200
Total Saturated HC	250,000	50,000

Bayou Rigaud

The benthic infauna of the Bayou Rigaud study area was a combination of marine and intermediate salinity organisms. The dominants of the benthic community were the polychaete, *Stroblospio benedicti*, and the bivalve, *Mulinia lateralis*. Numerous other polychaetes, bivalves, gastropods, nemertean worms, oligochaetes and pericaridean crustaceans were also collected.

While there were no severely depressed fauna, there were indications that some of the stations had fewer species and individuals than other stations. The benthic community parameters of number of species and individuals were not related to the grain size distribution. There was a relationship with the total organic carbon of the sediments; with the number of individuals being greater in sediments with higher organic content. The sample data was an influential factor on both the number of species and individuals, as shown in the co-analysis of variance. Numbers were higher in both of the February samples, reduced in May and lowest in October. In stepwise multiple regressions of environmental parameters important in determining the variation in number of species and individuals, the sample data was the first factor identified. With regards to number of species, there was only one more factor identified in the stepwise multiple regression as being important in explaining the variation among the means; this was sediment barium concentration. Sediment Ba concentration and total organic content were identified in the stepwise multiple regression for number of individuals as significant factors explaining the variation.

Non-linear relationships with several sediment chemical constituents identified threshold values above which number of species or number of individuals were usually depressed below background levels (Table 11-3). There were no observed threshold levels of chemical constituents in near-bottom waters above which the benthic fauna was depressed.

Table 11-3. Threshold concentrations of chemical constituents in surficial sediments at or above which benthic fauna was depressed at Bayou Rigaud (adapted from Rabalais *et al.,* 1991).

Chemical Constituent	Threshold (for species)	Threshold (for Individuals)
Surficial Sediments (μg/g)		
Parent PAH	N/A	740
Alkylated PAH	N/A	7,700
Total PAH	N/A	9,700
Total Saturated HC	300,000	240,000
Barium	N/A	440

East Timbalier Island

The benthic infauna community for the East Timbalier Island study area was characterized by marine fauna of mostly polychaetes and a few bivalves, gastropods, amphipods, isopods, cumaceans and nemerteans. There were large numbers of the opportunistic polychaetes *Streblospio benedicti*, *Mediomastus ambiseta* and *Capitella capitata.*

The benthic community parameters of number of species and number of individuals were related to some of the sedimentary characteristics and some sediment chemical constituents (Table 11-4), but the relationships were not consistent between the stations in the north-south transect and those in the east-west transect. In general there were more species and more individuals at stations in the north-south transect, with regard to increased sand content and reduced sediment organic content. For the east-west transect, there were more species and more individuals with reduced sand content and higher total organic carbon in the sediments. Where relationships were apparent with the benthic fauna and sediment hydrocarbon contamination, these relationships were stronger for stations in the north-south transect than the east-west.

Table 11-4. Threshold concentrations of chemical constituents in surficial sediments at or above which benthic fauna was depressed at East Timbalier Island (adapted from Rabalais *et al.,* 1991).

Chemical Constituent	Threshold (for species)	Threshold (for Individuals)
Surficial Sediments (µg/g)		
Alkylated PAH	5,000	5,000
Total PAH	5,700	4,700
Total Saturated HC	150,000	150,000
Zn	58	58

Emline Pass

The benthic macroinfaunal community of Emline Pass was characterized by the following groups: haustoriid amphipods, nemertean worms, and a few species of bivalves, gastropods and polychaetes. Haustoriid amphipods are typical infauna of high energy, sandy marine intertidal beach zones. The high energy, sandy environment at Emline Pass is the result of fast river currents. There were few or no individuals of benthic organisms at most Emline Pass stations (Rabalais *et al.,* 1991)

There were no relationships between the number of species and individuals in the benthic samples to various constituents of the sediments, either grain size distribution, organic content or hydrocarbon contamination. There was also little variability in the benthic community composition with overall low diversity and abundances throughout the transect. There was a slight depression in the number of species and individuals with elevated concentrations of Al and Ni in the surface sediments. The distribution of these elevated metals, however, was not related to distance from the discharge facility (Rabalais *et al.,* 1991).

Eugene Island

The benthic macrofauna was typical of a brackish to intermediate salinity fauna and composed primarily of polychaetes. The dominant polychaete was the opportunist, *Kediomastus ambiseta*. Other organisms included nemertean worms, oligochaetes, insect larvae, pericaridean crustaceans and a few bivalves and gastropods.

The general trend for Eugene Island was reduced numbers of species and individuals at the discharge point and at some stations near the discharge point. Differences among stations were minimal at distances greater than 300 m. There were reductions in both the number of species and number of individuals in relation to certain chemical constituents of the surface sediments. These relationships were stronger for number of individuals than for number of species, but the species richness for the study area was relatively low. The threshold for an effect in the benthic fauna was lower for the number of individuals than for the number of species (Table 11-5).

Table 11-5. Threshold concentrations of chemical constituents in surficial sediments at or above which benthic fauna was depressed at Eugene Island (adapted from Rabalais et al., 1991).

Chemical Constituent	Threshold (for species)	Threshold (for Individuals)
Surficial Sediments	**(µg/g)**	**(µg/g)**
Parent PAH	N/A	190
Alkylated PAH	430	430
Total PAH	2,100	520
Total Saturated HC	31,000	31,000
Barium	500	370
Aluminum	N/A	12,000

Romere Pass

The benthic infauna of the Romere Pass study area was typical of both fresh and brackish environments. Organisms were bivalves, nemertean worms, oligochaetes and a few amphipods, polychaetes and gastropods.

There were statistical differences among stations for three of the benthic community parameters calculated. The differences were not obviously related to proximity to a discharge point, and there was considerable overlap in the station means.

Empire Waterway

The benthic infauna of the Empire Waterway study area was a combination of marine and intermediate salinity organisms, primarily polychaetes. There were also oligochaetes, nemertean worms, bivalves, gastropods, ophiuroids and decapod crustaceans.

