HYDRODYNAMIC MODELING AND ECOLOGICAL RISK-BASED DESIGN OF PRODUCED WATER DISCHARGE FROM AN OFFSHORE PLATFORM

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by

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Abstract

This study has two major components: hydrodynamic modeling and ecological risk assessment (ERA) of produced water discharge. The general objective was to develop a framework for ecological risk-based design of produced water discharge from an offshore platform. This consisted of six more specific objectives: (1) developing an initial dilution model; (2) integrating the developed initial dilution model with a far field dilution model; (3) developing a methodology for probabilistic hydrodynamic modeling; (4) identifying methodologies for ERA of produced water discharge; (5) developing a framework for ecological risk-based design of a produced water outfall; and (6) applying the framework to a case study dealing with the discharge from an offshore oil platform.

Conceptual and numerical problems associated with presently available initial dilution models were elaborated in this study. A new approach to initial dilution modeling was proposed based on the hypothesis of additive shear and forced entrainment combined with nonlinear regression. Unlike the previous approach, which is typically "trial and error", the proposed approach is systematic and provides an objective means of evaluating the initial dilution model. Based on the proposed approach, an alternative initial dilution model was then developed. The developed model is more robust and justifiable conceptually and numerically. It gives a unique, continuous, solution of centerline dilution. A comparison with other available models shows that the proposed model is better in a number of ways: (1) it does not assume that the current has no effect in the buoyancy-dominated near field (BDNF), which other available models do; (2) in the buoyancy-dominated far field (BDFF) region the model has one parameter fewer than a previously available model yet it is no less accurate; (3) in the transition region it gives a unique solution which the asymptotic models do not; (4) unlike the previous models, the proposed model has approximately the same precision for all regions, i.e. the BDNF, the BDFF, and the transition; and (5) the proposed model can also be presented in a probabilistic form that permits calculation of failure probability for specified model inputs and a threshold dilution.

Hydrodynamic modeling was carried out by integrating near and far field models. The developed initial dilution model was used as the near field model. The far field model and the control volume approach for connecting near and far field models were adapted from published methods. A comparison using a case study showed that the proposed hydrodynamic model and the Cornell Mixing Zone Expert System (CORMIX) model are generally in good agreement, particularly in estimating average effluent concentrations. However, the proposed model also provides the concentration field in the X-Y directions so that it may be applicable for analysis of mixing zones, which in some cases is defined in terms of the horizontal area around the discharge location. The proposed model can also be readily used in a probabilistic analysis to take into account the uncertainty associated with the model inputs, model coefficients and error term. The probabilistic analysis was carried out using Monte Carlo (MC) simulations. A comparison between random sampling and Latin Hypercube Sampling (LHS) showed that LHS-based MC simulations were typically about 15% more efficient than the random sampling MC simulations.

In the context of produced water discharges, ERA has usually been directed at monitoring purposes. In the past, there is no consideration to the integration between ERA and engineering design of the produced water outfalls. In this research, an approach was identified to deal with specific problems relevant to design of produced water discharge in the marine environment. It consists of three phases, i.e. problem formulation, analysis, and risk characterization. A framework of ecological risk-based design was then developed by integrating the methodology of hydrodynamic modeling and ERA discussed above. The framework was presented systematically using a case study by evaluating design scenarios of produced water discharge relevant to an offshore oil production platform, the Terra Nova oil field, located on the Grand Banks, southeast of St. John's, Newfoundland, Canada. Instead of providing a solution for a particular problem of an existing oil production platform, the emphasis of the case study is to show how the risk-based design of produced water discharge could be undertaken.

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<u>List of symbols</u>:

a : coefficient of the Huang et al. (1998) initial dilution model.

a_1, a_2 : coefficient of the functional relationship of hydrofynacmic characteristics at various region, i.e. BDNF, transition, and BDFF.

$A(z)$: area of the Standard Normal Distribution from 0 to z along the abscissa.

b : coefficient of the Huang et al. (1998) initial dilution model.

B : discharge specific buoyancy flux (m^4/sec^3).

c : coefficient of the Huang et al. (1998) initial dilution model.

C_a : bulk pollutant concentration at the downstream end of the control volume.

C_1 : coefficient of the Lee and Cheung (1991) initial dilution model.

C_2 : coefficient of the Lee and Cheung (1991) initial dilution model.

C_3 : coefficient of the equation of the horizontal boil location.

C_4 : coefficient of the equation of the horizontal boil location, it is a function of the ratio of buoyancy and momentum length scales.

C_5 : coefficient of the equation of the horizontal boil location.

C_{D1} : coefficient of the equation of distance from the boil center to the downstream end of the control volume.

C_{D2} : coefficient of the equation of distance from the boil center to the downstream end of the control volume.

C_{S1} : coefficient of the equation of the bulk dilution at the downstream end of the control volume.

C_{S2} : coefficient of the equation of the bulk dilution at the downstream end of the control volume.

$C(x,y)$: pollutant concentration at a point (x,y).

d : diameter of the outfall port (m).

d_2 : coefficient of the Huang et al. (1998) initial dilution model.

e : coefficient of the proposed initial dilution model.

E_s : forced entraintment.

E_s : shear entraintment.

$erf(w)$: the error function of w.

EC_{50} : pollutant concentration resulting in observed effect in 50% test animals.

f : coefficient of the proposed initial dilution model.

f : count of the failure cases in Monte Carlo simulations.

F_o : jet densimetric Froude number (dimensionless).

g : gravitational acceleration (m/sec^2).

g' : reduced gravitational acceleration (m/sec^2).

h : coefficient of the proposed initial dilution model.

h_o : plume thickness (m).

$h(x)$: plume thickness (m) as a function of distance x.

H : depth of the ambient water (m).

$I_{0.1}$: regional variables (hydrodynamic characteristics) at the transitional regime define as $H/l_b > 0.1$.

I_{10} : regional variables (hydrodynamic characteristics) at the transitional regime define as $H/l_b < 10$.

I_{tr} : regional variables (hydrodynamic characteristics) at the transitional regime define at $0.1 \leq H/l_b \leq 10$.

l_b : length scale (m) as a measure of the vertical distance at which the velocity induced by the buoyancy has decayed to the value of the ambient velocity.

l_m : length scale (m) as a measure of the interaction of a momentum-dominated jet with a cross-flow.

l_M : length scale (m) as a measure of the distance at which the buoyancy becomes more important that the jet momentum.

l_Q : length scale (m) as a measure determining whether the jet geometry has a direct influence on the flow characteristic.

LC_{10} : pollutant concentration resulting in observed lethal effect in 10% test animals.

LC_{25} : pollutant concentration resulting in observed lethal effect in 25% test animals.

LC_{50} : pollutant concentration resulting in observed lethal effect in 50% test animals.

LC_{75} : pollutant concentration resulting in observed lethal effect in 75% test animals.

L_o : plume width at the downstream end of the control volume (m).

L_s : upstream intrusion length (m).

$L(x)$: plume width (x) as a function of distance x.

M : discharge momentum flux (m^4/s^2).

n : number of species for which toxicity data is available for a particular chemical.

N : number of sample (data points).

ns : number of simulations performed in Monte Carlo simulations.

p-value: the smallest level of significant at which hypothesis would be statistically rejected.

Q : outfall discharge (flow) rate (m^3/sec)

r : count of the reliable cases in Monte Carlo simulation.

R^2 : coefficient of determination.

R_f : flux Richardson number.

S : initial (centerline) dilution.

S_a : bulk dilution at the downstream end of the control volume (dimensionless).

TU_a : acute toxicity unit.

TU_c : chronic toxicity unit.

u : ambient current speeds (m/s).

u_* : shear velocity (m/s).

u_j : initial jet velocity (m/s).

w : coefficient of the proposed initial dilution model.

x : distance along the plume centerline starting from the center of the downstream end of the control volume (m).

x_b : horizontal distance of the boil location from the port (m).

x_D : distance from the boil center to the downstream end of the control volume (m).

X : global coordinate system in the horizontal direction.

Y : global coordinate system in the vertical direction.

z : depth above discharge (m).

α : entraintment coefficient.

β : constant in the equation of buoyant spreading.

ϵ : error term in the proposed initial dilution model.

γ_j : value of the *j*th parameter minus its starting value in the iteration process of the nonlinear regression.

ρ_a : density of ambient seawater (kg/m^3).

ρ_o : density of the effluent (kg/m^3).

κ : von Karman constant.

θ : angle between the rising buoyant jet axis and the water surface (radian).

ϕ : direction of the current with respect to the *X*-coordinate system (radian).

ω_{ji} : derivative of the nonlinear function with respect to the *j*th parameter, used in the nonlinear regression.

List of abbreviations:

AAN : Artificial neurol network.

ANZECC : Australia and New Zealand Environment and Conservation Council.

ARMCANZ : Agriculture and Resource Management Council of Australia and New Zealand.

BC : Benchmark concentration.

BDFF : Buoyancy-dominated far field.

BDNF : Buoyancy-dominated near field.

BTEX : Benzene, Toluene, Ethylene and Xylene.

CCME : Canadian Council of Ministers of the Environment.

CCB	: Critical body burden.
CCC	: Criterion continuous concentration.
CDF	: Cumulative distribution function.
CHARM	: Chemical Hazard Assessment and Risk Management.
CORMIX	: Cornell Mixing Zone Expert System.
CMC	: Continuous maximum concentration.
DFO	: Department Fisheries and Ocean.
DREAM	: Dose related Risk and Effects Assessment Models.
EC	: Exposure concentration.
ERA	: Ecological risk assessment.
FPSO	: Floating Production, Storage and Offloading.
FOSM	: First Order Second Moment.
GBS	: Gravity-based structures.
GM	: Geometric mean.
HHC	: Human health criterion.
HQ	: Hazard quotient.
LDEQ	: Louisiana Department of Environmental Quality.
LHS	: Latin hypercube sampling.
MC	: Monte Carlo, it is used to refer Monte Carlo simulations.
MSE	: Mean square error.
NOEC	: No observed effect concentration.
OOC	: Offshore Operators Committee.
PAH	: Polycyclic aromatic hydrocarbon.

PEC	: Predicted environmental concentration.
PDF	: Probability density function.
PNEC	: Predicted no effect concentration.
RSB	: Roberts, Snyder and Baumgartner.
RSM	: Response surface methodology.
TU	: Toxicity unit.
U.S. EPA	: United States Environmental Protection Agency.
VIF	: Variance inflation factor.

Chapter 1

Introduction

1.1. Background to Study

Associated with oil drilling and production are various types of wastes. These include drilling fluids, drill cuttings, produced water, produced sand, deck drainage. sewage, domestic wastes, and treatment chemicals. The major waste streams in terms of volumes and amount of pollutants are drilling fluids and drill cuttings from drilling operations and produced water from oil production operations. The term produced water refers to the water (brine) brought up from the hydrocarbon-bearing strata during the extraction of oil and gas, and can include formation water, injected water, and any chemical added downhole or during the oil/water separation process (U.S. EPA 1993).

The quantity of produced water from an oil field varies from case to case depending upon the characteristics of the oil reservoir and the age of the field. Typical examples of produced water discharge rates from offshore fields are on the order of 4,000 m^3/day in the Gulf of Mexico, USA, to 123,000 m^3/day in the Java Sea, Indonesia (Brandsma and Smith

1996, Smith et al. 1996, Somerville et al. 1987). Considering the rate of oil or gas production at a given platform, the flow rate of produced water is usually very substantial. From the EPA's 30-facility study (U.S. EPA 1993), it is reported that produced water flow rates range from 2 to 150,000 barrels per day, with associated production rates of 40 to 24,000 barrels per day and 0.1 to 150 million cubic feet per day for oil/condensate and gas, respectively. Generally, produced water can account for between 2 to 98% of the extracted fluids from the reservoir (Stephenson 1992, Wiedeman 1996). As a result, cost-effective and environmentally acceptable management and disposal of produced water is critical in the petroleum industries.

The chemical composition of produced waters is site specific, and includes a variety of inorganic, organic, and radioactive chemicals (Roe et al. 1996, Stephenson 1992). For offshore and coastal oil industries, produced water is often discharged into the ocean, following a treatment at the platform. The type and degree of the treatment depends on the end use of the water or disposal method. Although a treatment is provided before discharge, the produced water effluent commonly still contains toxic chemicals, making it an environmental concern.

Typical produced water from North Sea platforms has been associated with ecological impacts, which are reported in terms of effect concentration with 50% reduction in growth (EC_{50}, based on two-day exposure) of 45 to 535 ml/l for algae (Brendehaug et al. 1992). Lethal concentration with 50% mortality based on one-day exposure (LC_{50}) was 100 ml/l for the copepod *Calanus finmarchicus* (Sommerville et al. 1987). For fish, the lowest value registered of LC_{50} is for the guppy, *Poecilia retivulata*, at a value of 7.5-423 ml/l (Jacobs and Marquenie 1991). Based on the evidence of toxicity, environmental risk management is becoming increasingly important in offshore oil production (Ofjord et al. 1996).

When produced water is discharged into the ocean, the process is subject to compliance with relevant water quality standards. Recently, there has been a trend towards specifying pollutant limits from ecological and epidemiological viewpoints, in which pollutant concentrations are specified in terms of ecological and human health risks (ANZECC and ARMCANZ 1999, U.S. EPA 1999a). This raises the possibility that design of the produced water outfall could itself be looked at from the point of view of ecological risk due to exposure to produced water or specific toxic pollutants associated with it.

Ecological risks have been assessed for specific pollutants found in produced waters from offshore fields (Furuholt 1996, Karman et al. 1996, Neff and Sauer 1996, Ofjord et al. 1996). However, there are drawbacks associated with presently used approaches for ecological risk assessment of produced water discharges. These are that endpoints of the assessment are not well defined, and that uncertainty analysis is not carried out objectively. Furthermore, risk assessments are usually directed at monitoring or remediation purposes, rather than design. In particular, ecological risk assessment (ERA) has not been incorporated during the engineering design of produced water outfalls.

The risks associated with the offshore discharge of produced waters depend strongly on the contaminant distribution in the ambient seawater (Karman and Reerink 1998, Meinhold et al. 1996a, Smith et al. 1996, Stromgren et al. 1995, Girling 1989, Somerville et al. 1987). Hydrodynamic modeling plays an important role in assessing contaminant levels for ERA studies; however, there appears to be no generally accepted model for such a purpose.

Presently available approaches to hydrodynamic modeling have inherent problems. The first problem is related to the reliability of the initial dilution models. This includes assumptions taken in developing the models and the numerical accuracy of the models as discussed in more detail in Chapter Two. Another problem is that presently used approaches (e.g. Wasburn et al. 1999, Karman et al. 1996, Reed et al. 1996, Brandsma et al. 1992, Somerville et al. 1987) do not provide uncertainty analysis, and that exposure concentration at a fixed distance from the platform is calculated using a deterministic approach. Indeed, uncertainty is inherent and inevitable in the mixing processes between the produced water and the ambient seawater. Therefore, there is a need to develop a probabilistic hydrodynamic model, which could be integrated into an ERA model, for ecological risk-based design of produced water outfall.

1.2. Scope and Purpose of the Research

This study has two major components: hydrodynamic modeling and ERA. The previous section has briefly discussed the problems, which will be critically reviewed in subsequent chapters. Some limitations need to be established to ensure a realistic scope of the research project. The general objective of this study was to develop a methodology for an ecological risk-based design of produced water discharge from an offshore platform. This was carried out through integrating a probabilistic hydrodynamic model with an ERA model as shown schematically in Figure 1.1.

As indicated in Figure 1.1, the hydrodynamic modeling consists of the development of an initial dilution model and its integration with a far field model. The study was directed at the case of buoyant-jet discharge in unstratified moving waters. The deterministic far field models were adapted from the published models and their development is beyond the scope of this research. The integrated hydrodynamic model was used in the development of a methodology for ERA. A framework for ecological-risk based design of produced water outfall was then developed using the integrated hydrodynamic and ERA model. A case study was presented to highlight a potential application of the proposed methodology. Probabilistic and uncertainty analysis was applied throughout the modeling process.

Keeping in perspective the above problem formulation, this research has the following more specific objectives:

1. developing an initial dilution model;
2. integrating the developed initial dilution model with far field dilution models;
3. developing a methodology for probabilistic hydrodynamic modeling;
4. identifying methodologies for ecological risk assessment of produced water discharge;
5. developing a framework for ecological risk-based design of produced water outfalls;
6. Applying the framework of ecological-risk based design for a case study of outfall design for an offshore oil platform.

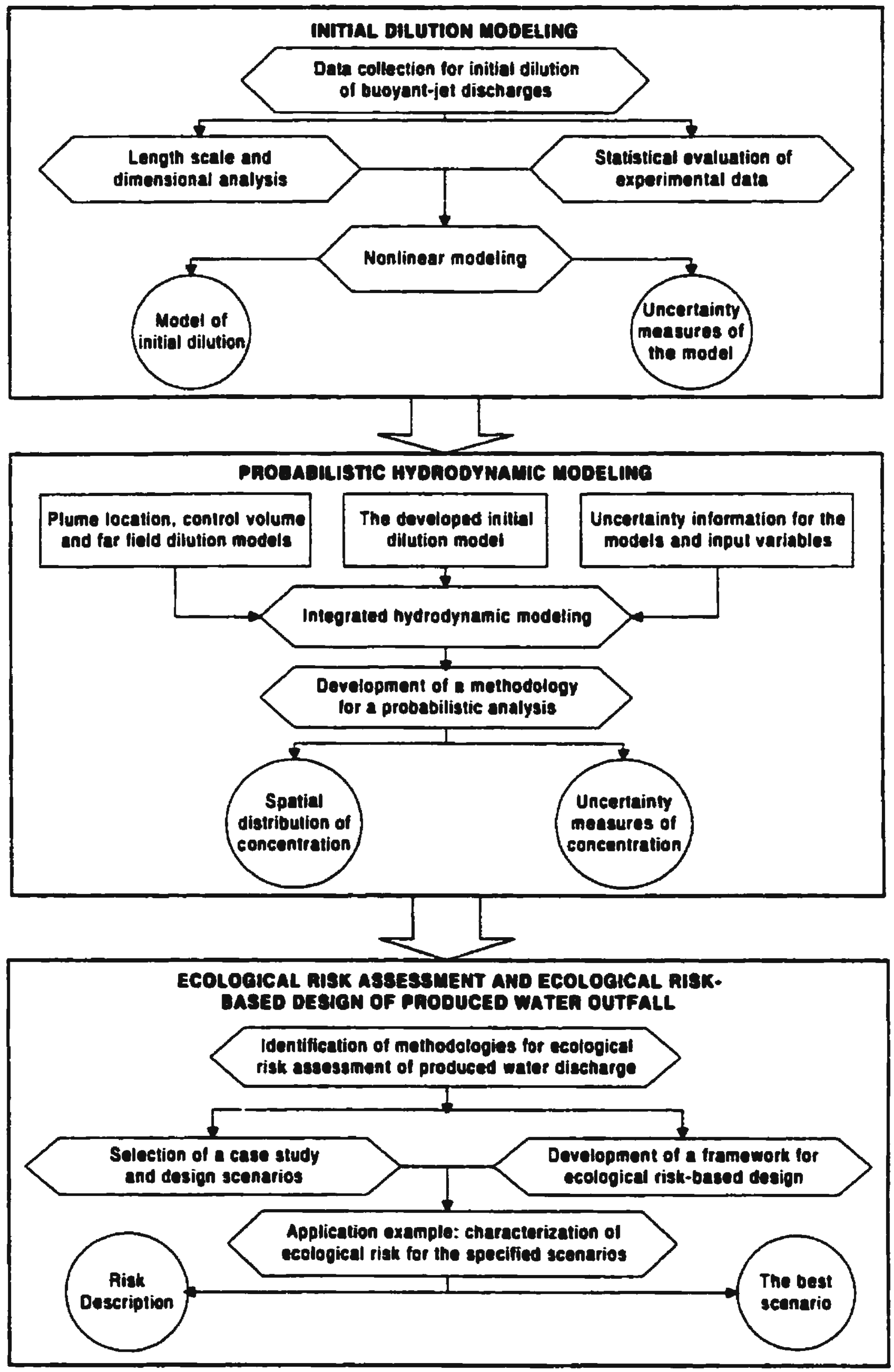


Figure 1.1. Schematic diagram of the research

1.3. Outline of the Thesis

The thesis consists of eight chapters. Background, objectives and outline of the thesis have been presented in this chapter. Chapter Two presents a critical review dealing with problems of presently available initial dilution models and discusses potential initial dilution modeling approaches, which may be useful to overcome the drawbacks discussed. Development and evaluation of an initial dilution model are presented in Chapter Three, in which a new approach to initial dilution modeling is proposed. A unique initial dilution model is presented in a deterministic and probabilistic form. An application example of the proposed model is also provided. Chapter Four provides reviews of approaches to integrating near and far field models in hydrodynamic modeling of produced water discharges. A probabilistic hydrodynamic modeling approach is formulated in this chapter. Chapter Five reviews available approaches to ecological risk assessment (ERA) and identifies methodologies of ERA in the context of produced water discharges.

Chapter Six provides a framework of ecological risk-based design for produced water outfall. A case study using data on potential discharge from the Terra Nova Floating, Production, Storage and Offloading (FPSO) system, located at the Grand Banks, Newfoundland, Canada (Petro-Canada 1996) is also given in this chapter. Different design scenarios are evaluated on the basis of ecological risks. This makes it possible to classify alternative designs (e.g. different geometries and/or different locations of outfalls) according to the ecological risks, which might arise from the discharge scenario, and to determine the degree to which one design is more appropriate than another. Conclusions and recommendations are presented in Chapter Seven, and the statement of originality of the thesis is given in Chapter Eight.

Chapter 2

Initial Dilution in Hydrodynamic Modeling: Problems of Presently Available Models and Potential Modeling Approaches

2.1. Introduction

Once produced water is discharged into the ocean, it mixes with the ambient seawater. The flow pattern of the discharge may be categorized as a buoyant jet flow as it is often found that the discharge has both initial momentum and a density difference between the effluent and the ambient seawater. Table 2.1 provides a summary of produced water discharges and receiving water conditions from different regions. A typical discharge from a North Sea platform has a discharge rate of 10,000 m^3 per hour and a density difference of 13 kg/m^3 less than ambient seawater (Somerville et al. 1987). Smith et al. (1996) noted that typical characteristics of a discharge result in a buoyant plume that comes to the surface within 10 meters of the open-ended outfall.

Hydrodynamic characteristics of the discharge of produced water play an important role in governing the fate of the effluent. Considerable attention has been given to modeling hydrodynamic mixing between the effluent and the ambient seawater for the assessment of the environmental impact (Smith et al. 1996, Brandsma and Smith 1996, Strømgren et al. 1995, Somerville et al. 1987) and ocean environmental risks (Karman and Reerink 1998, Meinhold et al. 1996). In addition to plume trajectory and turbulent diffusion, initial dilution is one of the most important measures in such a hydrodynamic modeling (Washburn et al. 1999, Smith et al. 1996, Strømgren et al. 1995, Somerville et al. 1987). This chapter outlines the definition of initial dilution and its use in design of effluent discharges. Critical reviews on presently available initial dilution models are presented. It also discusses potential modeling approaches in dealing with drawbacks of the models.

Table 2.1 Typical characteristics of discharge and receiving waters for different regions (data from Brandsma and Smith 1996, Smith et al. 1996, Somerville et al. 1987)

Parameters	Region			
	Bass Strait	Gulf of Mexico	Java Sea	North Sea
Discharge Rate (m^3/day)	14,000	3977.8	26,235 – 123,225	10,000
Effluent Temperature (°C)	90	29	62 – 90	30
Effluent Density (kg/m^3)	988	1088	-	1014
Ambient Density* (kg/m^3)	1026	1017	-	1027
Density Gradient (kg/m^4)	0	0.15	-	0
Port Diameter (m) or Discharge configuration	0.2	0.2	Holes of 2"x4"	0.76
Depth above Discharge (m)	12	0.3	3 – 15	5
Port Orientation	Downwards	Downwards	Radial	Horizontal
Sea Water Depth (m)	72	27.4	21.3 – 30.5	150
Sea Water Speed (m/s)	0.3	0.03 - 0.25	-	0.3

*ambient density in the area closed to the discharge point.

2.2. Initial Dilution in Hydrodynamic Modeling and Design

In modeling, hydrodynamics of produced water effluent from an ocean outfall can be conceptualized as a mixing process occurring in two separate regions. The first region is referred to as the "near field" in which the initial jet characteristics of momentum flux, buoyancy flux, and outfall geometry influence the jet trajectory and mixing. For this region, designers of the outfall may expect different characteristics of the initial mixing, such as the degree of dilution, through appropriate manipulation of design variables. The second region is referred to as the "far field" where the effluent plume travels farther away from the source, and the source characteristics become less important. The trajectory and dilution in the far field are mainly controlled by characteristics of ambient seawater, such as the strength and direction of seawater currents, through buoyant spreading motions and passive diffusion (Doneker and Jirka, 1990). A typical schematic depiction of the near and far fields is shown in Figure 2.1.

In hydrodynamic modeling, initial dilution has been widely used as a measure of the mixing in the near field. By definition, initial dilution is the dimensionless ratio of pollutant concentration in the wastewater effluent prior to discharge to the concentration at an equilibrium level, or the free surface, or seabed. Initial dilution can also be expressed in terms of centerline dilution, which is dilution at the centerline of the jet above, or below, the discharge port. Initial dilution occurs because of the entrainment of the surrounding fluid during the rise or sink of the effluent from the outfall ports. This rising or sinking motion occurs because of buoyancy resulting from the difference between the densities of wastewater and seawater. Figure 2.2 shows a typical depiction of a rising buoyant jet in a current for typical initial dilution modeling.

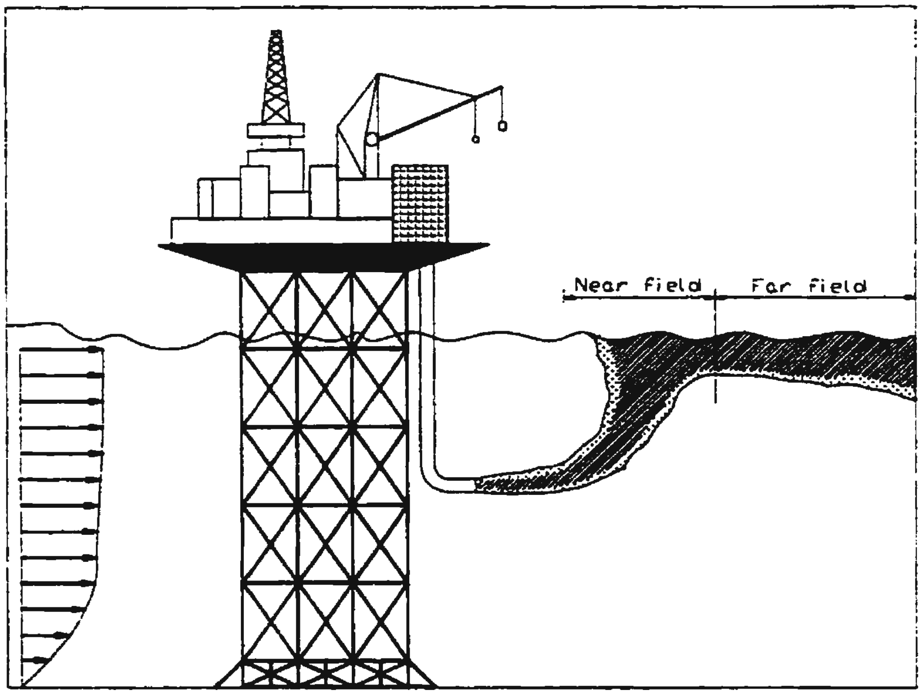


Figure 2.1. Schematic depiction of buoyant jet and plume following a produced water discharge from offshore oil fields (not to scale)

The use of initial dilution for the evaluation of discharge scenarios has been a traditional practice in the management of various wastewaters, including the release of sewage discharges, cooling water from a power plant, and produced water from oil production platforms. The discharge facility is usually designed in such a way that the effluent mixes effectively with ambient seawater. The design is not simply to dilute the effluent but, more importantly, to permit natural processes in the ocean to stabilize the waste with minimal environmental damage. In such a design, initial dilution is used as one important measure to investigate the degree of the mixing between the wastewater effluent and ambient seawater. Proper design ensures that the discharge results in sufficiently high initial dilution with minimal thickness of the effluent slick. High initial dilution is also required to maintain acceptably low ecological risks or to comply with relevant water quality standards within a designated mixing zone.

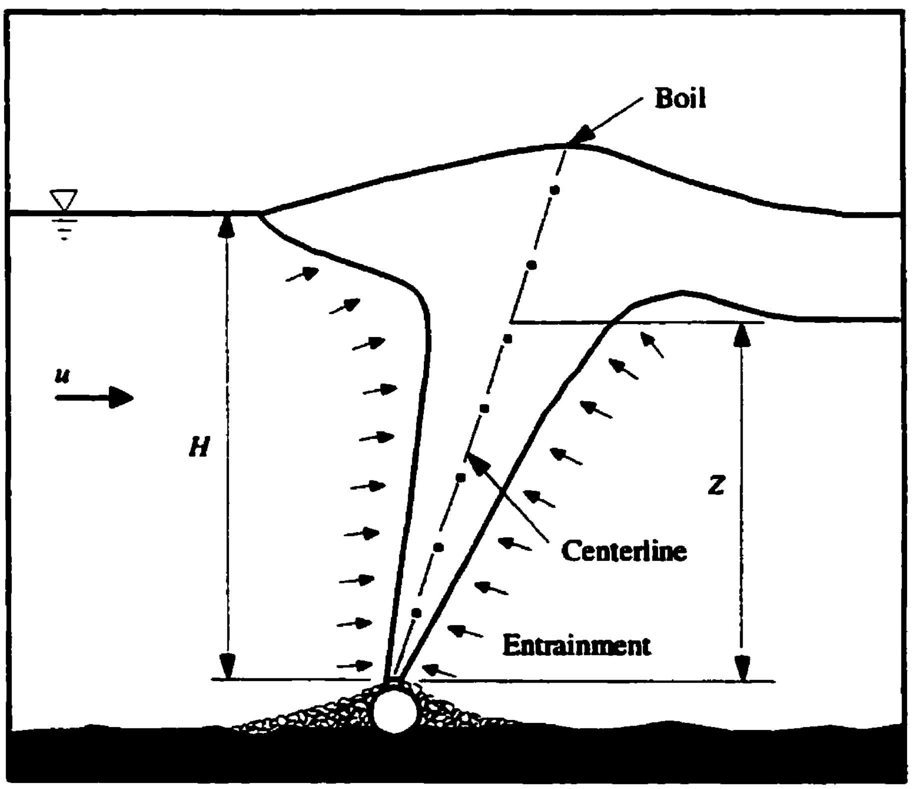


Figure 2.2. Sketch definition for typical initial dilution modeling

2.3. Previous Work on Initial Dilution Modeling

Many studies have been performed in the past for modeling initial dilution. For a stagnant ambient water condition, Cederwall (1968) provides a good, simple, empirical initial dilution model. This model is commonly accepted because its estimated dilution values generally agree with other theoretically and experimentally derived results (Sharp 1989a, Wood et al. 1993). In moving waters, however, there appears to be no universally accepted model for initial dilution calculations (Sharp and Moore 1989, Sharp and Moore 1987, Lee and Neville-Jones 1987a).

Mathematical modeling approaches based on the fundamental equations of motion have been employed for buoyant jets of drilling mud and produced water discharges (Brandsma et al. 1992, Brandsma et al. 1980, Reed et al. 1996, Skatun 1996). In developing the initial descent model, for example, the Offshore Operators Committee (OOC) model (Brandsma et al. 1992; Brandsma et al. 1980) was based on the equations of conservation of mass, momentum, buoyancy and constituent flux, and further based on the assumption of independent clouds of the Lagrangian advection-dispersion scheme. The mathematical models for initial dilution are theoretically sound, but suffer from a lack of well documented, detailed data for validation (Sharp and Moore 1987). When applying mathematical models, Andrade and Loder (1997) noted that the application should not be viewed as reliable unless they have been properly validated.

Andrade and Loder (1997) compared the performance of mathematical models for initial dilution calculations. They found that the OOC model provides similar qualitative features of the plume evolution and, in some cases, close agreement in values of the plume radius and dilution with those of the United State Environmental Protection Agency (U.S. EPA) models (Muellenhoff et al. 1985). The OOC model was originally developed for discharge evaluation of drilling wastes; and the U.S. EPA models are commonly employed for the analysis of sewage discharges. However, Sharp and Moore (1989) found that the same U.S. EPA mathematical models typically overestimated dilution by a factor of about 2 to 4.

An alternative approach to the modeling of a buoyant jet is to use empirical equations derived from experimental data. This approach has been applied for simulating initial dilution of produced water in the Santa Barbara Channel near Carpinteria, CA (Washburn et al. 1999) using the RSB (Roberts, Snyder and Baumgartner's) model, which is based on dimensional analysis and on laboratory experiments described by Roberts et al. (1989a-c). Empirical equations for initial dilution have also been employed for produced water discharge from the Krisna platform in the Java Sea, Indonesia (Smith et al. 1996). In this case, the Cornell Mixing Zone Expert System (CORMIX) model (Jirka et al. 1996) was calibrated and used for initial dilution calculations. CORMIX is computer software that compiles flow classifications and mixing behaviors of effluent discharges. For a given case of flow classification, mixing behavior is based on published empirical equations or experiments.

Applying empirical equations for dilution analysis does have the advantage of being based on physical data (Sharp and Moore 1987), so there is an increased confidence in the reliability of the modeling. In deriving empirical equations for initial dilution, an asymptotic approach has been widely used. The approach derives equations with the aid of dimensional analysis and data from laboratory or field experiments (Wright 1977a, Fisher et al. 1979, Lee and Neville-Jones 1987a, 1987b, Robert et al. 1989a, 1989b, 1989c, Lee and Cheung 1991, Wood 1993, and Proni et al. 1994).

Using asymptotic approaches, initial dilution of a round turbulent buoyant jet discharge in unstratified moving waters can be physically represented by the relevant parameters (Wright 1977a, Lee and Neville-Jones 1987a, Lee and Cheung 1991):

$$S = f(Q, M, B, u, z) \tag{2.1}$$

in which S is the initial dilution at depth above discharge z; u is the ambient current speed; and Q is the outfall discharge rate. M is the discharge momentum flux, defined as:

$$M = u_j Q \tag{2.2}$$

where u_j is the velocity of jet discharge. B is the buoyancy flux, defined as:

$$B = Q\, g\, \frac{\rho_a - \rho_o}{\rho_a} \tag{2.3}$$

where g is the gravitational acceleration; and ρ_a and ρ_o are densities of the ambient seawater and effluent, respectively. Since all these parameters have units of lengths and time only, Buckingham's π-theorem indicates that the phenomenon can be defined by only four dimensionless groups.

Following Wright (1977a) and Lee and Cheung (1991), the jet-ambient parameters can be combined into length scales, each of which characterizes a particular aspect of the general problem. The two length scales that characterize the jet discharge are l_M and l_Q, which are defined as:

$$l_M = \frac{M^{3/4}}{B^{1/2}} \tag{2.4}$$

$$l_Q = \frac{Q}{M^{1/2}} = d\left(\frac{\pi}{4}\right)^{1/2} \tag{2.5}$$

where d is the diameter of the port. The length scale l_M is a measure of the distance at which the buoyancy becomes more important than the jet momentum; the length scale l_Q is a measure determining whether the jet geometry will have a direct influence on the flow characteristics.

In the presence of an ambient velocity, two more length scales can be formed, i.e. l_m and l_b, which are defined as:

$$l_m = \frac{M^{1/2}}{u} \tag{2.6}$$

$$l_b = \frac{B}{u^3} \tag{2.7}$$

The length scale l_m relates to the interaction of a momentum-dominated jet with a cross-flow; and the length scale l_b represents the vertical distance at which the velocity induced by the buoyancy (proportional to $B^{1/3}/z^{1/3}$) has decayed to the ambient velocity value u.

If the functional relationship in equation (2.1) is expressed in non-dimensional parameters formed from the various length scales, one possible result is (Wright 1977a):

$$S = f\left(\frac{l_Q}{l_b}, \frac{l_m}{l_b}, \frac{z}{l_b}\right) \tag{2.8}$$

In dealing with this problem, a simplified solution using an asymptotic approach is usually adopted (Lee and Neville-Jones 1987a, Lee and Neville-Jones 1987b, Wright 1977a) because of the number of independent parameters that must be considered. In this approach, the number of independent factors affecting the system is reduced through physical reasoning. For instance, by considering the effects of the jet momentum and the buoyancy separately, the number of independent parameters is reduced. For buoyancy dominated discharges, $l_m/l_b << 1$, and for negligible volume flux, $l_Q/l_b << 1$, the relationship in equation (2.8) becomes (Lee and Cheung 1991):

$$S = f(z/l_b) \tag{2.9}$$

The relation developed using the asymptotic solutions is interpreted by examining the relative magnitude of the various length scales, primarily l_m and l_b, and by assuming that the analysis is to be applied for distances somewhat greater than l_Q from the source.

Large numbers of initial dilution models have been developed using this approach for different cases (e.g. Wright 1977a, Fischer et al. 1979, Wood et al. 1993). For buoyancy dominated discharges, e.g. freshwater discharges into the ocean, the following relationships are commonly used (Lee and Cheung 1991):

$$\frac{SQ}{ul_b^2}=C_1\left(\frac{z}{l_b}\right)^{5/3} \qquad \text{for BDNF} \qquad (2.10)$$

$$\frac{SQ}{ul_b^2}=C_2\left(\frac{z}{l_b}\right)^{2} \qquad \text{for BDFF} \qquad (2.11)$$

The terms BDNF and BDFF refer to the condition of the discharges, i.e. buoyancy-dominated near field (BDNF) for (z/l_b <<1) and buoyancy-dominated far field (BDFF) for (z/l_b >>1). The coefficients C_1 and C_2, were determined from experimental data, and are 0.1 and 0.51 for the BDNF and BDFF, respectively (Lee and Cheung 1991).

The reliability of using asymptotic solution-based initial dilution models has been addressed from several aspects. Analytically, Sharp (1989b) noted that rearranging these equations reveal that the ambient current speed is absent in the BDNF zone, and the effluent buoyancy will have no effect in the BDFF zone. The model is therefore conceptually questionable. Lee and Neville-Jones (1989) noted that for moderately small values of z/l_b of about 0.1, the dilution can exceed the dilution in still water by a factor of 2 or 3, and that this phenomena may be related to a marked change in flow structure (jet bifurcation). However, it was shown in the field observations of the Hollywood outfall that a low value of z/l_b of about 0.04 to 0.1 was associated with ambient seawater currents of about 8.5 to 10 cm/s (data from Proni et al. 1994). This suggests that effects of the ambient seawater current are not negligible even in cases with moderately small values of z/l_b, and thus should not be missing in the model formulation (BDNF).

Numerically, the asymptotic solutions (equations 2.10 and 2.11) can also be of concern. Figure 2.3. presents a curve fitting of equations 2.10 and 2.11 with the data from Lee and Cheung (1991). Traditionally, the dilution data were plotted in the form of SQ/ul_b^2 versus z/l_b (Lee and Neville-Jones 1987a, Lee and Cheung 1991, Wood 1993, Proni et al. 1994). Huang et al. (1998) suggest that the data may be plotted in the form of SQ/uz^2 versus z/l_b so that the transition between BDNF and BDFF is clearly identified. From Figure 2.3 it is shown that the transition region is evident at z/l_b about 0.05 to 0.5.

It can be seen in Figure 2.3 that in the BDNF region, in general, equation 2.10 underestimates dilution. There may be significant bias as the figure is presented in log-log form. It is also evident that using either the BDNF or BDFF equation may result in a substantial error when it is applied in the transitional region. Facing this problem, Lee and Cheung (1991) estimated a value of initial dilution for the transitional region using the BDNF equation with a modified coefficient of $C_I = 0.21$.

The use of the BDNF equation with the modified coefficient seems to be practical, but it is unrealistic when considering the nature of the data in the transition region. Consider data in the transition region, unlike the slope of the data in the BDNF region, the slope in the transition region is not negative as the data suggest. If continuous solution from the BDNF through the transition to the BDFF is expected, then the slope in the transition region should be positive. Not only does the modification (Lee and Cheung 1991) give unrealistic slope but also discontinuity of solutions between the regions (as shown in Figure 2.4).

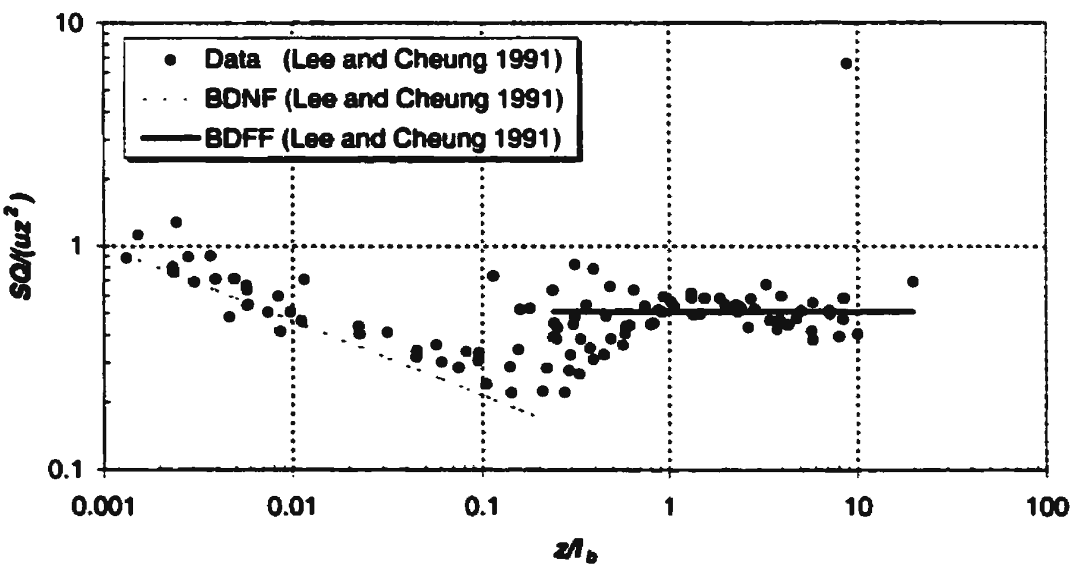


Figure 2.3. Curve fitting of asymptotic solutions and laboratory data

Lee and Neville-Jones (1989) noted that the concept of a BDNF and BDFF is strictly valid only for $z/l_b << 1$ and $>> 1$, respectively. However, in the field, not all cases of ocean outfall can exactly be classified into one of the regions (BDNF or BDFF). For example, field studies of South Florida outfalls reveal that the outfalls can be characterized as being in the transitional region, i.e. between BDNF and BDFF (Hazen and Sawyer 1994, Mukhtasor et al. 1999b, Proni et al. 1994). This raises more evidence of the need of developing an alternative initial dilution model applicable for such a case.

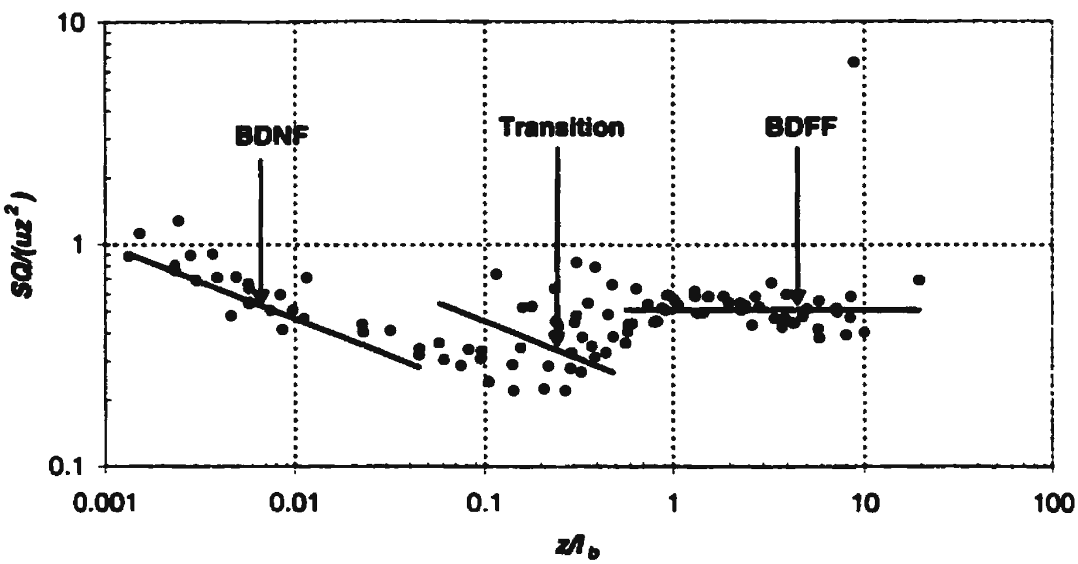


Figure 2.4. Discontinuity of asymptotic solutions proposed by Lee and Cheung (1991)

To overcome the above problems, alternative transitional initial dilution models have been proposed. Alternative models were developed using a dimensional analysis combined with a statistical analysis (Hazen and Sawyer 1994, Proni et al. 1994). The models were based on data from field studies of South Florida outfalls, specifically two single-port discharges (Hollywood and Broward outfalls) and two diffuser discharges (Miami-Central and Miami-North outfalls). A probabilistic initial dilution model using the same approach (i.e. a dimensional analysis combined with a statistical analysis) is also available for single port discharge based on data from the Hollywood outfall (Mukhtasor et al. 1999b). However, these models are limited for the transition region defined by z/l_b ranging from 0.04 to 5.

Huang et al. (1998) proposed a centerline initial dilution equation that spans all flow regimes, from the BDNF, through the transition, to the BDFF, providing continuous predictions for dilutions. The model was derived based on the continuity equation for the buoyant jet flow with a hypothesis of additive shear and forced entraintment. Holding this hypothesis, the Huang et al. (1998) presented the following model:

$$\frac{SQ}{uz^2} = a\left(\frac{z}{l_b}\right)^{-1/3} + \frac{b}{1+c\left(\frac{z}{l_b}\right)^{-d_2}} \tag{2.12}$$

where a, b, c, and d_2 are model constants, which were determined by "trial and error". In this approach, the coefficients from Lee and Cheung (1991) were used to determine two of the four constants in the equation 2.12 (a = 0.10 and b = 0.51). The other two constants were determined subjectively using the goodness of fit, which was evaluated by eye (c = 0.10 and d_2 = 2). The performance of the model was compared to the asymptotic-based solutions and data from Lee and Cheung (1991) as shown in Figure 2.5.

Despite inherent uncertainty because of physical instability as the data suggest, the Huang et al. (1998) model (equation 2.12) provides unique values of dilution in the transitional region. As can be seen in Figure 2.5, however, outside the transitional region, solutions given by the Huang et al. (1998) model are practically the same as those given by the Lee and Cheung (1991) model for BDNF. On the other hand, at $z/l_b > 0.5$ the values given by the Huang et al. (1998) model are somewhat higher that those given by the Lee and Cheung (1991) model for BDFF. Huang et al's (1998) solutions underestimate dilution at BDNF and overestimate dilution at BDFF. To investigate the problem more closely, residuals (data minus estimated value of y-axis) of Figure 2.5 can be evaluated. Based on the data from Lee and Cheung (1991), a residual plot at different regions of the BDNF, transition and BDFF, is shown in Figure 2.6. As shown in this figure, the residuals are positive, about zero, and negative for the BDNF, transition, and BDFF, respectively. This "structured bias", together with the use of the unsystematic approach (i.e. trial and error), leads to the need to develop a new approach to initial dilution modeling for a buoyancy-dominated jet.