There were no relationships between the number of species and number of individuals and sediment grain size and total organic content. The sediments, however, were uniformly composed of a majority of silts and TOC values were generally 2% for most samples. Similarly, there were no relationships of the benthic fauna with most of the sediment contaminants. Concentrations in the sediments, however, were generally low throughout the study area. There was a pronounced decrease in number of species and number of individuals at a station near an abandoned discharge site. These reductions were related to elevated levels of alkalyted PAH above 60 ppb. The vertical core for this station also indicated an accumulation of petrogenic hydrocarbons subsurface and at depth.

Lake Pelto (Neff *et al.,* 1989)

Neff *et al.* (1989) studied an estuarine site in Terrebonne Bay, Louisiana at Lake Pelto Tank Battery No. 1.

The benthic fauna at Lake Pelto was dominated by *Mediomastus ambiseta*. The dominant taxa at all stations were short lived, opportunistic species such as the polychaetes *M. ambiseta* and *Streblospio benedicti* and the bivalve mollusk *Mulinia lateralis*. These species are characteristic of communities under natural environmental or pollutant stress. The authors hypothesized that this condition was caused by factors such as high suspended sediment loads attributable to the Mississippi River outflow and periodic bottom water hypoxia, neither of which are related to discharges from coastal exploration/production operations.

Benthic communities at Lake Pelto, Tank Battery No. 1 were more diverse than those at an offshore platform sampled in the same study (see Eugene Island, Block 105). Stations nearest the platform with the coarsest sediment had the highest diversities. Similar analysis revealed that communities at two stations 20 m from the platform were very similar but unique in comparison from the other stations. The fauna from stations 1000 m from the platform also were distinct from fauna from other stations.

Neff *et al.* summarized that grain size was the most important parameter contributing to gradients of benthic community structure around the platform. Benthic communities within about 20 m of both platforms were influenced by sediment contamination probably derived from produced water discharges. Benthic communities located 100 to 1000 m from the platform exhibited no evidence of impacts attributable to platform discharges. Neff *et al.* concluded that relatively minor effects on benthic communities resulting from chronic produced water discharges to shallow estuarine and coastal waters of the northern Gulf of Mexico were restricted to small areas around the platforms.

Trinity Bay, Texas (Armstrong, *et al.*, 1977)

Armstrong *et al.,* (1977) performed a 21-month study on the effects of produced water effluent from a shallow water (eight ft) oil separator platform on the surrounding benthic fauna of Trinity Bay, a shallow Texas estuary (six to nine ft).

The effluent was discharged three feet from the bottom. Bottom samples were collected monthly and analyzed for naphthalene concentration, number of species and number of individuals. The concentration of total napthalenes (i.e., napthalenes, methylnapthalenes, dimethylnapthalenes) was used to indicate the presence of petroleum hydrocarbons in the water and sediments.

Fifteen stations were established in Trinity Bay along three transects radiating outward from the C-2 Separator Platform. The station and their distances ranged from 50 ft to 19,00 ft from the platform.

The study found the highest levels of naphthalenes were in sediments within 50 feet of the shallow water (8 ft). The mean concentration at station 1 over the 21 month study was 18-21 ppm. Stations located 250 ft from the outfall had mean total naphthalene concentrations 20-50% of those at station 1. Stations 1,500 feet from the outfall had total naphthalene concentrations at or near background levels. Benthos was shown to be depressed in the immediate vicinity of the C2 separator platform. The bottom was found to be almost completely devoid of benthos within 50 feet of the outfall and severely depressed for a radius of 500 feet.

Near the platform there was an inverse correlation between relative concentration of naphthalenes in the sediments and the number of benthic animals. Station 1 was almost completely devoid of benthos, and had an average naphthalene concentration almost twice that of any other station. Outer stations that were unaffected had 0 to <1 ppm concentrations of naphthalenes.

The authors concluded that other than seasonality, natural environmental factors did not appear to play a part in the distribution and abundance of benthos, since sediment type was the same for all stations, as was salinity, temperature and depth of water.

11.3 Offshore Benthic Communities

The Offshore Ecology Investigation (OEI)

The Offshore Ecology Investigation (OEI) was a field study to assess the cumulative ecological effects of normal oil and gas drilling and production operations on estuarine and near shore ecosystems. It was designed to study the effects of persistent exposure of an estuary and the adjacent offshore area to the low level discharge of contaminants and the physical disturbance associated with prolonged and intensive drilling and production operations (Menzies et al. 1979). The experimental design used to accomplish these objectives included the study of point source discharges based on data sets taken synoptically at active platform experimental sites and at control sites both in Timbalier Bay and offshore Louisiana (Bender et al., 1979).

All samples contained pelecypods, gastropods, amphipods and cumaceans. On the basis of the offshore benthic biological data the authors found no indication that the benthic macroinvertebrate community has been altered as a result of offshore oil production in Louisiana.

The study concluded that analysis of the benthic biological data in Timbalier Bay and offshore waters does not indicate any environmental stress resulting from oil drilling and production activities. Variation in biota are the result of variability in salinity or turbidity. The study presented the following four major conclusions. 1) Timbalier has not undergone significant ecological change as a result of petroleum drilling production since just prior to 1952 when other more limited baseline data were generated. 2) Every indication of good ecological health is present. The region of the sampling sites is a highly productive one from the biological standpoint, more so than other regions thus far studied in the eastern and open Gulf of Mexico. 3) Concentration of all compounds which are in any way related to drilling or production are sufficiently low to present no known persistent biological hazard. 4) Natural phenomena such as seasonality, floods, upwellings, and turbid layers have much greater impact upon the ecosystem than do petroleum drilling and production activities.

These conclusions were contested by Sanders (1981). With regard to benthic fauna Sanders argued that the studied area was not indicative of a healthy ecosystem. In comparing the densities of organisms in the OEI study with those found in shallow marine habitats elsewhere in the world, the OEI densities were significantly lower. Also, Sanders interpreted the presence of large numbers of two opportunistic species, *Mulinia lateralis* and *Spiochaetopterus oculatus,* as indicative of polluted conditions in the study area.

The Central Gulf Platform Study

Four primary sites and four control sites on the Louisiana shelf, were sampled in three consecutive seasons in 1978-79. The objectives of the biological data synthesis were to compare communities at control and platform sites with an emphasis on indicator species and to attempt to correlate biological parameters with various physical and chemical conditions related to offshore platforms. Problems including natural stress and anthropogenic contaminants from the Mississippi River were recognized and discussed.

The organic chemical analyses found low molecular weight hydrocarbons from several fold to approximately ten times open values. Sea water collected near two secondary platform sites in the second cruise had methane concentrations up to 24 mg/l.

High molecular weight hydrocarbons were always detected in sediment samples. Most of the measurements of tissue hydrocarbons were done on fish and macroepifauna. No unresolved complex mixtures of hydrocarbons were found in these tissues. The authors made some correlation between the amount of produced water and sand discharged and an increase in the transport of hydrocarbons to the benthos. However they suggest that the Mississippi river is the principle source of hydrocarbons in the study area.

It is strongly implied that hydrocarbons have a chronic sublethal effect on the fauna of the study area. Spies (1987) takes issue with this conclusion based on the fact that the authors extrapolated laboratory toxicity data to the hydrocarbons found in the study area. He asserts that most of the toxic effects described for marine organisms are from relatively low molecular weight aromatic hydrocarbons and that the sediment hydrocarbons found in the study were dominated by highly weathered mixtures.

Buccaneer Gas and Oil Field Study (BGOF)

The Buccaneer field consists of 18 structures of which 14 are satellite platforms, two are production platforms and two are crew quarters (see section 7). Harper *et al.* (1981) used data from a pilot study of the Buccaneer oil field to determine the possible impacts on the areal distribution of benthic biota, sediments, heavy metals, and hydrocarbons. The main source of hydrocarbon contamination came from discharge of about 600 bbl/day of produced water which contained about 2 ppm of extractable hydrocarbons. Oily sediments (306 ppm) were found adjacent to one of the production platforms, whereas mostly biogenic background levels of about 0.5 ppm were found at all other stations. The macrobenthic faunal densities in the oiled bottom were reduced and the meiobenthos had an inverse ratio of nematodes to foraminifera compared with other stations. There was also evidence of a depressed fauna to the northeast of the field which may have been caused by an increased percentage of clay in the bottom or by some aspect of the production activities.

The study found that the platforms caused a decrease in macrobenthic abundance that was restricted to an area within 100 m of the platforms and possibly within 50 m. It is not clear whether the effect was due to periodic contact with toxic substances in the bleedwater, substrate disturbance due to currents eddying around the platform leg and removing the substrate, or some other cause. There was no evidence that diversity was depressed around the platforms. Harper *et al.* (1981) found the area impacted to be relatively small when compared with the total area of the field. The surface sediments during certain seasons are in nearly continual motion over a somewhat more consolidated basement, and hydrocarbons reaching the benthos are quickly dispersed. It is therefore unclear whether the near platform effects were due to contamination or related to sediment variability (Spies, 1987).

Eugene Island (Neff *et al.*, 1989)

Neff *et al.* (1989) studied four oil-water separation platforms in the Gulf of Mexico off the central coast of Louisiana. They included an estuarine site in Terrebonne Bay at Lake Pelto Tank Battery No. 1 (section 11.2.3) and three offshore sites in 8 to 15 meters of water, Eugene Island, Block 105, Eugene Island, Block 120 and Ship Shoal, Block 114. Rates of produced water discharge from each platform were near the mean rate of discharge (1801 barrels/day) for the entire industry operation on the outer continental shelf of the Gulf of Mexico.

Bottom water and sediments in the vicinity of Ship Shoal, Block 114 contained little or no dissolved oxygen. As a result, the benthic fauna were depauperate throughout the area. Due to bad weather, an insufficient number of stations were sampled at Eugene Island, Block 120 to adequately characterize the local bottom environment. Therefore, of the offshore sites only Eugene Island, Block 105 was used for the detailed field study.

The benthic fauna are pioneer communities characteristic of disturbed marine and estuarine environments dominated by *Mediomastus ambiseta*. The dominant taxa at all stations were short lived, opportunistic species such as the polychaetes *M. ambiseta* and *Streblospio benedicti* and the bivalve mollusk *Mulinia lateralis*. The author hypothesized that this condition was caused by factors such as high suspended sediment loads attributable to the Mississippi River outflow and periodic bottom water hypoxia, neither of which are related to produced water discharges.

At Eugene Island Block 105, faunal density decreased with distance from the platform. However, diversity increased with distance. Benthic fauna at stations 20 m from the platform were all similar but differed from the fauna at stations further away.

The authors analyzed for sediment grain size for chemical and infaunal analysis, and concluded that grain size was the most important parameter contributing to gradients of benthic community structure around the platforms. Benthic communities within about 20 m of the platforms were influenced by sediment contamination probably derived from produced water discharges. Benthic communities located 100 to 1000 m from the platform exhibited no evidence of impacts attributable to platform discharges.

11.4 Summary of Benthic Effects

A number of studies have shown differences in benthic communities with distance from platforms discharging produced water. This was particularly true for coastal sites in contrast to offshore areas. Although some studies found correlations between the number of species and individuals and the chemical constituents in the near bottom waters or surficial sediments, the findings were site specific and not consistent across all studies. Some studies found little or no disturbance in the benthic communities. Information extracted from studies which found a correlation would be difficult for predicting effects because it is not clear whether the effect was due to periodic contact with toxic substrates in the produced water, substrate disturbance due to currents eddying around the platform leg and removing the substrate, or some other confounding factor (Harper et al., 1981).

11.5 Application to Ecological Risk Assessment for Produced Water Discharges

Site specific assessments of benthic effects from produced water discharges may have limited application for assessments at other sites. Available studies do, however, provide estimates of threshold levels for effects, and of community responses useful in the development of the problem formulation phase of an ecological risk assessment for produced water discharges.

12 SUMMARY -- DATA AND INFORMATION AVAILABLE FOR ECOLOGICAL RISK ASSESSMENTS FOR PRODUCED WATER DISCHARGES TO THE GULF OF MEXICO

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 United States Department of Energy (USDOE).

The report reviews important ecological risk assessment concepts and methods; describes important biological resources in the Gulf of Mexico of potential concern for produced water impacts; and summarizes data available to estimate exposure and effects of produced water discharges.