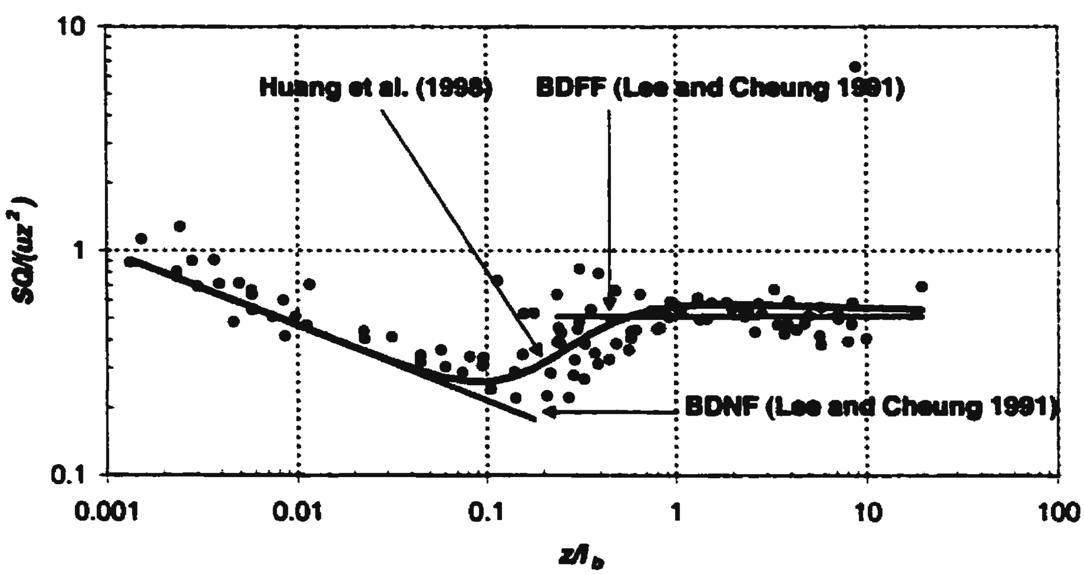


Figure 2.5. Comparison between initial dilution models and laboratory data (dot point)

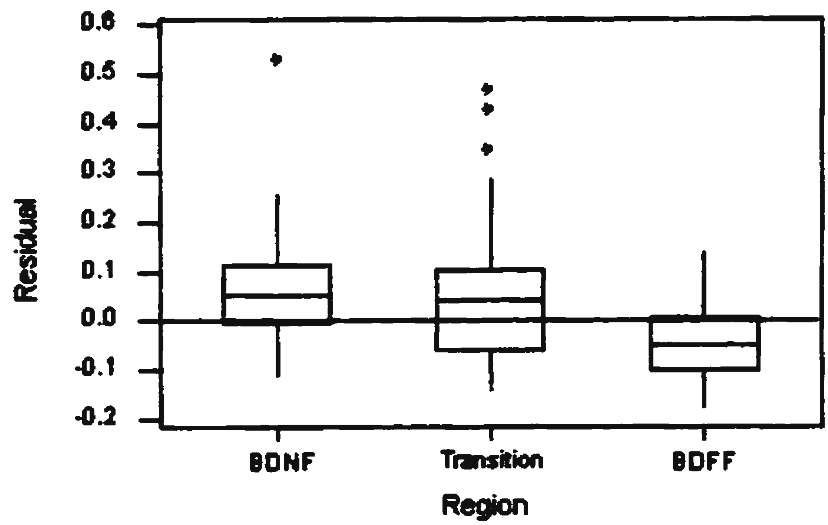


Figure 2.6. Residuals at different region for the Huang et al. (1998) model

2.4. Potential Modeling Approaches

Encountering the conceptual and accuracy problems discussed above, alternative approaches to initial dilution modeling have been proposed. An alternative approach of modeling was assessed by reanalyzing available experimental data, using methods of the Artificial Neural Network (ANN) combined with the Response Surface Methodology (RSM) (Mukhtasor et al. 2000d). ANN is an information processing system that consists of a number of interconnected computational elements called processing elements or neurons. By organizing the neurons into different layers and connecting them with proper weights, networks that are capable of "learning" can be developed (Malik 1993). In Mukhtasor et al. (2000d), after training and validation, the ANN performance was then modeled using RSM which is a collection of mathematical and statistical techniques for dealing with cases where several independent variables influence a dependent variable, or response, with a goal of optimizing this response (Montgomery 1976). Detailed description and discussion of ANN and RSM is not given here, but can be found elsewhere, e.g., Bishop (1994), Montgomery (1976), Myers and Montgomery (1995), and Smith (1993).

In Mukhtasor et al. (2000d), the ANN was trained using experimental data, and the length-scale ratios (z/l_Q, z/l_m, z/l_b) were used as input parameters of the network. After training, the network was validated using different sets of data. The RSM was then employed to predict the performance of the ANN in relating the initial dilution with associated parameters. The aim of finding a replacement model for the ANN is to come up with a simple initial dilution model that is easily integrated with other models for further analysis. It was concluded in that study that the ANN provides better results than asymptotic-based models in terms of accuracy. However, the ANN-based RSM model did not result in satisfactory results for replacement of the ANN. The reason for this problem is not clear, but it might be because the ANN only provided average estimates of initial dilution. The variability of the dilution data might be then reduced. Therefore, when the outputs of the ANN were used as inputs of the RSM, models developed by RSM cannot resemble actual initial dilution. These two methods (RSM and ANN) were then not used further because appropriate laboratory data specifically suitable for RSM modeling was not available, and because an ANN-based initial dilution model is not easily integrated with other models such as far field dispersion models and ecological risk assessment models.

Considering the nonlinear nature of the data (see Figures 2.3 to 2.5), it appears that nonlinear regression modeling combined with length scale analysis may be required to approach the problem (Mukhtasor et al. 2001a). The Huang et al. (1998) model is nonlinear in nature, but it does not provide a systematic approach to modeling neither objective measures of the goodness of the fit. Application of the nonlinear regression modeling for initial dilution is an alternative to the presently available models as discussed in the next chapter.

2.5. Summary

This chapter discussed the concept of hydrodynamic modeling in terms of near and far field models. At the beginning of the chapter, definition of initial dilution and its importance in the hydrodynamic modeling and outfall design are outlined. Then, presently used modeling approaches are reviewed and the focus is directed at critical review of conceptual and numerical problems associated with presently available initial dilution models. Potential modeling approaches to initial dilution are also discussed in this chapter.

Mathematical models based on the fundamental equations of motion have been proposed in the past for modeling initial dilution. They are theoretically sound but suffer from a lack of well documented, detailed data for validation. An alternative approach employs empirical equations based on physical data; this increases confidence in the reliability of the modeling. An approach that is widely used to develop empirical initial dilution models is an asymptotic solution, which derives equations with the aid of dimensional analysis and data from laboratory or field experiments. The asymptotic approach gives a solution that is limited for two different zones, namely the buoyancy-dominated near field (BDNF) and the buoyancy-dominated far field (BDFF).

The reliability of using asymptotic solution-based initial dilution models has been addressed from several aspects. Analytically, it has been conceptually questionable because rearranging the models reveals that the ambient current speed is absent in the BDNF zone, and that the effluent buoyancy has no effect in the BDFF zone. Field test data indicated that even at moderately small values of z/l_b the effects of the ambient seawater current can be quite significant, and should not thus be missing in the model formulation (BDNF). Numerically, based on laboratory data (Lee and Cheung 1991), the asymptotic solution underestimates dilution in the BDNF region and has no unique solution in the transitional region. Not only does the manipulation approach to the transitional region adopted in the previous studies give an unrealistic answer but also discontinuity of solutions between the regions (BDNF, transition and BDFF). Although it was believed that the concept of a BDNF and BDFF is strictly valid only for $z/l_b << 1$ and $>> 1$, respectively, in the field, some cases of the discharges cannot exactly be classified into one of the regions (BDNF or BDFF). This raises another question of the applicability of the asymptotic solution for such cases.

Alternative models have been proposed to overcome the above problems by using a dimensional analysis combined with a statistical analysis of field data. A probabilistic initial dilution model using the same approach is also available for single port discharge.

However, these models are limited for the transitional region defined by z/l_b ranging from 0.04 to 5. A model based on the continuity equation with a hypothesis of additive shear and forced entraintment was proposed as another alternative (Huang et al. 1998), which provides an equation that spans all flow regimes, from the BDNF, to the transition, to the BDFF. A systematic approach to modeling and objective measures of the goodness of the fit are not shown in the Huang et al. (1998) model. This model underestimates dilution in the BDNF and overestimates dilution in the BDFF. Beside the assumption of stagnant water adopted for BDNF region, residual analysis shows that this model suffers from the "structured bias".

Other potential modeling approaches are also discussed in this chapter, including Artificial Neural Network (ANN) and Response Surface Methodology (RSM). However, previous studies show that the ANN provides better results than asymptotic-based models in terms of accuracy but the ANN-based RSM models did not result in satisfactory results for the replacement of the ANN. Considering the nonlinear nature of the data, it appears in this chapter that nonlinear regression modeling combined with length scale analysis may be required to approach the problem.

Chapter 3

Development and Evaluation of Initial Dilution Models

3.1. Introduction

Conceptual and numerical problems associated with presently available initial dilution models have been discussed in the previous chapter. A nonlinear regression modeling combined with the additive entrainment hypothesis has been recommended as a potential alternative approach to modeling initial dilution. This chapter provides a more detailed description of this modeling. Following discussion on the characteristics of initial dilution, the approach used in the model development is described. After that, model evaluation and comparison with presently available models are presented. An application example is shown at the end of the chapter for deterministic and probabilistic initial dilution modeling.

3.2. Characteristics of Initial Dilution Data

An empirical initial dilution model was developed in this study based on data from Lee and Cheung (1991). The data was obtained from Lee and Cheung's (1991) series of 48 laboratory experiments (resulting in 107 sets of data) with a buoyancy-dominated vertical heated water jet in a steady crossflow. The experiments were carried out in a 10 m × 0.45 m by 0.3 m wide laboratory flume, from which the variation of characteristic jet dilution with z/l_b was studied. A statistical summary of parameters from the experiments is given in Table 3.1. More detailed description of the experiment and the data can be found in Lee and Cheung (1991).

Table 3.1. Characteristics of initial dilution data from Lee and Cheung (1991) experiments

Parameter	average	median	min	max	stdev	5%-tile	95%-tile
u_a/u_j	0.16	0.13	0.02	0.52	0.13	0.03	0.45
l_M (m)	0.02	0.02	0.01	0.03	0.01	0.01	0.03
l_b (m)	6.11	0.42	0.00	116.80	15.16	0.01	31.71
l_m (m)	0.08	0.05	0.01	0.38	0.07	0.01	0.22
l_Q (m)	0.007	0.007	0.007	0.007	0.000	0.007	0.007
z (m)	0.12	0.12	0.05	0.32	0.05	0.05	0.19
l_M/l_b	0.4033	0.0560	0.0002	12.0000	1.2768	0.0005	1.8000
l_Q/l_b	0.1253	0.0158	0.0001	2.6580	0.3160	0.0002	0.5809
l_m/l_b	0.3722	0.1470	0.0032	5.1597	0.6491	0.0065	1.5080
z/l_b	1.5416	0.3305	0.0013	19.6000	2.8246	0.0029	7.1377

Most hydrodynamic parameters shown in Table 3.1 have been defined in the previous chapter; and u_j is the jet velocity at the outlet of the port. The length scale ratios (the last four rows of the table) show that the experiment covers conditions of many operating ocean outfalls, including produced water outfalls that are shown in Table 3.2. Three of the four produced water outfalls shown in Table 3.2. are in operation but the last column, i.e. that for Terra Nova, is not in operation and the values given in the last column are estimated

(Petro-Canada 1996, Mukhtasor et al. 2000c). It is evident that the laboratory experiment covers a wide range of characteristics of produced water outfalls and that the data can be used to develop an initial dilution model for application to produced water discharges.

Table 3.2 Length scales for typical produced water discharges
(data from Brandsma and Smith 1996, Petro-Canada 1996, Somerville et al. 1987)

Parameters	Region			
	Bass Strait	Gulf of Mexico*	North Sea	Terra Nova**
Discharge Rate (m^3/day)	14000	3978	10000	18300
Effluent Temperature (°C)	90	29	30	96
Effluent Density (kg/m^3)	988	1088	1014	988
Ambient Density* (kg/m^3)	1026	1017	1027	1025
Density Gradient (kg/m^4)	0	0.15	0	-
Port Diameter (m)	0.2	0.2	0.76	0.305
Depth of Discharge (m)	12	0.3	5	10
Port Orientation	Downwards	Downwards	Horizontal	Horizontal
Sea Water Depth (m)	72	27	150	80
Sea Water Speed (m/s)	0.300	0.03 to 0.25	0.300	0.140
u_a/u_j	3.7	1.3	0.4	0.6
l_M (m)	3.6	0.7	0.6	2.5
l_b (m)	2.2	2 to 1168*	0.5	27.3
l_m (m)	3.05	1 to 8.7	0.57	5.6
l_Q (m)	0.177	0.177	0.673	0.270
z (m)	12	27	5	10
l_M/l_b	1.653	0.001 to 0.37*	1.117	0.093
l_Q/l_b	0.081	0.0002 to 0.088*	1.265	0.010
l_m/l_b	1.398	0.007 to 0.5*	1.076	0.205
z/l_b	5.503	0.023 to 13.4*	9.393	0.366

*The density of produced water from the Gulf of Mexico is heavier than that of ambient water, resulting in negative buoyancy and the plume sinking deep into the ocean.
**Data from this column is only estimated, at this time no produced water was expected (Petro-Canada 1996).

3.3. Preliminary Analysis of the Data

As discussed in the previous chapter, from dimensional analysis Buckingham's π-theorem indicates that there are four dimensionless parameters, which can be used to model initial dilution empirically. The length scale analysis (Lee and Cheung 1991, Lee and Neville-Jones 1987a, Wood 1993, Wright 1977a) shows that for buoyancy-dominated discharge with relatively negligible volume flux, only the length scale ratio of z/l_b may be employed to parameterize initial dilution. Huang et al. (1998) proposed a model that relates initial dilution in terms of SQ/uz^2 versus z/l_b only.

A statistical analysis was employed in this study to evaluate correlation among the independent variables used in equation 2.8 (i.e. using three independent variables of l_Q/l_b, l_m/l_b and z/l_b). The equation is rewritten as:

$$S = f\left(\frac{l_Q}{l_b}, \frac{l_m}{l_b}, \frac{z}{l_b}\right) \tag{3.1}$$

Applying a statistical analysis to the data from Lee and Cheung (1991), it was found that the functional relationship in equation 3.1 contains multicolinearity. This phenomenon is indicated by a high variance inflation factor (VIF) of about 15 to 39, suggesting that the input variables have quite a strong correlation between one another, and that one variable may be explained by another. Therefore, simplification like the presently used approach (i.e. using one independent variable of z/l_b only) is both physically and statistically acceptable.

Prior to developing a model from data, it is first necessary to take a closer look at the structure of the data. The data in this study was plotted in the form of SQ/uz^2 versus z/l_b as shown in Figure 3.1. It can be seen in the figure that five points are suspected to be far from the expected fit of the rest of the data. These points are subject to further analysis before being used in the modeling. It is observed that points # 1 to 4 seem to be close to the other data but when considering the log scale of the data, they are actually quite far off. Point # 5 is extremely far from the rest of the data and it is not shown in Huang et al's. (1998) paper. It is however not clear whether this point is removed in their analysis or just overlapped with the legend of their graph.

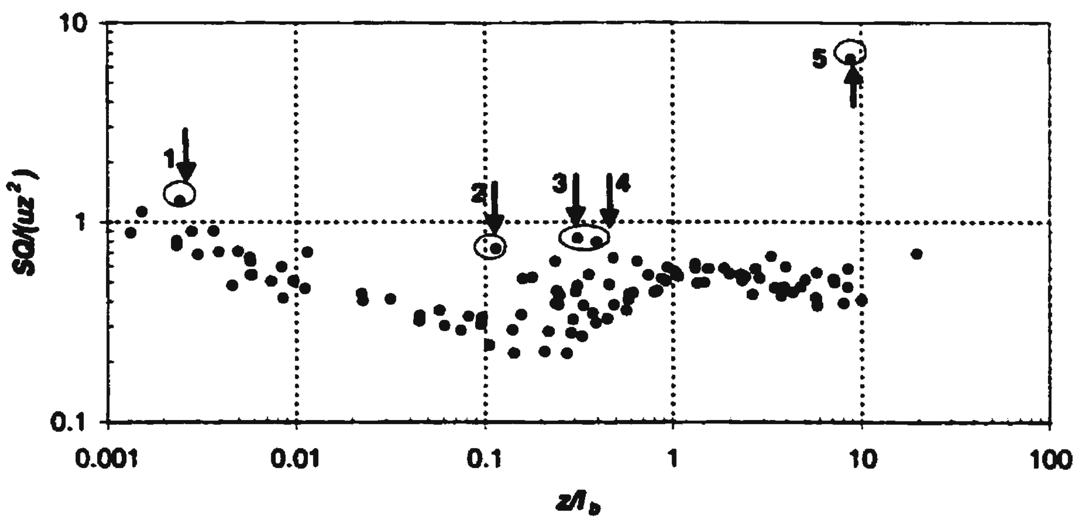


Figure 3.1. Plot SQ/uz^2 versus z/l_b for 107 sets of data from Lee and Cheung (1991)

From the plot itself it is difficult to determine if these five points are really outliers and whether they should be removed from the analysis. Upon rechecking the data, it was found that there is inconsistent data in row # 26 of 48 in Table 1 of Lee and Cheung's (1991) paper. This might be because of a recording, typing, or calculation error. As a result, point # 5 is extremely far from the others and was considered to be outlier. It was then removed in the subsequent analysis. As for the other four points, it was difficult to identify whether the same problem has been encountered so an effort of outlier identification was performed as discussed in the following sections.

3.4. Model Development

A centerline initial dilution model was developed based on the continuity equation for the buoyant jet flow with a hypothesis of additive shear and forced entraintment (Huang et al. 1998). In stagnant water or in a weak current like in the BDNF region, the shear between the boundary of the jet and the ambient seawater generates eddies that cause shear entrainment. On the other hand, when the ambient current is strong, a vortex pair is generated as a result of interaction between the jet and the current, resulting in forced entrainment. Holding the hypothesis of additive shear and forced entraintment, volume flux of the jet at the elevation z can be formulated as:

$$SQ = Q + E_s + E_f \tag{3.2}$$

where E_s and E_f are shear entrainment and forced entrainment, respectively. Assuming that the ambient current does not affect the shear entrainment, and that the forced entrainment factor can be derived from its asymptotic behaviors, as discussed in the previous chapter, Huang et al. (1998) proposed a centerline dilution:

$$\frac{SQ}{uz^2} = 0.1\left(\frac{z}{l_b}\right)^{-1/3} + \frac{0.51}{1+0.1\left(\frac{z}{l_b}\right)^{-2}} \tag{3.3}$$

Modifications to this approach were made in this study; and as a result, initial dilution model developed here has a different formulation compared to the Huang et al. (1998) model. Measures of the goodness of fit were provided by using the least squares approach and residual diagnostics. Unlike the Huang et al. (1998) approach, it is assumed that the ambient current must affect the shear entrainment. This implies that the power of the first part of the right hand side of equation 3.3 has to be higher than −1/3 so that dilution in a weak current (i.e BDNF) will be positively correlated with the current, i.e. increasing current should result in an increase in dilution.

An additional modification was made by reducing the number of coefficients in the second part of the right hand side of equation 3.3. If this part is replaced with a form of forced-entrainment factor, $\phi(z/l_b)$, and if it is assumed that the current affects the shear entrainment, equation 3.3 can be rewritten as:

$$\frac{SQ}{u z^2} = e\left(\frac{z}{l_b}\right)^f + \phi\left(\frac{z}{l_b}\right) \tag{3.4}$$

where e and f are model coefficients to be determined based on the data. The forced-entrainment factor, $\phi(z/l_b)$, is formulated by assuming that as z/l_b goes to infinity, equation 3.4 should converge to the advected thermal solution in which the value of the left hand side of equation 3.4 approaches a constant. This requires that:

$$\phi(z/l_b) \longrightarrow 0 \qquad \text{as } z/l_b \longrightarrow 0 \tag{3.5}$$

$$\phi(z/l_b) \longrightarrow \text{constant} \qquad \text{as } z/l_b \longrightarrow \infty \tag{3.6}$$

The form of $\phi(z/l_b)$ that satisfies these requirements is proposed as:

$$\phi\left(\frac{z}{l_b}\right) \propto w\exp\left(\frac{h}{z/l_b}\right) \tag{3.7}$$

where w and h are model coefficients. Using equation 3.7, equation 3.4 can be written as:

$$\frac{SQ}{u z^2} = e\left(\frac{z}{l_b}\right)^f + w\exp\left(\frac{h}{z/l_b}\right) \tag{3.8}$$

Equation 3.8 is a nonlinear equation so that a nonlinear regression can be employed to obtain the coefficients (e, f, w and h). The least squares approach was used so that estimates

($\hat{e}$, $\hat{f}$, $\hat{w}$ and $\hat{h}$) of the coefficients in equation 3.8 were determined by minimizing the sum square error (SS_{res}). Given a set of data $\left(\left(\frac{SQ}{uz^2}\right)_i, \left(\frac{z}{l_b}\right)_i\right)$ for i=1,2,3,...,n, the sum square error was formulated as:

$$SS_{res} = \sum_{i=1}^{n} \left(\left(\frac{SQ}{uz^2}\right)_i - \hat{e}\left(\frac{z}{l_b}\right)_i^{\hat{f}} - \hat{w}\exp\left(\frac{\hat{h}}{z/l_b}\right)_i \right)^2 \tag{3.9}$$

To minimize this, equation 3.9 was differentiated with respect to $\hat{e}$, $\hat{f}$, $\hat{w}$ and $\hat{h}$, and each derivative was set to zero, resulting in the following equations:

$$\sum_{i=1}^{n} \left(\left(\frac{SQ}{uz^2}\right)_i - \hat{e}\left(\frac{z}{l_b}\right)_i^{\hat{f}} - \hat{w}\exp\left(\frac{\hat{h}}{z/l_b}\right)_i \right)\left(-\left(\frac{z}{l_b}\right)_i^{\hat{f}} \right) = 0 \tag{3.10}$$

$$\sum_{i=1}^{n} \left(\left(\frac{SQ}{uz^2}\right)_i - \hat{e}\left(\frac{z}{l_b}\right)_i^{\hat{f}} - \hat{w}\exp\left(\frac{\hat{h}}{z/l_b}\right)_i \right)\left(-\hat{e}\left(\frac{z}{l_b}\right)_i^{\hat{f}} \ln\left(\frac{z}{l_b}\right)_i \right) = 0 \tag{3.11}$$

$$\sum_{i=1}^{n} \left(\left(\frac{SQ}{uz^2}\right)_i - \hat{e}\left(\frac{z}{l_b}\right)_i^{\hat{f}} - \hat{w}\exp\left(\frac{\hat{h}}{z/l_b}\right)_i \right)\left(-\exp\left(\frac{\hat{h}}{z/l_b}\right)_i \right) = 0 \tag{3.12}$$

$$\sum_{i=1}^{n} \left(\left(\frac{SQ}{uz^2}\right)_i - \hat{e}\left(\frac{z}{l_b}\right)_i^{\hat{f}} - \hat{w}\exp\left(\frac{\hat{h}}{z/l_b}\right)_i \right)\left(-\frac{\hat{w}}{\left(z/l_b\right)_i}\exp\left(\frac{\hat{h}}{z/l_b}\right)_i \right) = 0 \tag{3.13}$$

Equations 3.10 to 3.13 are also nonlinear in the parameter estimators $\hat{e}$, $\hat{f}$, $\hat{w}$ and $\hat{h}$. Therefore, they cannot be solved directly but require some type of iteration process. A method that is often used in finding the least square estimators in a nonlinear model is a *Gauss-Newton* procedure (Myers 1990). Essentially, the procedure is an iterative process which first expands the nonlinear function in equation 3.8 in a Taylor series around *a starting value* for the estimators - in this case the estimators are $\hat{e}$, $\hat{f}$, $\hat{w}$ and $\hat{h}$ - and which retains only linear terms. Thus:

$$\left(\frac{SQ}{u\,z^2}\right)_i \cong \hat{e}_0\left(\frac{z}{l_b}\right)_i^{\hat{f}_0} + \hat{w}_0 \exp\left(\frac{\hat{h}_0}{z/l_b}\right)_i + (\hat{e}-\hat{e}_0)\left(\frac{z}{l_b}\right)_i^{\hat{f}_0} + (\hat{f}-\hat{f}_0)\,\hat{e}_0\left(\frac{z}{l_b}\right)_i^{\hat{f}_0}\ln\left(\frac{z}{l_b}\right)_i$$

$$+ (\hat{w}-\hat{w}_0)\exp\left(\frac{\hat{w}_0}{z/l_b}\right)_i + (\hat{h}-\hat{h}_0)\frac{\hat{w}_0}{\left(z/l_b\right)_i}\exp\left(\frac{\hat{h}_0}{z/l_b}\right)_i \qquad (i=1,2,3,\ldots n) \qquad (3.14)$$

Equation 3.14 can be viewed as a linear approximation in the neighborhood of the starting values, $\hat{e}_0$, $\hat{f}_0$, $\hat{w}_0$ and $\hat{h}_0$. The equation 3.14 can be expressed in the form of:

$$\left(\frac{SQ}{u\,z^2}\right)_i - \hat{e}_0\left(\frac{z}{l_b}\right)_i^{\hat{f}_0} - \hat{w}_0 \exp\left(\frac{\hat{h}_0}{z/l_b}\right)_i \cong \gamma_1\omega_{1i} + \gamma_2\omega_{2i} + \gamma_3\omega_{3i} + \gamma_4\omega_{4i} \qquad (3.15)$$

where ω_{ji} is the derivative of the nonlinear function (equation 3.8) with respect to the *j*th parameter (coefficient). The γ_j is the *j*th parameter value minus the starting value. The left hand side of equation 3.15 can be considered as the residual when the parameters are replaced by starting values. For a given starting value, the ω_{ji} are known and can be considered as regressor variables in a linear regression, while the γ_j plays the role of a regression coefficient. As a result, the Gauss-Newton procedure builds on the linear regression structure:

$$y_i = \gamma_1\omega_{1i} + \gamma_2\omega_{2i} + \gamma_3\omega_{3i} + \gamma_4\omega_{4i} + \varepsilon_i \tag{3.16}$$

where,

$$y_i = \left(\frac{SQ}{uz^2}\right)_i - \hat{e}_0\left(\frac{z}{l_b}\right)_i^{\hat{f}_0} - \hat{w}_0 \exp\left(\frac{\hat{h}_0}{z/l_b}\right)_i \tag{3.17}$$

and ε_i is the error term of the regression model.

The *j*th parameter value, i.e. the value of $\hat{e}$, $\hat{f}$, $\hat{w}$ and $\hat{h}$, can be obtained from an iterative process as follows:

1. Determine the starting value of the *j*th parameter, $\hat{e}_0$, $\hat{f}_0$, $\hat{w}_0$ and $\hat{h}_0$.
2. Estimate γ_j in equation 3.16 by multiple linear least squares. Denote these first iteration estimates by $\gamma_{j,1}$.
3. Compute the new value of the *j*th parameter, e.g. $\hat{e}_1 = \hat{e}_0 + \gamma_{1,1}$.
4. Use the *j*th parameter from step 3 to replace the starting values.

5. Return to the first step, and continue the process until converegence is reached. The convergence implies that after, say r iterations, the residual sum squares and the jth parameter estimates are no longer changing.

This procedure can be carried out using a computer program or available statistical packages. Applying this procedure, the coefficients in equation 3.8 were determined using the statistical software SYSTAT from SPSS Science and DATAFIT from Oakdale Engineering. Both software packages gave the same answers to third or fourth decimal places; and for simplicity, second decimal places are given here, i.e. the mean ± standard error of 0.13 ± 0.02, -0.31 ± 0.03, 0.46 ± 0.02, and −0.22 ± 0.04 for $\hat{e}$, $\hat{f}$, $\hat{w}$ and $\hat{h}$, respectively. The coefficients in equation 3.8 (e, f, w and h) can be replaced with these estimates by introducing an error term, ε, so that the equation 3.8 can be expressed as:

$$\frac{SQ}{u\,z^2} = \hat{e}\left(\frac{z}{l_b}\right)^{\hat{f}} + \hat{w}\exp\left(\frac{\hat{h}}{z/l_b}\right) + \varepsilon \qquad (3.18)$$

where the error term, ε, is approximately normally distributed with a mean of about zero and standard deviation of 0.092.

3.5. Model Presentation

The proposed centerline initial dilution can be expressed in deterministic and probabilistic forms as shown in equation 3.19 and 3.20, respectively.

$$\frac{SQ}{u\,z^2} = 0.13\left(\frac{z}{l_b}\right)^{-0.31} + 0.46\exp\left(\frac{-0.22}{z/l_b}\right) \tag{3.19}$$

$$\frac{SQ}{u\,z^2} = (0.13 \pm 0.02)\left(\frac{z}{l_b}\right)^{(-0.31\pm 0.03)} + (0.46 \pm 0.02)\exp\left(\frac{(-0.22 \pm 0.04)}{z/l_b}\right) + N(0,\ 0.092) \tag{3.20}$$

where $N(0,\ 0.092)$ is a random quantity normally distributed with a mean of zero and a standard deviation of 0.092. This random quantity is basically the error term in the model given in equation 3.18. The above equations 3.19 and 3.20 are essentially the same except that equation 3.20 is specified probabilistically containing an error term, ε, and uncertainty measures of the model coefficients, i.e. the mean $\pm$ standard error. On the other hand, equation 3.19 is specified deterministically to estimate average dilution for a given value of input varibles. The advantage of using the probabilistic model is that uncertainty associated with the model itself can be explained. By doing so, the probabilistic presentation can contribute insight into the reliability of answers given by the model itself for a particular situation.

In outfall design, like hydraulic design in general, uncertainty could arise from various sources, including, but not limited to, data uncertainties, operational uncertainties, and model uncertainties (Tung 1994, Mukhtasor et al. 1999a). Data uncertainties are associated with inconsistency, and data measurement and recording errors. Operational uncertainties refer to human factors that are not accounted for in the analysis. Model uncertainties include the fact that the empirical laws do not describe the complete features of the true process and that some coefficients in the model cannot be quantified with absolute certainty.

However, empirical initial dilution models are commonly presented in a deterministic fashion without explicitly stating or elaborating the associated uncertainty (e.g. Lee and Cheung 1991, Wood 1993, Wright 1977a). As a result, applying this deterministic model may lead to loosing sight of the intrinsic uncertainty associated with the empirical equation. In that way, much of the information in the data is unused, resulting in unnecessary waste of information and perhaps flawed design decisions (Tung 1994). Therefore, the probabilistic model becomes a necessary alternative for initial dilution design problems. This is particularly important because a deterministic solution is only one special case of many of the probabilistic solutions, which encompass a wide spectrum of possible answers to a design problem (Tung 1994).

Figure 3.2 shows a comparison between deterministic and probabilistic fits of the models (equation 3.19 and 3.20) to the data. As can be seen from the figure, the deterministic solution (equation 3.19) provides a single answer for a given value of input variables. For example, at z/l_b of 0.1, SQ/uz^2 is 0.3. On the other hand, the probabilistic solution (equation 3.20) estimates that there is 50% certainty that SQ/uz^2 equal 0.3 when z/l_b is 0.1. At this z/l_b value, it is also possible to evaluate the probability of getting any possible value of SQ/uz^2 using the probabilistic model, for example, the probability of having SQ/uz^2 more than 0.5

when z/l_b is 0.1, or $p(SQ/uz^2 > 0.5$ given $z/l_b = 0.1)$. A more detailed explanation and calculation procedure for this probability can be given using an application example as presented in Section 3.7 of this chapter.

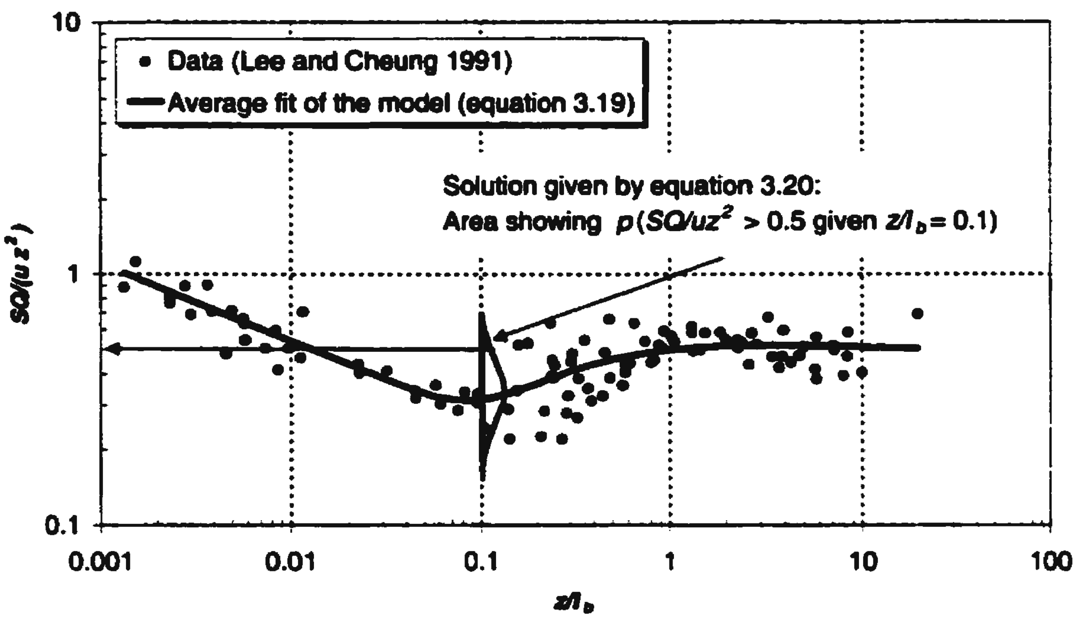


Figure 3.2. Deterministic and probabilistic fit of the models to the data

3.6. Model Evaluation

In this section, the performance of the proposed models (equation 3.19 and 3.20) is evaluated and compared with presently available models (Lee and Cheung 1991 and Huang et al. 1998). However, the proposed probabilistic model cannot be directly compared with presently available models, since they are deterministic. For a quantitative comparison purpose, only the general performance of the model is compared with presently available models, meaning that only equation 3.19 is compared with presently available models. Figure 3.3 shows a comparison between the Huang et al. (1998) model and equation 3.19.

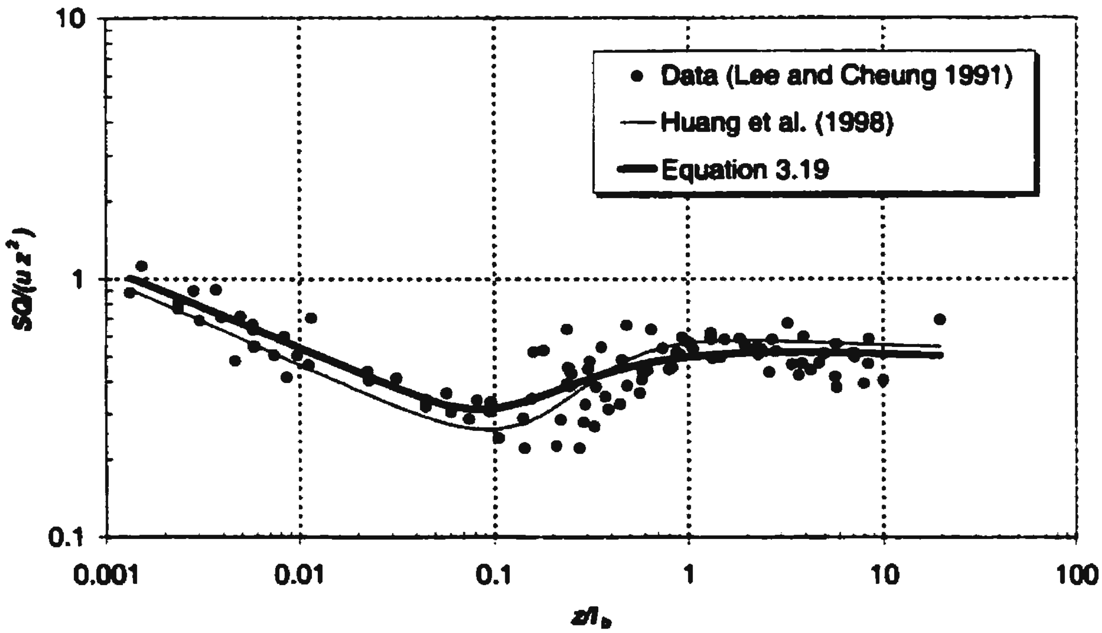


Figure 3.3. Comparison between the Huang et al. (1998) model and equation 3.19

Figure 3.3 shows that the performance of the proposed model is better than that of the Huang et al. (1998) model. The proposed model gives a *less-biased* solution in the whole range of the BDNF, the transition, and the BDFF regions. Compared to the Huang et al. (1998) model, the proposed model gives higher and lower dilutions for the BDNF and the BDFF regions, respectively. The problem of "structured bias" as shown in the previous chapter is not encountered in the proposed model. For any region, the mean residual of the proposed model is about zero. The comparison is shown in Figures 3.4 and 3.5.

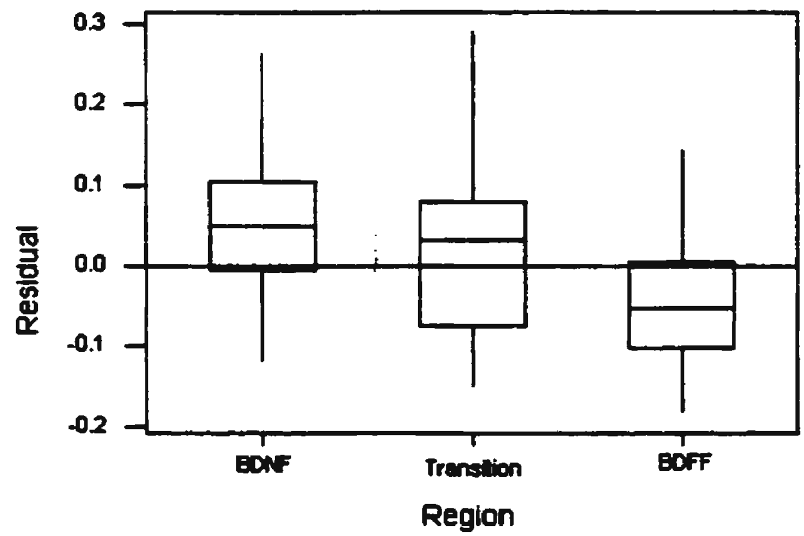


Figure 3.4. Residuals of the Huang et al. (1998) model

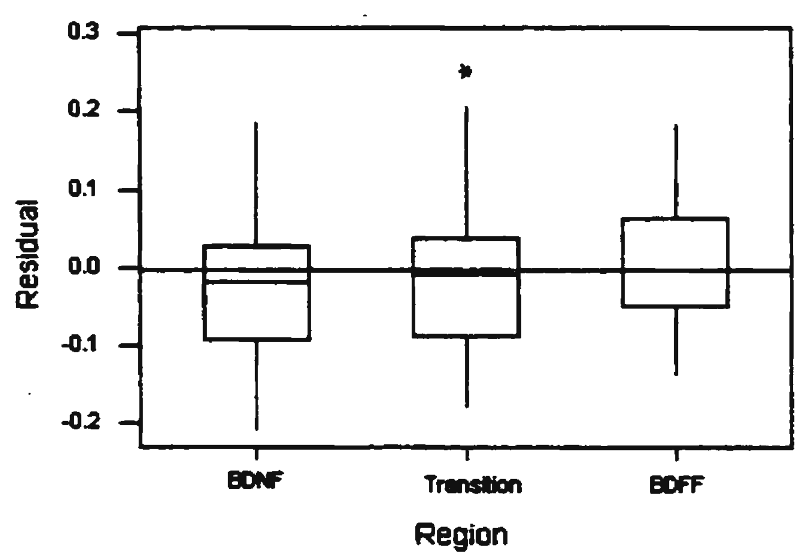


Figure 3.5. Residuals of equation 3.19

From a residual diagnostic, the proposed model has mean values of the residuals of –0.017, –0.007, and 0.004 for the BDNF, transition, and BDFF, respectively. These mean values indicate that the model is less biased than the Huang et al. (1998) model, which gives mean residuals of 0.065, 0.021, and –0.050 for the BDNF, transition, and BDFF, respectively.

Furthermore, it can be seen from Figure 3.5 that the spread of residuals is about the same for all cases, while Figure 3.4 shows that there is a wider spread of residuals in the transition region. This suggests that, compared with equation 3.19, the Huang et al. (1998) solution is relatively less precise in the transition region. The accuracy (bias and precision) comparisons may also be evaluated using a comparison between calculated dilution versus data, as shown in Figures 3.6 to 3.8. In those figures, a regression equation associated with the calculated dilution and data is provided for each model. Overall accuracy is also compared in terms of percentage error of calculated dilution. Statistics of the percentage error are presented in Figure 3.9.

From Figures 3.6 to 3.8, the performance of the three models can be assessed using the slope and intercept of the regression equation. For the unbiased model, the slope, intercept and coefficient of determination, R^2, are one, zero, and one, respectively. As can be seen from those regression equations, all models seem to be acceptable. However, Figure 3.9 provides more meaningful comparative interpretation using the percentage error of dilution, in which bias measure (mean, median), precision measure (standard deviation, 10- and 90-percentile), overall accuracy (mean square error, MSE) are provided. In general, equation

3.19 is relatively more accurate as indicated by the lowest value of mean, median, 10- and 90-percentiles, and MSE.

Considering the above comparisons and problems discussed in the previous chapter (i.e. the conceptual problem, non-unique solution in the transition region, and structured bias), equation 3.19 is considered to be more justifiable and preferable than presently available models, i.e. Lee and Cheung (1991) and Huang et al. (1998). The probabilistic form of the proposed model, equation 3.20, is also preferable for initial dilution calculations in uncertain situations.

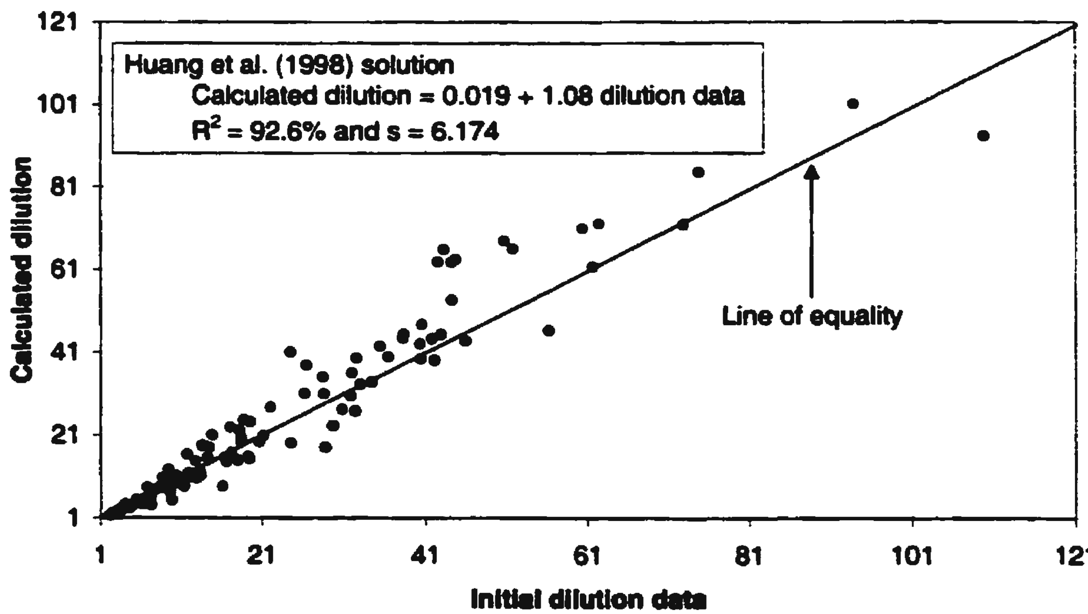


Figure 3.6. Comparison between calculated dilution (Huang et al. 1998) and the data

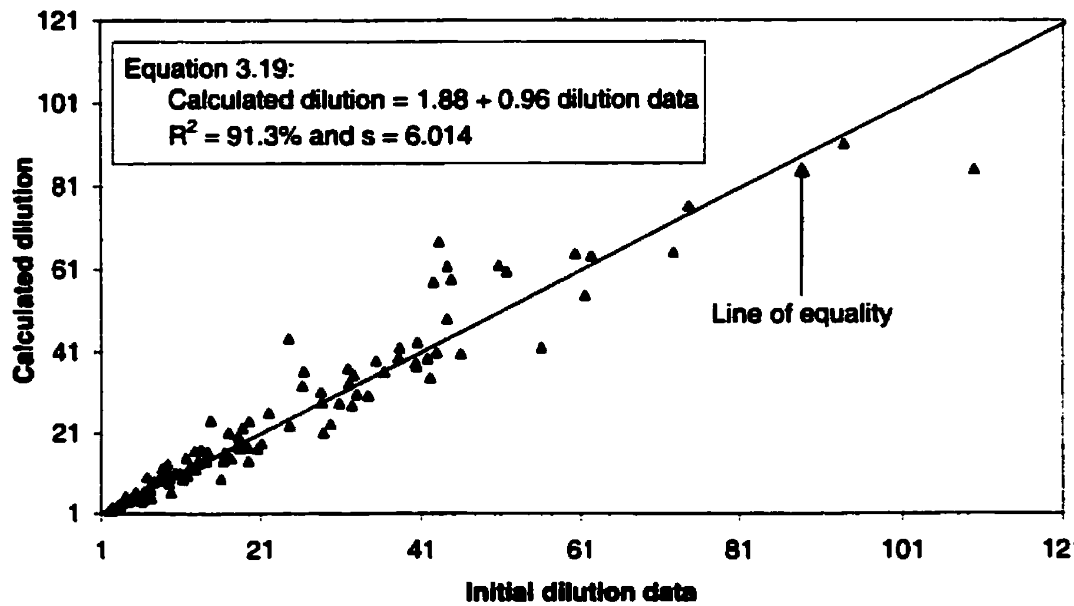


Figure 3.7. Comparison between calculated dilution (equation 3.19) and the data

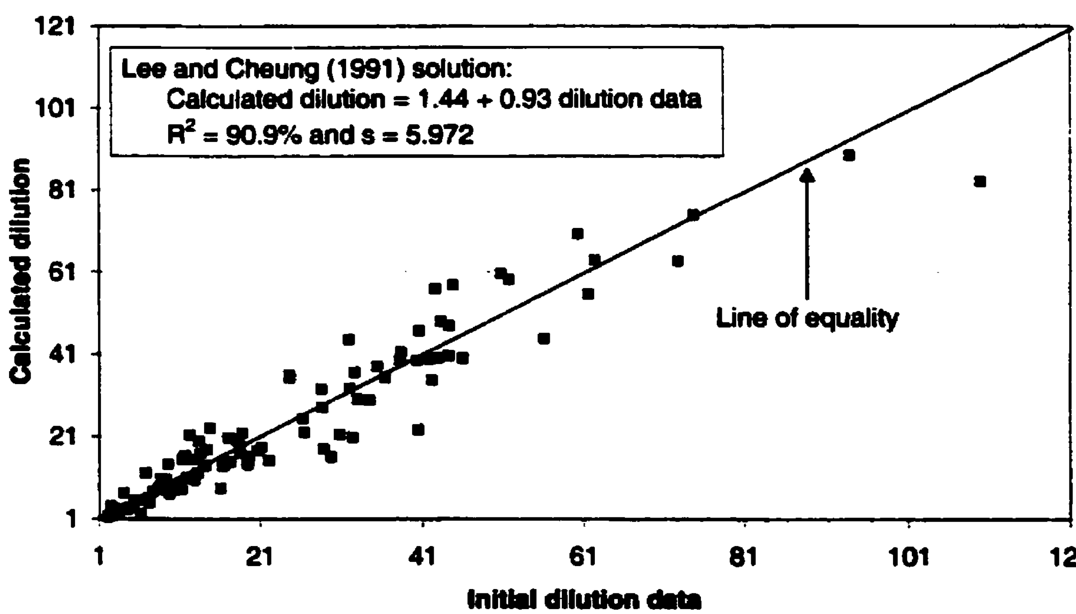
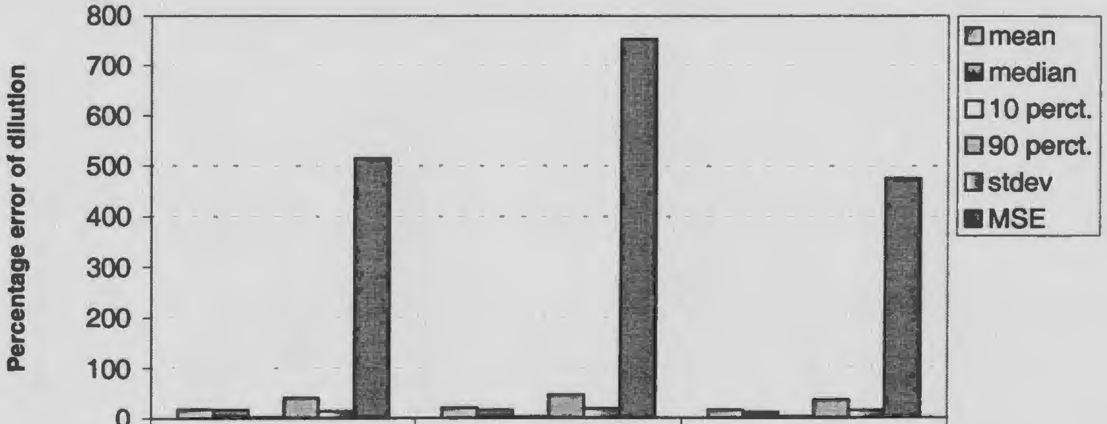


Figure 3.8. Comparison between calculated dilution (Lee and Cheung 1991) and the data



	Huang et al. (1998)	Lee and Cheung (1991)	Equation 3.19
mean	18.0	20.5	16.1
median	16.7	15.6	11.3
10 perct.	2.1	2.6	2.2
90 perct.	40.7	45.5	35.9
stdev	13.8	18.2	14.7
MSE	514.7	753.0	475.8

Figure 3.9. Comparison of percentage error of calculated dilution

It should be mentioned here that the proposed model was developed without taking the data outliers into account. Figure 3.10 shows the comparison of the models when the outlier points are included. As can be seen from this figure, there are five points lying far from the others. As discussed previously, there might be a recording, typing, or calculation error associated with point #5. It was therefore considered as an outlier and was removed in the subsequent analysis. In the Huang et al. (1998) paper, treatment of this point was not discussed.