No impact analysis or ecological risk assessment is presented. Environmental assessments relating to subsets of produced water discharges in the Gulf of Mexico (e.g., discharges in open Louisiana bays, offshore discharges) are being developed using the data and information presented here, combined with data collected in a field study conducted by the USDOE and results of environmental transport and exposure modeling.

Ecological Risk Assessment Concepts and Methods

Approaches to ecological risk assessment were reviewed in the context of USEPA's proposed framework for ecological risk assessment. Methods for exposure assessment include application of transport and fate models, and approaches to the estimation of dose and internal exposure. Methods and data for the effects assessment phase in an ecological risk assessment include data from acute and chronic toxicity tests at the individual and population level, and methods to extrapolate effects between species, genera and from acute to chronic effects. Methods for risk characterization include comparing exposure and effects values or distributions, and the application of population and ecosystem models.

Biological Resources In The Gulf Of Mexico

A description of the important fisheries resources in the Gulf of Mexico was provided to support the problem formulation phase of ecological risk assessments and to identify appropriate endpoints for a specific analysis. Additional site or region specific data may be needed for a specific analysis, and species with important social value other than commercial or recreational uses should also be considered.

The Gulf of Mexico includes a wide variety of habitats for marine biota, both in the water column and on the seafloor. Important coastal ecosystems associated with the Gulf of Mexico include extensive wetlands and estuaries. Wetlands provide habitat for a great number and diversity of invertebrates, fish, reptiles, birds and mammals, and are important nursery grounds for many species of fish.

The commercial fishery resources of Texas and Louisiana are of national importance, and the Gulf of Mexico provides almost 20 percent of commercial fish landings in the United States (MMS, 1993). Marine recreational fishing in the Gulf of Mexico accounts for an estimated $769 million in sales and employment for over 15,000 people (Sports Fishing Institute, 1987 as cited in MMS, 1993). Recreational fishing takes place from shore or within state waters, as well as offshore from private or charter boats.

Sportfishing in Louisiana and Texas is concentrated around oil and gas structures. Ditton and Auyong (1984) found heavy use of offshore platforms by private recreational fishing boats. Most private recreational boats were bottom fishing, with snapper and seatrout reported most frequently as the major target species. Croaker was reported as a major target species in the Delta Region.

Platform Communities

A description of the communities associated with coastal and offshore platforms in the Gulf of Mexico was given to support the problem formulation phase of an ecological risk assessment and describe the environment at risk. Community descriptions are also required in identifying assessment endpoints.

Oil and gas platform structures are colonized by microorganisms, algae, and sessile invertebrates that live attached to the structure and form the biofouling mat. These organisms provide food and habitat for many motile invertebrates and small fishes that live in close association with the biofouling mat. There is also a diverse assemblage of demersal and pelagic fishes associated with the platforms, some of which are residents. The composition of the biofouling community and assemblage of demersal and pelagic fishes associated with platforms varies with distance from shore, water depth, latitude and age of the platform (Gallaway and Lewbel, 1982).

Endangered Species And Sensitive Ecosystems

Descriptions of the major ecosystems and biota potentially at risk will support the problem formulation step in an ecological risk assessment. Endangered species and sensitive ecosystems represent unique social values and should be considered in identifying assessment endpoints.

Seven species of baleen whales have been reported in the Gulf of Mexico but are rare: the northern right whale, blue whale, fin whale, sei whale, Bryde's whale, minke whale and the humpback whale. Five of these species are listed as endangered. Twenty-five species of toothed whales and dolphins have been reported in the Gulf of Mexico. The sperm whale is the only one of the toothed whales and dolphins listed as endangered.

Endangered and threatened species of coastal and marine birds potentially impacted by produced water discharges include the brown pelican, bald eagle, arctic peregrine falcon, piping plover and the whooping crane. One species of fish listed as threatened is potentially affected by produced water discharges -- the Gulf Sturgeon (a subspecies of the Atlantic sturgeon).

Five species of marine turtles occur in the Gulf of Mexico, and all are listed as threatened or endangered (loggerhead turtle, green turtle, leatherback, hawksbill and Kemp's ridley).

Unique and sensitive biological resources of the Gulf of Mexico include coastal wetlands, the pinnacle trend live bottom features, topographic features inhabited by hard-bottom benthic communities, and deep water chemosynthetic benthic communities.

Chemical/Physical Characterization Of Produced Water

Data describing contaminant concentrations and discharge rates of produced water are needed to formulate the problem, identify potential impacts and describe the source term for an ecological risk assessment. Data presented here are limited and additional data derived from permit files and other sources may be needed in a site or area specific assessment.

Produced waters usually have high total dissolved solids (salinity) and total organic carbon, and are low in dissolved oxygen. Other components of potential concern include metals, dissolved and dispersed petroleum hydrocarbons, various treatment chemicals and radionuclides. Detected contaminants and contaminant concentrations in produced water vary widely, because the characteristics of the saline water and oil in the formation varies and because treatment methods and treatment efficiencies vary over time and space.

Produced Water Toxicity

Studies of the toxicity of produced water discharges can be used to assess potential effects on organisms. Problems with using these data include the importance of biocides in causing toxicity, and the change in toxicity that occurs with time and space. Site-specific toxicity tests are preferable to data derived from other sources.

Results of produced water bioassays conducted in the laboratory range from providing evidence of very low toxicities (Middleditch, 1984), to showing that produced water was highly toxic (Federal Register, 1992). This could be due to differences in the toxicity of the produced water, problems with protocols used in testing or the presence of biocides in some discharges.

Bioaccumulation Of Major Toxic Components Of Produced Waters

Data describing the bioaccumulation of produced water components can be used to formulate the problem, identify contaminants of potential concern and assess exposure. They can also be used to assess effects, although data relating body burdens to effects are limited. The BAF approach is commonly used in both human health and ecological risk assessments, but its reliability may be questionable.

The most direct estimates of bioaccumulation are derived from measurements of biotic and media specimens simultaneously obtained in the field. Although measurements on field-collected specimens are a preferred method for estimating site-specific bioaccumulation, they may be fiscally prohibitive (Lee, 1992). There is limited information on bioaccumulation of specific contaminants from produced waters in marine and estuarine organisms in the Gulf of Mexico. Because the data are limited, a modeling approach to estimating concentrations of organics, metals and radionuclides in animals is often used.