Points # 1 to 4 seem to be closer to the other data because Figure 3.10 is presented in a log-log scale. Actually these points are quite far off and affect the least squares solution significantly. The presence of these points results in non-normal residuals of the model using either the approach used in this study or the Huang et al. (1998) model. Removal of these points improves the normality of the residuals. This is shown in Figures 3.11 to 3.14. Even after removing outliers, the probability plot of the Huang et al. (1998) model gives *a p-value* less than the significant level of 0.05 (Figures 3.11 and 3.13) so that the test does not show that residuals are normally distributed. On the other hand, Figure 3,14 shows that residuals of equation 3.19 are normally distributed with *a p-value* significantly higher than the 5% significant level.

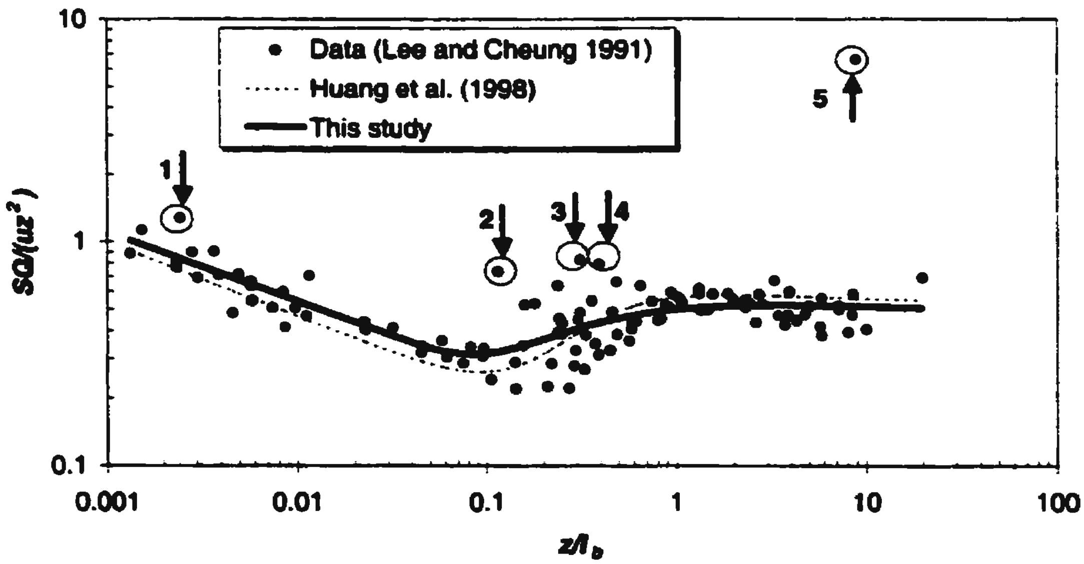


Figure 3.10. Influential points producing high residuals in the models

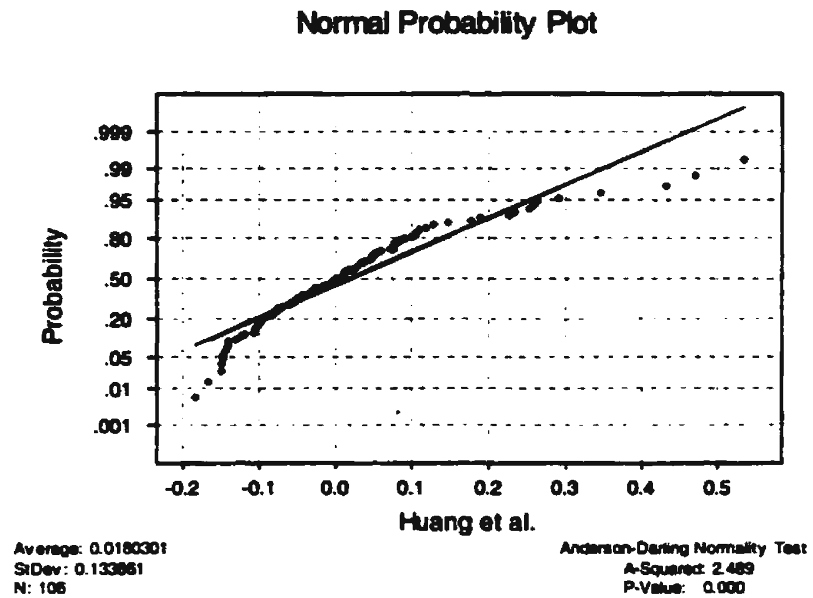


Figure 3.11. Probability plot of residuals for the Huang et al. (1998) model with outliers

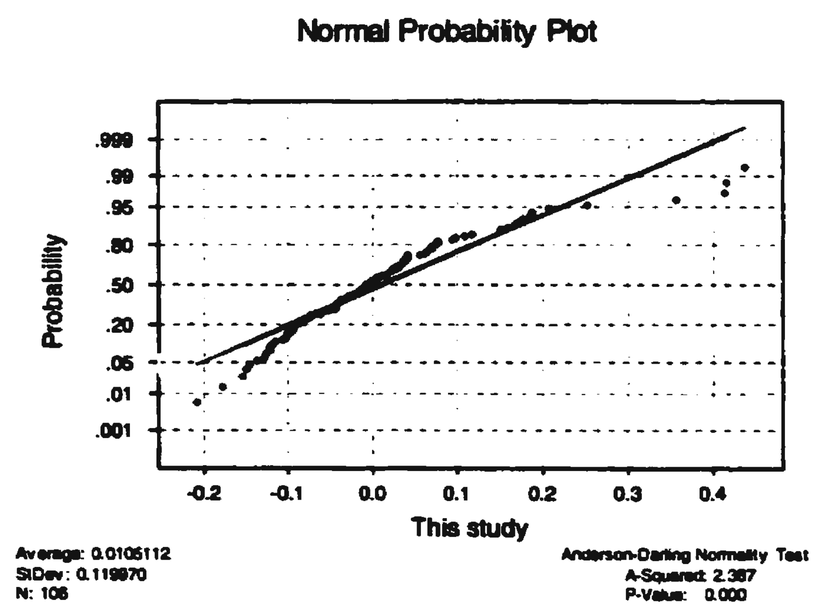


Figure 3.12. Probability plot of residuals for the proposed model with outliers

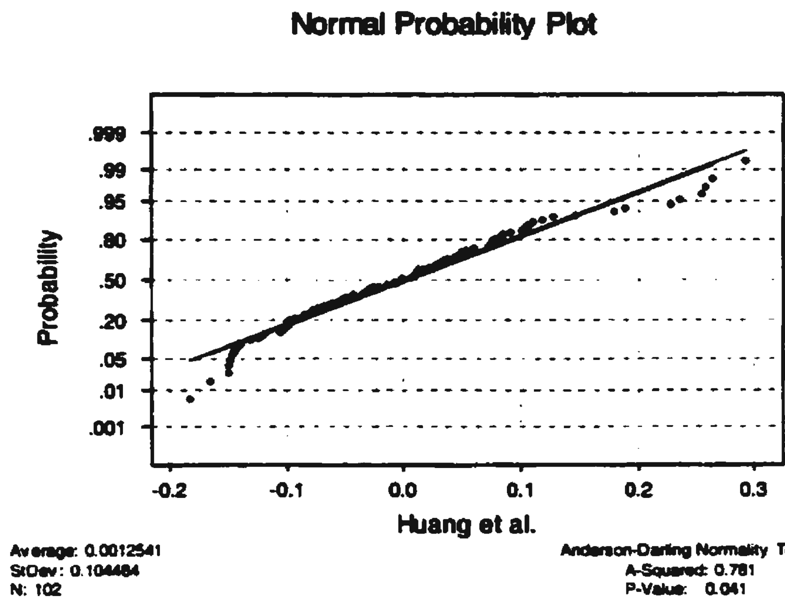


Figure 3.13. Probability plot of residuals for the Huang et al. (1998) model without outliers

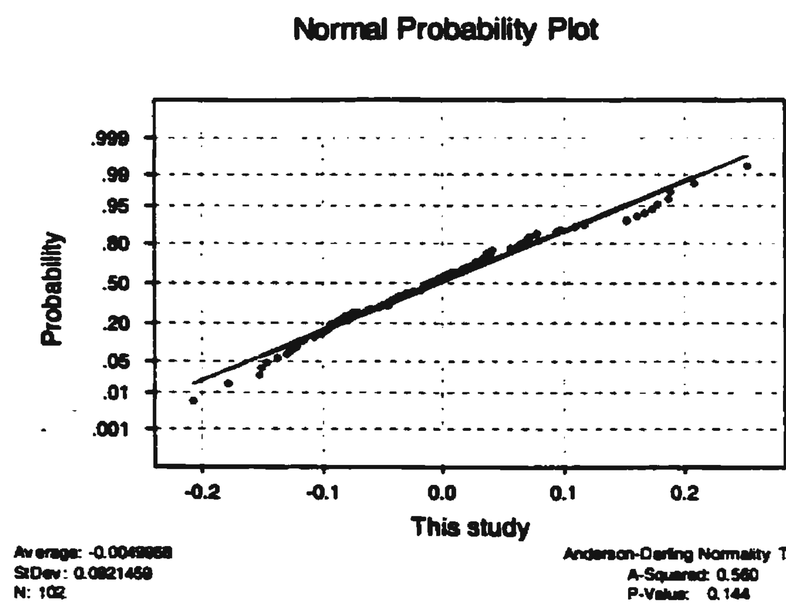


Figure 3.14. Probability plot of residuals for the proposed model without outliers

From the above discussions, it may be stated that a nonlinear regression modeling combined with a length scale analysis is a promising approach for modeling initial dilution. This study suggests that experimental work on dilution analysis may be usefully supplemented with a suitable data analysis. The approach discussed in this chapter may be useful for such a purpose.

3.7. Application Example

An example of an application is presented here using a case, which characterizes many operating outfalls, taken from Lee and Cheung (1991). The example considers a single port buoyant jet from an ocean outfall with a diameter (d) of 0.5 m, an initial jet velocity (u_j) of 0.6 m/s, a relative density difference ratio of 0.026, and a seawater depth above the discharge (z) of 10 m. With this situation, the outfall has a depth to diameter ratio (z/d) of 20. The volume, momentum, and specific buoyancy fluxes are computed to be 0.1178 m^3/s, 0.0707 m^4/s^2, and 0.03 m^4/s^3, respectively. With relevant length scales l_Q and l_M of 0.44 m and 0.79, respectively, relative to the depth, both the source geometry and the initial momentum are not significant, that is, $z >> l_Q$ and $z >> l_M$.

In order to evaluate the dilution in different regions, i.e. the BDNF, the transition, and the BDFF, three values of ambient current speed, u, are specified at 0.025, 0.1, and 0.3 m/s. This results in different buoyancy length scales, l_b of 1920, 30, 1.1 m, respectively, and the ratio of depth to buoyancy length scales, z/l_b, of 0.00521 (BDNF), 0.33 (transition), and 9 (BDFF), respectively. For the asymptotic solution (Lee and Cheung 1991), centerline dilutions were calculated for the BDNF and transition regions using equation 2.10 with coefficient values of 0.1 and 0.21, respectively. For the BDFF region, equation 2.11 with a coefficient value of 0.51 was used. Calculated centerline dilutions are shown in Table 3.3, in which equation 3.3 was used for Huang et al. (1998) centerline dilution.

This study employed equation 3.19 for calculating average dilution. The associated probability of failure, for a given case, was estimated using equation 3.20. Figure 3.15 illustrates a procedure for interpreting equation 3.20 using a simulation, from which a probability of failure associated with uncertainty in the model can be quantified. The probability of failure in this typical example is defined as the probability of not achieving a threshold design dilution and it is associated with uncertainty in the model itself, i.e. by assuming constant input variables. In this example the threshold dilution is arbitrarily assumed at $S_{design} = 20$.

As shown in Table 3.3, in the BDNF region, the proposed model provides higher dilutions than those given by the Huang et al. (1998) and Lee and Cheung (1991) models, by about 15%. This is expected because both the Huang et al. (1998) and Lee and Cheung (1991) models underestimate dilution at the BDNF as discussed previously. In the BDFF, the proposed model gives practically the same answer as the Lee and Cheung (1991) model, while the Huang et al. (1998) model overestimates dilution. In the transition, the Huang et al. (1998) model and the proposed model result practically in the same answer (the difference is only about 2%), while no unique solution is given by Lee and Cheung (1991). The solution of Lee and Cheung (1991) for the transition region can be obtained if it is assumed that it can be calculated using the BDNF equation with a modified coefficient (C_l = 0.21), resulting in different results of about 27%.

Table 3.3. Comparison of estimated centerline dilution among different models

Case #	Region	Lee and Cheung (1991)[1]	Huang et al. (1998)[2]	This study[3], $S(p_f)$
1	BDNF	12.2	12.2	14.1 (0.943)
2	Transition	25.8	34.9	35.6 (0.022)
3	BDFF	129.9	142.0	131.1 (0.000)

[1]Calculated using equations 2.10 and 2.11
[2]Calculated using equation 3.3
[3]Calculated using equation 3.19. The value in the brackets is the probability of failure (p_f), for which a calculation procedure is outlined in Figure 3.15.

Furthermore, the proposed model can be used to evaluate the reliability of the model by calculating the probability of not achieving a threshold design dilution for a given discharge scenario. If the threshold is set at S_{design} = 20, for example, then the probability of failure is 94.3%, 2.2%, 0% for cases where the ambient current speed, u, is at 0.025, 0.1, and 0.3 m/s, respectively. On the other hand, using the deterministic approach without considering uncertainties in the model (e.g. Lee and Cheung 1991 and Huang et al. 1998), the resulting dilution calculation would always correspond to a 50% failure probability.

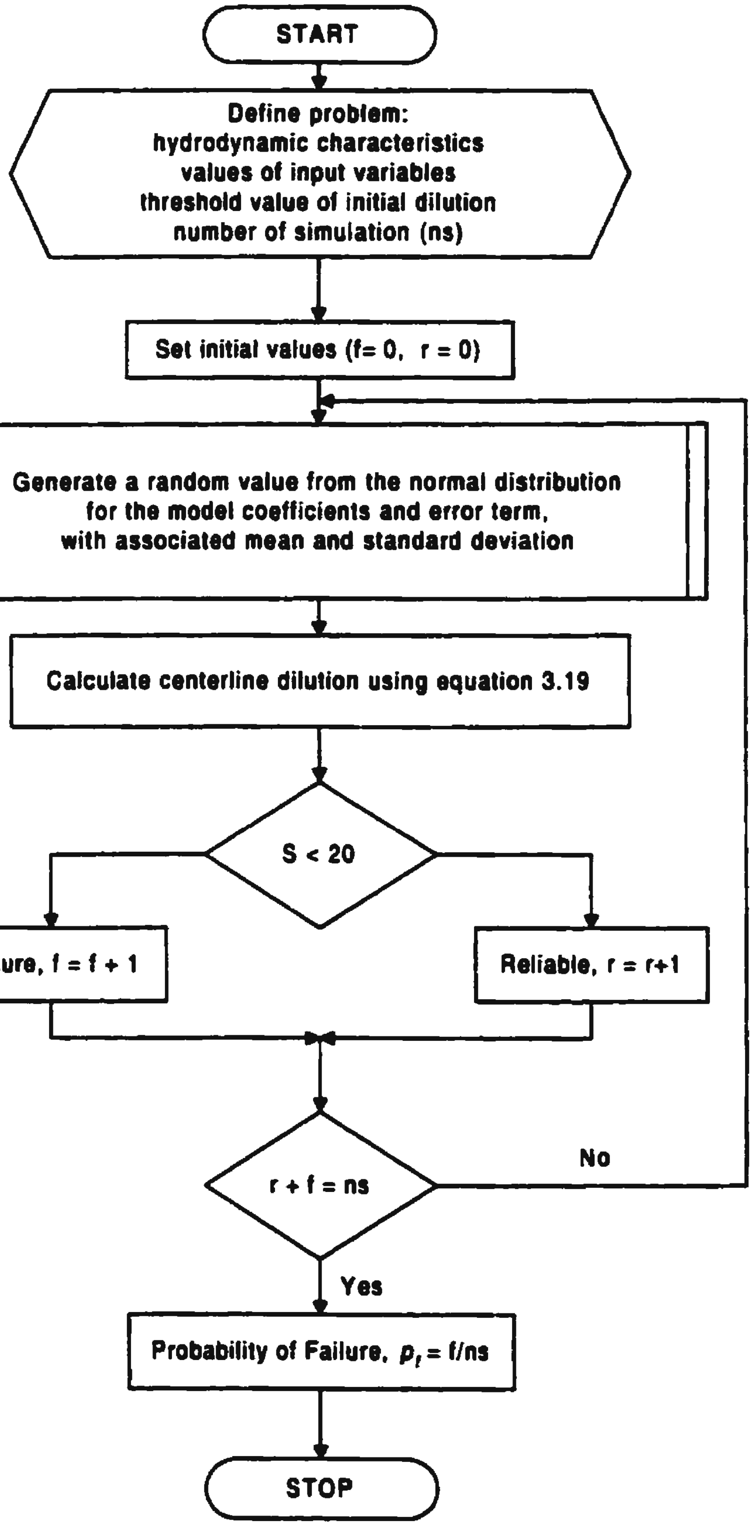


Figure 3.15. Flow chart of the procedure of evaluating probability of failure associated with model uncertainty.

3.8. Summary

A new approach to modeling initial dilution was proposed in this research. It was based on the hypothesis of additive shear and forced entrainment combined with nonlinear regression. The least squares approach and the Gauss-Newton numerical analysis were employed in this study. Unlike a previously available approach, which is typically "trial and error", the proposed approach provides an objective means of evaluating the model performance. Using the proposed approach, an alternative initial dilution model, which is more robust, is presented in this chapter. The model provides an unbiased solution for the whole range of the BDNF, the transition, and the BDFF. It gives a unique, continuous, solution of centerline dilution, which is presented in equation 3.19. A probabilistic form of the model is also given in equation 3.20. An example of an application for both deterministic and probabilistic evaluation of initial dilution is also provided.

Comparison of the proposed model with other available models shows that the proposed model is better in a number of ways: (1) it does not assume that the current has no effect in the BDNF, which asymptotic solutions do; (2) in the BDFF region the model has one parameter fewer than the Huang et al. (1998) model yet it is no less accurate; (3) in the transition region it gives a unique solution which the asymptotic models do not; (4) unlike the Huang et al. (1998) model, the proposed model has approximately the same precision for all regions, i.e. the BDNF, the BDFF, and the transition; and (5) the proposed model can also be presented in a probabilistic form that permits calculation of failure probability for specified model inputs and a threshold dilution.

Chapter 4

Hydrodynamic Modeling: Deterministic and Probabilistic Analyses

4.1. Introduction

Concepts of hydrodynamic modeling and the importance of initial dilution as an element of the modeling have been presented in Chapter Two. An alternative model for initial dilution in the near field has been proposed in Chapter Three. This chapter discusses the integration of near field and far field modeling, and an approach to evaluate the intermediate region between the near and far fields. Hydrodynamic modeling of typical mixing processes, which can occur as a result of discharging produced water into a marine environment, is presented in this chapter using deterministic and probabilistic analyses. Previous work related to hydrodynamic modeling of effluent discharges is reviewed in light of its application in the case of produced water discharge. Focus is directed at evaluating uncertainty associated with modeling and methods of dealing with this uncertainty. This chapter also describes an integrated hydrodynamic model using a deterministic approach. A methodology of probabilistic hydrodynamic modeling is then presented. An application example based on available information relevant to a typical offshore discharge of produced water is discussed in these sections.

4.2. Context

Hydrodynamic modeling using quantitative methods has been the most important tools for outfall design and environmental assessment purposes. Some of the quantitative methods have become quite elaborate and include sophisticated analysis; however, irrespective of the level of sophistication in the models, including experimental models, they are developed under some assumptions that simplify the problems. Consequently, they might not reflect the actual conditions of the problems under investigation and are associated with some degree of uncertainty. Furthermore, people have rarely provided *complete information*, such as values of parameters and uncertainties in data.

One cannot predict with certainty the occurrence (or nonoccurrence) of specific event. The underlying uncertainty may be due to: a) the inherent randomness of the natural phenomenon, b) the inaccuracies in estimating the parameters and in choosing the distribution, and c) the inaccuracies of modeling, which is based on idealized assumptions (Ang and Tang 1975). As a result, the use of a deterministic approach, which does not explicitly take these uncertainty factors into account in solving engineering problems, although sometimes useful, may in some cases lead to unrealistic results. This may result in a partial loss of information, misleading results, and incorrect solutions. Properly, the tools of engineering analysis should therefore include methods and concepts for evaluating the significance of uncertainty on system performance and design (Mukhtasor 1998).

In hydrodynamic modeling, uncertainty may be associated with a number of different factors, including uncertainty in model formulations and model inputs. Many model inputs used in hydrodynamic modeling, e.g. seawater current speeds and directions, are highly variable. In addition to this, as discussed in the previous chapter, model formulation may pose some degree of uncertainty, which may in part be reflected by uncertainty of its coefficients. However, hydrodynamic modeling of the discharge of produced water is unable to deal quantitatively with these uncertainties (e.g. Smith et al. 1996, Somerville et al. 1987). Previous efforts of modeling typically use simplified models (e.g. Rye et al. 1996) and assume a worst-case scenario by looking at dilution at 500 m from the platform (Karman and Reerink 1998). Other approaches of modeling are also deterministic in nature, such as the use of buoyant jet formulation (e.g. Skatun 1996) or far- field dispersion modeling (e.g. Murray-Smith et al. 1996).

Considering a worst-case scenario without explicitly taking uncertainty into account, a deterministic approach is compelled to rely on conservative assumptions. When these assumptions are compounded in estimating toxic chemical concentration, for example, the result becomes even more conservative, and can lead to misplaced control activities and environmental management priorities (Finley and Paustenbach 1994). In this context, presently widely used deterministic approaches need to be complemented by a probabilistic analysis. Results of a probabilistic analysis should help risk managers to make explicit decisions about how conservative their recommendations are (Finley and Paustenbach 1994).

4.3. Near Field Modeling

Hydrodynamic mixing of the effluent discharge in the near field, or initial dilution, can be increased by the proper design of the geometry of the outfall for specified characteristics of the effluent and ambient seawater. This includes selection of the shape, size and arrangement of effluent outlet, which may consist of a simple open end (single port) or a multiport diffuser containing a regularly spaced line of relatively small ports. This Section considers near field modeling of a single port discharge into a marine environment. The reason for choosing the single port type is discussed in Chapter Six (Subsection 6.4.2).

The near field mixing presented in the previous chapters is applicable for a deep-water condition, in which a distinct buoyant jet rises to the surface and dilution occurs because of turbulent jet entrainment (Jirka and Lee 1994). Upon impingement on the free surface, the effluent plume spreads laterally in the form of a stable density current (Hamdy 1981). A criterion for a deep-water condition is defined as:

$$\frac{H}{d} > 0.22 F_o \qquad (4.1)$$

where H is the depth of receiving water, and d is the diameter of port. F_o is the densimetric Froude number, which is defined as (William 1985):

$$F_o = \frac{V_j}{\sqrt{\left(g\,d\frac{(\rho_a - \rho_o)}{\rho_o}\right)}} \tag{4.2}$$

Equation 4.1 has little sensitivity to the discharge angle (Jirka and Lee 1994). When this criterion is not satisfied, the discharge falls in the category of shallow water condition, in which the discharge momentum may be sufficiently strong to cause a dynamic breakdown (instability) of the buoyant jet motion and create a local circulation zone as shown in Figure 4.1 (modified from Hamdy 1981). In order to maximize the dilution, unstable discharge is typically avoided because of recirculation of the plume and limited entrainment into the jet (Hamdy 1981).

For stable discharges, Chapter Two defines the concept of initial dilution and discusses problems associated with presently available initial dilution models. A unique, continuous, initial dilution model, which was derived from experimental data covering conditions of many operating ocean outfalls, including produced water outfalls, is presented in Chapter Three. The model is suitable for a round buoyant jet, i.e. discharge from an open-ended outfall into unstratified moving waters. The deterministic and probabilistic formulations of the initial dilution model are rewritten in equations 4.3a and 4.3b, respectively.

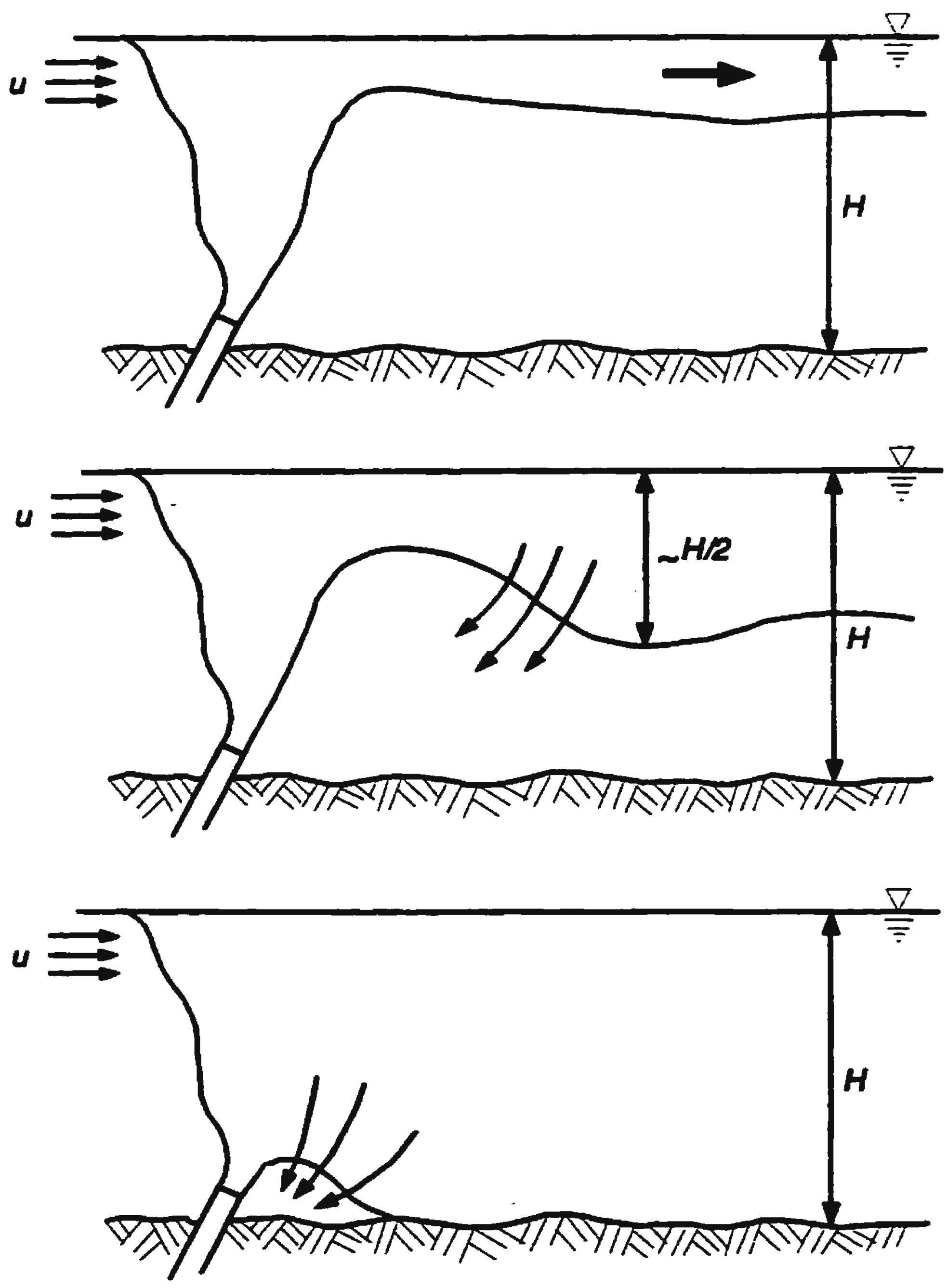


Figure 4.1. Various discharge characteristics (Top: stable, deep water discharge, middle: transition discharge, and bottom: unstable, shallow water discharge)

$$\frac{SQ}{u z^2} = 0.13\left(\frac{z}{l_b}\right)^{-0.31} + 0.46 \exp\left(\frac{-0.22}{z/l_b}\right) \quad (4.3a)$$

$$\frac{SQ}{u z^2} = (0.13 \pm 0.02)\left(\frac{z}{l_b}\right)^{(-0.31 \pm 0.03)} + (0.46 \pm 0.02) \exp\left(\frac{(-0.22 \pm 0.04)}{z/l_b}\right) + N(0,\ 0.092) \quad (4.3b)$$

where S is the initial (centerline) dilution (dimensionless) at an elevation z above discharge; Q is the outfall discharge rate, u is the ambient current speed (m/s); $N(0, 0.092)$ represents residuals of the model that are normally distributed with a mean of zero and standard deviation of 0.092; and l_b is the buoyancy length scale defined as:

$$l_b = \frac{Q\,g}{u^3}\left(\frac{\rho_a - \rho_o}{\rho_a}\right) \quad (4.4)$$

in which g is the gravitational acceleration; ρ_a and ρ_o are the densities of the ambient seawater and effluent, respectively.

Lee and Neville-Jones (1987a) provide a useful modeling approach to estimate the trajectory of the buoyant jet. For a typical horizontal discharge, the motion in the horizontal (x-y) plane is driven by the jet momentum, M, and the ambient current, u. Since the discharge imparts no vertical momentum, the motion in the vertical (x-z) plane is determined primarily by the interaction of the discharge buoyancy and the ambient flow. The trajectory equations have been reported in the literature (e.g. Lee and Neville-Jones 1987a, Wright 1977b) and models for the horizontal boil location x_b at the seawater surface can be written as:

$$x_b = C_3 \frac{H^{4/3}}{l_b^{1/3}} \qquad for\ H << l_b \tag{4.5}$$

$$x_b = C_4 \frac{H^{3/2}}{l_b^{1/2}} \qquad for\ H >> l_b \tag{4.6}$$

where H is the water depth above discharge, and C_3 and C_4 are coefficients determined based on field and laboratory data. Huang et al. (1996) proposed an interpolation method to deal with the nonlinearity in the transition between the two cases (equations 4.5 and 4.6). In this interpolation method, equations 4.5 and 4.6 are used within the region of H/l_b <0.1 and >10, respectively. Between these two ranges, the value of a variable is a function of those at other regions, defined as:

$$I_{tr} = a_1 I_{0.1} + a_2 I_{10} \tag{4.7}$$

where $I_{0.1}$, I_{10} and I_{tr} are regional variables for H/l_b <0.1, H/l_b >10, and $0.1 \leq H/l_b \leq 10$, respectively. The coefficients a_1 and a_2 are estimated from (Huang et al. 1996):

$$a_1 = 0.5 - 0.5\log_{10}\left(\frac{H}{l_b}\right) \tag{4.8}$$

$$a_2 = 0.5 + 0.5\log_{10}\left(\frac{H}{l_b}\right) \tag{4.9}$$

Wright (1977b) reported values of C_3 (equation 4.5) that depend on the method of obtaining the data, i.e. 0.6702 from photographic measurements and 0.4571 from concentration measurements. Huang et al. (1996) used a value of 0.5824 for C_3 based on the CORMIX model (Doneker and Jirka 1990). From Wright (1977a), a variation of C_3 from typical photographic measurements is in the range of 0.517 to 1.494.

While C_3 is usually treated as a constant, it seems that there is no common agreement whether C_4 (equation 4.6) is a constant or a variable that depends on other physical quantities. Lee and Neville-Jones (1987a) proposed a value of C_4 of 1.1 for estimating time-averaged boil location x_b based on field experiments at six outfalls. However, Wright (1977b) noted that C_4 varied depending upon the ratio of the buoyancy and momentum length scales i.e.

$$C_4 = C_5\left(\frac{l_b}{l_m}\right)^{1/4} \tag{4.10}$$

where l_m is defined as:

$$l_m = \frac{\left(u_j Q\right)^{1/2}}{u} \tag{4.11}$$

Values of C_5 also depend on the method of obtaining the data, i.e. 0.6037 and 1.2761 based on data from photographic and concentration measurements, respectively.

A constant value for C_4 might be acceptable for a site-specific outfall condition with a small variation in the ratio of the buoyancy and momentum scales (equation 4.10). Furthermore, it is not uncommon practice to treat C_4 as a constant. In produced water modeling, for example, this practice has been adopted by employing the CORMIX model (Smith et al. 1996), in which the value of C_4 is taken to be 1.0 (Huang et al. 1996, Doneker and Jirka 1990). Whether or not C_4 is actually a variable is one problem. Even if it is assumed to be a constant, there is uncertainty in defining its value, which is another problem. From the Wright (1977a) experiments, values of C_4 are typically in the range of 0.2254 to 1.7075. Methods of dealing with this uncertainty are discussed in Section 4.6.

4.4. Intermediate Modeling

Completing the plume rise in the near field region, surface impingement takes place. The jet is deflected and begins to spread horizontally. The process may result in the phenomena of the boil and the hydraulic jump, if a jump occurs. A control volume, which is a region where the surface impingement takes place, can be defined as the intermediate region, which connects the near and far fields. Figure 4.2 provides a schematic definition of the intermediate region (modified from Doneker and Jirka 1990).

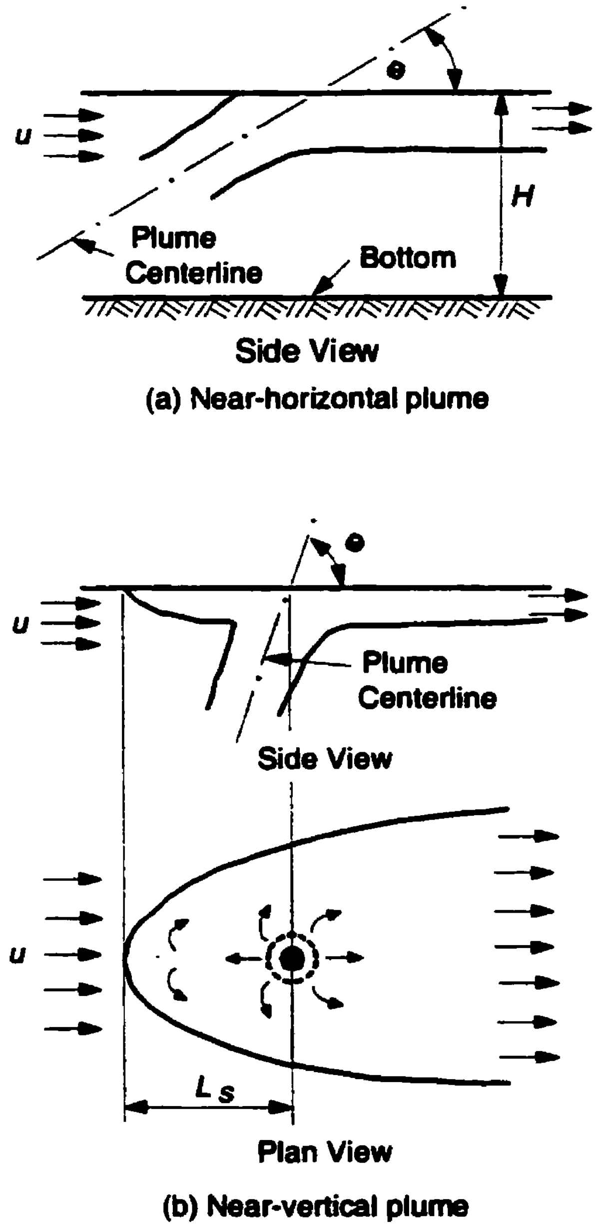


Figure 4.2. Typical intermediate region of discharges

For typical produced water modeling, the intermediate region has been neglected. Somerville et al. (1987), for example, adopted the Brooks (1960) model as a far field dispersion model. The inputs of the model are taken directly from the output of the initial dilution model. Smith et al. (1996) used a similar approach to that of Somerville et al. (1987): modification of the initial dilution outputs, before running into the far field dispersion model, is not carried out in these cases.

Formulation for the analysis of the intermediate region is available in the literature (Doneker and Jirka 1990, Wright et al. 1991 and Huang et al. 1996). These are based on a control volume approach, in which the inflow is the rising buoyant jet flow near the water surface and the outflow is the surface plume that is advected by the ambient current. The outflow characteristics required for connecting the near and far fields include the bulk dilution S_a (or alternatively the pollutant concentration of interest C_a, where $C_a = C_o/S_a$, and C_o is the pollutant concentration in the effluent prior to discharge), the plume width L_o, the plume thickness h_o, and the distance from the boil center to the upstream (L_s) and downstream end of the control volume (x_D). The formulations proposed by Doneker and Jirka (1990) and Huang et al. (1996) have been partly calibrated for typical field tests of an outfall plume. The deterministic components of these formulations are considered in this study for further modification into a probabilistic analysis of hydrodynamic modeling.

4.4.1. Bulk dilution

The bulk dilution at the downstream end of the control volume, S_a, is estimated (Wright et al. 1991, Doneker and Jirka 1990, Huang et al. 1996) as:

$$S_a = C_{S1}\, S \qquad \text{for } H/l_b < 0.1 \qquad (4.12)$$

$$S_a = C_{S2}\, S \qquad \text{for } H/l_b > 10 \qquad (4.13)$$

When $H/l_b < 0.1$, the rising buoyant jet is only weakly deflected by the ambient current and approaches the water surface at a near-vertical angle. In this case, an internal hydraulic jump is expected to occur. As relevant experimental data is unavailable, the coefficient C_{S1} is typically estimated based on experiments in stagnant water (Wright et al. 1991, Huang et al. 1996). This assumption, i.e. the applicability of a stagnant water experiment to a case with a weak ambient current, poses some degree of uncertainty, which is unquantifiable. Only the uncertainty associated with values of the coefficient C_{S1} may be quantified by specifying its values between 3 and 5 based on experiments from Wright et al. (1991).

When $H/l_b > 10$, the rising buoyant jet is strongly deflected by the ambient current and approaches the water surface at a near-horizontal angle. The flow is advected with the ambient velocity at the speed u of the surface plume layer. For this case, the constant C_{S2} in equation 4.13 is reported to be in the range of 1.5 to 2.0 (Doneker and Jirka 1990, Huang et al. 1996). Typical field test-based calibrated values of the coefficients are 2.01 and 1.74 for C_{S1} and C_{S2}, respectively (Huang et al. 1996).

4.4.2. Plume width and upstream intrusion length

The plume width at the downstream end of the control volume, L_o, is estimated (Doneker and Jirka 1990, Huang et al. 1996) as:

$$L_o = 5.2\, L_s \qquad \textit{for } H/l_b < 0.1 \qquad (4.14)$$

where L_s is the upstream intrusion length, which is the distance from the boil center to the upstream end of the control volume. For this case (i.e. $H/l_b < 0.1$), the parameter L_s is defined (Akar and Jirka 1994b, Doneker and Jirka 1990) as:

$$L_s = 2.12\, H^{3/2}\,(1-\cos\Theta)^{3/2}\, l_b^{-1/3} \qquad \text{for } l_b/H > 6.11(1-\cos\Theta) \qquad (4.15a)$$

$$L_s = 0.38\, l_b \qquad \text{for } l_b/H \le 6.11(1-\cos\Theta) \qquad (4.15b)$$

where Θ is the angle between the rising buoyant jet axis and the water surface, estimated from $\Theta = \tan^{-1}(H/x_b)$ (Huang et al. 1996).

When $H/l_b > 10$, the plume width at the downstream end of the control volume, L_o, and the distance from the boil center to the upstream end of the control volume, L_s, are estimated from equations 4.16 and 4.17, in which an equivalent cross-section aspect ratio for the outflow section of 2:1 is assumed (Doneker and Jirka 1990, Huang et al. 1996).

$$L_o = 2\sqrt{\frac{S_a Q}{2u}} \qquad \textit{for } H/l_b > 10 \qquad (4.16)$$

$$L_s = \frac{1}{\sin\theta}\sqrt{\frac{S_a Q}{\pi\, u}} \qquad \textit{for } H/l_b > 10 \qquad (4.17)$$

4.4.3. Distance of downstream end and plume thickness

The distance from the boil center to the downstream end of the control volume, x_D, is estimated by assuming that it is proportional to the depth above discharge, H, defined as:

$$x_D = C_{D1}\, H \qquad \textit{for } H/l_b < 0.1 \qquad (4.18)$$

$$x_D = C_{D2}\, H \qquad \textit{for } H/l_b > 10 \qquad (4.19)$$

where C_{D1} and C_{D2} are model coefficients. The value of C_{D1} is typically set at 3 (Huang et al. 1996, Wright et al. 1991) and C_{D2} at 0.6 (Huang et al. 1996, Doneker and Jirka 1990). In any case, the plume thickness h_o can be estimated from the continuity equation as:

$$h_o = \frac{S_a Q}{u L_o} \qquad (4.20)$$

4.4.4. Transitional regime

Characteristics of interest discussed above are defined as the regional solutions, i.e. in the regimes of $H/l_b < 0.1$ and $H/l_b > 10$. To have a smooth transition between these regimes, the same treatment interpolation method defined in equations 4.7 to 4.9 are applied as suggested by Huang et al. (1996). That is, a solution for a characteristic in the transitional regime ($0.1 \leq H/l_b \leq 10$) is taken to be a linear combination of the solutions of the other two regimes for that characteristic with a formulation presented in equation 4.7. Values for the characteristics S_a, L_o and h_o discussed in this section are then taken as the initial condition for the far field modeling.

4.5. Far Field Modeling

Hydrodynamic mixing of an effluent plume and ambient seawater in the far field is largely governed by two mixing mechanisms: buoyant spreading and turbulent diffusion. Buoyant spreading refers to a self-driven plume dispersion process, in which the buoyancy residual contained in the plume promotes the vertical collapse and horizontal transverse spreading of the plume. In addition to this self-driven process, oceanic turbulence disperses the effluent plume. This later process is referred to as turbulent diffusion. Both buoyant spreading and turbulent diffusion may be present in the dispersion processes of the effluent plume in the ocean; however, the relative importance of each mechanism depends upon the characteristics of the discharge and ambient waters (Akar and Jirka 1994a, Huang and Fergen 1997).

Typical field tests of outfall plumes indicate that the effluents were dominated by buoyant spreading over a range of several hundred meters from the outfall (Hazen and Sawyer 1994). Such spreading processes result from the buoyancy forces caused by the density difference of the mixed flow relative to the ambient density. If the discharge is not buoyant, or is weakly buoyant and the ambient is unstratified, there is no buoyant spreading (Doneker and Jirka 1990). This is, however, not the case for most produced water discharges as discussed in Chapter Two.

One approach to produced water dispersion typically neglects the buoyant spreading without evaluating whether or not the residual buoyancy is significantly absent (e.g.

Somerville et al.1987, Washburn et al. 1999). A deterministic far field analysis of dispersion processes considering the residual density difference is provided in the new version of the Offshore Operators Committee (OOC) model (Brandsma et al. 1992). The OOC model was developed analytically by employing mathematical models of conservation of mass, momentum, and energy.

A simpler formulation than the OOC model has been proposed by Doneker and Jirka (1990) and was implemented in the CORMIX model. The CORMIX model is widely used for offshore discharge analysis (e.g. Huang et al. 1996, U.S. EPA 1997) and was calibrated using many sets of laboratory and field data (Doneker and Jirka 1990, Huang et al. 1996). Figure 4.3 shows a sketch definition of a typical buoyant spreading process (modified from Doneker and Jirka 1990 and Huang et al. 1996). Typical model formulations for the spreading can be rewritten as:

$$h(x) = h_o \left(\frac{L(x)}{L_o} \right)^{\alpha - 1} \tag{4.21}$$

$$L(x) = L_o \left(3\beta \left[\frac{l_b}{L_o} \right]^{1/2} \frac{x}{L_o} + 1 \right)^{2/3} \tag{4.22}$$

where α is the entrainment coefficient ranging from 0.15 to 0.6, with a typical field test calibrated value of 0.59 (Huang et al. 1996, Doneker and Jirka 1990), β is the model constant ranging from 0.707 to 1.414, with a typical field test calibrated value of 1.33

(Huang et al. 1996, Doneker and Jirka 1990), l_b is the buoyancy length scale, typically evaluated for current speed at the 5 m depth (Hazen and Sawyer 1994), x is the distance along the plume centerline and $x = 0$ is set at the center of the downstream end of the control volume, and $L(x)$ is the plume width. The parameter $L(x)$ is assumed to be related to the standard deviation $\sigma(x)$ of the concentration distribution across the plume width by $L(x) = 2(3)^{1/2}\sigma(x)$, being consistent with Brooks (1960).

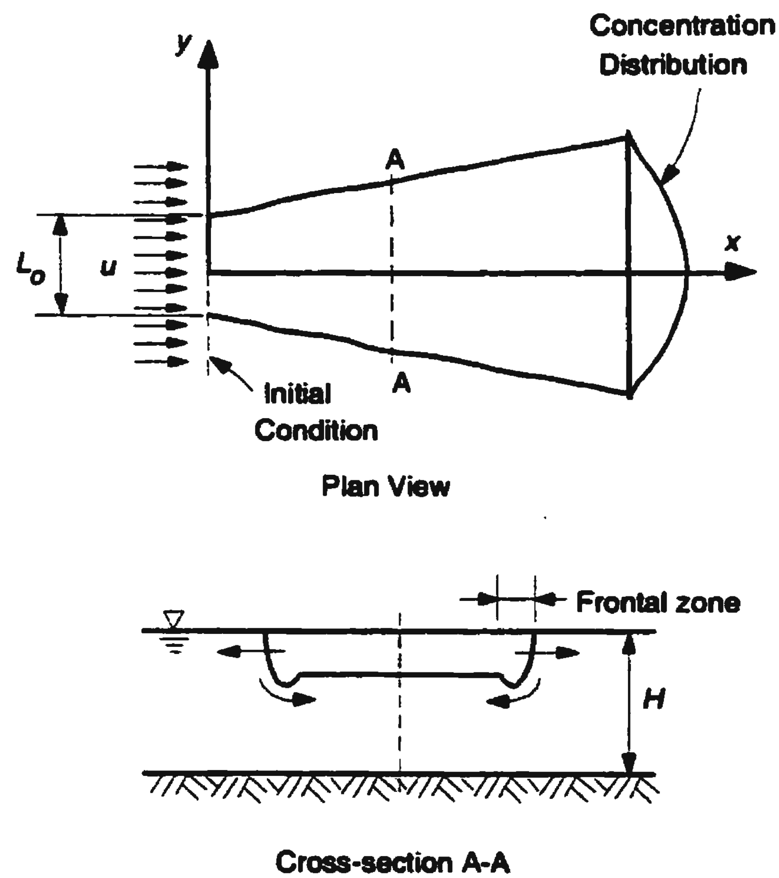


Figure 4.3. A typical sketch definition of buoyant spreading

Dilutions, or pollutant concentrations, associated with buoyant spreading processes are typically estimated by assuming that the concentration of a tracer in the surface plume has an error function distribution across the plume width and a uniform distribution across the plume thickness. Based on these assumptions and a mass balance, the pollutant concentration at a point (x, y) is estimated (Huang et al. 1996) as:

$$C(x,y) = 1.832\, C_a \frac{h_o}{h(x)} \frac{1}{2}\left[erf\left(\frac{0.273 L_o + y}{\sqrt{2}\,\sigma(x)}\right) + erf\left(\frac{0.273 L_o - y}{\sqrt{2}\,\sigma(x)}\right)\right] \quad \text{for } x \geq 0 \qquad (4.23)$$

where y is the horizontal coordinate perpendicular to the other horizontal coordinate x (which is along the plume centerline), C_a is the bulk pollutant concentration at the downstream end of the control volume ($x = 0$) estimated from the associated bulk dilution (equations 4.12 and 4.13), $erf(\ )$ is the error function defined as:

$$erf(w) = \frac{2}{\sqrt{\pi}} \int_0^w e^{-v^2}\, dv \qquad (4.24)$$

The error function can be solved approximately using Simpson's rule (Markham 1993). The function can also be evaluated from a statistical table of the area under the Normal Distribution curve by a change in variable such that (Williams 1985):

$$erf(w) = 2\, A(z) \qquad (4.25)$$

where z = 1.414 w, and $A(z)$ is the area of the Standard Normal Distribution from 0 to z along the abscissa.

As indicated, equation (4.23) may only be used for $x \geq 0$, otherwise it must be modified. It is typically assumed that the concentration is zero when $x < (-L_s - x_D)$. When $(-x_D + L_s) < x < 0$ the concentration is 1.2 C_a to be consistent with Huang et al. (1996). The average boil concentration $[C_o/(1.7\ S)]$ is defined when $(-L_s - x_D) \leq x \leq (-x_D + L_s)$, where C_o is the concentration prior to discharge and S is the centerline initial dilution (Hazen and Sawyer 1994, William 19985). A parabolic shape defined by Akar and Jirka (1995b) is adopted in this study to formulate a relationship of the width and the distance of the plume within $(-L_s - x_D) \leq x < 0$ such as:

$$L(x) = L_o \left(\frac{x + x_D + L_s}{x_D + L_s} \right)^{1/2} \qquad for\ (-L_s - x_D) \leq x < 0 \qquad (4.26)$$

This type of deterministic hydrodynamic model that assumes buoyant spreading has been calibrated based on data from laboratory and field tests, e.g. field tests on outfall discharges in the South Florida marine environment (Huang et al. 1996). The model is intended to estimate hydrodynamic characteristics of the plume in the vicinity of the discharges, in which effects of turbulent diffusion are less dominant than those of buoyant spreading. This approach may be valid only for a distance relatively close to the discharge. As the spreading plume travels downstream, the buoyancy effects gradually diminish, and at a particular distance the ambient turbulence of the receiving water is more dominant in the mixing process (Akar and Jirka 1995a). A criterion to characterize this transition is typically set by using the flux Richardson number R_f which may be approximated from (Akar and Jirka 1994a, Doneker and Jirka 1990):

$$R_f = \kappa^2 \frac{g' h}{u_*^2} \tag{4.27}$$

where κ is von Karman constant, with a value of 0.4, h is the plume depth, u_* is the shear velocity, and g' is the reduced gravitational acceleration defined as:

$$g' = g \frac{(\rho_a - \rho_o)}{\rho_o} \tag{4.28}$$

R_f can be used as a criterion in employing far field hydrodynamic models, i.e. buoyant spreading or turbulent diffusion models. When R_f falls below some critical value R_{fc} the buoyancy effects become relatively unimportant. Critical values of R_f between 0.1 and 0.2 have been reported from experimental tests, and an average value of 0.15 is typically adopted (Akar and Jirka 1994a).

It should be noted here that although the models formulated in this chapter may be applicable for only a limited case, i.e. relatively close to the discharge location, they are useful for evaluating discharge scenarios of produced water from offshore platforms. Despite the fact that there are field variations in produced water effects, studies show that ecological effects of produced water can generally be associated with the distance from the outfall and that the effects are usually limited close to the discharge location (within 500 to 1000 m radii). This may be because of rapid dilution in the marine environment (Frost et al. 1998, Somerville et al. 1987, Stromgren et al. 1995). Besides that, the design of the discharge scenario is usually directed at evaluating regulatory ambient water quality criteria which are in turn typically specified using a mixing zone concept, e.g. about 100 m or 200 m from the discharge location. It is obvious that if the analysis is extended in the range larger than that considered in this study, both of the phenomena (i.e. buoyant spreading and turbulent diffusion) may have to be considered for the problem of interest.

4.6. Integrated Model

The models discussed in the previous sections are deterministic steady state models based on physical principles. Local concentrations of produced water near an ocean outfall following a discharge may vary continuously both in space and time, mainly due to variability of ocean currents that advect the effluent plume. To simulate the variation, a coordinate system defined by Huang et al. (1996) is shown in Figure 4.4. This coordinate definition can be used to locate and simulate plume movement around the outfall discharge. In that figure, a fixed global coordinate system X,Y is defined; X is to the right (horizontal direction), Y is to the top (vertical direction), and the origin is set at the outfall location. A translating local coordinate system x,y for the surface plume is defined so that it varies depending on the variation of governing parameters, such as ambient current speeds and directions. A transformation between the translating coordinate system and the fixed coordinate system can be defined as:

$$x = X\cos\phi + Y\sin\phi - x_b - x_D \tag{4.29a}$$

$$y = Y\cos\phi - X\sin\phi \tag{4.29b}$$

where ϕ is the current direction (radian) with respect to the X-coordinate direction.

If simulated concentrations at points of concern can be assumed to be a representative sample of produced water concentrations, a simulated concentration field at an instance may be regarded as one possible "snapshot" of an outfall plume. The concentration field can be defined by dividing an area around the outfall into grids (Figure 4.5). Concentration at every grid point is calculated by employing near, intermediate, and far field models discussed above. By doing this, the model may be regarded as a quasi steady state model.

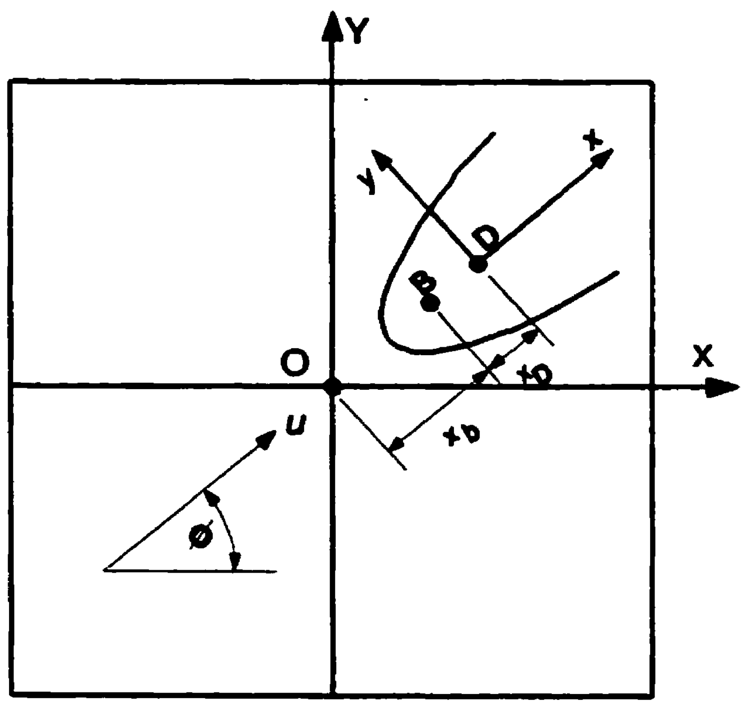


Figure 4.4 Coordinate definition for locating plume movement

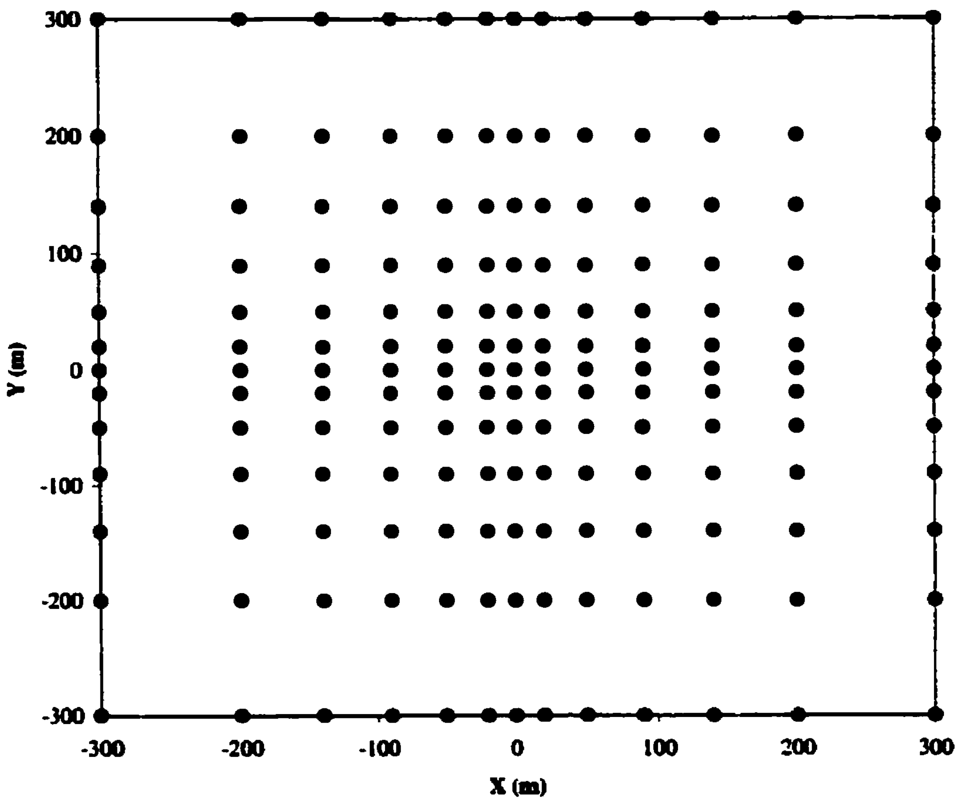


Figure 4.5. Typical grid points showing nodes for the simulation

An application example of an analysis is presented here by considering a hypothetical study associated with a potential discharge of produced water from an offshore platform on the Grand Banks, southeast of St. John's, Newfoundland, Canada (Petro-Canada 1996, Mukhtasor 2000, Mukhtasor et al. 2000c). The focus in this analysis is to show a potential application of the methodology outlined above using a deterministic approach. A comparison is given using a presently available model, i.e. the CORMIX model (Jirka et al. 1996), which is recommended for use for a typical dilution analysis of produced water (U.S. EPA 1997).

For this purpose, consider a potential discharge of produced water with a flow rate of approximately 0.212 m^3/s, and a relative density difference of about 0.025. A discharge design is specified with a single port located about 11 m below the sea surface. With these parameters, hydrodynamic characteristics of the discharge would likely be in the range of those calculated based on data from other produced water discharges worldwide (Brandsma and Smith 1996, Smith et al. 1996, and Somerville et al. 1987). Using this information, simulations were carried out using the methodology discussed above and the concentration distribution up to 300 m downstream is shown in Figure 4.6.

This study estimated that the far field dispersion region begins at about 22.9 m downstream from the boil location, with the centerline effluent concentration at the edge of the mixing zone, 100 m downstream, of approximately 2.3% and average effluent concentration of about 1.4%. The CORMIX model (DOS version 3.20, Jirka et al. 1996) was also used in this study; and its typical plume evaluation result is graphically shown in Figure 4.7. The CORMIX model estimated the edge of the near field region to be at 23.5 m downstream, with an average effluent concentration at the edge of mixing zone, 100 m downstream, of approximately 1.5%. The comparison shows that the two approaches are generally in good agreement, particularly in estimating average effluent concentrations.

It should be noted here that the scale of the horizontal axis in Figures 4.6 and 4.7. are different because of restriction in the CORMIX model, in which the horizontal distance must be specified at least 100 times the average depth. If the depth of the water at the Grand Banks site is about 80 m, for example, the minimum horizontal distance that has to be specified in the CORMIX model is at least 8 km, which is typically beyond the distance of interest in the case of produced water discharge. To manipulate this restriction, the water depth of 15 m, i.e. 4 m below the discharge port, and the horizontal distance of 2 km were specified for the CORMIX simulations.

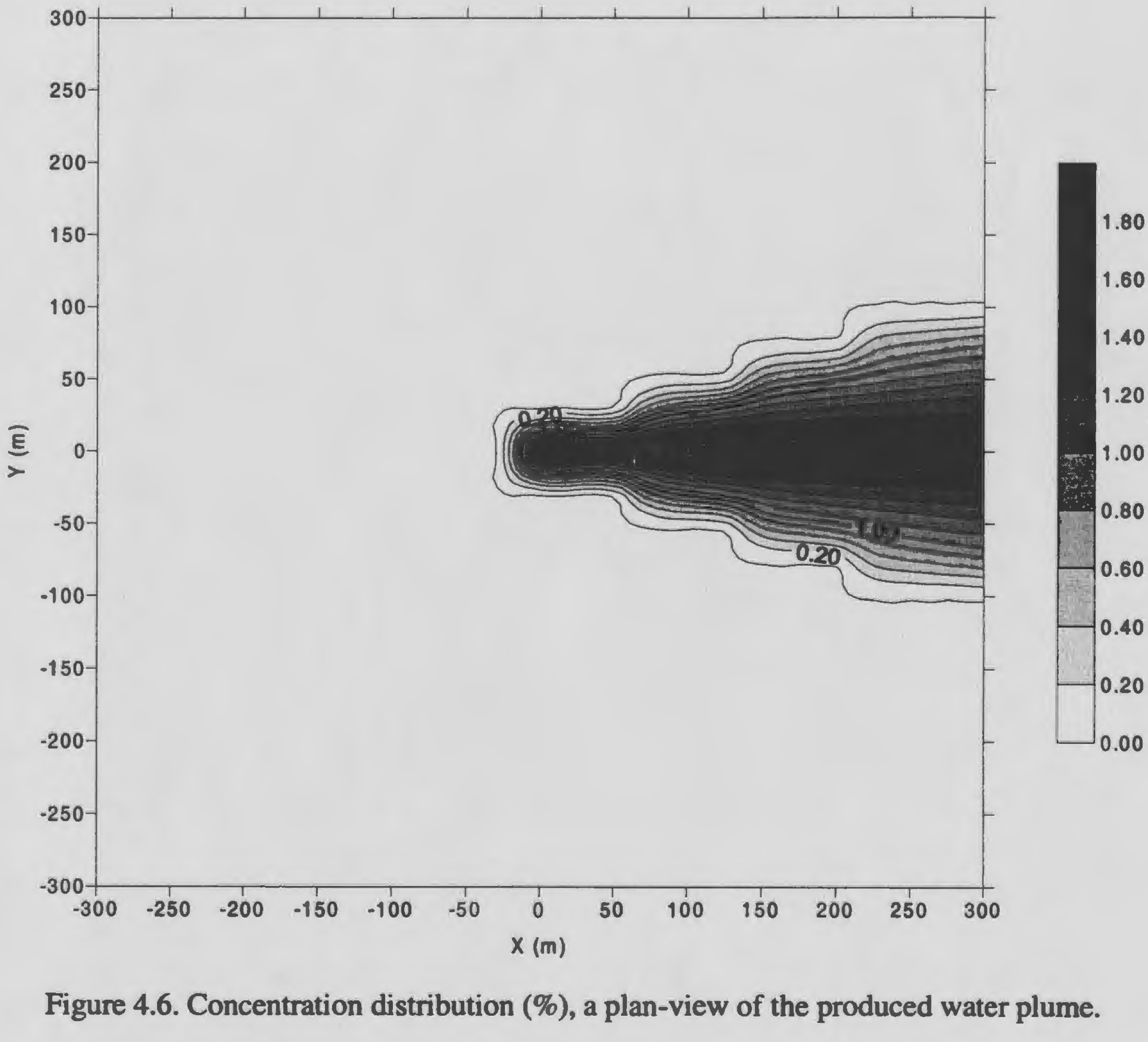


Figure 4.6. Concentration distribution (%), a plan-view of the produced water plume.

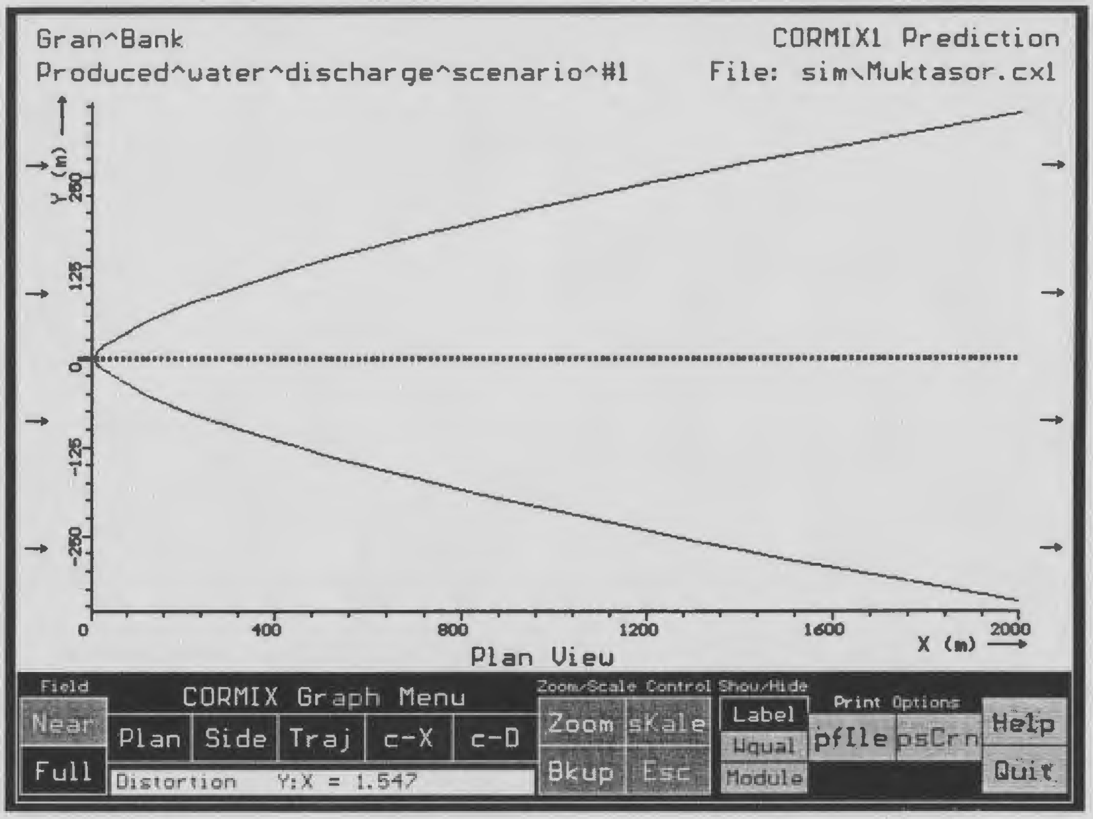


Figure 4.7. Typical CORMIX output, a plan-view of the produced water plume

In contrast to the methodology presented here, the CORMIX model provides only averaged concentrations at different distances downstream, even though the width of the plume can typically be as much as about 65% of the downstream distance as shown in Figures 4.6 and 4.7. Furthermore, since a regulatory mixing zone may also be defined in terms of horizontal area (Doneker and Jirka 1990, Huang et al. 1996), the distribution of effluent concentration in both *X*- and *Y*-directions are important to analyze so that the area of impact zones can be estimated. In this situation, the approach employed in this study, which provides concentration distribution in the horizontal (*X-Y*) plane, is more appropriate, particularly when this is combined with an analysis of plume location around the boil as a result of variation of ambient current speeds and directions. In addition, the approach described here is readily modified into a probabilistic analysis to estimate, for instance, concentration distributions or exceedance probability fields for prescribed threshold toxic concentrations. Uncertainty associated with modeling can also be evaluated for better assurance of the reliability of the modeling as discussed in the following section.

4.7. Probabilistic Analysis

The immediate objective in a probabilistic analysis is to present a systematic approach to deal with uncertainty, which is inevitable in hydrodynamic modeling. The term uncertainty is sometimes associated specifically with partial ignorance or lack of perfect information and it is different from variability (U.S. EPA 1996a, Frey and Burmaster 1999). In other cases, like in this study, the term uncertainty may also be used to refer to either variability or lack of perfect information about phenomena or model variables (Ferson et al. 1999, Mukhtasor et al. 1999a). Variability represents diversity or heterogeneity in a well-characterized population, and is a property of nature and usually not reducible through further measurement or study (Frey and Burmaster 1999). It may include temporal and spatial variation, and heterogeneity among individuals (Ferson et al. 1999). For example, at different seasons an offshore site may have different ambient currents, no matter how carefully they are measured. Furthermore, when a model is developed under partial ignorance or lack of perfect information about poorly-characterized phenomena being investigated, uncertainty is also exposed. The partial ignorance is a property associated with the risk analyst and is sometimes reducible through further measurement or study (Frey and

Burmaster 1999). For example, even though a true daily current speed at a particular site is not known, more samples can be taken to gain additional (but still imperfect) information about that daily current speed.

One type of model uncertainty in hydrodynamic modeling can be associated with assumptions on which the analysis is based upon, e.g. the assumption of equivalent cross-section aspect ratio discussed in the previous section. This type of uncertainty is difficult to take into account quantitatively in the analysis. Other model uncertainty is also raised because of the difficulty to accurately specify values of coefficients in a model. This latter type of uncertainty may be quantified by using uncertainty measures of these coefficients. Another issue in hydrodynamic modeling is how to cope with variability in model inputs, such as variability in the ambient current speed and directions.

Different approaches have been employed to deal with uncertainty, as described, for example, by Ferson et al. (1999). Deterministic or so-called worst case analysis is a traditional approach, which recognizes the fact that uncertainty exists but does not try to model it explicitly. In this approach, uncertainty is typically accounted for by selecting values for uncertain parameters so as to come up with a conservative answer, meaning that it is intended to be "safe". For example, values for uncertain parameters in hydrodynamic modeling may be selected so that an estimated toxic concentration is not to be less than the true concentration, although the true concentration is not known, so as to be "environmentally protective". Figures 4.6 and 4.7 above are typical results of deterministic analysis.

The worst case analysis, considering its simplicity, is remarkably effective. It is also useful as a preliminary screening procedure (U.S. EPA 1996a). Despite the fact that the worst case analysis is useful and championed for some cases of engineering applications, Ferson et al. (1999) highlighted problems associated with it. The main problem is that the degree of conservatism is not regulated in the worst case analysis. Furthermore, extreme values are not always selected for uncertain parameters involved in the analysis; instead, *a mixed approach* that uses mean estimates for some parameters and extreme values for other parameters is often found. Which parameters are estimated by which values is more a product of tradition than the result of serious justification (Ferson et al. 1999). These problems lead to situations where ecological risk assessment from different agencies, or focused on different potential hazards, are difficult to compare in the context of environmental management. Results of worst case analyses for different assessments may not be indicative of the likely actual outcomes or their rank order. This, as a result, limits their usefulness in planning and decision making (Ferson et al. 1999).

One approach to dealing with uncertainty is to use a sensitivity analysis. This approach is the most straightforward way to figure out what effect uncertainty has on a model by repeating the calculation for each of several possible values of an uncertain parameter of interest and depicting the final answer as a function of the uncertain parameter. This approach may be reasonably simple for some cases, however, when there are multiple numbers of uncertain parameters and a complex modeling formulation, this can be computationally prohibitive, and sometimes practically cumbersome even on computers.

Worst of all, the results can be difficult to interpret and hard to explain when there is correlation among uncertain parameters (Ferson et. al. 1999). The sensitivity analysis will always be an important tool in modeling, particularly, for instance, for evaluating the relative importance of uncertainty in parameters of interest. However, it needs to be complemented with methods that can provide an explanation about effects of several uncertain parameters on modeling results, such as a probabilistic analysis. This section presents a methodology for the probabilistic analysis of hydrodynamic modeling to better understand effects of uncertainty on modeling results.

4.7.1. Uncertainty measures

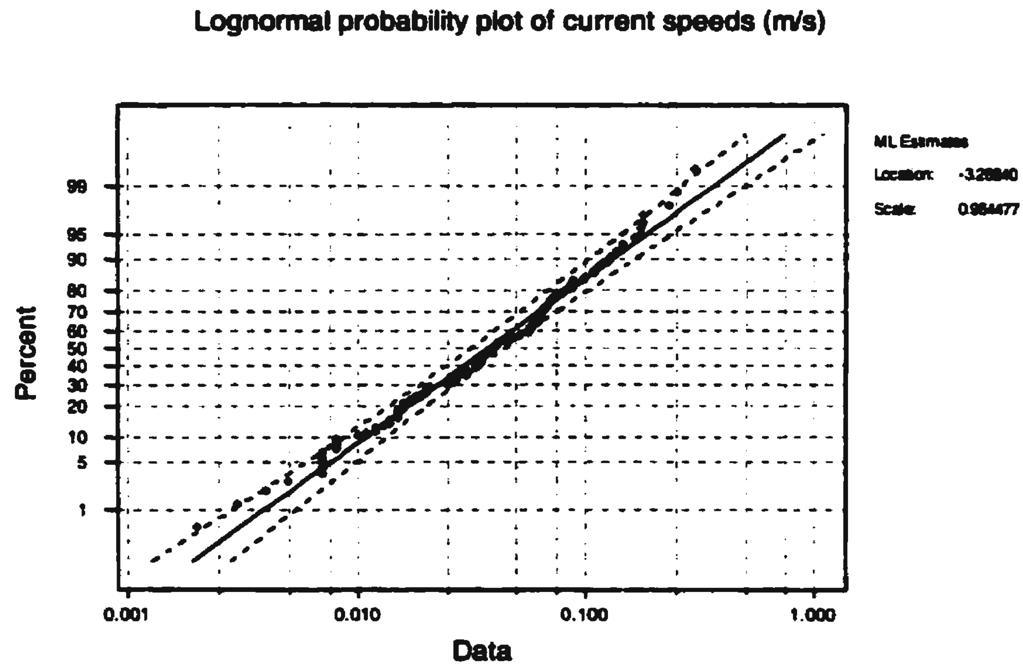
In order to proceed to a probabilistic analysis, uncertainty measures need to be first determined so that they can be taken into account quantitatively. The uncertainty measures may include statistics of the parameters, such as the mean, variance, minimum, maximum, and in some cases, the probability density function (*PDF*) or the cumulative distribution function (*CDF*). Table 4.1 presents uncertainty measures associated with model coefficients summarized from the modeling description presented in previous sections. As discussed previously, beside uncertainty in the model, which is reflected by uncertain values of model coefficients, uncertainty may also be exposed because of variability in model inputs. The model inputs include (1) ambient parameters, e.g. seawater current speed and directions, seawater depth above discharge, and density of ambient water; and (2) discharge parameters, e.g. flow rate of the produced water discharge and density of effluent.

Table 4.1. Uncertainty measures associated with hydrodynamic models

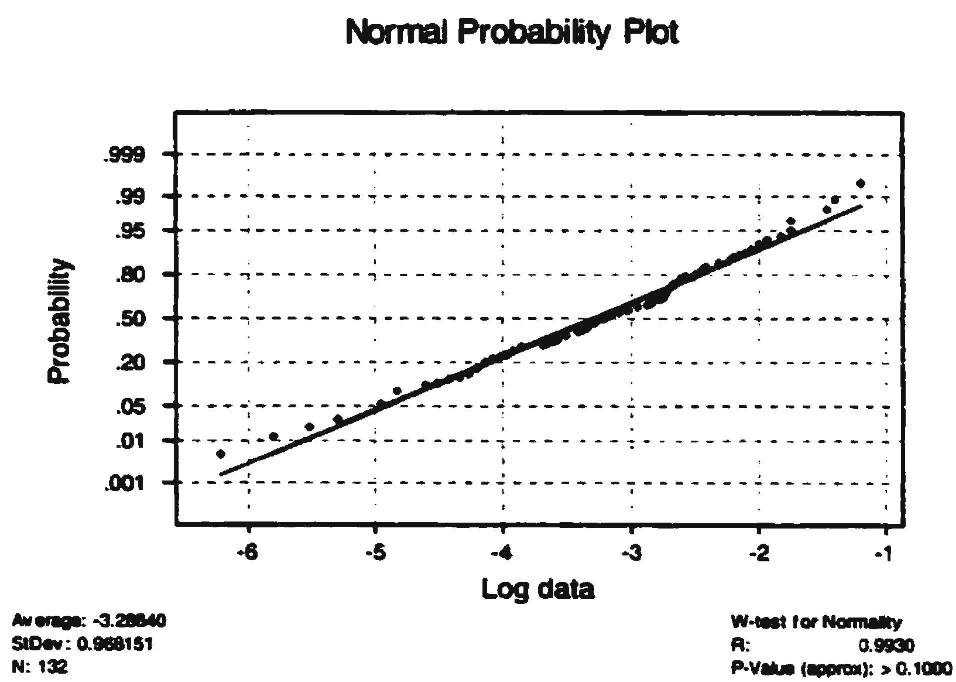
Parameter	Uncertainty measures	References
Near Field:		
Initia dilution, S (eq. 4.3)		
Model coefficients	$e = 0.13 \pm 0.02$	a) This thesis, Section 3.4
	$f = -0.31 \pm 0.03$	See a)
	$w = 0.46 \pm 0.02$	See a)
	$h = -0.22 \pm 0.04$	See a)
Error term, ϵ	$\epsilon = 0 \pm 0.092$	See a)
Boil location, x_b		
Model coefficients	$C_3 = 0.5824$	b) Huang et al. (1996) Doneker and Jirka (1990)
	$C_3 = 0.4571 - 0.6702$	c) Wright (1977b)
	C_4 = a function of l_b/l_m	See c)
	$C_5 = 0.6037 - 1.2761$	See c)
Intermediate Field:		
Bulk dilution, S_a (eqs. 4.5 - 4.7)		
Model coefficients	$C_{S1} = 3 - 5$	d) Wright et al. (1991)
	$C_{S2} = 1.5 - 2.0$	See b)
Far Field:		
Buoyant spreading (eqs. 4.21 - 4.22)		
Model coefficients	$\alpha = 0.15 - 0.6$	See b)
	$\beta = 0.707 - 1.414$	See b)

Uncertainty measures associated with ambient seawater currents have been reported in the literature, e.g. Wood et al. (1993), Webb (1987, and Orlob and Tumeo (1986). Although, when data are not available, the surface currents might be assumed to be induced by winds (Sullivan and Vithanage 1994), seawater current for design of the discharge should ideally be estimated based on site specific data. For example, when presenting discharge scenarios of produced water, Petro-Canada (1996) reported ambient characteristics including ambient current speeds and directions. The current speed of 0.14 m/s used in the deterministic analysis above was one of possible values based on information reported by Petro-Canada (1996). The Department of Fisheries and Ocean (DFO, 1999) provides information on ambient water characteristics for the East Coast of Canada. Data from DFO (1999) were analyzed in this study to fit probability distributions to those data. Figures 4.8 and 4.9 show the best fit of the daily-averaged ambient current speeds (Lognormal distribution, *p-value* > 0.10) and their directions (Beta distribution, *p-value* > 0.15) for moor depth of 20 m or less at the Grand Banks, located at 48-N 48-W to 47-N, 49-W.

Uncertainty measures for seawater depth may be required when there is a significant variability in the depth because of tide, which rises and falls gradually. Huang et al. (1994) took tidal variation into account when calculating initial dilution using a time domain simulation. Although they were unable to fit the data for tidal height into a theoretical probability density function, they noted that the distribution of the tidal height might be bimodal, and that the distribution might be approximated using the uniform distribution. They found that the difference between the mean water level and the mean lower low water level was typically 1.4 m for the Miami-Central Outfall at the east coast of South Florida, and that the 10 percentile on the cumulative distribution for the tidal height was 1.0 m. However, Lee and Koenig (1995) suggested that it was unnecessary to consider tidal variations because the variation in the Huang et al. (1994) study was small relative to the total depth and because the deterministic calculations for tide is often accurate.

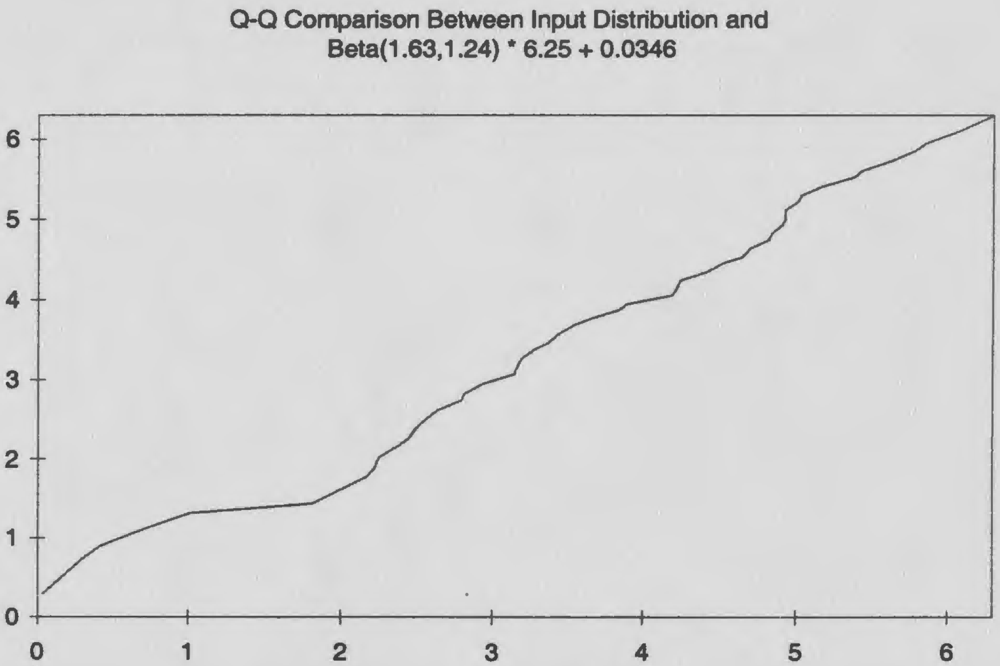


(a) Original averaged daily data

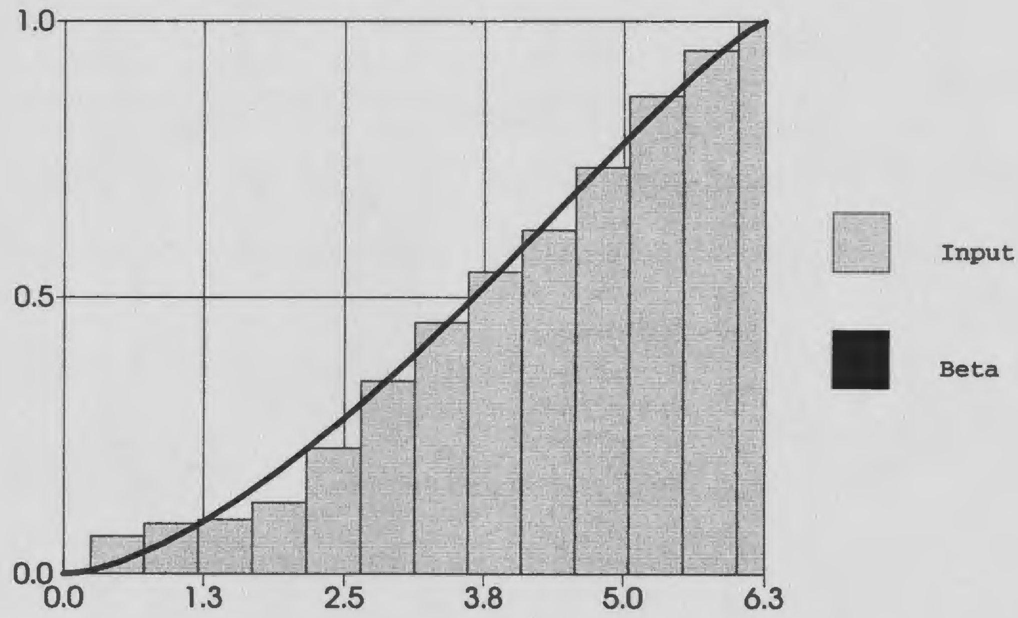


(b) Log-transformed averaged daily data

Figure 4.8. Lognormal distribution of daily-averaged current speeds (m/s)



(a) Q-Q plot of Beta distribution and the input data (unit in radian)



(b) Cumulative distributions of direction of currents (radian)

Figure 4.9. Beta distribution of daily-averaged ambient current directions (radian)

Petro-Canada (1996) noted that the tide along the east coast of Newfoundland and over the Grand Banks was associated with an amplitude of approximately 0.4 m. For a specific case where produced water is discharged from Floating Production, Storage and Offloading (FPSO) systems, effects of the variation in tidal height on the degree of the effluent dilution might be negligible. If the pipe of the outfall is attached to the floating system, the rises and falls of the tide may be followed by those of the port of the jet so that changes in the depth of the discharge, and thus in the dilution of the effluent, may be negligible.

Another ambient characteristic that is associated with the mixing process is the density of seawater. For example, it is noted (Petro-Canada 1996) that the density of seawater at 8 km east of St. John's, Newfoundland, is typically subject to seasonal variability with the largest seasonal cycles occurring at the surface, where maximum temperatures (greater than 12 °C) and minimum densities (salinities of approximately 31.1 ppt) occur in late August. The annual minimum in temperature (less than –1 °C), and maxima in density (salinity of about 32.3 ppt) occur in March. The development of a stratified water column in spring and summer is evident in the monthly temperature and salinity. During the winter, stratification throughout the water column is typically low. A similar pattern to the above seasonal temperature-density variability is observed over the central portion of the Grand Banks.

The density of ambient water is an important parameter in governing mixing processes, particularly initial dilution and buoyant spreading processes. However, its effects in the mixing processes are relative, in that they are usually attributed to the relative density difference between densities of ambient water and produced water, instead of the absolute value of the ambient density. The relative density difference is usually expressed as $\frac{(\rho_a - \rho_o)}{\rho_o}$, where ρ_a and ρ_o are the densities of the ambient seawater and effluent, respectively. During design of the discharge, however, data on effluent density is not available; and an estimate is the only information which can used in the design. Studies (Brandsma and Smith 1996, Somerville et al. 1987) show that the relative density difference in offshore discharges of produced water varies significantly, typically 0.037 (Bass Strait, Australia), –0.069 (Gulf of Mexico, U.S.A.), and 0.013 (North Sea, Europe).

Similarly to the case of effluent density, uncertainty measures for produced water flow rate is very difficult to define accurately during the design stage. The flow rate may, however, be determined using estimates; and for evaluation purposes, produced water flow rates from other offshore sites can be considered. Studies show that the discharge rates of produced water from offshore oil fields are on the order of 4,000 m^3/day in the Gulf of Mexico, U.S.A. to 123,000 m^3/day in the Java Sea, Indonesia (Brandsma and Smith1996, Smith et al. 1996, Somerville et al. 1987). This confirms results from the 30-platform study (U.S. EPA (1993), indicating that the ratio of produced water to oil production rates is typically in the range of 0.1 to more than 12, with the mean of the ratio of 3.5. The flow rate typically ranges from 2 to 150,000 bbl/day. The flow rate varies from time to time and from field to field; however, it is generally very significant in magnitude.

4.7.2. Probabilistic methods

Probabilistic methods have been used in the past for assessing ocean discharge of wastewater; however, for produced water discharges, the assessment is usually based on a deterministic approach, e.g. Somerville et al. (1987), Smith et al. (1996). Meinhold et al. (1996a) employed a probabilistic analysis in assessing human health risk associated with produced water discharge, but uncertainty in hydrodynamic modeling was left unevaluated. In the case of hydrodynamic modeling, Huang et al. (1994) proposed an approach, which employs a Time Domain (TD) simulation using field data sets to generate a time series of initial dilution of sewage discharge. They presented input parameters including ambient seawater currents, seawater depth above discharge and wastewater flow rate, in terms of time series, hourly data sets. These were then used as inputs into a deterministic empirical equation to produce a time series of hourly initial dilution. Values of the parameters at a given time were used to calculate the initial dilution at that time. Other possible combinations were not considered. As a result, although this approach takes into account the variability of the input parameters, its application for estimating the extreme events can be misleading.

Another method for addressing uncertainty is the first order second moment (FOSM) method (Mukhtasor et al. 1998). This method may be applied in a very limited case, in which the performance function of the system under consideration (e.g. initial dilution equation) is simple. FOSM is a useful method for cases in which information on the uncertainty of the parameters is limited to the mean and variance of the input parameters and the probability distributions of the parameters are left undetermined (Ang and Tang

1975, 1984). The problem with this method is that its performance becomes unacceptably poor when it is used for complex systems (Melching 1995; Mukhtasor et al. 1998, 2001b) and therefore, this may not be applicable for complex formulations such as hydrodynamic modeling described in the previous sections.

An alternative to these approaches is to evaluate the available information in a way that reveals just how probable each of the possible outcomes actually is and that typically involves complex probability analysis, which can in turn be very difficult analytically (Ferson et al. 1999). A practical approach to this problem is to use a numerical analysis called Monte Carlo (MC) simulation, which involves random sampling from each of the probability distributions characterizing uncertainty. The MC simulations can be applied to a wide variety of problems involving uncertainty analysis. Driven by advancing computational power, MC simulations for uncertainty analysis have been commonly used; and software packages have become available providing general access to MC simulations (Palisade 1997, U.S. EPA 1996a). These software packages make MC simulations computationally practical and have been greeted with much enthusiasm in the risk assessment community (Ferson et al. 1999, U.S. EPA 1996a).

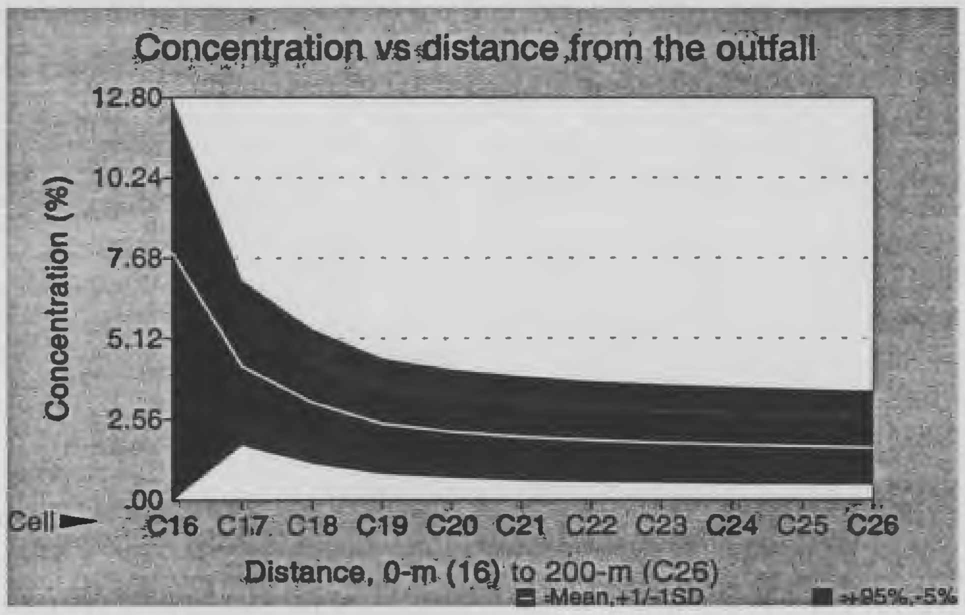
MC simulations have been used to consider uncertainty associated with the variability of model inputs of sewage discharge in the ocean environment, e.g. Orlob and Tumeo (1986), Webb (1987), Bale et al. (1990). MC simulation are performed by replicating the real world based on a set of assumptions and conceived models of reality. In each simulation, the MC simulation uses a particular set of random values generated in accordance with the corresponding probability distribution function of the input parameters. Then, for each simulation, the performance function is calculated using the appropriate values of these parameters. This simulation process is repeated many times and the results are recorded in a form of output statistics or distributions. In other words, the main task in MC simulations is to generate random values from a prescribed probability distribution. For a given set of generated random values, the simulation is deterministic (Ang and Tang 1984).

One of the most important steps in MC simulations is a sampling process, which is a process by which values of a model input of interest are randomly drawn from a prescribed probability distribution (Palisade 1997). Accurate results for output distributions depend on a complete sampling of input distributions. MC sampling refers to a technique for using random or pseudo-random numbers to sample from a probability distribution. The sampling technique is random sampling in a sense that any given sample may fall anywhere within the range of the input distribution. Samples are more likely to be drawn in areas of the distributions, which have higher probabilities of occurrence. In some cases involving complex systems, the number of iterations that is required in MC simulation to "recreate" the input distributions through sampling is typically very large (in the order of tens of thousands) and is sometimes computationally cumbersome or prohibitive. If only a small number of iterations is performed, a problem of clustering may arise. The problem of clustering becomes especially pronounced when the case includes skewed probability distributions (Palisade 1997). That is the reason why MC sampling often requires a large number of samples to approximate an input distribution, especially if the input distribution is highly skewed.

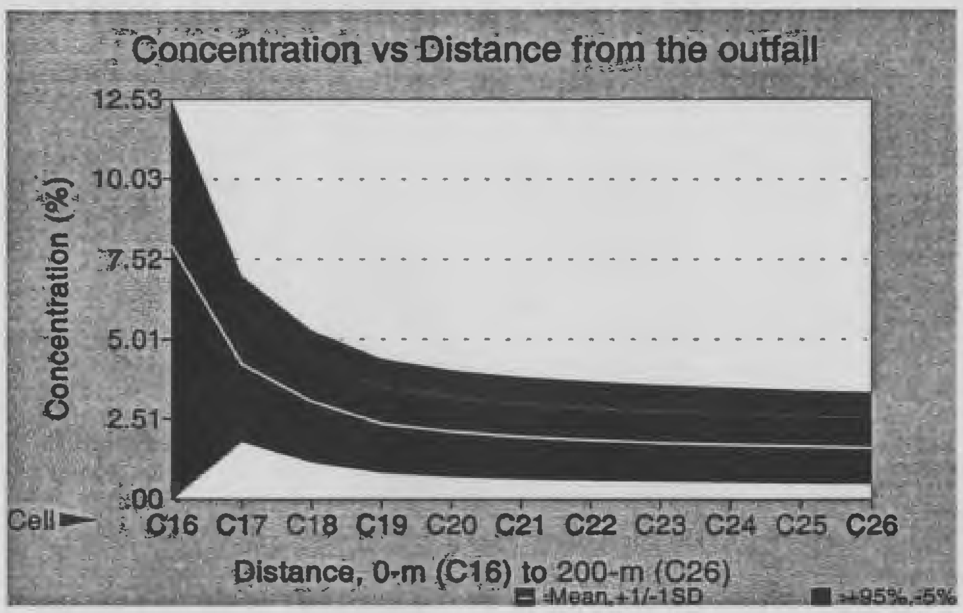
An improvement in the sampling technique is developed by using a method of Latin Hypercube Sampling (LHS), which is designed to accurately recreate the input distribution through sampling in fewer iterations when compared with random sampling in the MC simulation. In the LHS approach, stratification of the input probability distributions is employed by dividing the cumulative curve into equal intervals on the cumulative probability scales (0 to 1.0). A sample is then randomly drawn from each interval so that sampling is forced to represent values in each interval and thus the input probability distribution. The number of stratifications of the cumulative distribution is equal to the number of iterations performed. By this approach, LHS offers great benefits in terms of increased sampling efficiency, faster runtimes because of fewer iterations, and assuring the representation of the input probability by forcing the sampling to include the outlying events (Palisade 1997).

The traditional approach of using MC simulation considers only uncertainty in model inputs (e.g. Bale et al. 1990, Mukhtasor et al. 1999a, Orlob and Tumeo 1986, Webb 1987) but the significance of uncertainty associated with model coefficients and error term is left unevaluated. As discussed in Chapter Three, Tung (1994) suggested that information on this type of uncertainty should be considered in the risk analysis. In this study, MC simulations were employed using random sampling and LHS for the case of the produced water discharge at the Grand Banks area, as considered in the previous sections. Uncertainty measures discussed in Subsection 4.7.1 were used in this analysis, particularly those related to the model uncertainty (Table 4.1) and the variability in model inputs relevant for the Grand Banks discharge. Those uncertainty measures are defined and summarized in Table 4.2 for MC simulations.

This analysis is typical in that it considers only one of the possible scenarios in a discharge design; discussion on evaluating different scenarios of discharge is given in Chapter Six. Figure 4.10 shows a typical comparison of MC simulation results using the random sampling and LHS approaches. The simulations were performed using @RISK software (Palisade 1997). As can be seen in Figure 4.10, using a "sufficient" number of simulations, both the approaches provide the same answer. A sufficient number of simulations was determined by specifying a convergence criterion. To monitor this convergence, a set of statistics (typically mean, median, skewness and percentile probabilities) was calculated for each output every 100 iterations (or interval) and compared with the same statistics calculated at the prior interval during the simulation. As more iterations (simulations) are run, statistics describing each distribution change less and less with additional iterations. The processes continue until they "converge" or change less than a specified threshold. In this study, a typical threshold is set at 0.5%. For a simple case, i.e. for a given ambient current direction, the LHS approach is typically about 15% more efficient than the random sampling MC simulation, i.e. the time required to perform simulation using the LHS approach is 15% less than that using random sampling MC simulation. The simulations were performed using a medium type of computer (Celeron 333, 64 MB RAM). For this reason, further probabilistic analysis was performed using the LHS-based MC simulations.



(a) LHS-based MC simulations (ns = 2400)



(b) Random sampling-based MC simulations (ns = 2600)

Figure 4.10. A typical comparison of random sampling and LHS-based MC simulations (a simple case, i.e. for a given ambient current direction)

The deterministic analysis presented in the previous section shows that the produced water concentration at 100 m downstream is 1.5%. Unlike the deterministic analysis, MC simulations provide uncertainty measures of the concentration as shown in Figure 4.11, which presents the distribution of the concentration. Figure 4.11 shows the likelihood of the concentrations with the mean, median, standard deviation and 95% percentile of approximately 2.0, 1.9, 1.0, and 3.9%, respectively. The concentration of 1.5% from the deterministic analysis is associated with a cumulative probability of approximately 36%, meaning that there is 64% probability that this concentration is exceeded. This is one advantage of the probabilistic analysis, that is the reliability of the calculation can be estimated. Probabilistic analysis not only provides distributions shapes, but also takes into account the uncertainty factors simultaneously. Sensitivity and deterministic analyses do neither. The probability analysis can also be presented in term of exceedance probability of a given threshold toxic concentration as shown in Figure 4.12.