Only limited data are available for BAFs for organic and metals in saltwater organisms. BCFs available in the literature should be reviewed in the context of their relevance and appropriateness for application to a specific organism and specific circumstance. Generic values are often used in screening-models (Strenge and Peterson, 1989) and for organics may be calculated from octanol-partition coefficients. BCFs for some contaminants are available in the USEPA AQUIRE database (Russom *et al.,* 1991).

Toxicity of Chemical Components of Produced Water

Data on the toxicity of produced water chemical components are needed to support an effects assessment. Limited data are available to describe toxicity to marine organisms native to the Gulf of Mexico, and extrapolating from laboratory studies performed on a standard test organisms adds uncertainty to an assessment.

A good place to start in developing the toxicity data needed in an analysis are the USEPA water quality criteria developed to protect saltwater animals as well as other values (USEPA, 1986). Additional data are available in the documents that support these criteria, in the open literature, and in electronic data bases. Another important concern in terms of potential impacts from produced water are effects on animals living on or in the sediment. There are limited data available to describe toxicities to these organisms, but sediment quality criteria have been derived for some contaminants. Limited data are also available to suggest a relationship between exposure to contaminants in water and sediment and genotoxic and histopathologic effects in fish.

Radionuclide Effects

Estimates of external and external dose and doses associated with individual and population effects are needed in an effects assessment for radionuclides discharged in produced water. Generic dose-factors can be used in a screening assessment to identify potential important contaminants and exposure pathways. There is uncertainty in the doses to marine organisms that may be expected to result in effects on individuals or populations, but estimates are available.

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.

NCRP (1991) reviewed several models useful for estimating dose to aquatic animals. Models described include CRITR (Soldat et al., 1974), EXREM III, and BIORAD (Trubey and Kaye, 1973).

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

IAEA (1976) presents a method for estimating the dose to aquatic organisms for radionuclides in water, sediment and accumulated in tissue.

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).

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). NCRP (1991) suggested 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) came to similar conclusions, but expressed their reference levels in terms of dose equivalent rather than absorbed dose. IAEA (1988) concluded that some somatic effects which would be eliminated by natural selection could occur between 0.004 and 1 mSv/hr (0.1-24 mSv/d).

Effects On Benthic Communities

Site specific assessments of benthic effects from produced water discharges may have limited application for assessments at other sites. Available studies do, however, provide estimates of threshold levels for effects, and of community responses useful in the development of the problem formulation phase of an ecological risk assessment for produced water discharges.

Effects of platforms and platform discharges on benthic communities can vary from reductions in diversity and populations of particular species to increases in diversity and populations. This variation includes an increase in diversity at the expense of preexisting species, or an increase in numbers of particular species while total diversity declines. Some of these results are attributable to the introduction of a new structure (i.e., the platform), as well as the accumulation of contaminants in sediments. In the latter case relatively low levels of contaminants may increase populations (hormesis) of opportunistic species that can either metabolically adapt to the contaminants, or compensate for toxic effects on particular life-stages by increases of more resistant stages in the populations.

A number of studies have shown differences in benthic communities with distance from platforms discharging produced water. This was particularly true for coastal sites in contrast to offshore areas. Although some studies found correlations between the number of species and individuals and the chemical constituents in the near bottom waters or surficial sediments, the findings were site specific and not consistent across all studies. Some studies had even found little or no disturbance in the benthic communities. Information extracted from studies which found a correlation would be difficult for predicting effects because it is not clear whether the effect was due to periodic contact with toxic substrates in the produced water, substrate disturbance due to currents eddying around the platform leg and removing the substrate, or some other confounding factor (Harper et al., 1981).

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APPENDIX A

SPECIES LISTS FOR THE GULF OF MEXICO
(modified after Darnell and Schmidly, 1988)

page

Table A-1. Common nekton species of the Texas-Louisiana continental shelf.

INVERTEBRATES	
Medusae	
Cyanea capillata	lion's mane
Dactylometra quinquecirrha	sea nettle
Stomolophus meleagris	cabbage head
Cephalopods	
Doryteuthis plei	arrow squid
Loligo pealei	common squid
Lolliguncula brevis	little squid
CARTILAGINOUS FISHES	
Carcharhinidae	
Carcharhinus brevipinna	spinner shark
Carcharhinus falciformis	silky shark
Carcharhinus isodon	finetooth shark
Carcharhinus leucas	bull shark
Carcharhinus limbatus	blacktip shark
Galeocerdo cuvieri	tiger shark
Mustelis canis	smooth dogfish
Rhizoprinodon terraenovae	Atlantic sharpnose shark
Sphyrnidae	
Sphyrna lewini	scalloped hammerhead
Sphyrna tiburo	bonnethead
Myliobatidae	
Rhinoptera bonasus	cownose ray
Mobulidae	
Manta birostris	Atlantic manta
BONY FISHES	
Elopidae	
Elops saurus	ladyfish
Megalops atlanticus	tarpon
Clupeidae	
Brevoortia gunteri	finescale menhaden
Brevoortia patronus	Gulf menhaden
Harengula jaguana	scaled sardine
Opisthonema oglinum	Atlantic thread herring

Table A-1 (continued).

Engraulidae	
Anchoa hepsetus	striped anchovy
Anchoa mitchilli	bay anchovy
Exocoetidae	
Cypselurus cyanopterus	margined flying fish
Hirundichthys rondeleti	blackwing flyingfish
Hyporhampus unifasciatus	halfbeak
Belonidae	
Strongylura marina	Atlantic needlefish
Atherinidae	
Membras martinica	rough silverside
Menidia peninsulae	tidewater silverside
Pomatomidae	
Pomatomus saltatrix	bluefish
Rachycentridae	
Rachycentron canadum	cobia
Echeneidae	
Echeneis naucrates	sharksucker
Carangidae	
Carnax hippos	crevalle jack
Carnax latus	horse-eye jack
Chloroscombrus chrysurus	Atlantic bumber
Decapterus punctatus	round scad
Oligoplites sarus	leatherjacket
Selene setapinnis	Atlantic moonfish
Seriola dumerili	greater amberjack
Seriola zonata	banded rudderfish
Trachinotus carolinus	Florida pompano
Coryphaenidae	
Coryphaena equisetis	pompano dolphin
Coryphaena hippurus	dolphin
Mugilidae	
Mugil cephalus	striped mullet
Mugil curema	white mullet
Sphyraenidae	
Sphyraena barracuda	great barracuda
Sphyraena guachancho	guaguanche

Table A-1 (continued).