Table 4.2. Uncertainty measures used in typical MC simulations

Parameter	Value	Uncertainty measure in typical MC simulations
Near Field:		
Initia dilution, S (eq. 4.3)		
Model coefficients [1]	$e = 0.13 \pm 0.02$	Normal (0.13, 0.02)
	$f = -0.31 \pm 0.03$	Normal (-0.31, 0.03)
	$w = 0.46 \pm 0.02$	Normal (0.46, 0.02)
	$h = -0.22 \pm 0.04$	Normal (-0.22, 0.04)
Error term, ϵ [1]	$\epsilon = 0 \pm 0.092$	Normal (0, 0.092)
Boil location, x_b		
Model coefficients	$C_3 = 0.5824$ [2]	Triangle (0.46, 0.58, 0.67)
	0.4571 - 0.6702 [3]	
	C_4 = a function of l_b/l_m [3]	
	$C_5 = 0.6037 - 1.2761$ [3]	Uniform (0.60, 1.28)
Between Near and Far Fields:		
Bulk dilution, S_a (eqs. 4.5 - 4.7)		
Model coefficients	$C_{S1} = 3 - 5$ [4]	Uniform (3, 5)
	$C_{S2} = 1.5 - 2.0$ [2]	
Far Field:		
Buoyant spreading (eqs. 4.21 - 4.22)		
Model coefficients	$\alpha = 0.15 - 0.6$ [2]	Uniform (0.15, 0.6)
	$\beta = 0.707 - 1.414$ [2]	Uniform (0.71, 1.41)
Variability of model input:		
Ambient seawater currents [5]	Daily mean current speeds (m/s)	Lognormal (-3.29, 0.96)
	Direction of currents (radian)	Beta (1.63, 1.24) * 6.25 + 0.0346

Note:
1. From Chapter Three of this thesis
2. From Huang et al. (1996) and Doneker and Jirka (1990)
3. From Wright (1977b)
4. From Wright et al. (1991)
5. Data analyzed from DFO (1999)

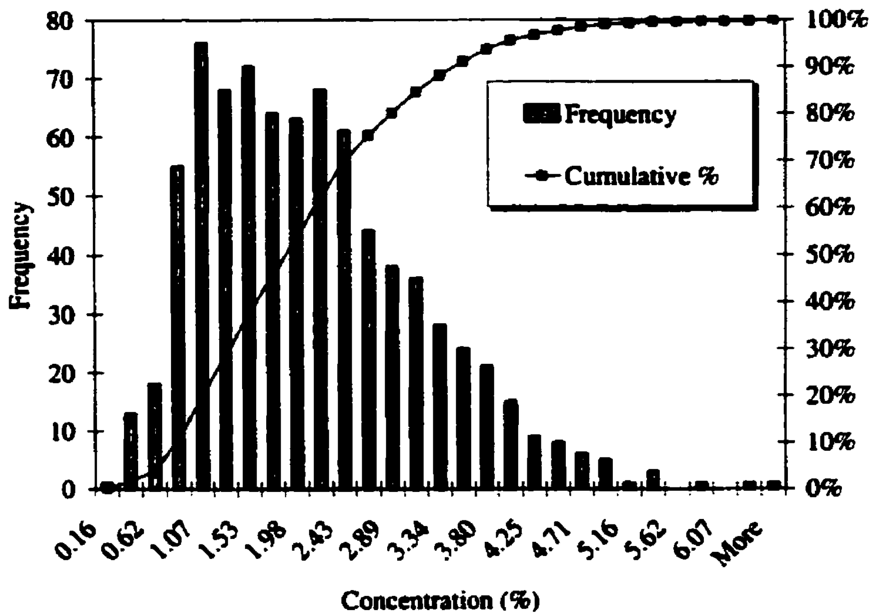


Figure 4.11. Distribution of produced water concentration at 100 m downstream

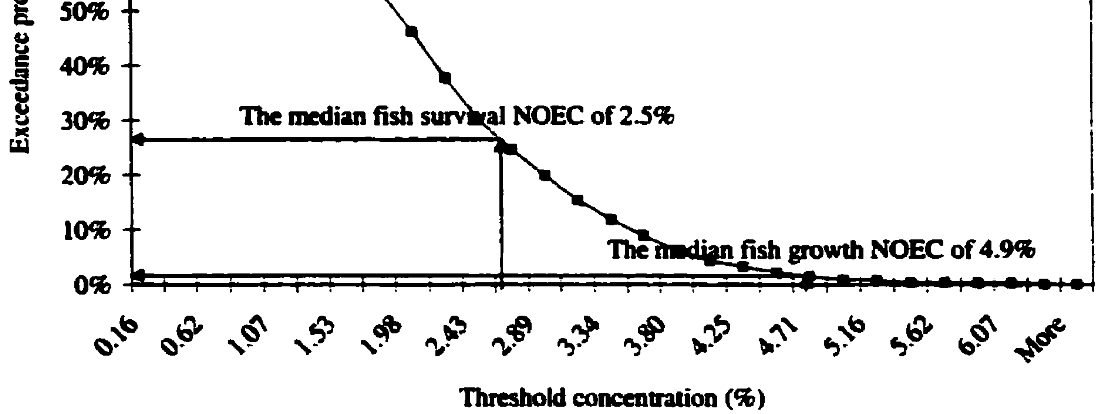


Figure 4.12. Exceedance probability for typical threshold concentrations.

As can be seen in Figure 4.12, for example, if the threshold concentration is set using the median fish survival and growth NOECs (No observed effect concentrations) of 2.5% and 4.9% (data from Meinhold et al. 1996c), the exceedance probability is approximately 27.8% and 0.8% for fish survival and growth, respectively. The effects of the direction of currents on the concentration distribution can also be taken into account in this study by presenting several statistics of produced water concentrations as shown in Figures 4.13 to 4.15. These figures were developed by taking the probability distribution of the ambient current direction and other relevant parameters as defined in Table 4.2. The results in terms of the concentration fields are useful for ecological risk assessment and ecological risk-based design as discussed in Chapters Five and Six.

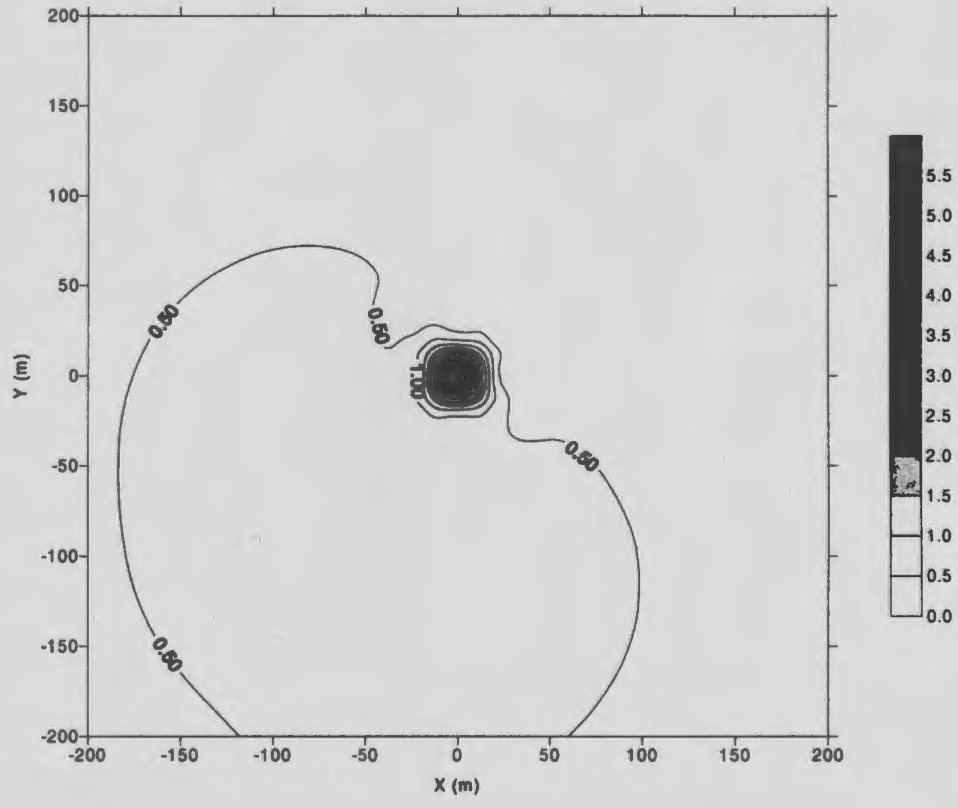


Figure 4.13. Distribution of the mean concentrations (%)

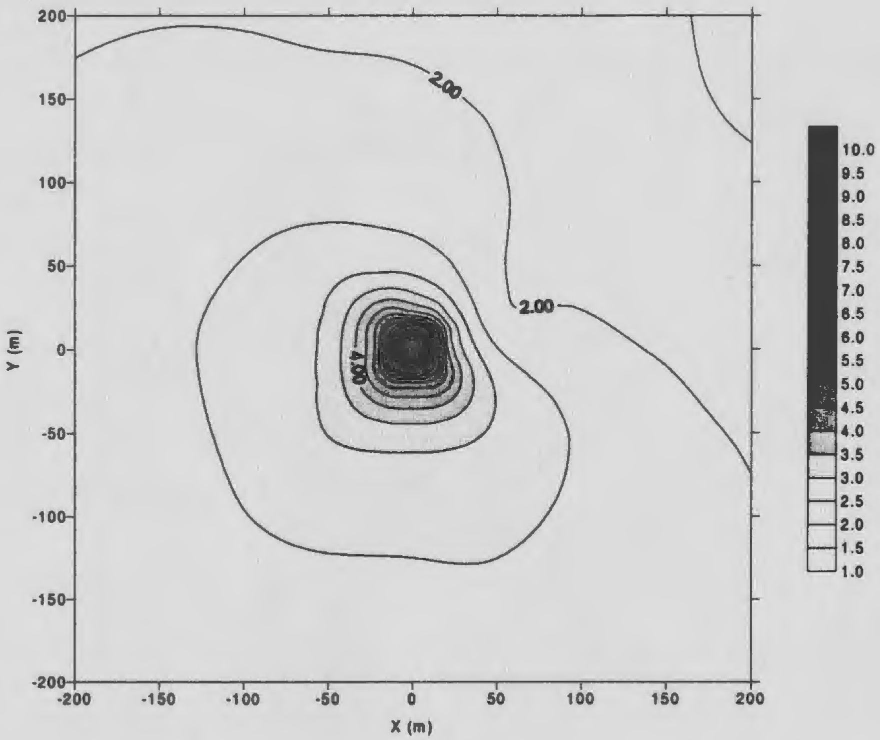


Figure 4.14. Distribution of the 95%-tile concentration (%)

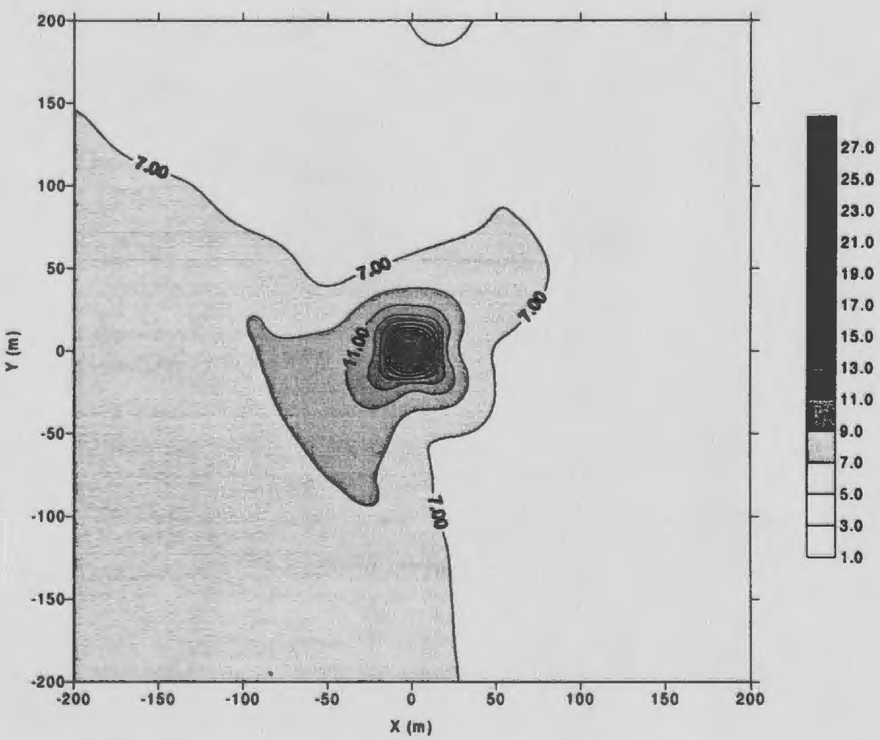


Figure 4.15. Distribution of the maximum concentrations (%)

4.8. Summary

This chapter presents analyses of hydrodynamic modeling for produced water discharge from an offshore platform. A mixing process occurring in two separate regions, near and far fields are discussed. Modeling of the intermediate region connecting the near- and far-fields is provided using a control volume approach. An application example is discussed using deterministic and probabilistic analysis and a comparison with a presently available model, i.e. the CORMIX model, is also presented. The probabilistic analysis presented in this chapter considers uncertainty measures associated with model coefficients and model inputs.

The deterministic components of the proposed model presented in this chapter are not entirely new in that the initial dilution model is based on the model developed in Chapter Three and the far field model and the control volume approach are adapted from published models. However, the methodology presented in this chapter has not been applied for the probabilistic analysis of produced water discharge. Although the integrated model may only be applicable for the limited area close to the discharge location, it is useful for assessing discharge scenarios of produced water from an offshore platform. This is because ecological effects of produced water are usually close to the discharge location (within 500 to 1000 m) and because regulatory mixing zones are usually defined at a distance of typically about 100 m or 200 m.

Comparison using a case study shows that the proposed model and the CORMIX model are generally in good agreement, particularly in estimating average effluent concentrations. However, the proposed model also provides the concentration field in the X-Y directions so that it may be applicable for the analysis of the mixing zone, which in some cases is defined in terms of horizontal area around the discharge location. The proposed model is also readily modified into a probabilistic analysis to take into account uncertainty associated with model inputs and model coefficients.

The probabilistic analysis was carried out in this chapter using Monte Carlo (MC) simulations. Concern regarding the "excessive" number of simulation was addressed by comparing two methods of sampling in the simulations, i.e. random sampling and Latin Hypercube Sampling (LHS) methods. A comparison between random sampling and LHS for MC simulations of a case of hydrodynamic modeling shows that LHS-based MC simulations are typically about 15% more efficient than the random sampling MS simulations. This chapter also shows that probabilistic analysis not only provides distribution shapes, but also takes into account the uncertainty factors simultaneously. The probabilistic analysis can also be presented in term of exceedance probability for a specified threshold toxic concentrations, which may be used for further study of ecological-risk based design of produced water discharge.

Chapter 5

Ecological Risk Assessment

5.1. Introduction

The term ecological risk assessment (ERA) is typically defined as "*a process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors*" (U.S. EPA 1998). The purpose of ERA is to contribute to the protection and management of the environment through scientifically credible evaluation of the ecological effects of man-made activities such as disposal of wastes from offshore oil production (Suter II 1993, Mukhtasor et al. 2000c). In the last two decades, interest in ecological risk assessment has increased significantly and guidelines for the assessment have been made available from regulatory agencies, e.g. Canadian Council of Minister of Environment, CCME (1996a, 1996b, 1997) and U.S. EPA (1998). This chapter reviews current approaches used for ERA of produced water discharges. Problems associated with presently used approaches are discussed and a methodology relevant to design of produced water discharges in the marine environment is identified. Application of the methodology is provided in Chapter Six.

5.2. Review of ERA of produced water discharges

Ecological risks have been assessed for specific pollutants associated with produced water discharges from offshore oil fields. Approaches used in the assessment vary from simple to quite comprehensive. Neff and Sauer (1996) assessed ecological risks associated with polycyclic aromatic hydrocarbon (PAH) from produced water discharges into the Western Gulf of Mexico. In this assessment, concentrations of individual and total PAHs in ambient water, sediment, and whole tissue of marine animals were compared to the highest no observable effects concentrations, or threshold concentrations. The conclusion was that risk of harm from PAHs in the produced water was minimal. However, the level of the minimal risk was not quantitatively defined in terms of, for example, the probability of exceedance of one in a million.

A quantitative ERA was performed to evaluate risks associated with produced water discharged from the Statfjord and Gullfaks fields (Karman et al. 1996). An approach, which is based on the Chemical Hazard Assessment and Risk Management (CHARM, Thatcher et al. 1999) model, was employed in the Karman et al. (1996) study. The CHARM (Thatcher et al. 1999) model was originally developed for ERA of discharges related to offshore oil operations in the North Sea. In this model, the ecological risk is calculated by taking the ratio of predicted environmental concentration to predicted no-effect concentration, (*PEC/PNEC*). For calculating *PNEC* in water, the *NOEC* (No Observed Effect Concentration) for the most sensitive effect parameter (e.g., growth, reproduction) is considered when data are available. When toxicity data for several species are available, *PNEC* is defined as (Karman et al. 1996):

$$PNEC = \frac{GM}{1000/\sqrt{n}} \tag{5.1}$$

where *GM* is the geometric mean of available EC_{50} values (i.e. chemical concentration resulting in observed effects in 50% of test animals), *n* is the number of species for which toxicity data is available for a particular chemical. In the above equation, the coefficient of 1000 is a subjective factor (French 1999).

Karman and Reerink (1998) proposed a dynamic assessment of the ecological risk by assuming that risks can be estimated from the ratio of time-integrated predicted environmental concentration (*PEC*) to time-adjusted predicted no effect concentration (*PNEC*). With this modification, they improved the current practice of using the CHARM model by taking into account the variability of exposure concentration. However, like the CHARM approach, Karman and Reerink (1998) used the hazard quotient approach without making any consideration of a probabilistic ERA. A recent version of the CHARM (Thatcher et al. 1999) model vaguely addresses uncertainty associated with hazard or risk quotients by simply dividing and multiplying the calculated quotient by 3 to define the lower and upper 90% confidence level, respectively.

Unlike studies on acute effects, very limited studies have been conducted on chronic effects of produced water discharges. Because of this, Reed (1996) suggested that the focus of future research in environmental risk of produced water should be on potential chronic effects. Reed et al. (1996) presented a model called PROVANN for assessing potential chronic effects of produced water. The model consists of four components: a near-field release model, a far-field transport model, a biological exposure model, and a bioaccumulation and biomagnification model. PROVANN was modified into DREAM (Dose related Risk and Effects Assessment Model) (Reed 1999). DREAM (Reed 1999) addresses several problems in ERA, including time-space variations of discharge concentration fields, exposure of organisms with different behavior patterns, assessment of mixture of chemicals, and assessment of sub-lethal chronic effects in terms of body burdens.

A comparative summary among different risk assessment models is presented in Table 5.1. The models differ in the degree of sophistication of fate modeling as well as assumptions in characterizing exposures, effects and risks. In general, however, all the models do not specify uncertainty associated with modeling. An uncertainty evaluation of produced water discharges was presented by Meinhold et al. (1996a). They proposed an approach for the assessment of human health risks associated with produced water discharges to open bays in Louisiana, U.S.A. Monte Carlo simulations were used in that approach by focusing on the human health effects of the two contaminants: radium and lead. However, uncertainty associated with hydrodynamic modeling was not evaluated in that approach.

Table 5.1. Summary of risk assessment models for produced water discharges

Module	CHARM	PROVANN	DREAM
FATES	Fixed dilution factor	Mathematical equations for contaminant fates	Mathematical equations for contaminant fates
EXPOSE	Gross exposure estimate through water, sediment and food chain.	Exposure from water only. Exposure to single chemical.	Exposure from water, sediments, and user-defined food web.
EFFECT	No effect calculation	Single-component critical body burden (CBB).	Critical body burden (CBB) defined for short and long-term exposures.
RISK	PEC/PNEC > 1 implies non negligible risk	Risk threshold set for BB/CBB > 1	Risk distributed over the local populations.

As discussed above, considerable effort has been devoted in the past to assess ecological risks of produced water discharges. The effort was usually directed towards monitoring without specifically considering the integration between ERA and engineering design of the produced water outfalls. Futhermore, in the presently used approaches, the endpoint of the assessment is not well defined. Defining assessment endpoint is critical because it is an explicit expression of the environmental value to be protected. "*Ecological risk assessment will not be influential until regulatory agencies say that some ecological entities are worth protecting*" (Suter II 2000).

5.3. Methodology for ERA

The objective of ERA in this study is to evaluate the likelihood that adverse ecological effects may occur as a result of exposure to produced water from a designed outfall. Guidelines for ERA are presently available but they are commonly intended for a wide range of environmental issues. This section considers available ERA guidelines and applications such as Efroymson et al. (1996), CCME (1996a, 1996b, 1997), and U.S. EPA (1998), particularly those relevant for produced water discharges in the marine environment. A typical framework for ERA is presented in Figure 5.1, which consists of two major elements: characterization of effects and characterization of exposure. It provides a focus for conducting three phases of risk assessment, i.e. problem formulation, analysis, and risk characterization, enclosed by a dark solid line in that figure. Adaptation of this framework to design of produced water discharge is discussed in the following subsections. A compilation of information relevant for ERA of produced water discharge is also provided.

5.3.1. Problem formulation

Problem formulation is the first step in the risk assessment framework. It provides the foundation for the ERA processes and covers description of sources of contamination with relevant features of the environment, identification of ecological endpoints, summary of that information in terms of a conceptual model of the hazard posed by the contaminants to the endpoint biota, and analysis plans.

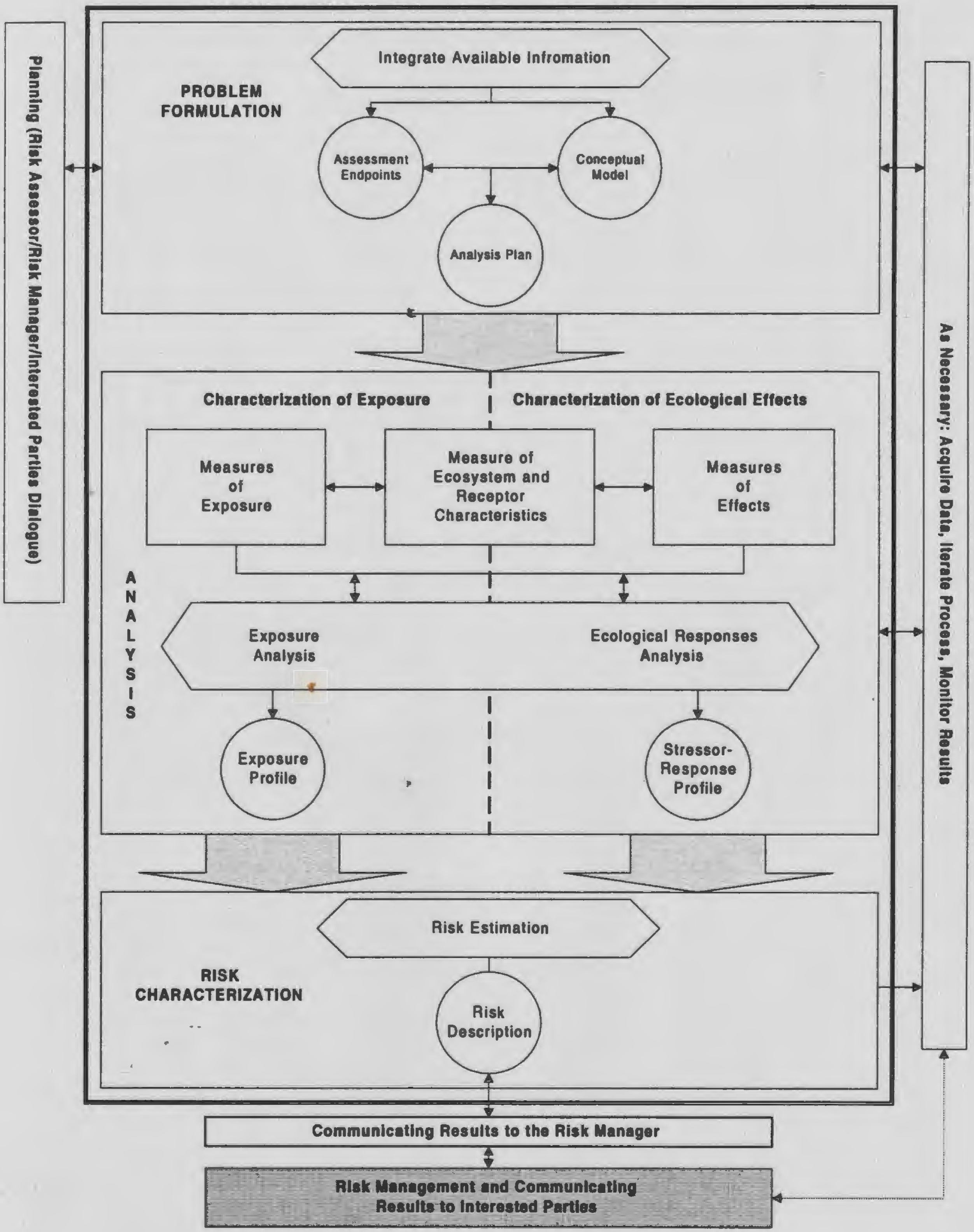


Figure 5.1. General framework of ecological risk assessment (after U.S. EPA 1998) (rectangles designate inputs, hexagons indicate actions, and circles represent outputs)

(i) Source of contaminants and the environment

Produced water is the major waste stream during oil production, in terms of volumes and amount of pollutants. It includes formation water, injected water, and any chemical added downhole or during the oil/water separation process. The quantity of produced water from an oil field varies considerably and depends on the characteristics of the oil reservoir and the age of the field. A typical variation between 2 to 150,000 bbl/day has been reported in the literature for associated production rates of 40 to 24,000 bbl/day for oil/condensate and 0.1 to 150 MMCF/day for gas (U.S. EPA 1993). Generally, produced water can account for between 2 to 98% of the extracted fluids from the reservoir (Stephenson 1992, Wiedeman 1996).

Following treatment at the platform, produced water is often discharged into the ocean. Although it is subject to treatment before discharge, produced water effluent commonly still contains toxic chemicals. The composition of the effluent varies from place to place and includes various inorganic, organic, and radioactive chemicals (Roe et al. 1996, Stephenson 1992). Tables 5.2 and 5.3 show chemical concentrations in produced water compiled from different regions as well as a summary of the range of chemical concentration in produced water worldwide. These chemicals have been identified to be of potential ecological concern for ecological risks and have been subject to many environmental studies, e.g. Frost et al. (1998), Neff (1997), and Roe et al. (1996).

Figure 5.2. Typical chemical concentration in produced water from different regions
(in µg/L or otherwise stated, data compiled from Roe et al. (1996), Smith et al. (1996), Stephenson (1992))

Parameter	North Sea (6 platforms)			Gulf Mexico (42 platforms)			Java Sea (6 platforms)			Bass Straits
	minimum	average	maximum	minimum	average	maximum	minimum	average	maximum	(3 platforms)
As	nr	nr	nr	nr	nr	nr	1.5	4.7	9	<1.5
Ba	12000	27430	42100	nr	nr	nr	nr	nr	nr	nr
Cd	20	6670	10000	0	27	98	nd	0,5	nd	<5
Cr	0.08	13.2	40	0	186	390	7.5	124	185	<5
Cu	2	128.8	600	0	104	1455	nd	5.2	nd	<5
Fe	4	20.57	23	nr	nr	nr	nr	nr	nr	nr
Hg	1.9	4	9	nr	nr	nr	0.004	0.006	0.0012	0.044
Ni	nr	nr	nr	0	192	1674	45	95	143	<5
Pb	50	112.5	270	2	670	5700	12	193	260	23
Zn	0.26	47	200	17	170	1600	nd	nd	nd	<30
Benzene	nr	nr	nr	2	1318	8722	69.3	1720	3000	24
Toluene	nr	nr	nr	60	1065	4902	90.8	650	1300	nr
BTX	1100	15740	66900	nr	nr	nr	nr	nr	nr	nr
Napthalene	38	185	398	0	132	1179	8.4	35	99	1.6
phenol	33	1617	5100	0	1049	3660	nr	nr	nr	nr
^{226}Ra (pCi/l)	nr	nr	nr	4	262	584	nr	nr	nr	nr
^{228}Ra (pCi/l)	nr	nr	nr	18	277	586	nr	nr	nr	nr

Note nr : data were not reported.
nd: concentration was not detected.

Table 5.3. Range of concentration of organic chemicals and metals in produced water worldwide (after Neff 1997)

Parameter	Concentration (μg/l)
Total Organic Carbon	≤100 to 2,100,000
Total Saturated Hydrocarbons	17,000 to 30,000
Total Benzene, Toluene, Ethylbenzene, Xylenes (BTEX)	68 to 578,000
Total Polycyclic Aromatic Hydrocarbons (PAHs)	80 to 3,000
Steranes/Triterpanes	140 to 175
Ketones	1,000 to 2,000
Phenols	600 to 21,500
Organic Acids	≤1.0 to 10,000,000
Sulfates	≤1,000 to 8,000,000
Arsenic	0.004 to 320
Barium	≤1.0 to 2,000,000
Cadmium	0.0005 to 490
Chromium	≤0.001 to 390
Copper	≤ 0.001 to 55,000
Lead	≤ 0.001 to 18,000
Mercury	≤ 0.001 to 33
Nickel	≤ 0.01 to 1,674
Zinc	0.005 to 150,000

Neff (1997) discusses in detail the environmental hazard of these contaminants in the marine environment worldwide, based on a chemical specific basis. Table 5.4 provides an example of environmental hazards associated with specific chemicals. A whole effluent toxicity evaluation has also been reported for produced water. For example, typical produced water from North Sea platforms has been associated with ecological impacts, which are reported in terms of effect concentration with 50% reduction in growth (EC_{50}) of algae (based on two-day exposure) of 45 to 535 ml/l (Brendehaug et al. 1992). Lethal concentration with 50% mortality based on one-day exposure (LC_{50}) was 100 ml/l to the copepod *Calanus finmarchicus* (Sommerville et al. 1987). For fish, the lowest value registered of LC_{50} is for guppy, *Poecilia retivulata*, at a value of 7.5-423 ml/l (Jacobs and Marquenie 1991).

Table 5.4. Environmental hazards associated with specific chemicals (data from Middleditch 1984)

Substance	Concentration (ppm)	Sublethal effect
Cd	0.01	Copepod reduction reduced
	0.028-0.11	Hydroid growth rate reduced
	0.05	Decapod larval development retarded
	0.078	Scallop growth rate reduced
	0.1	Polychaete reproduction enhanced
	0.56-2.5	Polychaete reproduction suppressed
	0.76	Shrimp gills blackened
	>2-10	Fish hatch rate decreased
	100	Fiddler crab regeneration retarded
Cr (VI)	0.03-0.1	Polychaete spawning inhibited
	0.05-0.1	Polychaete reproduction suppressed
	0.1	Polychaete reproduction halted
Cu	0.01-0.4	Phytoplankton growth rate reduced
	0.01-0.013	Hydroid growth inhibited
	0.012-0.05	Algal growth reduced
	0.02-0.05	Dinoflagellate growth reduced
	0.04	Shrimp growth rate reduced
	0.1-0.25	Polychaete reproduction suppressed
	0.1	Barnacle larvae photokinesis reduced
	0.25	Clam inhalant siphon contracts
Pb	0.2-5	Polychaete reproduction suppressed
	1.0-10	Fish hatch rate decreased
Hg	0.0016-0.0017	Hydroid growth inhibited
	0.01	Phytoplankton growth rate reduced
	0.05-0.1	Polychaete reproduction suppressed
	0.1-0.5	Crab melanogenesis inhibited
Zn	0.32-0.56	Polychaete reproduction suppressed

Once produced water is discharged into the ocean, it mixes with the ambient seawater and its concentration decreases. Characteristics of the ambient water and the effluent vary considerably from place to place as discussed in the previous chapter. For example, the density of produced water from different oil production fields typically varies from 977 to 1088 kg/m^3 (Bransma and Smith 1996; U.S. EPA 1996b). Considering the density difference between effluent and ambient seawater, some produced water discharges result in a positive buoyant plume that comes up to the sea surface, while others produce a negative buoyant plume that sinks deep into the water. Variation in density stratification of ambient water makes the environmental effects assessment more complex in term of which ecological entities are exposed to the produced water discharges.

(ii) Selection of endpoints

Assessment endpoints are selected to provide an explicit expression of the environment value to be protected. The selection is based on ecological relevance, susceptibility to known potential stressors (pollutants) and relevance to management goals. Ecologically relevant endpoints reflect important characteristics of the system and are functionally related to other endpoints (U.S. EPA 1998). These endpoints may be identified at any level of organization, e.g., individual, population, community and ecosystem, as discussed in a subsequent phase, i.e. characterization of effects.

The relevance of an endpoint in the assessment can be related to appropriateness of scale. An endpoint has appropriate scale for a site if toxic effects on the site could have a significant effect on the endpoint. For example, a site under assessment supports only a few particular fish, which forms a very small fraction of the biological population to which they belong. In this case, individual fish of this kind have an appropriate scale, but the fish population does not. Ecological resources are considered susceptible when they are sensitive to the stressor to which they are exposed. Sensitivity refers to how readily a particular stressor affects entities. It is related to the mode action of the stressor and is also influenced by life history characteristics. Measures of sensitivity may include mortality, growth, or adverse reproductive effects from exposure to the stressor.

As discussed earlier, the interaction between the effluent and the ambient seawater determines which ecological entities may be potentially exposed to the contaminants from the effluent plume. Considering variation in the characteristics of produced water and ambient water to which produced water is discharged, selection of an endpoint is site specific. Typically, effects on survival and growth of pelagic (e.g. fish) and benthic (e.g. scallop) species are considered to be an appropriate assessment endpoint. This is because of their ecological and societal importance and their susceptibility, and because of availability of data on those endpoints reported from laboratory experiments (U.S. EPA 1993). The ecological significance is due to the fact that much of the energy flow passes through these species; the societal importance comes from economic (e.g. fishery) activities. Pelagic and benthic species are sensitive to a variety of contaminants contained in produced water as reported by Neff (1997).

Assessment endpoints are explicit expressions of the actual environmental value that is to be protected. Measurement endpoints have to be defined to enable estimation of changes in the assessment endpoints. Measurement endpoints are thus measurable responses to stressors that can be correlated with the assessment endpoints. Typically, they can be a lethal concentration of 50% of the species (LC_{50}), or a No-observed effect concentration (NOEC).

(iii) <u>Conceptual models</u>

A conceptual model in the problem formulation is to identify relationships between ecological entities and stressors. The major emphasis is the development of a series of hypotheses regarding how produced water might affect exposed ecosystems. Under the conceptual model, a wide range of hypotheses about the effects of produced water on a marine ecosystem could be considered, including interactions with abiotic environment and impacts on ecosystem structure and function. Which hypothesis needs to be evaluated during the discharge design may depend on specific problems under investigation. Typical hypotheses which can be considered in the assessment might be that "produced water may cause adverse effects on survival and growth of fish and shrimp species. If exposures are long, and the periods between exposures are short enough, a significant number of species may be killed." These hypotheses can be tested during the analysis by assessing exposures and effects based on laboratory or field data, or modeling, as discussed in the following subsections.

(iv) <u>Analysis plans</u>

An analysis plan includes a delineation of the assessment design, data needs, measures, and methods for conducting the analysis phases of the risk assessment. It can be viewed as an assessment checkpoint to ensure that the analyses will provide information useable for decision making. When ecological risk assessment is performed during design of produced water discharges, the interest is not only on the quantification of potential ecological risks, but also on comparative evaluation of different design scenarions. During design, actual information relevant to the case under assessment is usually limited, or even not available. For example, no information is known about the quantity and quality of produced water during design of the facility, until it is actually produced in the field. Therefore, assumptions or methods of obtaining such information need to be carefully considered. Typically the information can be obtained from sites that are assumed to have similar characteristics to those of the case under consideration.

The analysis plan also includes the analytical methods planned and the nature of the risk characterization options and considerations to be generated, e.g., quotients, narrative discussion, stressor-response curve with probabilities. In the design stage, a quantitative expression of risk is preferable as it is easier to compare among different design alternatives in terms of ecological risks associated with such designs.

5.3.2. Analysis phase

The analysis phase covers the two primary components of risk assessment: characterization of exposures and characterization of effects. The analysis connects problem formulation with risk characterization. The assessment endpoints and conceptual models developed during problem formulation provide the focus and structure for the analysis. Uncertainty is considered throughout the analysis phase. The objective is to describe, and where possible quantify, the knowns and unknowns about exposures and effects.

(i) Characterization of exposures

Characterizing exposure describes the potential or actual contact of stressors with endpoint biota. It is based on the measures of exposure and the ecosystem, and also on characteristics of the endpoints. It analyzes sources of pollution, distribution of contaminants, and modes of contact between stressors and endpoints. In this stage, the focus is directed at the identification of pollutant sources, the exposure pathway, and the intensity and distribution of stressors.

Chemical contaminants may come into a marine environment from many different sources, including produced water, sewage, drilling mud and so on. However, assessing potential ecological risk associated with a scenario of produced water discharge may focus on a single type of source, i.e. the produced water outfall itself. Produced water discharged from the outfall may consist of formation water, injected water, and any chemical added downhole or during the oil/water separation processes. Typically, the source of the discharge can be associated with well and deck drainage-based effluent as shown schematically in Figures 5.2 (modified from U.S. EPA 1993). Table 5.5 shows typical flow rates of produced water from oil production platforms. The table indicates that the flow rate of produced water is substantial with a water-to-oil ratio ranging from 0.1 to more than 12.

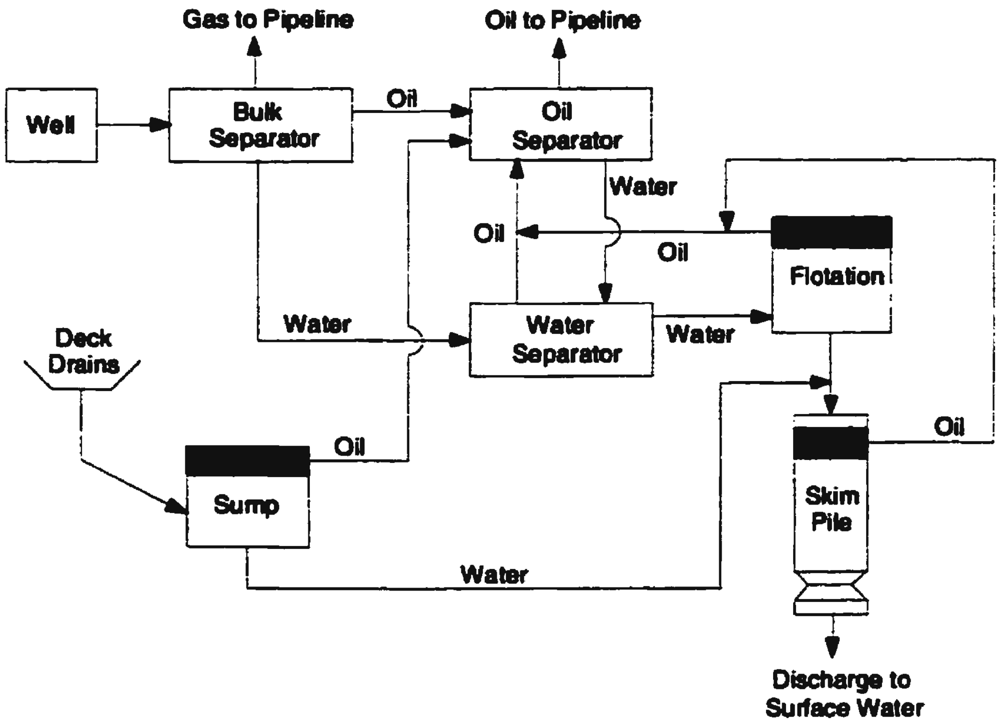


Figure 5.2. Typical produced water treatment system

The second objective in the characterization of exposure is to describe the exposure pathway and thus the contact between stressor and receptor. Stressor distribution in the environment is examined by evaluating pathways from sources. Ecological entities in the water column may be most affected by the effluent plume when the interaction between the effluent and the ambient seawater results in a positive buoyant plume. On the other hand, a negative buoyant plume may pose higher risk to biota living at the sediment. For shallow discharges, both pelagic (water column) and benthic (sediment) community might be exposed at a comparable intensity. In the case of deep and stratified density of ambient water, the effluent plume may be trapped at a water depth and animals living at this depth may be exposed significantly.

Table 5.5. Produced water in oil and gas production (data from U.S. EPA 1993)

No	Company	Oil or condensate (bbl/day)	Gas (MMCF/day)	Produced water (bbl/day)	Water to Oil ratio*
1	Conoco	76.6	15.2	62.0	0.8
2	Mobil	807.0	13.1	2005.0	2.5
3	Conoco	890.0	3.4	2817.0	3.2
4	Shell	950.0	14.0	1298.0	1.4
5	Gulf	228.0	13.8	495.0	2.2
6	Shell	395.0	38.0	634.0	1.6
7	Exxon	250.0	0.2	625.0	2.5
8	Marathon	1200.0	150.0	500 - 2000	0.42 - 1.67
9	Shell	750.0	45.0	1200.0	1.6
10	Mobil	3500.0	5.0	2000.0	0.6
11	Shell	21500.0	63.0	9733.0	0.5
12	Conoco	1501.0	0.2	350.0	0.2
13	Shell	2000.0	30.0	22000.0	11.0
14	Gulf	40.0	6.0	2.0	0.1
15	Placid	1500.0	100.0	1470.0	1.0
16	Chevron	501.0	1.2	4610.0	9.2
17	Chevron	2875.0	5.0	12500.0	4.3
18	Amoco	3000.0	7.0	800 - 1000	0.27 - 0.33
19	Gulf	2800.0	10.0	1072.0	0.4
20	Shell	10794.0	11.7	6590.0	0.6
21	Texaco	873.0	2.8	11028.0	12.6
22	Gulf	6000.0	18.0	8400.0	1.4
23	Amoco	2244.0	10.7	15000.0	6.7
24	Conoco	745.0	2.3	1578.0	2.1
25	Conoco	5273.0	15.5	10721.0	2.0
26	Texaco	554.0	0.1	3796.0	6.9
27	Shell	2091.0	12.1	7532	3.6
28	Shell	1800.0	1.3	3100.0	1.7
29	Shell	24000.0	40.0	150000.0	6.3
30	Shell	5000.0	8.0	3000.0	0.6

*The water-to-oil ratio has a mean, median, minimum and maximum of 3.5, 1.7, 0.1 and 12.6, respectively. This suggests that the rate of the produced water is generally very significant.

In general, pelagic fish are exposed primarily to contaminants in water, whereas benthic organisms are exposed to those in water and sediments (i.e. pore water in the sediments). Those benthic organisms that live on rocks and organic debris are primarily exposed to contaminants in water. Total concentrations may be used as conservative estimates of the exposure concentration (Efroymson et al. 1996). Alternatively, it is typically assumed that aquatic biota are exposed to the dissolved fraction of the chemicals in water because that is the bioavailable form. A leaching factor (LF) is usually used to convert concentration of a chemical from total concentrations into dissolved fractions (e.g. Meinhold et al. 1996a, U.S. EPA 1999b).

Contact between contaminant and ecological entities may be quantified as the amount of the chemical ingested, inhaled, or material applied to the skin. Some stressors must not only be contacted but also must be internally adsorbed to be able to result in effects. In that case uptake is evaluated by considering the amount of stressor internally adsorbed by an organism. For aquatic systems, organisms are continuously exposed to dissolved contaminants in the water column (CCME 1997). Therefore, in its simplest form, contact may be quatified as an environmental concentraton, assuming that contaminants are well mixed or that organisms move randomly through the medium (U.S. EPA 1998). In the absence of complete knowledge about the contact, the approach (U.S. EPA 1998) may be employed for a conservative assessment.

The third objective of exposure analysis is to describe the distribution of stressors in the environment. Ecosystem characteristics influence the transport of all types of stressors; the challenge is to determine the particular aspects of the ecosystem that are most important. In the marine environment water moves very rapidly and it is therefore likely to be more variable in time than in space. Efroymson et al. (1996) suggest that the mean water concentration in a sub-region is an appropriate estimate of chronic exposures experienced by fishes, and the upper 95% confidence limit on the mean concentration is an appropriately conservative estimate of this exposure. Unlike water, sediment is likely to be more variable in space than in time due to its relative immobility. The organisms living at sediments are also relatively immobile, and it is therefore more appropriate to use the median sediment concentration as a central tendency of the contaminant data than averaging their exposures to sediment over space or time (Efroymson et al. 1996). Furthermore, Efroymson et al. (1996) suggest that an appropriate conservative estimate of this exposure is the maximum concentration.

The final product of exposure analysis is an exposure profile, which can be combined with effect assessment to characterize ecological risk. A typical exposure profile may be in terms of distribution of effluent concentration at particular organism habitats, following discharge from a produced water outfall. The analysis should take impact of uncertainty on exposure estimates into consideration, for example using methods described in Chapter Four. In general, the distribution of contaminants may be assessed by means of field monitoring or modeling, or a combination of the two. Models are very important if a quantitative relationship between sources and stressors is desired. In the case of design of the discharge, the modeling approach is the only means possible to estimate the distribution of contaminants. Hydrodynamic modeling to estimate the distribution of the contaminant concentrations is required in this step of ecological risk assessment. Chapters Three and Four provide a methodology for modeling the concentration distribution using deterministic and probabilistic approaches. This study uses this methodology to characterize the contaminant distributions.

(ii) <u>Characterization of ecological effects</u>

Characterization of ecological effects includes describing the effects elicited by a stressor(s), liking the effects to the assessment endpoints, and evaluating how they change with varying stressor level. In general, ecological effects of produced water may be categorized as acute and chronic effects. Acute means a stimulus severe enough to rapidly induce an effect usually measured in terms of lethality. In aquatic systems, an effect observed in 96 hours or less is considered acute (U.S. EPA 1991). On the other hand, a chronic effect or so-called "*long-term effect*" is defined as a stimulus that is lingering or continues for a long time, often one-tenth of the life span or more. It depends on the life cycle of the species. Chronic effects include growth, reduced reproduction, etc., in addition to lethality (U.S. EPA 1991).

Many investigators, e.g. Frost et al. (1998), have reviewed various studies on ecological effects of produced water. These studies show that there are field variations in the toxicity of produced water. However, in general, ecological effects can be associated with the distance from the outfall discharge points. Osenberg et al. (1992) evaluated infaunal density at different distances from produced water outfall by surveying infauna at a total of 20 sites along a gradient upcoast and downcoast of the outfall. In Osenberg et al. (1992), organisms were picked from the sediments, counted and identified according to broad taxonomic categories. There was evidence that infaunal densities were strongly associated with distance from the outfall. They also found that mussels near the produced water outfall tended to grow more slowly than those far from the outfall. Tissue production for mussel was also correlated with the distance from the outfall. Sites farthest from the outfall had production about two to three times greater than those near the outfall.

Ecological effects may be measured at individual level, e.g. growth of species, or at population level, e.g. population density. Frost et al. (1998) summarize responses and effects at different level of the ecosystem as shown in Table 5.6. Measures of effects are required to define their state of changes associated with the discharge. As data gaps between the assessment endpoints and measures of effects are usually encountered due to limited resources or a practical means to acquire more data, extrapolations may be the only way to bridge the gaps (U.S. EPA 1998). Extrapolation may be between taxa (e.g. among different kinds of shrimp), between responses (e.g. mortality to reproduction), from laboratory to field, between geographic areas, and from data collected over a short time frame to longer-term effects (U.S. EPA 1998).

The CCME (1997) provides examples of the extrapolation, such as the earthworm test, which represents soil invertebrates and the rainbow trout, which represents freshwater fish. Following the CCME (1997) and U.S. EPA (1998) approach of extrapolation, and taking survival and growth of fish and shrimp as typical assessment endpoints for produced water discharge in the marine environment, toxicity information that is available for the

Sheepshead minnow, *Cyprinodon variegatus*, and the Mysid shrimp, *Mysidopsis bahia*, may be used as measures of effects. Measurement endpoints may then be defined by, for example, LC_{50} or NOEC of these species. Table 5.7 shows typical results of toxicity tests associated with produced water.

Table 5.6. Responses and effects at next level at different levels of the ecosystem (after Frost et al. 1998)

Level	Types of response	Effects at next level
Biochemical level	Impairment of metabolic pathways Detoxification	Disruption in energetics Reduction in energy stores Adaptation of organism
Organism	Metabolic changes Behavioral changes Increased incidence of disease Reduction in growth and reproduction Adjustments in rate functions Disease defence	Reduction in performance of populations
Population	Changes in population dynamics Adaptations of populations to stress	Effects on coexisting organisms and community
Community	Changes in species composition Reduced energy flow Ecosystem adaptation	Deterioration of community Reduced secondary production No change in community stability

As chemical composition of the produced water is different from case to case, there is concern if toxicity tests from one site might be applicable to another site. It might be applicable if it is assumed that the produced water from the two sites have similar toxicity characteristics. A typical study on toxicity evaluation from different platforms with various discharge rates and sampling times (Moffitt et al. 1992) found that no significant differences were observed between results from samples collected at different time periods or from different offshore platforms with varied discharges rates or between any combinations. A typical compilation of toxicity information associated with 96-hour LC_{50} of mysid shrimp is shown in Table 5.8.