Polynemidae	
Polydactylus octonemus	Atlantic threadfin
Scombridae	
Acanthocybium solanderi	wahoo
Euthynnus alletteratus	little tunny
Scomberomorus cavalla	king mackerel
Scomberomorus maculatus	Spanish mackerel
Thunnus albacares	yellowfin tuna
Thunnus atlanticus	blackfin tuna
Xiphiidae	
Xiphias gladius	swordfish
Istiphoridae	
Istiophorus platypterus	sailfish
Makaira nigricans	blue marlin
Tetrapterus albidus	white marlin
Stromateidae	
Peprilus alepidotus	harvestfish
Peprilus burti	Gulf butterfish
SEA TURTLES	
Caretta caretta	loggerhead
Lepidochelys kempi	Kemp's ridley
CETACEANS	
Globicephala macrorhynchus	short-finned pilot whale
Kogia breviceps	pygmy sperm whale
Stenella plagiodon	Atlantic spotted dolphin
Tursiops truncatus	Atlantic bottlenose dolphin

Table A-2. Common demersal species of the Texas-Louisiana continental shelf (0-200 m).

SQUIDS	
*Loligo peali**	
Lolliguncula brevis	
SHRIMP	
*Peneaus aztecus**	
*Peanaeus setiferus**	
Sicyonia brevirostris	
Sicyonia dorsalis	
Solenocera vioscai	
Trachypenaeus similis	
Xiphopenaeus kroyeri	
CRABS	
Porcellana sayana	
Raninoides louisianensis	
Calappa sulcata	
*Callinectes sapidus**	
Callinectes similis	
Portunus gibbesi	
Portunus spinicarpus	
Portunus spinimanus	
STOMATOPODS	
Squilla chydaea	
Squilla empusa	
FISHES	
Clupeidae	
*Brevoortia patronus**	Gulf menhaden
Harengula jaguana	scaled sardine
Synodontidae	
Synodus foetens	inshore lizardfish
Ariidae	
*Arius felis**	hardhead catfish
Batrachoididae	
*Porichthys plectrodon**	Atlantic midshipman
Ogcocephalidae	
Halieutichthys aculeatus	pancake batfish
Gadidae	
Urophycis floridana	southern hake

* estuary related species

Table A-2 (cont).

Serranidae	
Centropristis philadelphica	rock sea bass
Diplectum bivittatum	dwarf sand perch
Diplectum formosum	sand perch
Serranus atrobranchus	blackear bass
Carangidae	
Chloroscombrus chrysurus	Atlantic bumber
Selena setapinnis	Atlantic moonfish
Trachurus lathami	rough scad
Lujanidae	
Lutjanus campechanus	red snapper
Pristipomoides aquilonaris	wenchman
Rhomboplites aurorubens	vermilion snapper
Haemulidae	
Orthopristis chrysoptera*	pigfish
Sparidae	
Lagodon rhomboides*	pinfish
Stenotomus caprinus	longspine porgy
Sciaenidae	
Cynoscion arenarius*	sand seatrout
Cynoscion nothus	silver seatrout
Larimus fasciatus*	banded drum
Leiostomus xanthurus*	spot
Menticirrhus americanus	southern kingfish
Micropogonias undulatus*	Atlantic croaker
	star drum
Mullidae	
Mulloidichthys martinicus	yellow goatfish
Mullus auratus	red goatfish
Upeneus parvus	dwarf goatfish
Labridae	
Hemipteronotus novacula	pearly razorfish
Polynemidae	
Polydactylus octonemus*	Atlantic threadfin
Trichiuridae	
Trichiurus lepturus	Atlantic cutlassfish

* estuary related species

Table A-2. (cont).

Stromateidae	
Peprilus alepidotus	harvestfish
Peprilus burti	Gulf butterfish
Triglidae	
Bellator militaris	horned searobin
Prionotus paralatus	Mexican searobin
Prionotus rubio	blackfin searobin
Prionotus stearnsi	shortwing searobin
*Prionotus tribulus**	bighead searobin
Bothidae	
Ancyclopsetta dilecta	three-eye flounder
*Citharichthys spilopterus**	bay whiff
Cyclopsetta chittendeni	Mexican flounder
*Etropus crossotus**	fringed flounder
Syacium gunteri	shoal flounder
Trichopsetta ventralis	sash flounder
Soleidae	
Trinectes maculatus	hogchoker
Balistidae	
Monacanthus hispidus	planehead filefish
Tetraodontidae	
Lagocephalus laevigatus	smooth puffer
Spoeroides parvus	least puffer

* estuary related species

Table A-3. Common demersal species of the Texas-Louisiana upper continental slope (200-500 m).

SHRIMP
Acanthephyra purpurea
Bentheogennema intermedia
Gennadas valens
Heterocarpus ensifer
Hymenopenaeus debilis
Hymenopenaeus robustus
Parapandalis willisi
Parapenaeus longirostris
Penaeopsis serrata
Plesionika tenuipes
Solenocera vioscai
Systellaspis pellucida
GALATHEIDS
Munida forceps
Munida longipes
CRABS
Acanthocarpus alexandri
Bathyplax typhla
Benthochascon schmitti
Cyclodorripoe antennaria
Ethusa macrophthalma
Lyreidus bairdii
Myropsis quinquespinosa
Portunus spinicarpus
Pyromaia arachna
Raninoides louisianensis
Thalassoplax angusta
FISHES
Ancyclopsetta dilecta
Benbrops anatirostris
Benbrops gobioides
Coelorynchus caribbaeus
Coelorynchus coelorhynchus
Dibranchus atlanticus
Halieuticthys aculeatus
Hymenocephalus italicus
Parasudis truculenta
Peristedion greyae
Poecilopsetta beani
Pontinus longispinis
Pristipomoides aquilonaris
Steindachneria argentea
Trichopsetta ventralis
Urophycis cirrata
Urophycis regia
Ventrifossa occidentalis

Table A-4. Some of the major macrofaunal taxa reported from the Texas-Louisiana continental shelf and slope.