Table 5.7. Toxicity of produced water from different platforms at Gulf of Mexico (concentration in % effluent, data from Moffitt et al. 1992)

Source of produced water sample	96-hr. LC_{50}		7-day LC_{50}		Survival NOEC		Growth NOEC	
	Mb[1]	Cv[2]	Mb[1]	Cv[2]	Mb[1]	Cv[2]	Mb[1]	Cv[2]
Platform A, sample # 1	8.27	27.3	7.29	>28	3	14	3	7
sample # 2	4.66	25.7	4.48	21.7	1.5	14	1.5	7
sample # 3	7.2	15.1	7.47	18.1	3	14	1.5	1.5
Platform B, sample # 1	9.87	>28	8.72	>28	3	28	1.5	1.5
sample # 2	3.95	>28	2.8	>28	1.5	14	1.5	3
sample # 3	5.65	N/A	4.38	N/A	3	N/A	0.75	N/A
Platform C, sample # 1	5.58	9.61	5.92	9.61	3	7	<0.75	<0.75
sample # 2	2.9	18.14	2.55	17.56	1.5	14	0.75	1.5
sample # 3	2.78	5.44	2.81	4.49	1.5	1.5	<0.75	1.5
Platform D, sample # 1	6.18	>28	4.45	7.72	3	7	3	3
sample # 2	6.34	>28	7.37	>28	3	28	1.5	<0.75
sample # 3	3.15	>28	2.76	>28	1.5	7	<0.75	1.5
Platform E, sample # 1	4.76	11.3	6.21	3.68	1.5	1.5	1.5	0.75
sample # 2	7.12	25.5	4.49	19.43	3	7	<0.75	<0.75
sample # 3	3.88	>28	3	24.77	1.5	14	<0.75	3
Platform F, sample # 1	9.9	16.6	4.82	14.58	7	7	<0.75	3
sample # 2	6.15	13.3	6.97	15.95	3	7	<0.75	1.5
sample # 3	9.61	20.6	9.61	17.46	7	14	1.5	1.5
Platform G, sample # 1	7.47	19.8	6.78	19.8	3	14	3	7
sample # 2	16.6	23	10.05	21.84	3	7	0.75	3
sample # 3	16	>28	5.22	>28	3	14	3	7
Platform H, sample # 1	7.65	17.6	9.03	18.88	7	14	7	<0.75
sample # 2	14	>28	9.16	>28	7	7	0.75	3
sample # 3	2.3	23	2.05	21.96	0.75	14	0.75	3

Notes:
[1]Mb: Mysid shrimp, *Mysidopsis bahia*
[2]Cv: Sheephead minnow, *Cyprinodon variegatus*

Table 5.8. Compilation of toxicity of produced water on Mysid Shrimp (units in % effluent)

No	Region	Facility/Platform/Notes	96-h LC_{50}	Refferences
1	Cook Inlet, Alaska[1]	Amoco Platform Bruce	0.91	U.S. EPA (1996b)
		Amoco Platform Baker	23.99	
		Marathon Granite Point	13.5	
		Marathon Trading Bay	18	
		Philips Platform A	63.69	
		Shell East Forelands	21.66	
2	Gulf of Mexico	Platform # 1	15.8	Brown et al. (1992)
		Platform # 2	4.8	
3	Mississippi Delta, U.S.A. (1988-1989)	Platform # 1	4.9	Schiff et al. (1992)
		Platform # 2	11.8	
		Platform # 3	11.4	
4	Gulf of Mexico[2]	Platform A, sample # 1	8.27	Moffitt et al. (1992)
		sample # 2	4.66	
		sample # 3	7.2	
		Platform B, sample # 1	9.87	
		sample # 2	3.95	
		sample # 3	5.65	
		Platform C, sample # 1	5.58	
		sample # 2	2.9	
		sample # 3	2.78	
		Platform D, sample # 1	6.18	
		sample # 2	6.34	
		sample # 3	3.15	
		Platform E, sample # 1	4.76	
		sample # 2	7.12	
		sample # 3	3.88	
		Platform F, sample # 1	9.9	
		sample # 2	6.15	
		sample # 3	9.61	
		Platform G, sample # 1	7.47	
		sample # 2	16.6	
		sample # 3	16	
		Platform H, sample # 1	7.65	
		sample # 2	>14	
		sample # 3	2.3	
5	Louisiana Coastal, U.S.A.[3]	Summary of 222 outfalls study (mean, median)	(12.4; 8.24)	U.S. EPA (1996b)
6	Louisiana State[4]	Summary of 400 samples (mean, standard deviation)	(10.05; 10.36)	Moffitt et al. (1992)
7	Texas Coastal, U.S.A.[5]	Brown shrimp	37	U.S. EPA (1996b)
		White shrimp	36	

Note: [1]Mean of two seasonal LC_{50} values; [2]Single grab samples were collected three times from each platform, with sampling times separated by approximately two weeks; [3,4]Raw data was not available at the time of this study; [3,4,5]This information is included in this table for visual comparison purposes.

Characterization of effects can also be more detailed by presenting the entire relationship between concentration and one or more responses. In that way, a broad range of effect magnitudes, e.g. LC_{10}, LC_{25}, LC_{50}, LC_{75} etc., is related to different levels of stressor concentrations. Figure 5.3 shows a typical dose-response relationship for toxicity of produced water on Mysid shrimp and Sheepshead minnow. Beside whole effluent-based effects, chemical specific effects may also be considered (Middleditch 1984, Neff 1997). As shown in Table 5.4 above, sublethal effects of several metals associated with produced waters are observed.

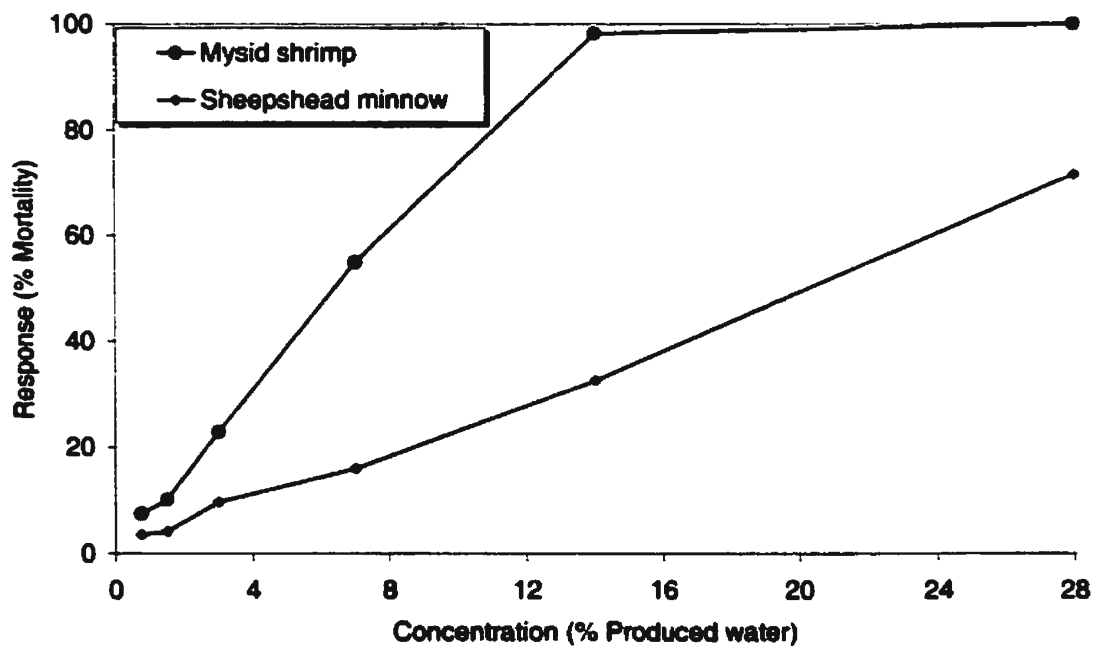


Figure 5.3. Typical dose-response relationship of produced water toxicity (data from Moffitt et al. 1992)

Characterization of effects is to some extent subject to uncertainty because of the difficulty in obtaining complete information required in the analysis of effects. For example, modifying factors, which are any characteristics of an organism or the surrounding water affecting toxicity, can contribute to uncertainty by either increasing or decreasing the concentration of a contaminant required to produce a biological response or effect. Another source of uncertainty is the extrapolation in modeling discussed previously. Models have been developed for extrapolating among taxa, endpoints, and laboratory and field data with some degrees of uncertainty (CCME 1997). Use of the models is the only way possible for conducting ERA, particularly in predictive risk analysis like in this study.

5.3.3. Risk characterization

Risk characterization is the final step in ERA and is the combination of problem formulation and analysis of estimated adverse effects associated with assessment endpoints. Risk characterization clarifies the relationships between the stressors (i.e. produced water or associated contaminants) and effects on endpoints to reach the conclusions (i.e. estimated magnitude and probability of the effects). It combines the results of characterization of exposure, which estimates the concentrations of contaminants in the environment, and characterization of effects, which estimates the effects associated with various concentrations. The risk estimate in the context of the significance of adverse effects is described, and uncertainities and assumptions in the risk assessment are discussed. The conclusions explained in the risk characterization should provide information for environmental decision making (CCME 1996b, U.S. EPA 1998).

(i) Methods of Characterizing risk

Ecological risks may be described qualitatively or quantitatively. Qualitative methods do not quantify the magnitude or probability of effects, and in many cases, depend on professional judgement. Qualitative methods are usually used as a preliminary means of identifying problems of concern (CCME 1996b). The CCME (1996b) provides examples of methods in use. In a produced water study, a qualitative method was used by Neff and Sauer (1996) to study ecological risks associated with polycyclic aromatic hydrocarbon (PAH) by comparing concentrations of individual and total PAHs in ambient water, sediment, and whole tissue of marine animals to published concentrations in these media equivalent to the highest no observable effects concentrations, or threshold concentrations.

Since the magnitude or probability of effects are not quantified in the qualitative approach, it is not readily applicable to engineering design for providing relative merits of different design scenarios of a produced water outfall. In this case, a quantitative approach is required. Basically, there are two methods of quantitative approach: quotient methods and continuous exposure-response methods (CCME 1996b; U.S. EPA 1998). Quotient methods require input of benchmark concentration (BC) and exposuse concentration (EC), and may be expressed as:

$$Quotient = \frac{EC}{BC} \tag{5.2}$$

The quotient method identifies the presence of potential risk by, for example, defining a quotient less than one to indicate low or extremely low risk or probability of effects and a quotient equal to or more than one to indicate potential risk or effects. There are several types of quotient methods. The first, simplest, type is a deterministic quotient method, which charaterizes relative risks by comparing point estimates of EC (e.g. mean concentration) to point estimates of BC (e.g. NOEC). This method has been adopted for produced water discharges, e.g., the CHARM (Thatcher et al. 1999) and the PROVANN (Reed et al. 1996) models. Being based on the deterministic method, uncertainty associated with ecological risks - or in this case hazard quotients – is left unevaluated. Furthermore, Thatcher et al. (1999) acknowledged the uncertainty in the CHARM model, but it is vaguely addressed by simply dividing and multiplying the calculated hazard or risk quotient by 3 to define the lower and upper 90% confidence levels, respectively.

The second type of quotient method is a probabilistic approach, in which uncertainty is evaluated in the analysis. Figure 5.4 shows degrees of quantification of uncertainty for quotient risk characterization methods (CCME 1999b, U.S. EPA 1998). As seen in this figure, the analysis may consider uncertainty by defining probability distributions in BC or EC or both (CCME 1996b).

Another version of the quotient method has also been used in other fields of ERA by using different values of BCs associated with species representing the community under assessment. Lenwood et al. (1998) assessed ecological risks associated with metal contamination in the surface waters of the Chesapeake Bay Watershed. They compared the probability distributions of environmental exposure concentrations with probability distributions of species response data determined from laboratory studies. The objective of the analysis was to protect at least 90% of the species 90% of the time. They repeated this exercise for both chronic and acute data separately. Risk was defined on the basis of exceedence of the 90^{th} percentile of exposure to the lowest 10^{th} percentile of response value (to protect 90% of the organisms 90% of the time). The U.S. EPA (1998) provides an illustration of this approach. This approach can be shown graphically in Figures 5.5.

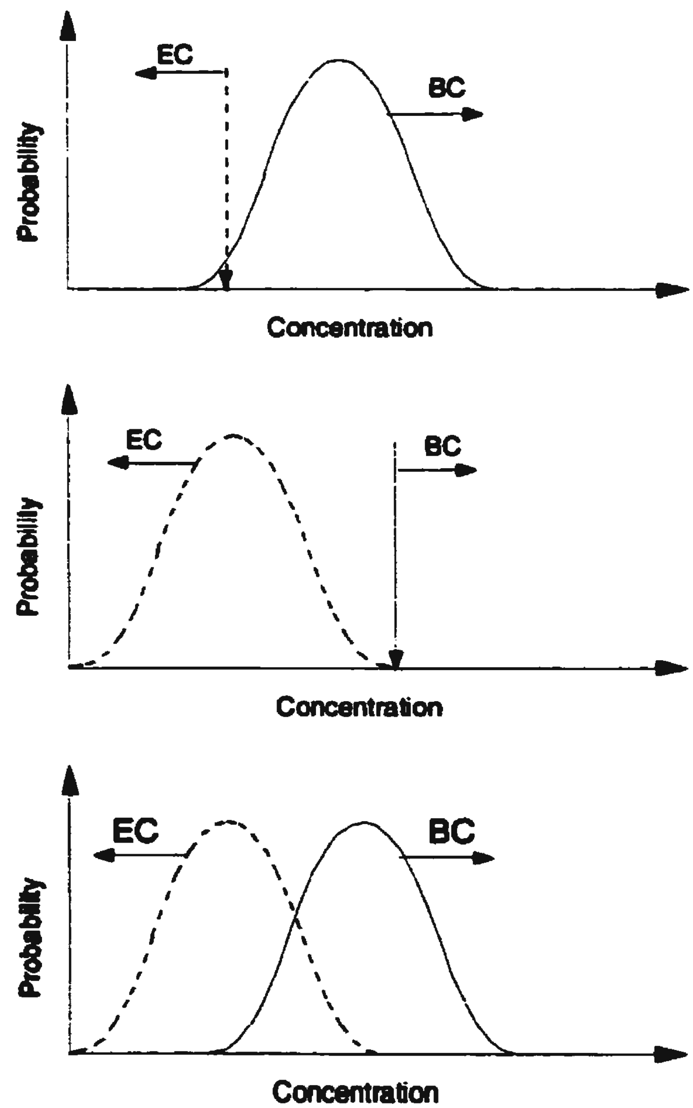


Figure 5.4. Degrees of quantification of uncertainty in risk characterization (Curves show probability distribution and straigt errows show point estimate)

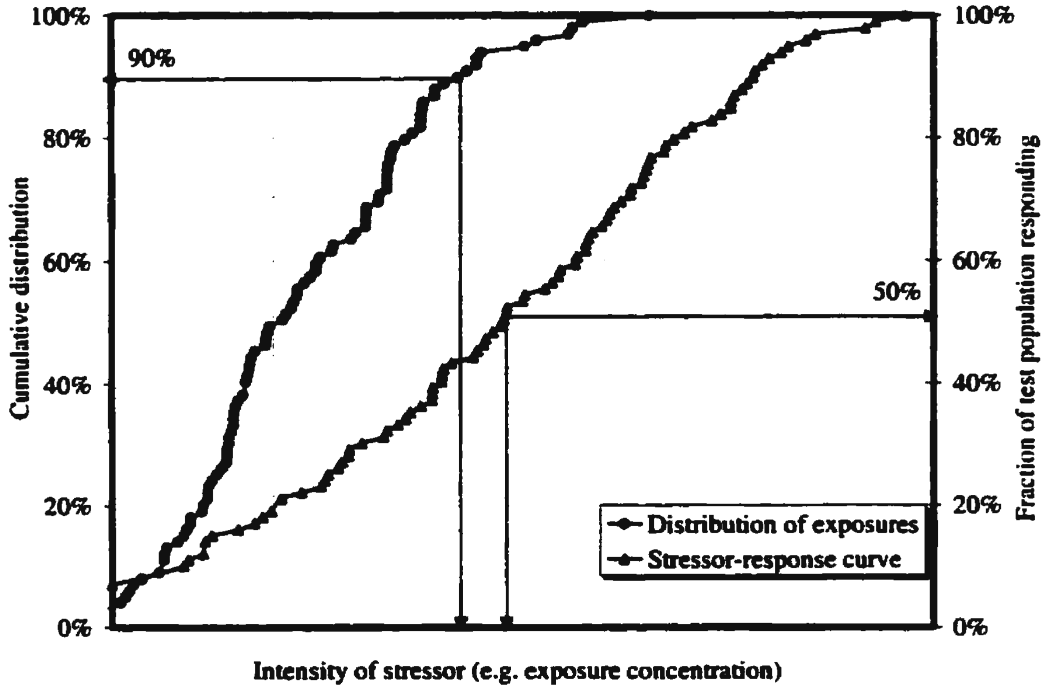


Figure 5.5. Typical risk estimation technique relating stressor-response curve with a cumulative distribution of exposure (comparison of 90th percentile exposure with EC_{50})

The second type of quantitative approach is continuous exposure-response methods. Unlike quotient methods, they do not rely on single BC, e.g. EC_{50}, but use the entire relationship between concentration and one or more responses (CCME 1996b). Thus, a broad range of effect magnitudes, e.g. EC_{10}, EC_{25}, $EC_{50,}$ EC_{75} etc., is considered in characterizing risk. The continuous exposure-response methods are particularly useful when the risk assessment outcome is not based on exceedance of a predetermined decision rule like a toxicity bechmark level (U.S. EPA 1998). Comparing a stressor-response curve with an exposure distribution (Figure 5.5) can increase the capability of estimating changes in the magnitude and likelihood of effects for different exposure scenarios (U.S. EPA 1998).

Many choices of approaches and methods are available to characterize ecological risks. For the case of produced water discharge, particularly in design of an outfall, a quantitative approach is required for doing ERA. The complexity of the methods that can be employed depends on the availability of the required data. The information compiled in this study indicates that both methods of quantitative approach, i.e. quotient methods and continuous exposure-response methods, may be employed depending on the details of the assessment required. An example of application of these methods is presented in Chapter Six.

(ii) Dealing with effluent containing a mixture of chemicals

Another issue that is relevant for discussion is how to deal with effluent consisting of multiple chemicals or mixtures. This is particularly important because produced water contains various chemicals as shown previously in Table 5.2, making the characterization of effects very complex. Each chemical in produced water might be associated with different degrees of effects. Developing models addressing multiple chemicals is theoretically possible but might be technically difficult in practice (CCME 1997). In this situation, there are two approaches to characterizing ecological effects: chemical specific and whole effluent toxicity approaches (U.S. EPA 1991). In the chemical specific toxicity approach, each chemical component is evaluated based on its dose-response relationships. The whole effluent toxicity approach considers the effluent as "one entity" that has a specific dose-response relationship. Evaluating the potential toxicity of the effluent does not necessarily evaluate all chemicals contained in the produced water.

Each of the approaches has its own advantages and limitations. Conducting a chemical specific study is sound in terms of identifying cause-and-effect relationship. However, it is difficult to identify which chemicals contribute more to the toxicity of produced water. Frost et al. (1998) summarize toxicity studies indicating that the major contributors to the acute toxicity in produced water might be associated with the aromatic and phenol fractions. Polycyclic aromatic hydrocarbons (PAHs) may lead to cancer in fish, while alkylated phenols are potential endocrine disrupters. In another case, metal, particularly Zn, was considered relatively important so that produced water containing relatively high Zn was more toxic than that with low Zn (Stromgren et al. 1995). Furthermore, Sauer et al. (1997) argue that for most produced water samples, toxicity to any one fraction represented only part of the toxicity of the whole sample.

This complexity poses difficulty in conducting predictive risk assessment of produced water discharge on the basis of a chemical specific approach. Nevertheless, Neff and Sauer (1996) conducted qualitative ecological risk assessment for produced water by using a specific chemical approach, in which risks associated with individual and total PAHs are studied. Quantitative ERA also commonly employ a chemical-specific approach in risk characterization, and defines total risk by summing up all elemental risk associated with each chemical. The problem in this approach is that it is difficult to be sure how the resultant toxicity may be influenced by the combination of the different chemicals. The overall effluent toxicity could be equal to the sum of each chemical's toxicity (additivity), less than the sum (antagonism), or greater than the sum (synergism).

Unlike the chemical specific approach, the whole effluent toxicity approach does not require assumptions regarding the resultant toxicity because the toxicity tests are conducted in terms of the whole effluent. Water quality associated with wastewater discharge can also be evaluated using the whole effluent toxicity approach. The U.S. EPA (1991) provides examples on the use of water quality standards, which are specified in terms of whole effluent toxicity. For the case of produced water, many toxicity studies are conducted using the whole effluent approach, for example, Brendehaug et al. (1992) and Moffitt et al. (1992). In this approach, results of the toxicity test may be used for further analysis of ERA. Concern in doing risk characterization on the basis of the whole effluent approach arises because the toxicity test in this approach is performed on the effluent before it is discharged, while when discharging it in the ambient water, the effluent composition may change, and individual substances may partition according to their physico-chemical properties (Thatcher et al. 1999).

Because of this, many studies used a chemical-specific basis, e.g. CHARM (Thatcher et al. 1999). However, there is inconsistency in their approach. On one side the preference of using the specific chemical approach is based on an acknowledgment that individual substances may partition according to their physico-chemical properties; however, when calculating concentration of a specific chemical at a distance from the discharge, the composition of effluent is assumed to remain unchanged so that concentration of the chemical can be calculated by dividing the concentration of the chemical before discharge by the dilution factor of the effluent (Thatcher et al. 1999). The same approach of calculating chemical concentration is used to estimate chemical concentrations associated with discharge of drilling fluids (U.S. EPA 1999b). On the basis of this discussion, the presently used chemical-specific approach for analysis of produced water discharge is not more scientifically sound than the whole effluent approach.

Use of a whole effluent approach may be possible in ERA of produced water, particularly for design of the discharge facility, because it does not require assumptions regarding the resultant toxicity and because many toxicity studies of produced water present their results in terms of whole effluent toxicity (Brendehaug et al. 1992; Moffitt et al. 1992). Another consideration in designing a discharge facility is that chemical composition of produced water is not known and that a large number of chemicals are present in produced water with a great variability in quality and quantity among produced waters from different fields. In this situation, chemical specific analysis is seriously subject to uncertainty.

Similar uncertainty is also faced when doing whole effluent analysis because of variability of toxicity data among produced water from different oil production fields. However, problems associated with variability of such toxicity data as presented in Table 5.8, may be handled by using a probabilistic approach. It is found in this study that toxicity data shown in Table 5.8. was lognormally distributed. This is graphically shown in Figure 5.6. Until a better and more scientifically sound method is available, the specific-chemical and whole effluent approaches may be employed depending on their suitability to the case under consideration, e.g. availability of data.

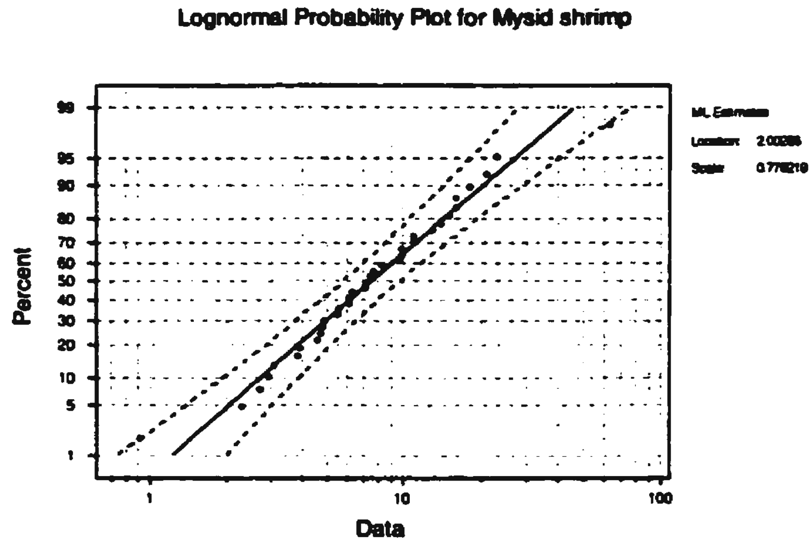


Figure 5.6. Log-normal probability plot of Mysid shrimp LC_{50} (95% CI is also shown)

(iii) Uncertainty Analysis

In general, there are several sources of uncertainty, including inherent variability, parameter uncertainty and model errors (CCME 1997). Inherent variability may be associated with the natural variability such as variability in ambient water characteristics possibly affecting different biological responses for a given discharge of produced waters. Parameter uncertainty may be associated with estimation of parameters such as chronic benchmark concentration from LC_{50}s. Model uncertainty may include uncertainty associated with using a few variables to model many complex phenomena, or using inappropriate boundaries to define the system under investigation, or employing assumptions to simplify the analysis. An example of this is the use of the risk assessment approach by employing an LC_{50} derived from a 96-hour laboratory test using constant exposure levels, which may not be the most appropriate for an assessment of effects on reproduction resulting from short-term, pulse exposures (U.S. EPA 1998).

The relative importance of these sources of uncertainty may vary among cases. Inherent variability may be the most important source of uncertainty for retrospective and empirical ERA, whereas parameter uncertainty may be a more important source for predictive and theoretical ERA (CCME 1997). Approaches to dealing with uncertainty have been discussed in Chapter Four. The U.S. EPA (1996a) provides guidance for use of Monte Carlo (MC) simulations for risk assessment particularly relevant to human health risk assessment. Mukhtasor et al. (1999a, 2001b) use MC simulations for dealing with uncertainty in ocean outfall design and analysis. Use of MC simulations in ecological risk assessment associated with soot deposition in the marine environment is shown in Mukhtasor et al. (2000a). This method may be employed in ecological risk assessment of produced water discharges, particularly in discharge design as shown in Chapter Six.

5.4. Summary

This chapter reviews current approaches used for ecological risk assessment of produced water discharges. Problems associated with presently used approaches are discussed. Substantial efforts have been devoted in the past for assessing ecological risks of produced water discharges, and several models are now available for that purpose. The efforts were however usually directed at monitoring purposes, making no consideration for the integration between ERA and engineering design of the produced water outfalls. In the presently used approaches, the endpoint of the assessment is not well defined; and uncertainty associated with the assessment is evaluated only vaguely, or not at all. Approaches are identified to deal with specific problems relevant to design of produced water discharges in a marine environment. The approaches are adapted from the literature and consist of three phases of ERA, i.e. problem formulation, analysis and risk characterization. Uncertainty associated with each phase is also identified. Discussion of the approaches is directed at how to adapt the state-of-art of ERA to cope with specific problems in produced water discharges, particularly in design of ocean outfalls.

Chapter 6

A Framework and Case Study for Ecological Risk-based Design of a Produced Water Outfall

6.1. Introduction

Ocean outfall design is a very small subset of the engineering designs necessary to make the world a more environmentally safe place to live in. The main purpose of ocean outfall design is to optimize the mixing process so that the wastewater effluent, i.e. produced water, is reduced to a level that is acceptable to the environment by utilizing natural processes which are available in the ocean to dilute, disperse and assimilate the wastes. A great deal of work must be conducted to properly design an ocean outfall system. The work lies on a range from economic and ecological studies to technical evaluations, including the selection of construction methods, the design parameters, the effluent dilution calculations and the evaluation of potential environmental effects.

One of the most important tasks in the design is to mitigate any harmful local ecological effects and to anticipate the global large-scale degradation and transformation processes associated with the effluent discharge under a particular design scenario. This chapter provides a framework for the design on the basis of potential ecological risks. The methodology of hydrodynamic modeling and ecological risk assessment discussed in the previous chapters is integrated here in the context of the design. The applicability of the approach is presented by evaluating scenarios of produced water discharge relevant to an offshore oil production platform located on the Grand Banks, southeast of St. John's, Newfoundland, Canada. Instead of offering a solution for a particular problem of an existing oil field, the emphasis of the case study is to show how a risk-based engineering design could be potentially undertaken.

6.2. Relevance

Risk-based design of ocean outfalls has been a subject of international discussions, e.g. Mukhtasor (2000), Mukhtasor et al. (2000b, 2000c, 2001c), Mukhtasor and Husain (1999). The primary consideration in such a design is to ensure that the outfall effluent is well assimilated in the ocean by maintaining the assimilative capacity of the ocean. Referring to Goldberg, Wolfe (1988) defined the assimilative capacity as "*a concept for waste management in which the waste inputs to an environment are balanced against natural environmental processes of dilution, dispersion, and degradation to maintain the potentially adverse environmental impacts within acceptable bounds*." Thus, the assimilative capacity of the ocean reflects the extent to which the ocean can receive wastes discharged from the outfall without unacceptable impacts such as extremes in oxygen concentration deficit, aesthetic impacts and ecotoxicological problems.

Outfall design has traditionally been undertaken on a deterministic basis where worst, normal, and best condition scenarios would be analyzed to ensure compliance with regulations under all operating conditions. However, many environmental standards (guidelines or criteria) include probabilistic elements and that has spurred an interest in probabilistic design (Christoulas and Andreadakis 1994, Huang et al. 1996, Mukhtasor et al. 1999a). The standards applicable to the discharge are generally set in two ways: based on the quality of the discharge (end-of-pipe approach), or based on the quality of the ambient water. The end-of-pipe approach is applied by specifying the physical and chemical quality of the effluent. The second approach is usually applied using the concept of a mixing zone, that is an "allocated impact zone" where numeric water quality criteria can be exceeded as long as toxic conditions are prevented (Doneker and Jirka 1990).

The end-of-pipe approach is commonly expressed in terms of oil and grease concentrations in the effluent to be discharged. Canadian guidelines (National Energy Board et al. 1996) specify that produced water should be treated to reduce the concentrations of dispersed oil to 40 mg/L or less, as averaged over a 30 day period. Similar guidelines are applied in oil industries worldwide, but the level of concentration and frequency of monitoring may be different from place to place, depending on the local regulatory bodies. The Norwegian sector of the North Sea specifies the oil and grease concentration at 42 mg/L (Ray 1996). The maximum permissible concentration of oil and grease in produced water discharged in Australian marine water is in the range of 40 to 50 mg/L (Neff and Sauer 1996). The United States Environmental Protection Agency (U.S. EPA 1993) provides guidelines limiting oil and grease to a 29 mg/L monthly average and a 42 mg/L daily maximum. Oil and grease are used as contaminant measures partly because they serve as indicators for toxic pollutants in the effluent, including phenol, naphthalene, ethylbenzene, and toluene, and partly because it is not technically feasible to control these toxic pollutants (U.S. EPA 1993).

The standards for the second type of approach specify the quality of ambient water being protected. In this approach, the critical condition of ambient water is specified, and the oil industries are required to study or monitor prior to and during the production to ensure that the effluent discharge is in compliance with the standards. The marine water quality standards vary in terms of "environmental protective measures" and acceptable levels of measures. Table 6.1 shows the variation in ambient water quality standards from several countries. As shown in Table 6.1, Australian and New Zealand guidelines (ANZECC and ARMCANZ 1999) use measures associated with toxic chemicals expressed in terms of a protection level of (a,b%). The protection level (a,b%) is the concentration of chemical that should not be exceeded in order to protect a% species with b% confidence. In a slightly different way, the U.S. EPA (1999a) uses measures of the criterion maximum concentration (CMC), the criterion continuous concentration (CCC), and the human health criterion (HHC). CMC is an estimate of the highest concentration of material in the surface water to which an aquatic community can be exposed briefly without resulting in an unacceptable effect. CCC is an estimate of the highest concentration of material in surface water to which an aquatic community can be exposed indefinitely without resulting in an unacceptable effect. HHC is based on an individual life-time cancer risk of one in one million following consumption of organisms from the polluted water. Relatively simple guidelines are found in Canadian water quality guidelines (CCME 1999), in which a single threshold concentration is given for each of a number of specified toxic chemicals.

Table 6.1. Ambient water quality standards (μg/l) from different countries applicable to several chemicals often found in produced waters (ANZECC and ARMCANZ 1999, CCME 1999, U.S. EPA 1999a)

Parameter	Australia and New Zealand		Canada	United States		
	95,50%	90,50%		CMC	CCC	HHC
As	-	-	12.5	69	36	0.14
Cd	5	12	0.12	42	9.3	-
Cr	10	25	1.5	1,100	50	-
Cu	1.3	3	-	4.8	3.1	-
Hg	0.1	0.4	-	1.8	0.94	0.94
Ni	190	380	-	74	8.2	4,600
Pb	-	-	-	210	8.1	
Zn	10	21	-	90	81	69,000
Benzene	-	-	110	-	-	71
Toluene	-	-	215	-	-	200,000
Napthalene	40	60	1.4	-	-	

In addition to the chemical-specific based criteria discussed above, water quality standards are also available in terms of whole effluent toxicity. In this case, they are usually given in terms of toxic units (*TU*'s), namely toxic unit acute and toxic unit chronic. The U.S. EPA (1991) defines the toxic unit acute (TU_a) as the reciprocal of the effluent concentration that causes 50% of the organisms to die by the end of the acute exposure period (i.e. $100/LC_{50}$); and the toxic unit chronic (TU_c) as the reciprocal of the effluent concentration that causes no observable effect on the test organisms by the end of the chronic exposure period (i.e. 100/NOEC). The use of toxic units makes it easy to quantify the toxicity of an effluent and to specify water quality criteria based upon toxicity. For example, an effluent that has 20 TU_c is twice as toxic as that with 10 TU_c (U.S. EPA 1991). Based on the whole effluent toxicity approach, the U.S. EPA (1991) noted that for acute and chronic protection, the CMC and CCC should be set at 0.3 TU_a and 1.0 TU_c, respectively, to the most sensitive of at least three test species (e.g. a fish, an invertebrate, and a plant).

Each of the approaches discussed above has advantages and disadvantages. In the end-of-pipe approach, the operators (dischargers) know exactly where they stand and sampling for compliance is relatively simple. However, the protection of ambient water quality is not explicitly considered in this approach and measures to protect the ambient water into which the effluent is being discharged are missing. By adopting the end-of-pipe approach alone, the protection of ambient water quality becomes the responsibility of the regulatory authority. On the other hand, if criteria or guidelines are set for ambient water, then the responsibility for meeting these criteria or guidelines rests with the dischargers. Nowadays, there are cases where both the approaches are used prior to issuing a permit for produced water discharges (U.S. EPA 1997).

Development of water quality standards is usually based on scientific toxicity data combined with acceptable risks. For example, as discussed above, the HHC assumes that there would likely be a carcinogenicity risk of no more than one in one million with consumption of organisms from contaminated water (U.S. EPA 1991, 1999a). The difference in chemical concentration levels specified by different regulations reflects a willingness to accept different degrees of risks. As a result, a produced water discharge that is in compliance with the standards specified by one regulatory body may not be in compliance with those of others. Under this situation, environmental (ecological or human health) risk assessment provides more direct measures of ecological effects.

In many cases, environmental risk assessment was applied for the purposes of monitoring or mitigating of wastewater discharges, e.g. Meinhold et al. (1996b, 1996c). On the other hand, in engineering design of such a discharge, efforts are conventionally directed at compliance with relevant water quality standards, which, as discussed above, are in turn commonly specified upon an epidemiological and ecological viewpoints. This raises the possibility that the design of produced water outfalls could itself be looked at from the point of view of the ecological risks from exposure to produced water.

In addition, and of more importance in the near future, the approach of ecological risk-based design should be sufficient to provide a guide to the relative merits of different designs from an ecological risk viewpoint. It will therefore permit designers to classify alternative designs (e.g. different geometries and/or different locations) according to the ecological risks, which might arise from their construction, and to determine the degree to which one design is more appropriate than another. In this context, there is a possibility that the outfall design criteria themselves might be changed to reflect an awareness of ecological risks by incorporating engineering principles and ecotoxicological studies. In light of the advances of the methodology in engineering design and ecotoxicological studies, the approach described in this chapter provides a means for the better understanding of ecological risks associated with a potential discharge of produced water under particular designs.

6.3. Framework

The framework of ecological risk-based design is based on the integration of hydrodynamic modeling and ecological risk assessment. The focus of the framework is directed at providing design recommendations on the basis of ecological risk perspectives. The framework consists of six steps, namely:

1. Formulate a problem of ecological risk-based design of produced water discharge;
2. Identify and evaluate preliminary design scenarios;
3. Screen the preliminary design scenarios, and if potentially acceptable scenarios are not identified in the screening, return to step 2;
4. Perform analysis of exposures and ecological effects associated with potentially acceptable scenarios;
5. Characterize ecological risks associated with potentially acceptable scenarios;
6. Provide discussions and design recommendations on the basis of ecological risks.

The first step in this framework is to define a problem of ecological risk-based design of produced water discharge. As characteristics of produced water discharge and ambient seawater are site-specific, the problem formulation may also be site specific. Once the problem has been formulated, steps 2 and 3 are conducted to identify and screen preliminary design scenarios. These two steps rely heavily on principles of hydrodynamic modeling and engineering performance as well as information about ambient water standards or benchmark concentrations of the produced water toxicity. The hydrodynamic modeling is then integrated with ecological risk assessment in steps 4 and 5. The last step is to provide discussions and design recommendations, which are based on descriptions of ecological risks and principles of outfall design. A more detailed description and an example of an application of the framework is provided in the next section.

6.4. Description of the framework and case study

A description of the framework for ecological risk-based design is presented in this section using a case study. The case study is based on information relevant to a potential discharge of produced water from an offshore platform, the Terra Nova FPSO (Floating Production Storage and Offloading) vessel, located at the Grand Banks, Newfoundland, Canada (Petro-Canada 1996). Figure 6.1 shows the location map of the Terra Nova project (modified from MUN 2001 and CNOPB 2001). The FPSO vessel is located about 350 km east-southeast of St. John's, 35 km southeast of Hibernia, a Gravity-based Structure (GBS) oil production platform. The reason for choosing the Terra Nova is that information related to estimates of the potential discharge of produced water were available (e.g. Petro-Canada 1996), and that, based on the development application (Petro-Canada 1996), no similar framework using an ecological risk-based design has been undertaken for the produced water outfall. The primary objective of the analysis is to provide an example of an application of the framework. Thus, this example might not reflect actual problems of the operational oil production platform because assumptions were made when information was not available.

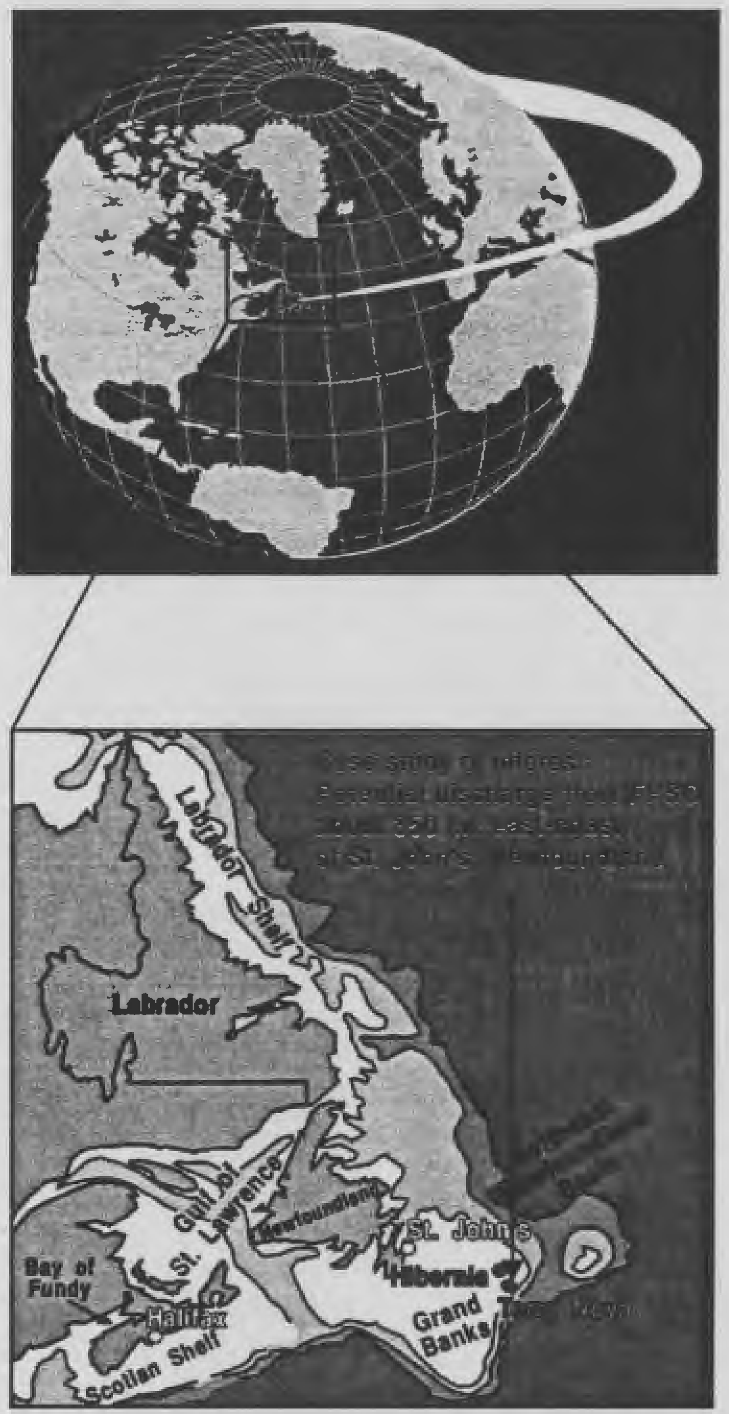


Figure 6.1. Location map of the case study (modified from MUN 2001 and CNOPB 2001)

The produced water discharge under consideration is from the FPSO, which is a ship-shaped vessel with integrated oil storage from which oil will be offloaded onto a shuttle tanker. The vessel is about 292.2 m long and 45.5 m wide. Crude oil storage capacity is about 960,000 barrels (7.5 days storage at peak production). It has a Topside Processing Unit that is designed to produce a maximum of 125,000 barrels per day (BPD) of stabilized crude oil, and to treat and discharge produced water that is generated during the production. In a typical offshore oil production, a module of produced water/glycol takes up a significant space in the FPSO. Table 6.2 provides typical module weights, showing that produced water handling takes considerable attention on the FPSO.

Table 6.2. Module weights in the FPSO (Terra Nova Project 2000)

Modules	Weights (tonnes)
T08 Flare Tower	600
M09 Power Generator	1484
M05 Separation Low Pressure Compression	1790
M04 Produced Water/Glycol	1400
M03 Separation High Pressure Compression	2167
M02 Water Injection	1086

6.4.1 Formulating a problem of ecological risk-based design

Problem formulation for ecological risk assessment (ERA) has been discussed in Section 5.3.1 (Chapter Five). Two primary objectives of the problem formulation in this framework can be highlighted here: (1) defining a problem of ecological risk-based design, and (2) outlining a plan for analyzing the problem. Problem definition is a process of developing hypotheses about what ecological effects might occur if a discharge of produced water is introduced into the marine environment. This includes describing characteristics of the discharge and ambient seawater, and identifying potential risks to ecological resources. A plan for analyzing the problem includes determining methods of conducting the design on the basis of the ecological risks, which will be used in the analysis.

(i) <u>Discharge characteristics</u>

At the Terra Nova FPSO platform, the produced water that is separated from the crude oil during the production process will be passed through a produced water treatment system to reduce its oil content to meet the guidelines (National Energy Board et al. 1996, Petro-Canada 1996). During the life of the field, about 46.5×10^6 m^3 of produced water, which contains a maximum of 1863 m^3 of oil, will be discharged. The Floating Production Facility will be designed to treat 18,300 m^3/d (0.2118 m^3/s) of produced water (Petro-Canada 1996). The produced water was estimated to consist mostly of "breakthrough" injected seawater. Estimates of produced water characteristics indicate that it will be warmer and less dense than the receiving seawater and, if discharged, would form a buoyant plume. It has been decided to enhance dispersion of the produced water by discharging it 10 m or more below the sea surface (Petro-Canada 1996).

Since produced water has not been actually generated from the Terra Nova project yet, its chemical composition is presently not known. The previous chapter indicated that the chemical composition of the produced water is site specific (Chapter Five, Table 5.2), and in general includes various organic, inorganic and radioactive chemicals (Roe et al. 1996, Stephenson 1992). In the absence of site specific data, particularly during the design, estimates may be based on chemical concentrations from other existing oil fields. For example, when discussing potential effects of produced water from the Terra Nova project, Petro-Canada (1996) considers polycyclic aromatic hydrocarbons (PAH) as the "most toxic" components of produced water based on a study done in Australia. Therefore, chemical compositions of produced water in Table 5.2 (Chapter Five) might be used as estimates for the case study under consideration.

(ii) Ambient water characteristics

Ambient characteristics at the Terra Nova area (the Grand Banks) have been described briefly when discussing uncertainty measures of ambient parameters for probabilistic analysis of hydrodynamic modeling (Chapter Four). In general, characteristics of the environment of the Terra Nova project can be summarized as follows (Terra Nova Project 2000):

- Water depth: 95 m.
- Air temperature: ranging from-17.3 to 26.8 to °C with a mean of 5 °C.
- Wind speeds: 35 km/h on average.
- Water temperature: ranging from -1.7 to 15.4 °C.
- Fog: seasonal (May-July)
- Sea ice & icebergs: seasonal (April-June)

As discussed in Chapter Four, analysis of the available data (DFO 1999) shows that averaged daily data of current speeds and directions fit Lognormal and Beta distributions, respectively (Chapter Four, Table 4.2). As for tide data, Petro-Canada (1996) reported that tide in the area had a typical range of about 1 m each day, with a maximum tidal amplitude above mean water level of about 0.53 m and a minimum below mean water level of –0.51 m. Seasonal variation is observed for temperature and density of the water. The vertical profiles show that the water column is a two-layer system over most of the year, except in winter when the water column is uniformly cold. The upper portion of the water column is most stratified in August, when the thickness of the upper mixed layer is about 15 m deep.

(iii) Potential ecological risks

Despite the possibility that some parts of the chemicals in the effluent, particularly heavy metals, might leach, it is estimated that the produced water distributes mainly into the upper part of the water column since the density of produced water from the Terra Nova project is expected to be less than that of ambient water. Once it is in the seawater, ecological entities, particularly those in the water column in the vicinity of the discharge are potentially exposed to produced water, which is toxic or contains toxic chemicals. As discussed in the previous chapter, produced water has been associated with a number of ecological problems. Examples of the problems include inhibition of growth and survival of fish, shrimp, algae and mussel (Brendehaug et al. 1992, Moffit et al. 1992, U.S. EPA 1996b, Osenberg et al. 1992).

In the context of the Terra Nova project, fish and shellfish might be potentially at risk. Fish and shellfish species that occur in the Terra Nova area are not unique to the area and include both pelagic (e.g. capelin, mackerel and tuna), demersal (e.g. skate, flatfish and cod), and shellfish (e.g. northern shrimp and snow crab) (Petro-Canada 1996). Fish and shellfish are important not only economically for humans but also ecologically as predators and food for other species. As an example, one of the economically important fishery resources on the Grand Banks is Atlantic cod, which inhabits cool-temperate to subarctic waters from inshore regions to the edge of the Continental Shelf. They may be found from the surface to depths of greater than 400 m. For the Grand Banks study area, exploitable biomass in the early 1980s was estimated to be about 100,000 to 220,000 ton, and the estimated trawlable biomass in the early 1990s was about 10,000 ton (Petro-Canada 1996). Table 6.2 summarizes species caught commercially on Grank Banks and landed at Newfoundland ports in 1992-1994.

Based on produced water studies from different oil fields as discussed in Chapter Five, the hypotheses may be that produced water from the Terra Nova project might cause adverse effects on survival and growth of fish species. Other ecological entities may also have adverse responses to produced water. As the ecological effects are a function of concentration of produced water or toxic chemicals associated with it, the effectiveness of the mixing processes of the effluent and ambient seawater determines the degree of ecological effects. The level of the toxic concentrations and the extent of the distribution area, which is a function of the discharge scenarios, should be considered in evaluating discharge scenarios on the basis of ecological risks. Therefore, the problem in an ecological risk based design is to find design scenarios associated with the least ecological risks.

Table 6.3. Species caught commercially in Grand Banks and landed at Newfoundland ports between 1992-1994 (after Petro-Canada 1996)

Common Name	Scientific Name
Stimpson surf clam	*Mactromeris polynyma*
Snow crab	*Chionecetes opilio*
Scallops	*Chlamys islandica (predom.)*
Skate	*Raja radiata (predom.)*
Redfish	*Sebastes spp.*
Capelin	*Mallotus villosus*
Herring	*Clupea harengus*
Winter flounder	*Pleuronectes americanus*
Atlantic cod	*Gadus morhua*
Quahogs	*Mercenaria mercenaria*
Monkfish	*Lophius americanus*
Turbot (Greenland halibut)	*Rheinhardtius hippoglossoides*
Lobster	*Homarus americanus*
Swordfish	*Xiphias gladius*
Witch flounder	*Glytocephalus cynoglossus*
American plaice	*Hippoglossoides platessoides*
Squid	*Illex illecebrosus*
White hake	*Urophycis tenuis*
Halibut	*Hippoglossus hippoglossus*
Pollock	*Pollachius virens*
Haddock	*Melanogrammus aeglefinus*
Wolffish	*Anarchichas lupus (predom.)*
Eels	*Anguilla rostrata*
Mackerel	*Scomber scombrus*
Bluefin tuna	*Thunnus thynnus*
Roundnose grenadier	*Coryphaenoides rupestris*
Mussels	*Mytilus edulis*
Rock cod	*Gadus ogac*
Yellowtail flounder	*Limanda ferruginea*
Silver hake	*Merluccius bilinearis*
Lumpfish	*Cyclopterus lumpus*
Dogfish	*Squalus acanthias*
Trout	*Salmo trutta, Salvelinus fontinalis*
Bar clams	*Various*

(iv) Analysis plan

In the following sections, the problem of ecological risk-based design will be analyzed by integrating the methodology of hydrodynamic modeling (Chapter Four) and the principles of ecological risk assessment (Chapter Five). A quantitative risk characterization is used in the analysis in order to enable comparative evaluation among different discharge scenarios on the basis of potential ecological risks. The hazard quotient methods will be employed using available toxicity data of single and multiple species. Whole effluent and chemical specific approaches will be employed in the analysis. Before characterizing ecological risks, hydrodynamic characteristics will be analyzed to ensure that the discharge scenarios are acceptable from an engineering viewpoint. In addition, a preliminary analysis to screen scenarios will be undertaken so that only the most attractive scenarios will be carried out in further analysis. The final outputs of the analysis will be descriptions of ecological risks of the attractive scenarios, and recommendations for the design on the basis of potential ecological risks.

A toxicology study of produced water to site-specific ecological entities from the Grand Banks is not readily available for the analysis. In this situation, toxicity data (NOEC) on survival and growth of Sheephead minnows and Mysid shrimps will be used as measurement endpoints because they are economically and ecologically important to the area of interest as discussed above and because standard toxicity tests for fish are usually performed using these species, e.g. Klemm et al. (1994). Available information related to the case study was considered in the analysis, including information from the Environmental Impact Statement of the Terra Nova project (Petro-Canada 1996) and DFO

(1996). In absence of the data, assumptions will be used as discussed in the following sections.

6.4.2 Identifying and evaluating preliminary design scenarios

Preliminary scenarios of the discharge design are identified and evaluated in this second step, based on principles of hydrodynamic modeling and engineering performance. Guidelines on analysis of the type of the outfall (single open-end and diffuser), length scales, criterion of the deep water discharge, depth of the discharge, orientation of the discharge, and diameter of the port(s) are used to identify and evaluate the scenarios. General guidelines associated with engineering aspects of outfall design should include (Sharp 1989a):

1. If required to maximize initial dilution, several ports may be used such that the flow distribution is nearly uniform along the diffuser when ambient velocities are zero or approximately constants. If there is a variation in ambient velocities along the diffuser (ports), the discharge effluent should be approximately proportional to the ambient discharge distribution;
2. Flushing velocities (0.6 to 1.0 m/s) should be obtained in the manifold pipe at least once per day to inhibit settlement of solid (particularly for designs involving horizontal pipes and considerable suspended solids);
3. Arrangement of the outfall should be made to permit periodic flushing if it is required;

4. Ports of the outfall should be designed to flow full in order to prohibit seawater intrusion which can lead to clogging because of marine growth. This requires the jet densimetric Froude number (F_o) to be in excess of 1.0;
5. It is good practice to ensure that the ratio of the sum of the areas of all ports is less than (preferably between 1/3 to 2/3) the area of the manifold pipe;
6. If possible, ports should discharge horizontally and should be separated by about one third of the depth above discharge.

The geometry of the jet may be designed by selecting the type of outfall outlet, i.e. a simple open-end or a multiport diffuser containing a regularly spaced line of relatively small ports. For large-diameter outfalls, the multiport diffuser has become a conventional design feature. In this design, the end of the pipe is capped off and wastewater flow enters the sea through a series of small holes spaced along the sides of the outfall. The length of pipe through which effluent leaves the outfall is called the diffuser (Grace 1978). The use of a multiport diffuser for produced water discharge is reported in Washburn et al. (1999). The purpose of such multiport diffusers is to ensure a much greater initial interception of ambient water by the effluent stream in order to obtain greater initial dilution.

However, a multiport diffuser provides increased initial dilution only within a small mixing zone near the diffuser. At the distance of a few lengths downstream, particularly for density unstratified conditions, the plume dilution distribution becomes independent of the diffuser length. Unlike a multiport diffuser, a simple open end is the easiest terminus to build and maintain. Therefore, use of a simple open end is recommended in cases where it will provide adequate initial dilution to meet design requirements, and in cases where plume submergence due to a diffuser is undesirable.

Furthermore, if the ratio of the port spacing to the discharge depth is greater than about one third (1/3), diffuser discharges could be considered equivalent to single port discharges with a flow rate taken to be a flow rate of one port (Proni et al. 1994, William 1985). This ratio criterion is to ensure that jets do not interfere or merge with each other before they reach the water surface. This particular case is usually referred to as "adequately spaced ports or diffuser". Many operational and designed produced water outfalls are open-ended (Brandsma and Smith 1996, Somerville et al. 1987, Petro-Canada 1996). Based on the above discussions, whenever possible, this study focuses on a single port discharge and if necessary, a modification into "adequately spaced ports or diffuser" may be considered. With this modification, the hydrodynamic modeling presented in the previous chapter may still be employed even though the case study involves an outfall with the outlet consisting of more than one port.

For the analysis of the length scales, the case study considers data from Petro-Canada (1996), which estimated a produced water flow rate of 0.2118 m^3/s and a discharge depth of about 10 m or more below the sea surface. If it is assumed for now that the diameter of the port can be estimated to be 0.305 m (single open-ended pipe), the same as that used at the Hibernia oil producing platform (Hibernia Management and Development Company 1996), various length scale ratios defined in Chapters Two and Three can be calculated for the case study under consideration, i.e. l_Q/l_b of 0.01, l_m/l_b of 0.205, and z/l_b of 0.366. These characteristics are in the range of those calculated based on data from other produced water discharges (Brandsma and Smith 1996, Smith et al. 1996, and Somerville et al. 1987), or field sewage discharges (Bennet 1981, Bettess and Munro 1980, Lee and Neville-Jones 1987a, and Proni et al. 1994). The laboratory data used in developing the initial dilution models (Chapter Three, and Lee and Cheung 1991) and those of Wright (1977a) are also in a range that accommodates these values.

Design scenarios are identified and evaluated by further considering two types of parameters: (1) discharge parameters and (2) design parameters. Scenarios for the first type of parameters are required when the produced water has not actually been generated in the production, meaning that actual numerical values for the discharge parameters are not known. Therefore, scenarios of the discharge parameters reflect assumptions about the discharge characteristics, e.g. scenarios of the flow rate and the relative density difference. On the other hand, scenarios of the design parameters are identified and evaluated so as to be in compliance with specified guidelines and criteria, e.g. selection of the depth above discharge and the diameter of the port.

Table 6.4 shows possible preliminary design scenarios, which could be identified for the Terra Nova case study. The single round open-ended outfall is considered with three possible discharge scenarios. The flow rate is set at a fixed value of 0.2118 m^3/s to be consistent with the estimate from Petro-Canada (1996). Since information on the relative density difference is not available, three different values are assumed based on produced water discharges from other oil fields discussed in Chapter Two. The depth of the discharge is varied, ranging from 8 to 20 m. The estimate of the discharge depth from Petro-Canada (1996) is 10 m or more, which is within the range considered in this table. The diameter of the port is given in the form of a range, where the minimum diameter is to satisfy the deepwater criterion, and the maximum diameter is to satisfy the criterion for the densimetric Froude number and the flushing velocity.

Table 6.4. Preliminary discharge scenarios for the case study

Scenario #	Discharge #1[a]		Discharge #2[b]		Discharge #3[c]	
	H (m)	d (m) range	H (m)	d (m) range	H (m)	d (m) range
1	8	0.028 - 0.483	8	0.024 - 0.446	8	0.020 - 0.391
2	10	0.024 - 0.483	10	0.021 - 0.446	10	0.017 - 0.391
3	11	0.022 - 0.483	11	0.020 - 0.446	11	0.016 - 0.391
4	14	0.019 - 0.483	14	0.017 - 0.446	14	0.014 - 0.391
5	17	0.017 - 0.483	17	0.015 - 0.446	17	0.012 - 0.391
6	20	0.015 - 0.483	20	0.013 - 0.446	20	0.011 - 0.391

Note: Discharge flow rate is taken at 0.2118 m^3/s (Petro-Canada 1996).
Relative density difference is specified for each discharge scenario, i.e.:
[a] 0.037 (based on the discharge at the Bass Strait, see Table 2.1, Chapter 2).
[b] 0.025 (the mean of the uniformly distributed relative density difference ranging from 0.013 to 0.037)
[c] 0.013 (based on the discharge at the North Sea, see Table 2.1, Chapter 2).

The scenarios identified in Table 6.4 need to be further evaluated. As an example, consider a case of evaluating the diameter of the port. Although the range of the diameter satisfies the criteria discussed above, not all values in the range may be technically and economically possible. For instance, setting a diameter at the minimum value of 0.01 m (discharge #3, design scenario #6) is technically difficult in operation and maintenance.

Furthermore, with this diameter and the specified flow rate, the system requires an extremely high discharge velocity, i.e. more than 2600 m/s, which could be economically expensive because of the excessive energy required to maintain the discharge.

It can also be seen from Table 6.4 that the estimate based on data from Hibernia, i.e. port diameter of 0.305 m (Hibernia Management and Development Company 1996), is on the upper side of the range. Being on the upper side, setting a diameter of 0.305 m is suitable only if the flow rate does not fall below 9.7 m^3/d (0.1123 m^3/s). However, a lower discharge rate is possible particularly in the early stages of production. Based on the production and injection forecast (Petro-Canada 1996), the flow rate of about 0.1123 m^3/s is not reached until 2013 (i.e. 13 year old production). In order for the outfall to flow full and prohibit seawater intrusion, which can lead to clogging because of marine growth, the possibility of having a densimetric Froude number below unity should be minimized. An example of a design that might be associated with a densimetric Froude number below unity and thus potentially be at risk of seawater intrusion, is that from the North Sea oil field presented in the last column of Table 2.1 (Chapter Two).

Based on the above discussion, the diameter of 0.305 m may not be the most suitable for the case study at hand. A diameter of about 0.2 m may be set and may still be suitable even at a flow rate as low as 4.1 m^3/d (0.0475 m^3/s), which is the estimate for the earlier stage of oil production at the Terra Nova project (Petro-Canada 1996). The diameter of 0.2 m is also comparable to that from other oil fields, i.e. Bass Strait, Australia, and the Gulf of Mexico, U.S.A. (Table 2.1, Chapter Two).

6.4.3 Screening the preliminary scenarios

The third step in the framework is a screening process prior to performing further analysis of ecological risks. Initial dilution and effluent concentration of one-dimensional (1-D) cases, e.g. concentration as a function of distance downstream, can be used as screening measures. The term 1-D case implies that only one direction of ambient current, and thus effluent plume, is considered and it is towards the location of interest, e.g. 100 m downstream. In the evaluation, concentrations of whole effluent or specific chemicals at the location of interest are compared with threshold concentrations, typically benchmark concentrations of the produced water toxicity (e.g. LC_{50} and NOEC), or ambient water quality standards. Scenarios that are not in compliance with the threshold concentrations are screened off and are not considered for further analysis.

The ambient water quality standards can be used as threshold concentrations in the screening based on the chemical specific approach. The standards applicable for chemicals that are often found in produced water have been discussed and presented in Table 6.1 (Section 6.2). As can be seen in Table 6.1, the level of the concentrations varies for different chemicals. If several chemicals are subject to consideration, the number of the analysis can be reduced by a modification. For example, water quality standards associated with these chemicals may be modified by converting concentrations specified in the standards into equivalent dilutions, which are calculated by dividing the concentration specified in the standards for a given chemical with the concentration prior to discharge for the same chemical. The highest equivalent dilutions can be used as representative thresholds to evaluate different design scenarios based on the initial dilutions or chemical concentrations, which are also in terms of equivalent dilutions.

When the whole effluent toxicity approach is employed in the screening, the standards are typically given in terms of toxicity units (as discussed in section 6.2), or alternatively, they may be replaced by using benchmark concentrations of the produced water toxicity, e.g. LC_{50} and NOEC. Toxicity of produced water for shrimp and fish from various studies at different oil fields has been discussed in Chapter Five (Tables 5.7 and 5.8). A larger data base of toxicity data for produced water from more than 220 outfalls is available at the Louisiana Department of Environmental Quality (LDEQ), in which the median NOEC to fish (Sheephead minnow) is 2.5% and 4.9% for survival and growth, respectively (Meinhold et al. 1996c).

For the case study, preliminary scenarios presented in the previous subsection are evaluated using the whole effluent analyses. For each scenario in Table 6.4, the initial dilution was calculated using equations 4.2 (Chapter Four). The effective depth above discharge, z, for the equation was taken at 75% of the total depth above discharge, H (William 1985). The ambient current speed as an input parameter was based on the value of the mean and maximum daily-averaged ambient current speeds of 0.056 m/s and 0.30 m/s, respectively (data from the Department Fisheries and Ocean, DFO 1999). Results of the calculated initial dilution are shown in Table 6.5 and Figures 6.2 and 6.3.

Table 6.5. Initial dilution associated with the preliminary discharge scenarios

Scenario	H (d)	Discharge #1[a]		Discharge #2[b]		Discharge #3[c]	
#	m (m)	S (at mean u)	S (at max u)	S (at mean u)	S (at max u)	S (at mean u)	S (at max u)
1	8 (0.2)	4.7	26.4	4.1	26.5	3.4	26.4
2	10 (0.2)	6.8	41.4	6.0	41.4	5.0	41.4
3	11 (0.2)	8.0	50.1	7.1	50.1	5.9	49.7
4	14 (0.2)	12.1	81.2	10.7	80.9	9.2	80.1
5	17 (0.2)	16.7	119.7	14.9	119.1	13.5	117.7
6	20 (0.2)	22.1	165.4	19.9	164.4	18.8	162.4

Note: Discharge flow rate is taken at 0.2118 m^3/s (Petro-Canada 1996).
Relative density difference is specified for each discharge scenario, i.e.:
[a] 0.037 (based on the discharge at the Bass Strait, see Table 2.1, Chapter 2).
[b] 0.025 (the mean of the uniformly distributed relative density difference ranging from 0.013 to 0.037)
[c] 0.013 (based on the discharge at the North Sea, see Table 2.1, Chapter 2).
Mean u is the mean daily-averaged ambient current speed of 0.056 m/s.
Max u is the maximum daily-averaged ambient current speed of 0.30 m/s.

Effluent concentrations at different distances downstream are calculated based on the methodology of hydrodynamic modeling (Chapter Four). Results of evaluating preliminary discharge scenarios (Table 6.4) are presented in Figures 6.4 to 6.6. The mean daily-averaged ambient current speed of 0.056 m/s was taken for the evaluation, and the effluent concentrations downstream associated with the maximum ambient current speed of 0.3 m/s are expected to be less than those presented in Figures 6.4 and 6.6. Those figures also show threshold concentrations, i.e. benchmark concentrations of the produced water toxicity using the median fish survival NOEC of 2.5% and the median fish growth NOEC of 4.9%, upon which acceptability of the preliminary scenarios may be evaluated.

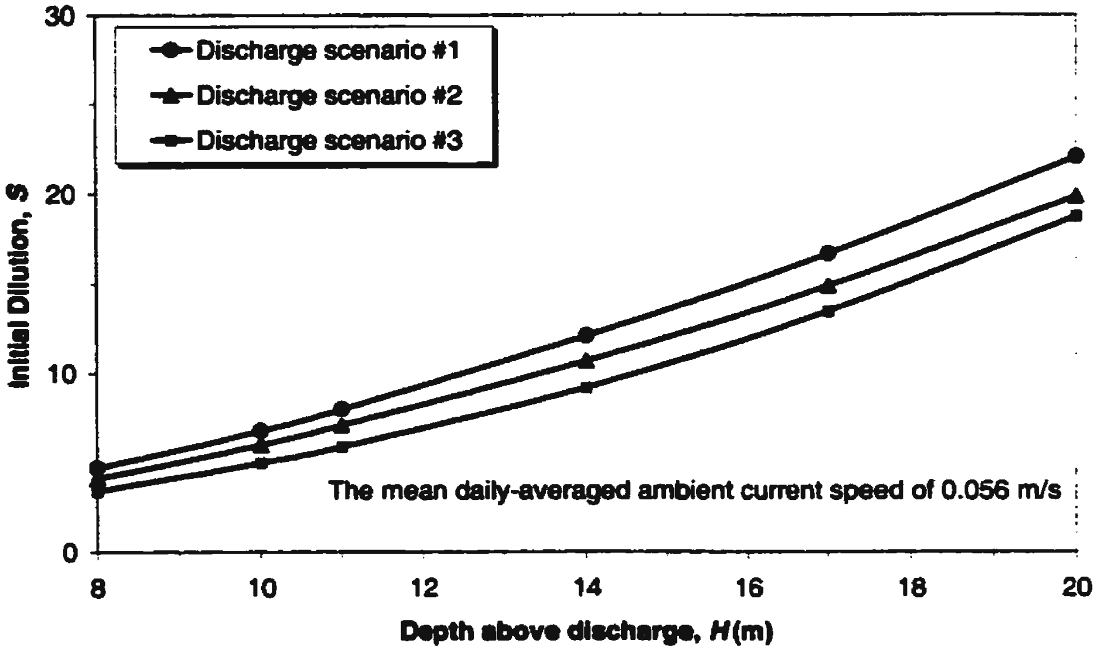


Figure 6.2. Initial dilution as a function of depth above discharge
(at the mean daily-averaged ambient current speed)

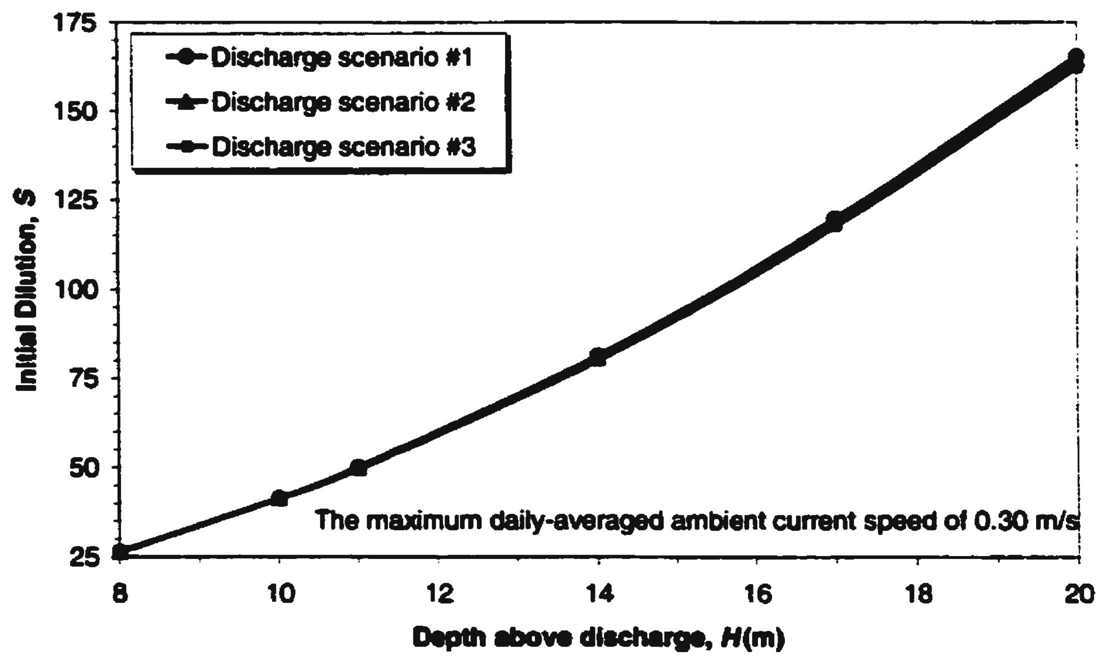


Figure 6.3. Initial dilution as a function of depth above discharge
(at the maximum daily-averaged ambient current speed)

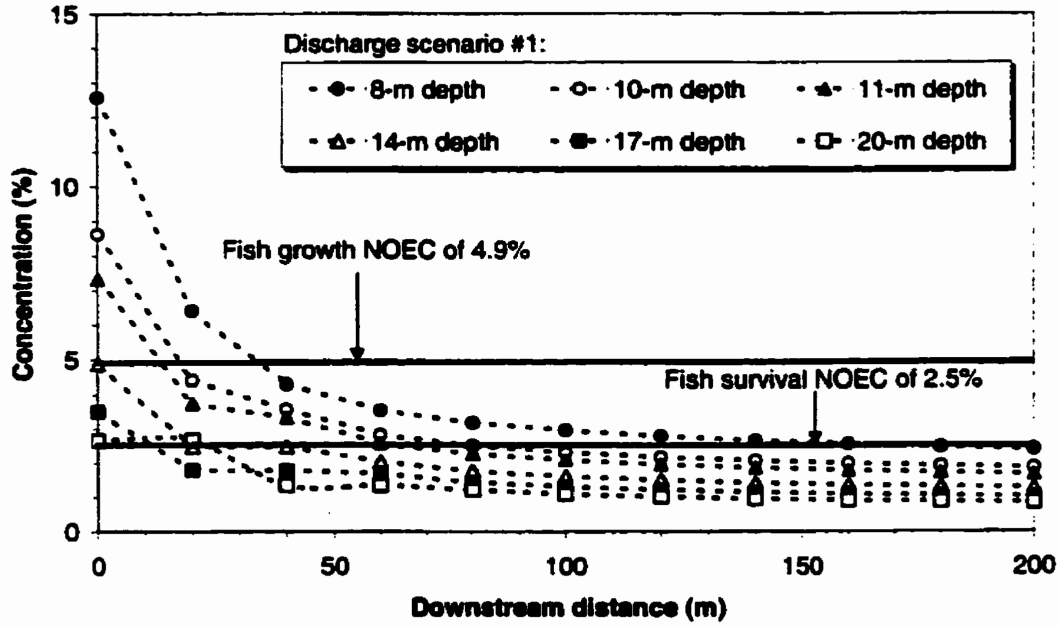


Figure 6.4. Centerline concentration as a function of distance downstream (relative density difference of 0.037)

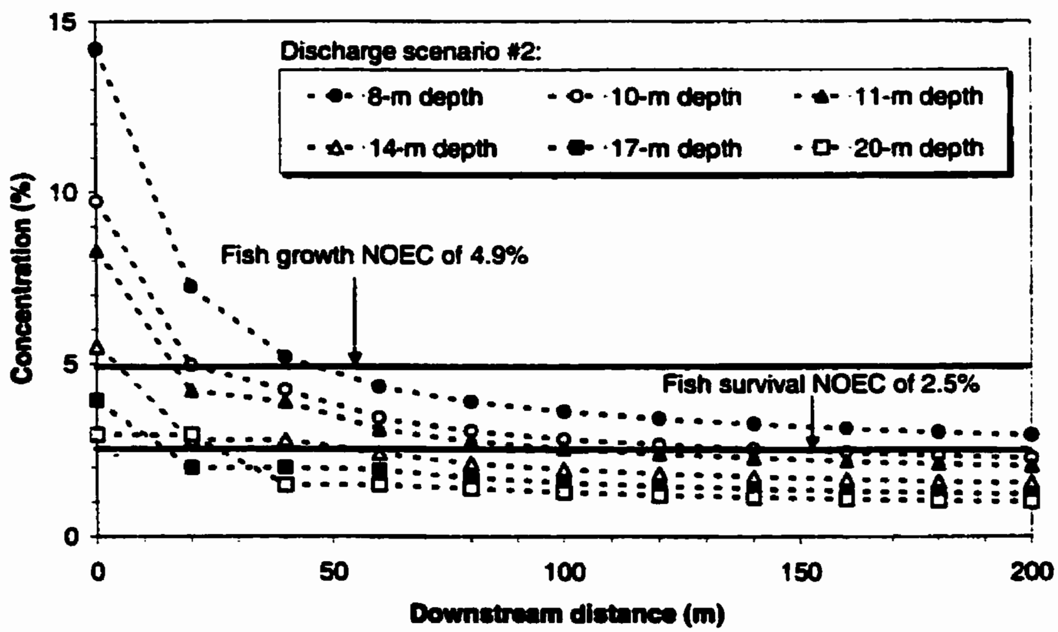


Figure 6.5. Centerline concentration as a function of distance downstream (relative density difference of 0.025)

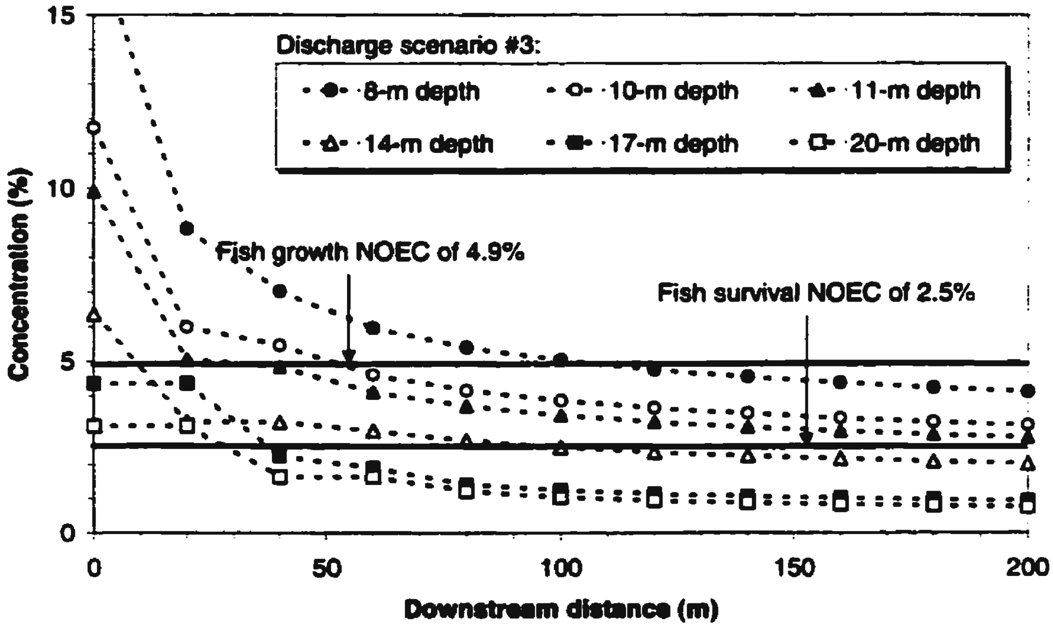


Figure 6.6. Centerline concentration as a function of distance downstream (relative density difference of 0.013)

As discussed in section 2.3 (Chapter Two), sufficiently high initial dilutions are required to maintain minimal thickness of the effluent slick. Unlike mixing in the far field, which is mainly governed by ambient characteristics, initial dilution is highly dependent on the design of the discharges. Despite its importance in design, however, no fixed threshold value has been defined as an "acceptable initial dilution". This might be because it likely depends on the type of the discharges (e.g. sewage, cooling water, produced water etc.), the quantity and quality of the discharge, and the environment where the discharge takes place. For example, an ocean outfall discharging sewage from a small town in Newfoundland into Spaniard's Bay was designed to achieve initial dilution of about 30 (Gowda 1992); while a diffuser design for sewage discharge from a metropolitan city of Boston was based on an experiment, in which initial dilution was about 60 to 80 (Roberts and Snyder 1993). For produced water, Hodgins and Hodgins (2000) estimated that a potential produced water discharge from the White Rose oil field, Grand Banks, Newfoundland, would have initial dilution of 35.

The above calculations show that the range of dilution values associated with different ambient current speeds for a given scenario is very wide. For instance, the initial dilution for the discharge #1 and scenario #6 (Table 6.5) ranges from 22.1 (at the mean current speed) to 165.4 (at the maximum current speed), showing a more than 7-fold difference. Initial dilution is also sensitive to the depth above discharge. The effect of the density difference is less dominant than that of the ambient current speeds as presented in Figures 6.2 and 6.3. Those figures show that, in this typical analysis, at the ambient current speed of 0.3 m/s the initial dilutions are practically the same irrespective of the variation in the relative density difference. Because of this, the number of discharge scenarios associated with the relative density difference may be reduced in the next analysis by considering the minimum and maximum estimates, i.e. 0.013 and 0.037. Another treatment may also be employed by taking the probability distribution of the relative density difference, instead of using only its mean value of 0.025.

Although the initial dilution is relatively low at the ambient current speed of 0.056 m/s, the centerline concentrations are reduced very fast within 100 m downstream as shown in Figures 6.4 to 6.6. When the relative density difference is high, i.e. 0.037 (Figure 6.4), at 100 m downstream all scenarios of depth above discharge result in centerline concentrations below the fish growth NOEC of 4.9%. The majority of the scenarios (except the 8-m depth) is associated with centerline concentrations below the fish survival NOEC of 2.5%. In Figure 6.6, however, half of the scenarios are not in compliance with the threshold concentration for fish survival at 100 m downstream.

From the above analysis, it can be seen that the deeper the discharge port, the better the performance of the outfall in terms of the initial dilutions and effluent concentrations downstream. However, for the reason of construction, operation, maintenance and cost, an extremely deep outfall may not be appropriate. If the distance of 100 m downstream is assumed as the length of the mixing zone at which the effluent concentration should not exceed the most stringent threshold value, i.e. the fish survival NOEC of 2.5%, scenarios #1 and #2 (the depth above discharge of 8 and 10 m) may then be screened off. The scenario # 3 (the depth above discharge of 11 m) exceeds the threshold value only when the relative density difference is low, i.e. 0.013 (Figure 6.6), but not in the other two cases. Because of this, and because the above analysis is very conservative (i.e. effluent plume is assumed always to be spreading towards the location of interest, no other direction is considered), the scenario #3 (the depth above discharge of 11 m) will still be considered for further analysis. On the other hand, the scenarios #5 and # 6 (the depth above discharge of 17 and 20 m) have almost the same value of the effluent concentrations at 100 m downstream in all three cases, which are well below both the threshold concentrations. Only one of them needs to be considered for further analysis. Therefore, scenarios #3, #4 and #5 (i.e. the depth above discharge of 11, 14 and 17 m) will be further evaluated as discussed in the following steps.

6.4.4 Analysis of exposures and ecological effects

The analysis of exposure and ecological effects is based on the principles of ecological risk assessment discussed in Chapter Five (Section 5.3.1). The analysis of exposure considers the source of the pollutant, distribution of the contaminants, and modes of contact between the pollutants and the endpoint biota. No other source of produced water is identified near the Terra Nova project; the nearest is that from Hibernia (at about 35 km distance). Further developments at White Rose (85 km away) and Hebron (20 km away) are also beyond the distance of interest considered in the mixing zone analysis. It is therefore assumed that the source is a single source of produced water outfall.

The U.S. EPA (1998) approach, which is discussed in Chapter Five (Section 5.3.2), is adopted in the case study to estimate the exposure. In this approach, the exposure is quantified using an environmental concentration of pollutant, assuming that the effluent is well mixed in the ocean or that organisms move randomly through the water. Therefore, distribution of effluent concentrations can be referred to as "exposure concentrations" (U.S. EPA 1998). The exposure concentrations may be evaluated for each scenario under consideration. The distribution of the exposure concentrations for the case study was estimated based on hydrodynamic modeling discussed in the previous chapter (Chapter Four). For the case study, the statistics of the concentrations are presented in Figures 6.7 to 6.12, showing the mean and the 95%-tile exposure concentrations at different scenarios of depth above discharge (i.e. 11, 14 and 17 m) and relative density difference (i.e. 0.013 and 0.037). The 95%-tile exposure concentrations based on uniformly distributed relative density difference (referred to as discharge #4) are also shown in Figures 6.13 to 6.15.

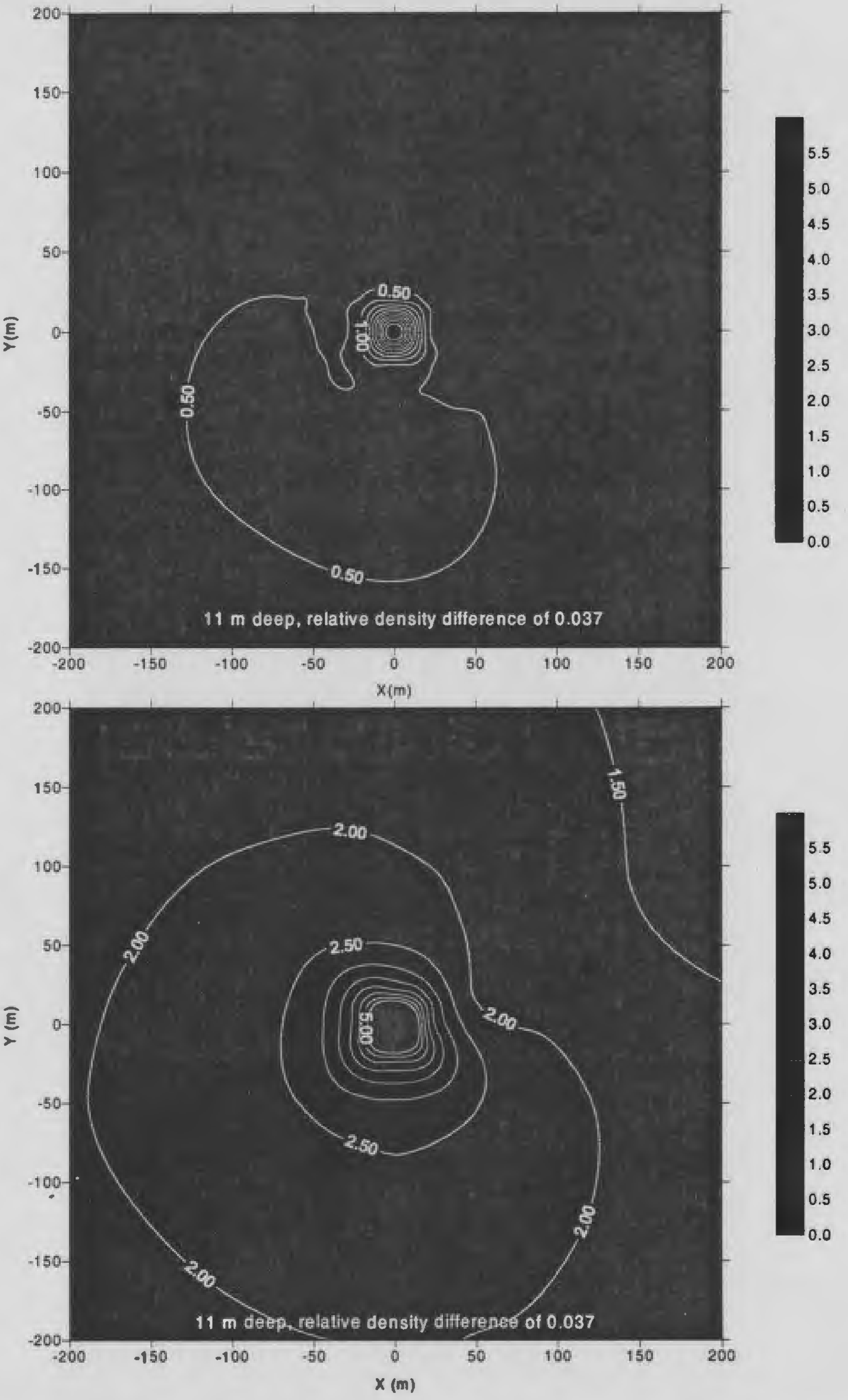


Figure 6.7. Exposure concentrations (%) associated with discharge #1, design #3 (Top is the mean concentrations, bottom is the 95%-tile exposure concentrations)

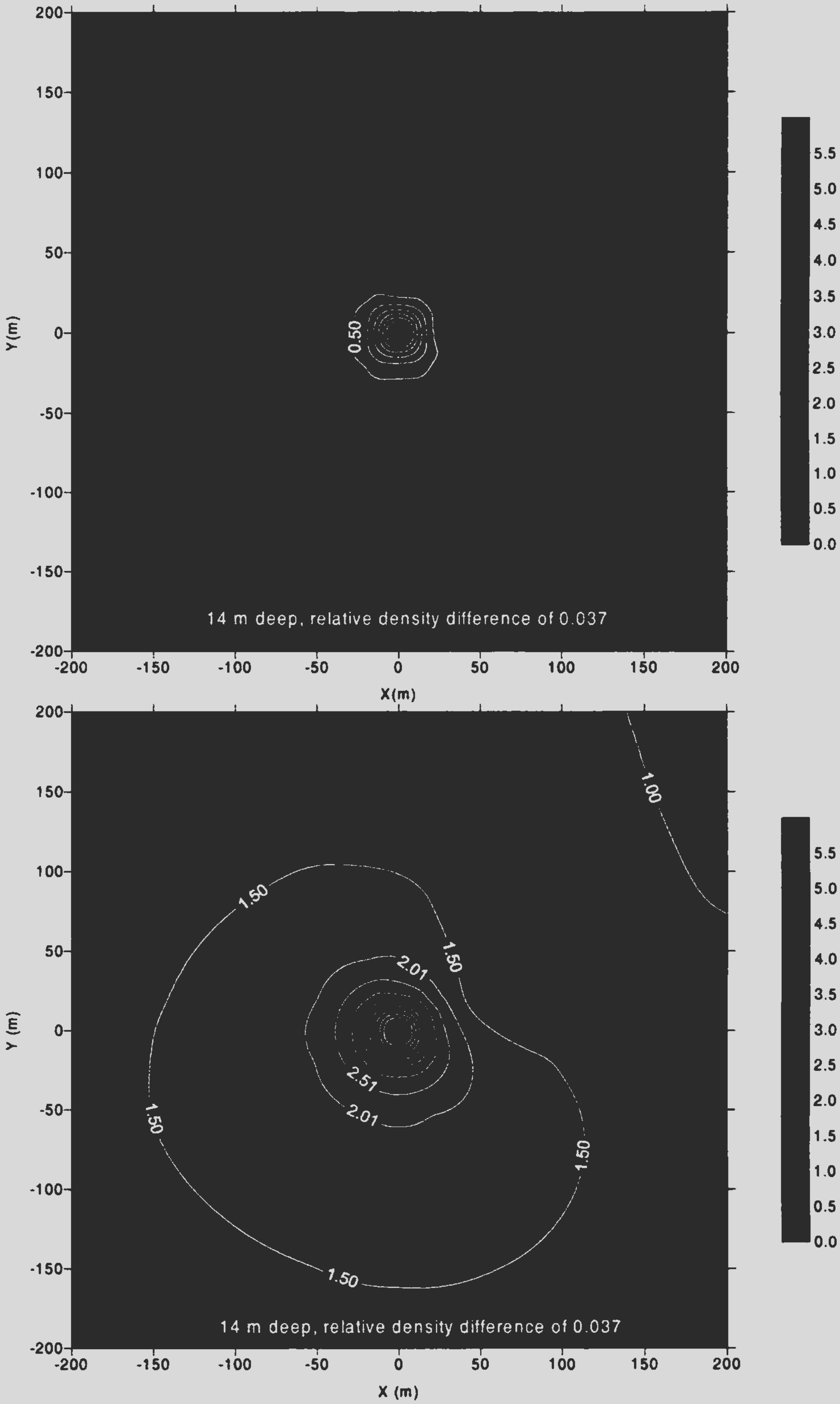


Figure 6.8. Exposure concentrations (%) associated with discharge #1, design #4 (Top is the mean concentrations, bottom is the 95%-tile exposure concentrations)

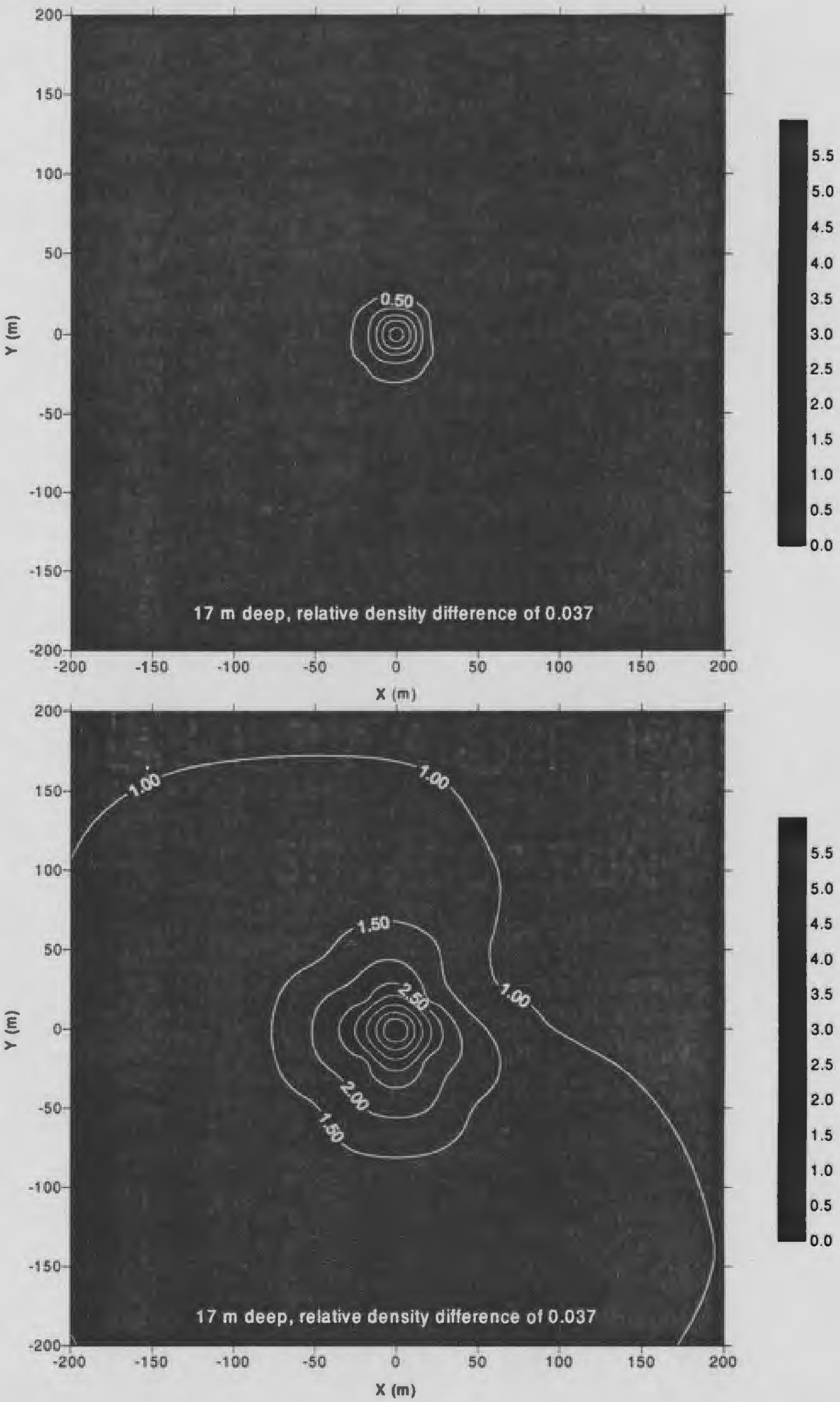


Figure 6.9. Exposure concentrations (%) associated with discharge #1, design #5 (Top is the mean concentrations, bottom is the 95%-tile exposure concentrations)

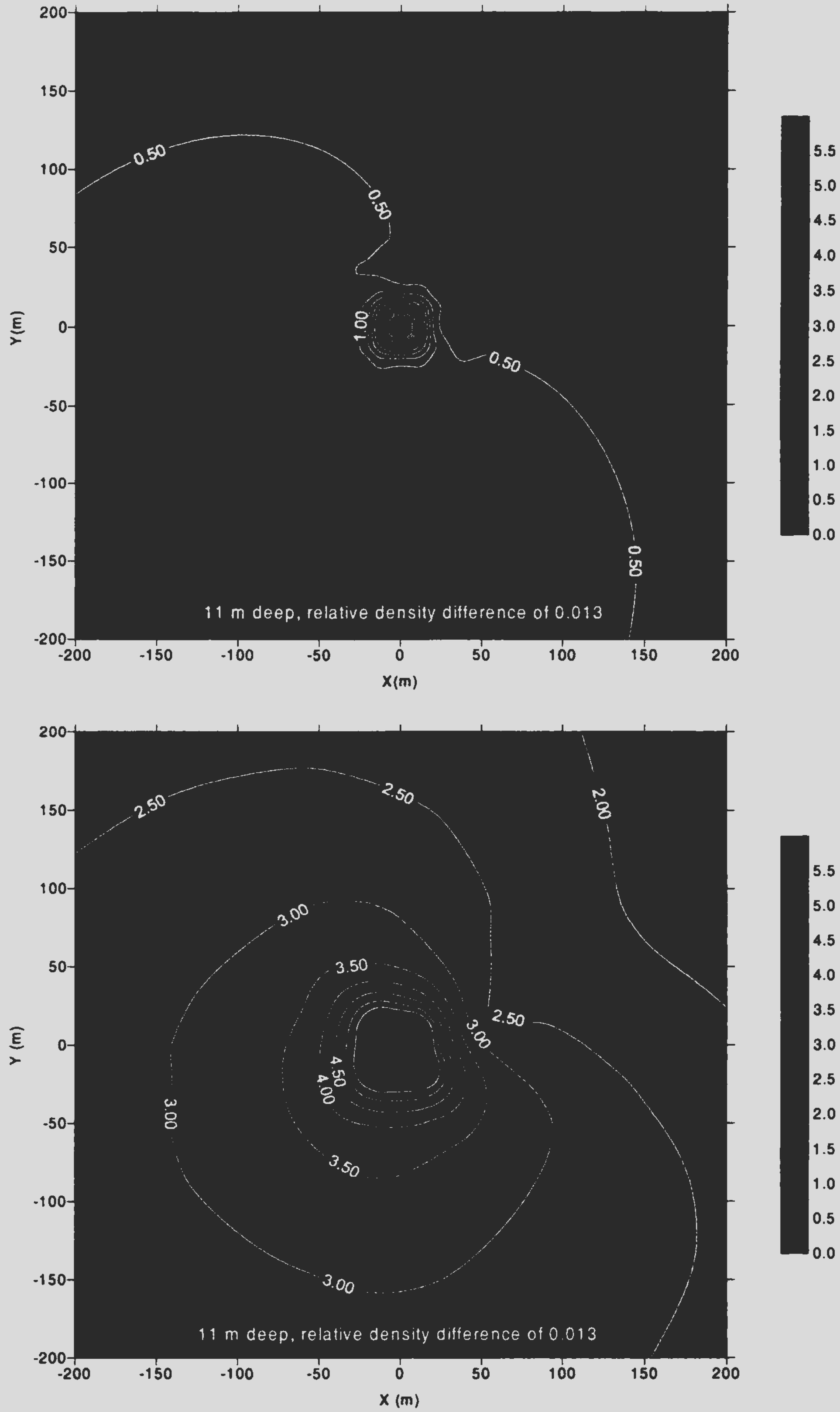


Figure 6.10. Exposure concentrations (%) associated with discharge #3, design #3 (Top is the mean concentrations, bottom is the 95%-tile exposure concentrations)