PORIFERA	**MOLLUSCA**
	Gastropoda
CNIDARIA	Bivalvia
Hydrozoa	Scaphopoda
Scyphozoa	
Anthozoa	**ARTHROPOD**
Actinaria	Acarina
Alcyonaria	Ostracoda
Scleractinia	Copepoda
	Mysidacea
NEMERTA	Cumacea
	Tanaidacea
ASCHELMINTHES	Isopoda
Kinorhyncha	Amphipoda
Nemadoda	Decapoda
Priapulida	
	ECHINODERM
LOPHOPHORA	Echinoidea
Brachiopoda	Holothuroidea
Bryozoa	Ophiuroidea
Phoronida	
	POGONOPHORA
ECHIURIDA	
	CHORDATA
SIPUNCULIDA	Urochordata
	Ascidiacea
ANNELIDA	
Oligochaeta	
Polychaeta	

Table A-5. Depth-related faunal assemblages of the Texas-Louisiana continental shelf and upper slope based upon the larger benthic invertebrates.

INNER SHELF ASSEMBLAGE

Cnidaria
Renilla mlleri
Astrangia astreiformis
Palythoa texaensis

Annelida
Diopatra cuprea
Onuphis eremita oculata

Gastropoda
Architectonica nobilis
Polinices duplicatus
Phalium granulatum
Ihais haemostoma canaliculata
Anachis obesa
Cantharus cancellarius
Busycon spiratum plagosum
Nassarius acutus
Oliva sayana
Olivella mutica
Terebra dislocata
Terebra protexta

Bivalvia
Nuculana concentrica
Anadara ovalis
Anadara transversa
Noetia ponderosa
Atrina serrata
Dinocardium robustum
Dosinia discus
Corbula swiftiana

Natantia
Penaeus aztecus
Penaeus setiferus
Sicyonia brevirostris
Sicyonia dorsalis
Trachypenaeus similis

Reptantia
Pagurus pollicaris
Persephona aquilonaris
Persephona crinata
Persephona punctata
Calappa sulcata
Hepatus epheliticus
Callinectes sapidus
Callinectes similis
Portunus gibbesi
Portunus spinimanus
Libina emarginata

Stomatopoda
Squilla emousa

Echinodermata
Luidia clathrata
Ophioleopis elegans
Mellita quinquiesperforata

Table A-5. (continued)

PRO-DELTA FAN ASSEMBLAGE

Cnidaria
Renilla mulleri

Gastropoda
Cantharus cancellarius
Nassarius acutus

Bivalvia
Nuculana concentrica
Macoma tageliformis
Abra loica

Natantia
Peneaus aztecus
Peanaeus setiferus
Sicyonia dorsalis
Trachypenaeus similis

Reptantia
Persephona crinata
Callinectes similis
Portunus gibbesi
Portunus spinimanus

Stomatopoda
Squilla empusa

INTERMEDIATE SHELF ASSEMBLAGE

Annelida
Diopatra cuprea

Gastropoda
Strombus alatus
Distorsio clathrata
Tonna galea
Murex fulvescens
Busycon contrarium
Fasciolaris lilium hunteri
Conus austini
Polystira albida
Pleurobrachea hedgpethi

Natantia
Peneaus aztecus
Peanaeus setiferus
Sicyonia brevirostris
Sicyonia dorsalis
Trachypenaeus similis

Reptantia
Petrochirus diogenes
Persephona crinata
Calappa sulcata
Hepatus epheliticus
Callinectes similis
Portunus gibbesi
Portunus spinicarpus
Portunus spinimanus
Anasimus latus
Libina emarginata
Parthenope serrata

Stomatopoda
Squilla chydaea
Squilla empusa

Echinodermata
Luidia alternata
Luidia clathrata
Astropecten duplicatus
Ophiolepis elegans
Clypeaster ravenelli
Encope michelini
Echinaster sp.
Stylocidaris affinis

Table A-5. (continued)

OUTER SHELF ASSEMBLAGE

Gastropoda	**Reptantia**
Turritella exoleta	*Munida forceps*
Distorsio clathrata macgintyi	*Raninoides louisianensis*
Polystira albida	*Myropsis quinquespinosa*
	Calappa springeri
Bivalvia	*Calappa sulcata*
Anadara baughmani	*Portunus spinicarpus*
Anadara floridana	*Anasimus latus*
Amusium papyraceus	*Leiolambrus nitidus*
Argopecten gibbus	
Pitar cordatus	**Stomatopoda**
Verticordia ornata	*Squilla chydaea*
Natantia	**Echniodermata**
Parapenaeus longirostris	*Astropecten cingulatus*
Peneaus aztecus	*Astropecten duplicatus*
Sicyonia brevirostris	*Brissopsis atlantica*
Trachypenaeus similis	*Echinocardium fulvescens*

UPPER SLOPE ASSEMBLAGE

Cnidaria	**Reptantia**
sea anemone (unidentified)	*Munida forceps*
	Raninoides louisianensis
Annelida	*Myropsis quinquespinosa*
Protula tubularia	*Iliacantha subglobosa*
	Pyromaia arachna
Gastropoda	*Acanthocarpus alexandri*
Sconsia striata	*Calappa sulcata*
Murex beauii	*Portunus spinicarpus*
	Thalassoplax angusta
Bivalvia	*Anasimus latus*
Yoldia solenoides	*Stenocionops spinimana*
Limopsis sulcata	*Parthenope agona*
Cyclopecten nanus	
Cyclocardia armilla	**Stomatopoda**
Nemocardium perambile	*Squilla chydaea*
Natantia	**Echinodermata**
Parapenaeus longirostris	*Anthenoides piercei*
Solenocera vioscai	*Luidia elegans*
Hymenopenaeus tropicalis	*Astropecten nitidus*
	Chieraster echinulatus
	Echinocardium fulvescens
	Brissopsis alta
	Brissopsis atlantica

Table A-6. Partial list of algae and invertebrates reported from drilling rigs and platforms of the Texas-Louisiana continental shelf.

ALGAE

Cyanophyta - *Microcoleus, Oscillatoria, Schizothrix*
Rhodophyta - *Acrochaetum, Callithamnion, Ceramium, Polysiphonia*
Phaeophyta - *Ectocarpus, Giffordia, Sargassum*
Chlorophyta - *Bryopsis, Chaetomorpha, Cladophora, Enteromorpha*

INVERTEBRATES

Porifera - *Cliona, Haliclona, Halichondria, Verongia*

Cnidaria
  Hydrozoa - *Aglaophenia, Bougainvillea, Obelia, Tubularia*
  Anthoza - *Astrangea, Leptogorgia, Oculina, Telesto*

Nemerteans

Platyhelminthes - *Leptoplana*

Bryozoans - *Acanthodesmia, Bugula, Membranipora*

Sipunculids

Annelida
  Polychaetes - *Eunice, Haplosyllus, Neanthes, Nereis*

Mollusks
  Gastropods - *Cantharus, Crepidula, Murex, Thais*
  Bivalves - *Arca, Crassostrea, Isognomon, Ostrea*
  Pycnogonids - *Tanystylum*

Crustaceans
  Copepods - *Acartia, Labidocera*
  Cirripedes - *Balanus, Lepas, Megabalanus*
  Amphipods - *Caprella, Corophium, Jassa, Stenothoe*
  Tanaids - *Tanais*
  Isopods - *Limnoria, Sphaeroma*
  Decapods - *Callinectes, Dromidia, Eurypanopeus, Hexapaneopeus, Menippe, Neopanope, Pachygrapsus, Pagurus, Panopeus, Panulirus, Petrochirus, Pilumus, Porcellana, Portunus, Stenorhynchus, Synalpheus*

Echinoderms - *Arbacea, Ophiactis, Ophiothrix*

Urochordata
  Ascidiaceans - *Enterogona*

Table A-7. Fish species reported around drilling rigs and platforms of the Texas-Louisiana a continental shelf.

Orectolobidae	carpet sharks
Ginglymostoma cirratum	nurse shark
Sphyrnidae	hammerhead sharks
Sphyrna americana	scalloped hammerhead
Dasyatidae	stingrays
Dasyatis americana	southern stingray
Clupeidae	herrings
Harengula jaguana	scaled sardine
Sardinella aurita	Spanish sardine
Serranidae	sea basses
Epinephelus adscensionis	rock hind
Epinephelus itajara	Jewfish
Epinephelus nigritus	Warsaw grouper
Mycteroperca phenax	scamp
Mycteroperca rubra	comb grouper
Paranthias furcifer	Creole-fish
Serranus subligarius	belted sandfish
Grammistidae	soapfishes
Rypticus maculatus	whitespotted soapfish
Priacanthidae	bigeyes
Pricanthus arenatus	bigeye
Apogonidae	cardinalfishes
Apogon maculatus	flamefish
Pomatomidae	bluefishes
Pomatomus saltatrix	bluefish
Rachycentridae	cobias
Rachycentron canadum	cobia
Carangidae	jacks
Caranx crysos	blue runner
Caranx hippos	crevalle jack
Caranx latus	horse-eye jack
Chloroscombrus chrysurus	Atlantic bumber
Decapterus punctatus	round scad
Elagatis bipinnulata	rainbow runner
Selene setapinnis	Atlantic moonfish
Selene vomer	lookdown
Seriola dumerili	greater amberjack
Seriola rivoliana	almaco jack
Trachurus lathami	rough scad

Table A-7. (continued).

Coryphaenidae	dolphins
Coryphaena hippurus	dolphin
Lutjanidae	snappers
Lutjanus campechanus	red snapper
Lutjanus cyanopterus	cubera snapper
Lutjanus griseus	gray snapper
Lutjanus synagris	lane snapper
Ocyurus chrysurus	yellowtail snapper
Rhomboplites aurorubens	vermiliom snapper
Haemulidae	grunts
Haemulon aurolineatum	tomtate
Sparidae	porgies
Archosargus probatocephalus	sheepshead
Sciaenidae	drums
Cynoscion arenarius	sand seatrout
Cynoscion nebulosus	spotted seatrout
Equetus umbrosus	cubbyu
Micropogonias undulatus	Atlantic croaker
Kyphosiadae	sea chubs
Kyphosus sectatrix	Bermuda chub
Ephippidae	spadefishes
Chaetodipterus faber	Atlantic spadefish
Chaetodontidae	butterfly fishes
Chaetodnon ocellatus	spotfin butterflyfish
Pomacanthidae	angelfishes
Holacanthus bermudensis	blue angelfish
Holacanthus ciliaris	queen angelfish
Holacanthus tricolor	rock beauty
Pomacanthus arcuatus	gray angelfish
Pomacanthus paru	French angelfish
Pomacentridae	damselfishes
Abudefduf saxatilis	sergeant major
Chromis multilineatus	brown chromis
Pomacentrus variabilis	cocoa damselfish
Cirrhitidae	hawkfishes
Amblycirrhitus pinos	redspotted hawkfish
Labridae	wrasses
Bodianus pulchellus	spotted hogfish
Bodianus rufus	Spanish hogfish
Thalassoma bifasciatum	bluehead

Table A-7. (continued).

Sphyraenidae	barracuda
Sphyraene barracuda	great barracuda
Blenniidae	combtooth blennies
Hypleurochilus geminatus	crested blenny
Hypleurochiuis springeri	orangespotted blenny
Parablennius marmoreus	seaweed blenny
Scartella cristata	molly miller
Acanthuridae	surgeonfishes
Acanthurus coeruleus	blue tang
Scombridae	mackerels
Euthynnus alleteratus	little tunny
Scomberomorus cavalla	king mackerel
Balistidae	leatherjackets
Aluterus schoepfi	orange filefish
Aluterus scriptus	scrawled filefish
Balistes capriscus	gray triggerfish
Balistes vetula	queen triggerfish
Cantherhines pullus	orangespotted filefish
Cantherhines sufflamen	ocean triggerfish
Monacanthus hispidus	planehead filefish
Tetracodontidae	puffers
Canthigaster rostrata	sharpnose puffer