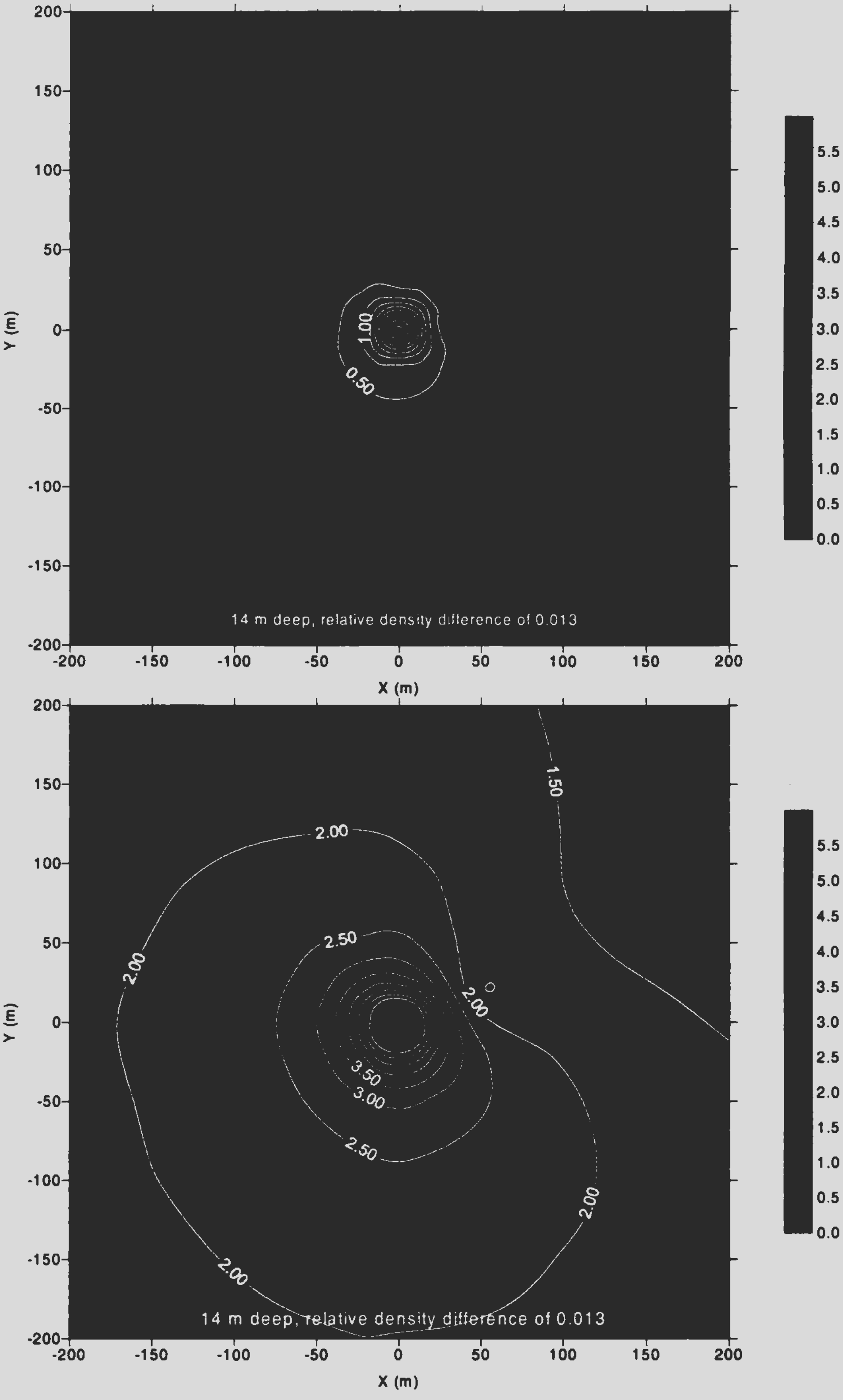


Figure 6.11. Exposure concentrations (%) associated with discharge #3, design #4
(Top is the mean concentrations, bottom is the 95%-tile exposure concentrations)

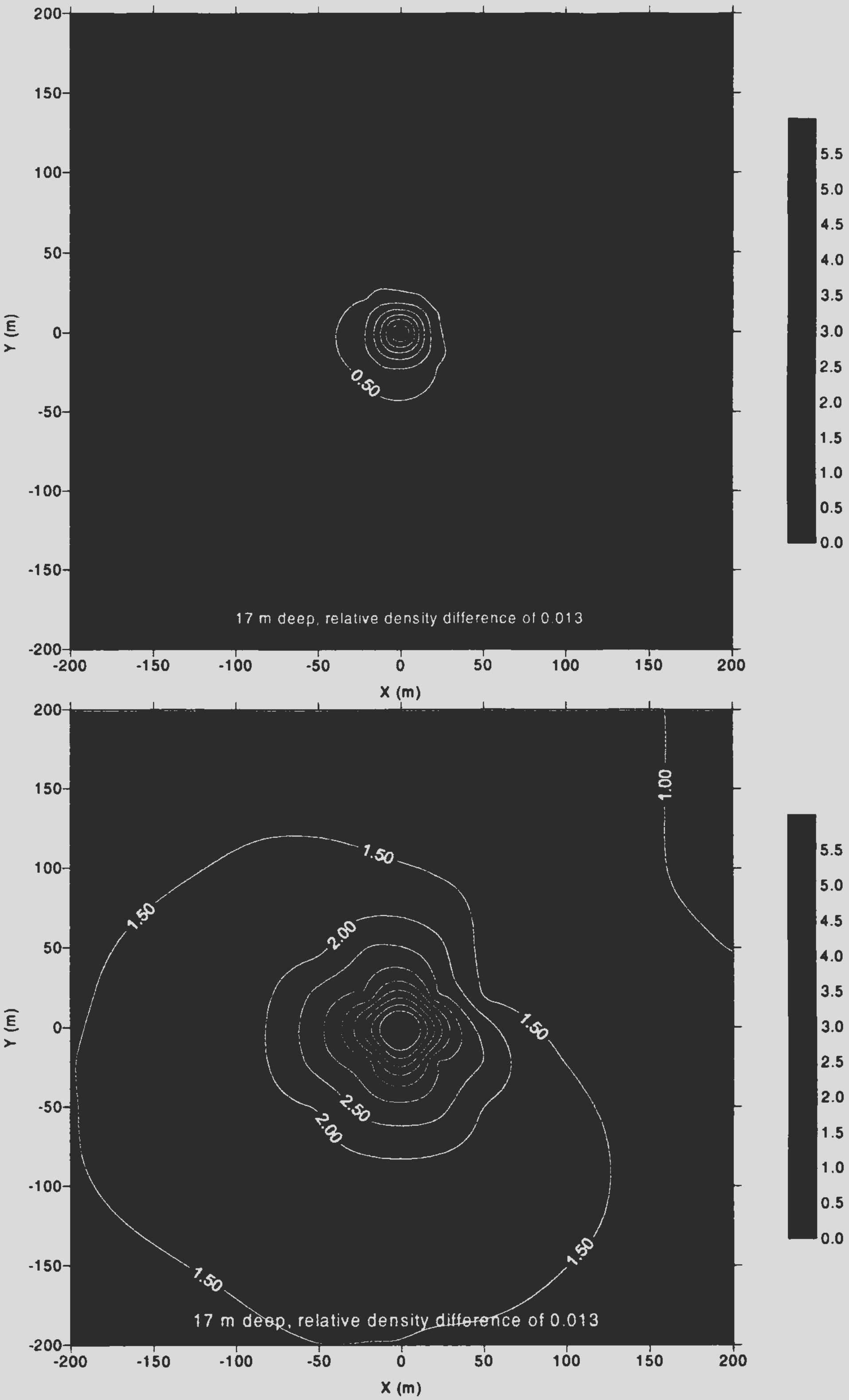


Figure 6.12. Exposure concentrations (%) associated with discharge #3, design #5 (Top is the mean concentrations, bottom is the 95%-tile exposure concentrations)

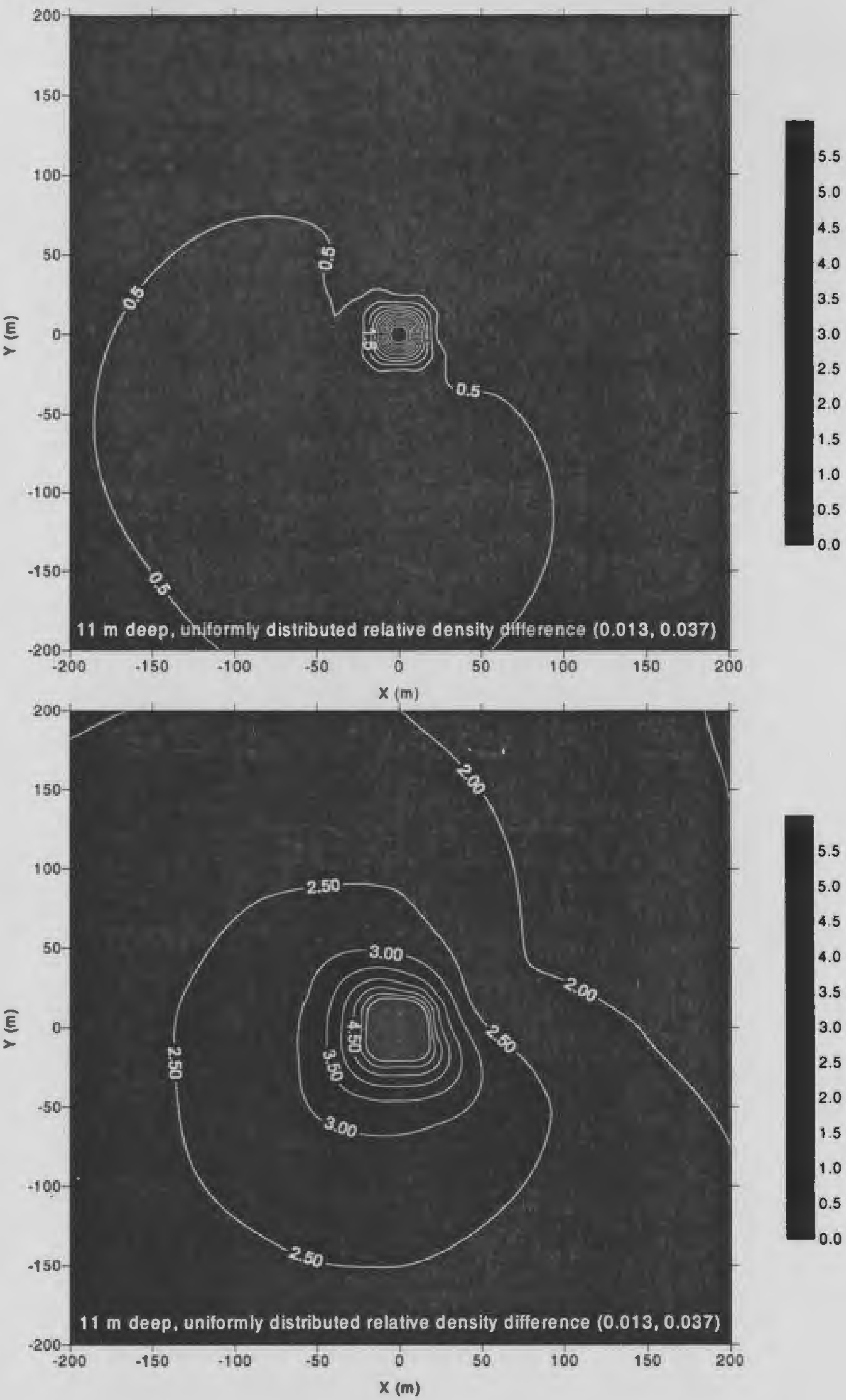


Figure 6.13. Exposure concentrations (%) associated with discharge #2, design #3 (Top is the mean concentrations, bottom is the 95%-tile exposure concentrations)

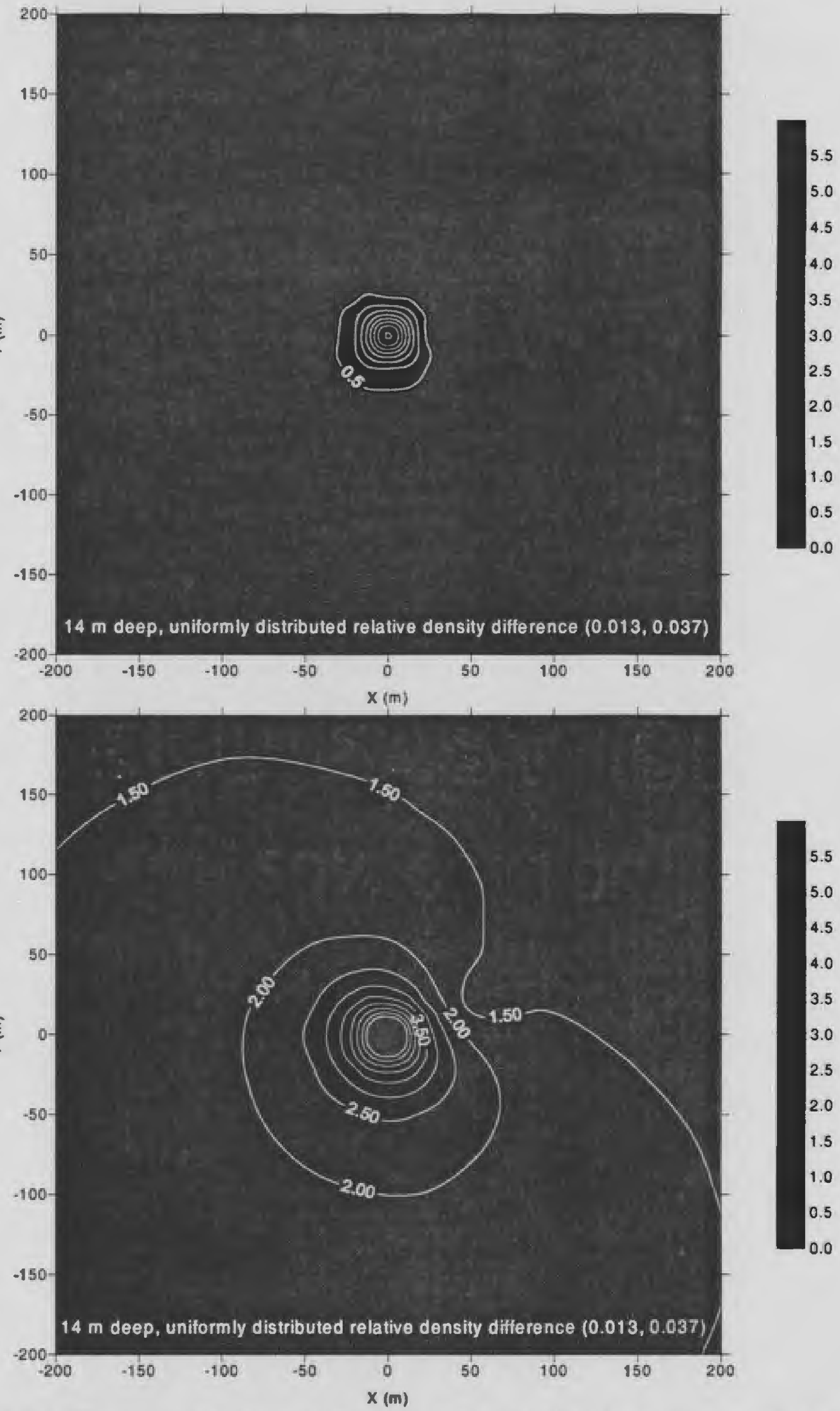


Figure 6.14. Exposure concentrations (%) associated with discharge #2, design #4 (Top is the mean concentrations, bottom is the 95%-tile exposure concentrations)

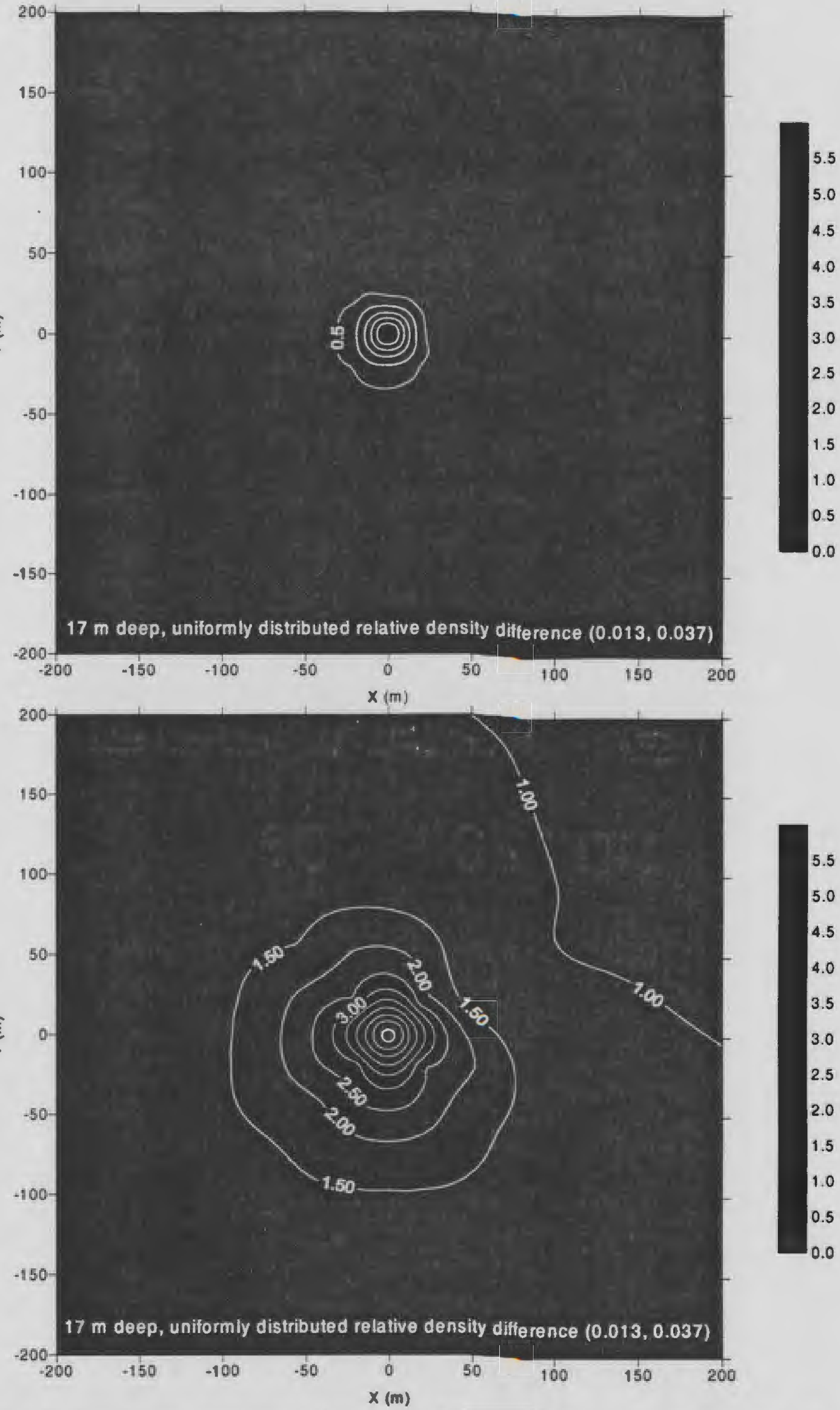


Figure 6.15. Exposure concentrations (%) associated with discharge #2, design #5 (Top is the mean concentrations, bottom is the 95%-tile exposure concentrations)

It can be seen from Figures 6.7 to 6.15 that the effect of the depth above discharge to exposure concentrations is generally significant. For example, for the relative density difference of 0.037, the area of the mean exposure concentrations of 0.5% or more typically reduces from approximately 17,000 m^2 (Figure 6.7, top: 11 m depth) to approximately 1,900 m^2 (Figure 6.9, top: 17 m depth), or about 88% in reduction. Similar evidence is also observed for the area of the mean exposure concentrations of 0.5% or more at the relative density difference of 0.013 (Figures 6.10 to 6.12, top), and for the area of the 95%-tile exposure concentrations at the relative density difference of 0.013 and 0.037 (Figures 6.7 to 6.12, bottom).

The effect of the density difference is also observed from those figures but in general is relatively less than that from the depth above discharge. From the previous section, it appears that the effect of the relative density difference on the exposure concentrations is less dominant than that of the ambient current speeds (shown in Figures 6.2 and 6.3). Unlike the depth above discharge, which is a design scenario and is specified upon the decision of the designer, the relative density difference is a discharge characteristic depending on the physical characteristics of produced water, which are not controllable. In addition, only a range was available for the estimates of the relative density difference and there was no evidence to support whether one value has more likelihood than another does. For these reasons, uniform distribution may be a reasonable assumption to represent both cases, the lowest and highest estimates of the relative density difference. Therefore, the following risk characterization considers that the relative density difference is uniformly distributed.

Once exposure concentrations have been defined, they may be integrated with the analysis of ecological effects to enable characterization of ecological risks. The analysis of ecological effects determines the relationships between the exposure to the contaminant and effects on the measurement endpoint, and it is usually based on results of toxicity studies as discussed in section 5.3.2 (Chapter Five). Ecological effects of produced water at the organism level have been reported in the literature, e.g. Brown et al. (1992), Meinhold et al. (1996b, 1996c) and U.S EPA (1996b). Although field tests may be possible, most toxicity tests of individual organisms are performed in the laboratory. Results of toxicity tests of individual animals can be used as the basis for the effects assessment (Meinhold et al. 1996c). Acute and chronic effects are usually reported in terms of the 96-hours median lethal concentration LC_{50} and the survival or growth NOEC, respectively. Table 6.6 shows typical results from toxicity studies of produced water on two organisms, i.e. Sheephead minnows (fish) and Mysid shrimps (aquatic invertebrate).

Table 6.6. Typical results from whole effluent toxicity tests (after Meinhold et al. 1996b)

Statistics	Mysids (*Mysidopsis bahia*)			Sheepshead Minnows (*Cyprinodon variegatus*)		
	96-hour LC_{50}	7-d NOEC Survival	7-d NOEC Growth	96-hour LC_{50}	7-d NOEC Survival	7-d NOEC Growth
Mean	9.5	2.9	4.0	24.4	7.1	9.0
Standard deviation	11	2.9	3.5	38.2	5.7	6.9
Minimum	0.2	0.0004	0.1	2.4	0.2	0.2
Maximum	71.2	11.4	12.1	-	19.1	25.2
No. of outfalls	41	43	42	39	41	39

As can be seen from Table 6.6, the aquatic organisms respond differently at various levels of produced water concentration. As discussed in the problem formulation, the survival and growth NOEC for both Sheephead minnows and Mysid shrimps are considered as measurement endpoints. However, since the survival NOECs are more sensitive measures than the growth NOECs (see Table 6.6), the use of survival NOECs as measurement endpoints can reflect the protection level of both survival and growth NOECs for these organisms.

Other ecological entities, which may also have adverse responses to produced water, may also be considered. Aquatic ecological entities in the area under consideration are various, which may be reflected by feeding relationships among species as shown in Table 6.7. For produced water as a whole effluent. reports of toxicity effects on various aquatic organisms are not readily available. However, they may be evaluated in terms of chemical specifics usually found in produced waters. As discussed previously, various chemicals reported in produced waters worldwide are shown in Tables 5.2 and 5.3 (Chapter Five). For the Terra Nova project, studies have considered hydrocarbon chemicals (i.e. PAH) during the evaluation of produced water effects in the environmental impact statement (Petro-Canada 1996). This analysis will therefore focus on other specific chemicals, i.e. a metal and a radioactive chemical.

Table 6.7. Feeding relationships among species in terms of stomach contents of common fish species (in percentage volume, from Petro-Canada 1996)

	Atlantic Wolffish	Spotted Wolffish	Atlantic Cod	Longfin Hake	Common Grenadier	Roughhead Grenadier	Witch Flounder	American Plaice	Greenland Halibut	Arctic Eelpout	Thorny Skate	Acadian Redfish	Golden Redfish	Deepwater Redfish
Gammarid Amphipods			0.1		3.5	0.2	6.7	0.0			1.2			
Cumacea					0.2		0.5							
Decapods	3.7	28.6	9.1	53.7	24.5	29.6		1.2	13.2	33.4	35.7	11.0	3.7	35.4
Isopods							0.1				0.0			
Echinoderms	28.8	30.3	1.3		1.0	5.4	5.7	86.3	0.2	52.8	0.1			
Molluscs	2.2	0.3			1.7	1.9	4.6	0.4	1.7	0.8	4.2			4.1
Anthozoa	1.0	0.2				0.3								
Polychaetes	0.5	0.3	0.4		22.3	4.8	80.1	0.8	0.1	3.4	2.2			0.2
Sponges		2.0												
Tunicates						0.1								
Sipunculids							1.8			0.2				
Total fish and pelagic	63.8	38.3	89.1	46.3	46.8	57.7	0.5	11.3	84.8	9.4	56.6	89.0	96.3	60.3
Total benthic animals	36.2	61.7	10.9	53.7	53.2	42.3	99.5	88.7	15.2	90.6	43.4	11.0	3.7	39.7

A detailed review of ecological effects of petroleum hydrocarbons, metals, and radioactive materials has been reported in Neff (1997). For metals, Neff (1997) noted that Barium (Ba), Cadmium (Cd), Lead (Pb) and Zinc (Zn) typically have the highest enrichment factor (i.e. the ratio of concentration in produced water and that in solution in seawater of approximately on the order of 1000). In the context of ocean discharge of produced waters, the review showed that Cd is generally the most potentially hazardous to the environment, compared with the other three chemicals. Dissolved, ionic Cd is bioavailable and highly toxic to marine organisms: even relatively low concentrations in sediments are considered toxic (Neff 1997). Table 6.8 summarizes available toxicity information of Cd for different species. Figure 6.16 presents the toxicity data from Table 6.8 in a form that is suitable for assessing effects on the ecological community of interest. This figure employs a plotting position approach so that the data can be assumed as a probability distribution of species responses consistent with a methodology presented by Lenwood et al. (1998) and the U.S. EPA (1998).

Other specific chemicals associated with produced water, which have not been assessed in detail in the Terra Nova environmental impact statement for their potential ecological effects, are radioactive materials (radionuclides). Radionuclides are known to occur in produced water with typically very high enrichment, which can be up to about 6000 (Neff 1997). In addition to other decay products, ^{226}Ra, ^{228}Ra and ^{210}Pb may be expected in produced water at relatively high concentration compared with other radioactive materials (Meinhold et al. 1996b, Neff 1997).

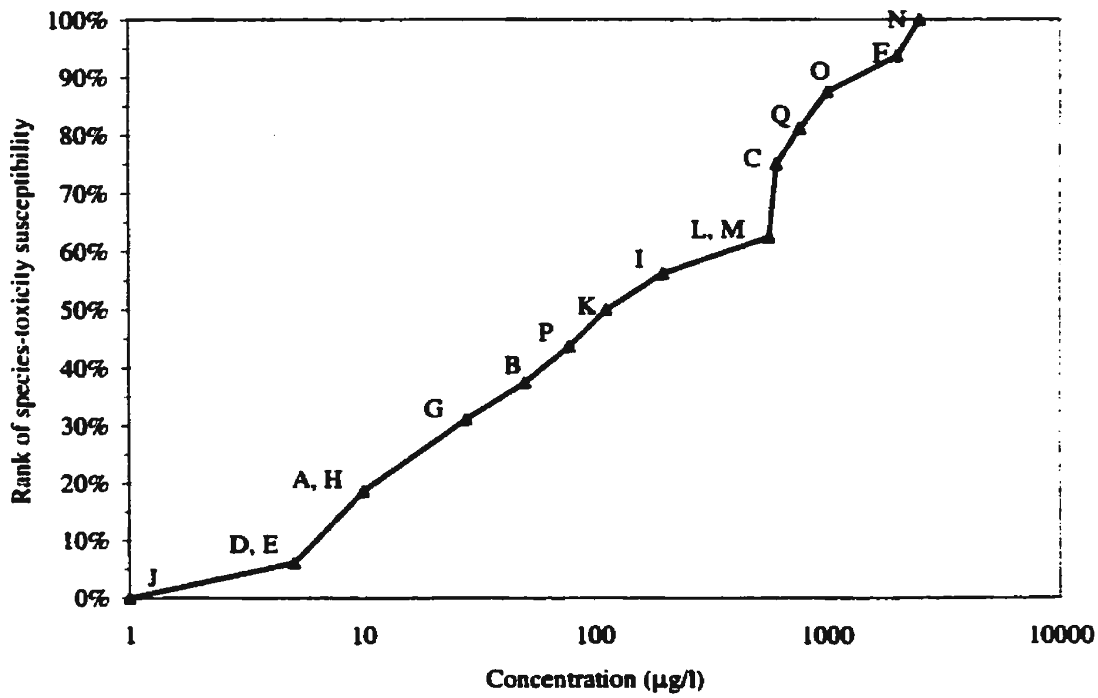


Figure 6.16. Plotting position for the toxicity data summarized in Table 6.8 below

Table 6.8. A summary of available data on ecological effects of Cd on different species (adapted from Kennish 1997, Middleditch 1984)

Code	Sublethal effects on various organisms	Concentration (µg/l)
A	Copepod reduction reduced	10
B	Decapod larval development retarded	50
C	Enchinoderms (arrested fertilization and development of sea urchin eggs)	600
D	Fish (reduced fertilization of *Spring-spawning herring*)	5
E	Fish (reduced growth of *Pleuronectes platessa*)	5
F	Fish hatch rate decreased	2000
G	Hydroid growth rate reduced	28
H	Hydroids (altered hydranth morphology of *Eirene viridula*)	10
I	Hydroids (reduced growth of *Campanularia flexuosa*)	195
J	Phytoplankton (reduced ^{14}C fixation)	1
K	Phytoplankton (reduced growth)	112
L	Polychaete reproduction suppressed	560
M	Polychates (reductions in reproduction of *Capitella capitata*)	560
N	Polychates (reductions in reproduction of *Ctenodrilus serratus*)	2500
O	Polychates (reductions in reproduction of *Neanthes arenaceodentata*)	1000
P	Scallop growth rate reduced	78
Q	Shrimp gills blackened	760

Radioactivity has been 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 ray. The unit of activity has historically been the Curie (Ci), in which one Ci is equal to 3.7×10^{10} disintegrations per second. In the SI system, the unit has been redefined as one disintegration per second, known as the Becquerel (Bq). One Curie is then equal to 3.7×10^{10} Bq.

Exposure to ionizing radiation can result in injury at the molecular, cellular and whole body level (Meinhold et al. 1996c). The exposure to radionuclides is related to the absorbed dose and dose rate. The absorbed dose is a measure of the energy imparted to matter and has the SI unit of the Gray (Gy, 1 Joule/kg). The ecological effects depend not only on the absorbed dose but also the type and energy of radiation. Radiation weighting factors are used to account for the differences in biological effectiveness of different radiation, such as gamma and alpha particles. The absorbed dose modified by the weighting factor is referred to as the equivalent dose expressed in units of Joule/kg or Sievert (Sv). Meinhold et al. (1996b, 1996c) have reviewed that thresholds of effects on aquatic biota are expected at the equivalent dose of 0.004, 1, and 10 mSv/hr for no adverse effects, reduced reproductive success, and increased mortality threshold, respectively. Based on the International Atomic Energy Agency (IAEA) dose conversion factors for typical fish species in ^{226}Ra contaminated water (Meinhold et al. 1996c), these doses are associated with radium concentration in the water of 0.783, 195.65, and 1956.52 pCi/l for no adverse effects, reduced reproductive success, and increased mortality threshold, respectively.

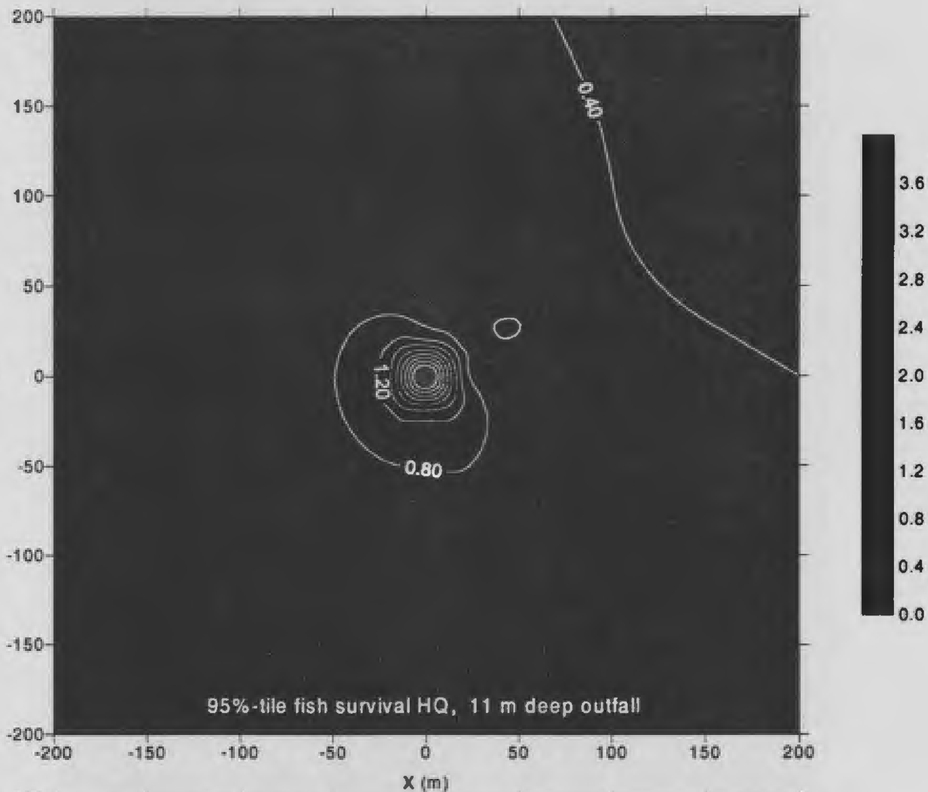
6.4.5 Characterizing ecological risks

Characterization of ecological risks is based on the integration of the analysis of exposures and ecological effects. The methods for characterizing ecological risks have been discussed in Chapter Five (Section 5.3.3) and include qualitative and quantitative methods. For the purpose of evaluating different design scenarios, the quantitative methods were employed in this case study in a number of ways, including (1) the hazard quotients based on the chronic benchmark concentrations of the whole effluent toxicity to individual species, (2) the exceedance probability of the critical hazard quotient based on the whole effluent toxicity approach, (3) the hazard quotients based on the no effect radium concentration (chemical specific approach), (4) the exceedance probability of the no ecological effect threshold of the chemical specific (radium), (5) and the protection level in terms of percentage of ecological species affected as well as associated probability, based on the chemical specific (cadmium) toxicity approach. For each of the above-mentioned analyses, probability distributions of the parameters in the hydrodynamic models, i.e. model inputs and coefficients, were taken from Table 4.2. As discussed in the previous section, the discharge scenario #2, uniformly distributed relative density difference was considered.

For the case study at hand, the first characterization of risks was performed by evaluating hazard quotients (HQs) associated with each design scenario. The characterization was based on the chronic benchmark concentration of the whole effluent toxicity to individual fish and shrimp species. From the analysis of ecological effects, measurement endpoints of the survival NOECs for fish (Sheephead minnows) and shrimp (Mysids) were considered in the analysis. From Table 6.6, the mean and standard deviation of the survival NOEC are 7.1% and 5.7%, respectively, for fish, and 2.9% and 2.9%, respectively, for shrimp. The analysis presented in Chapter Five shows that the toxicity data (Mysid LC_{50}) was lognormally distributed. Neither raw data on the NOECs nor their probability distributions were available in this study; it was therefore assumed that the survival NOECs were also lognormally distributed. Using these inputs, MC simulations were performed and results of the HQs associated with each design scenario are shown in Figures 6.17 to 6.22.

Hazard quotients (HQs) presented in Figures 6.17 to 6.22 can be viewed as "severity measures" of risks as they show how far the exposure concentrations from the specified benchmark concentrations are. When the hazard quotient is more than unity, ecological effects, e.g. fish growth, may be expected. The higher the value of HQ above unity, the more the ecological risks expected. Since uncertainty is taken into account in the analysis, each HQ for a specified point near the discharge location is also uncertain and those figures show the statistics of the HQ in terms of 95% and 99%-tiles.

To evaluate the probability that ecological risks may be expected, probabilistic analysis was performed to calculate the exceedance probability of HQs more than unity. An exceedance probability is the probability of the exposure concentrations exceeding the chronic benchmark concentrations of the whole effluent toxicity to individual fish and shrimp species. For the case study at hand, exceedance probabilities associated with each design scenario are presented in Figures 6.23 to 6.28.



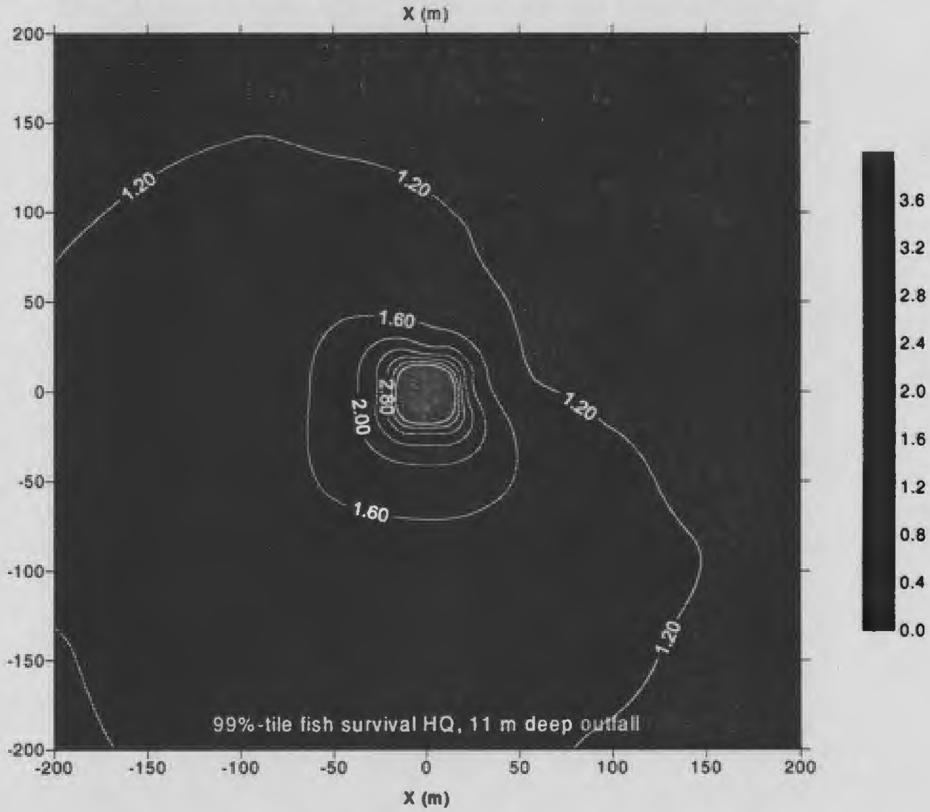


Figure 6.17. Whole effluent chronic hazard quotients, design #3 (Fish survival risks: top is 95%-tile HQ and bottom is 99%-tile HQ)

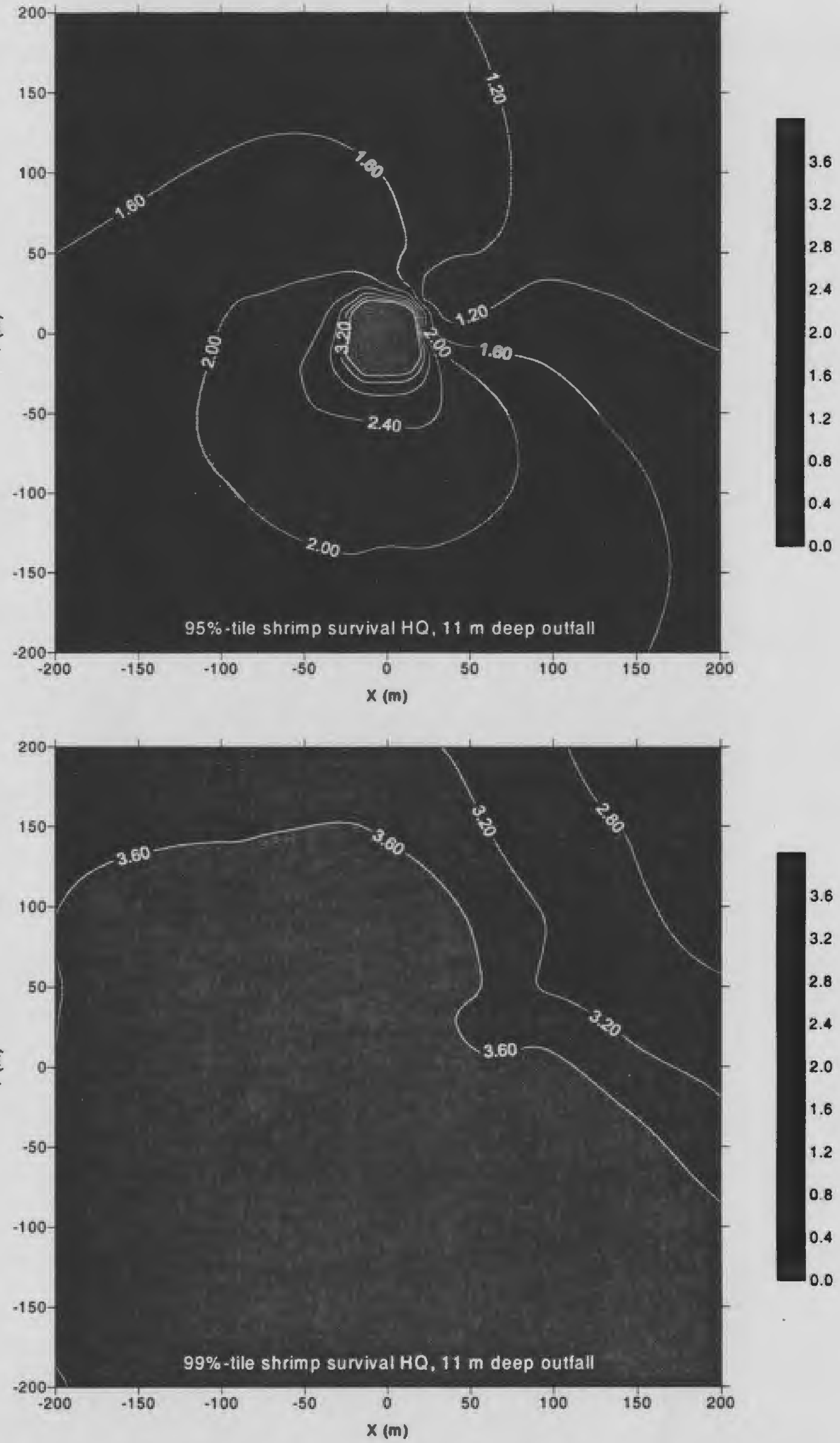


Figure 6.18. Whole effluent chronic hazard quotients, design #3 (Shrimp survival risks: top is 95%-tile HQ and bottom is 99%-tile HQ)

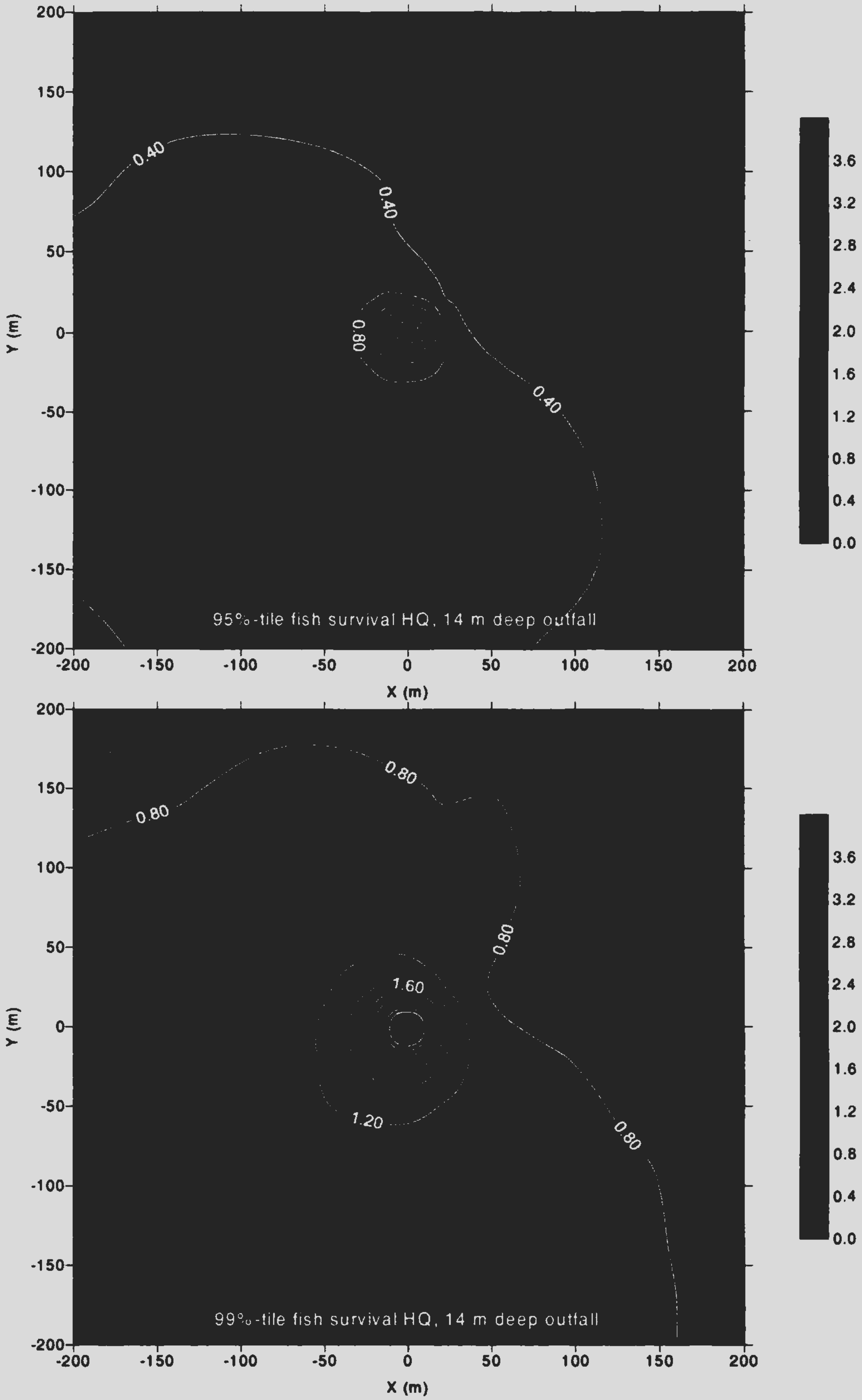


Figure 6.19. Whole effluent chronic hazard quotients, design #4
(Fish survival risks: top is 95%-tile HQ and bottom is 99%-tile HQ)

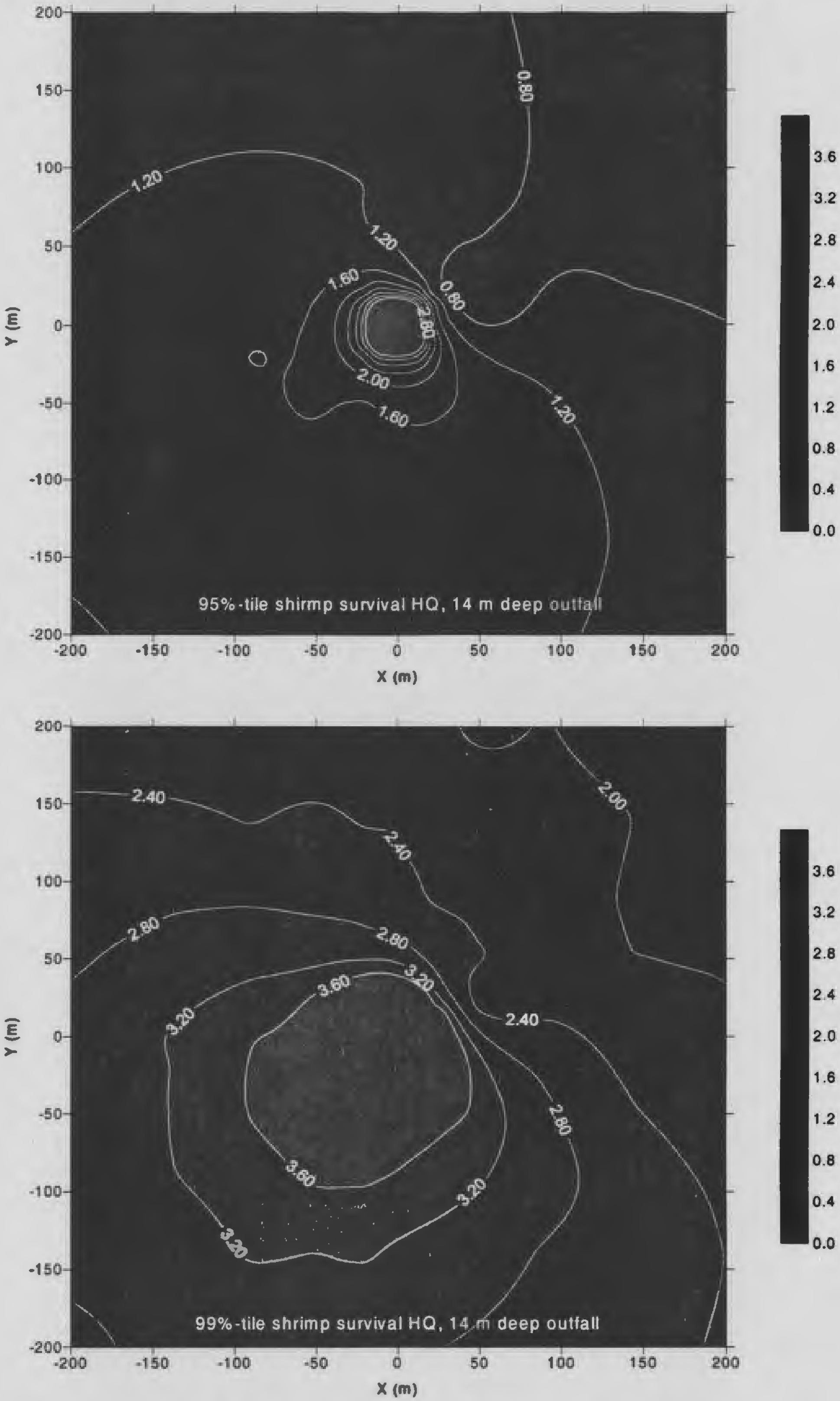


Figure 6.20. Whole effluent chronic hazard quotients, design #4 (Shrimp survival risks: top is 95%-tile HQ and bottom is 99%-tile HQ)

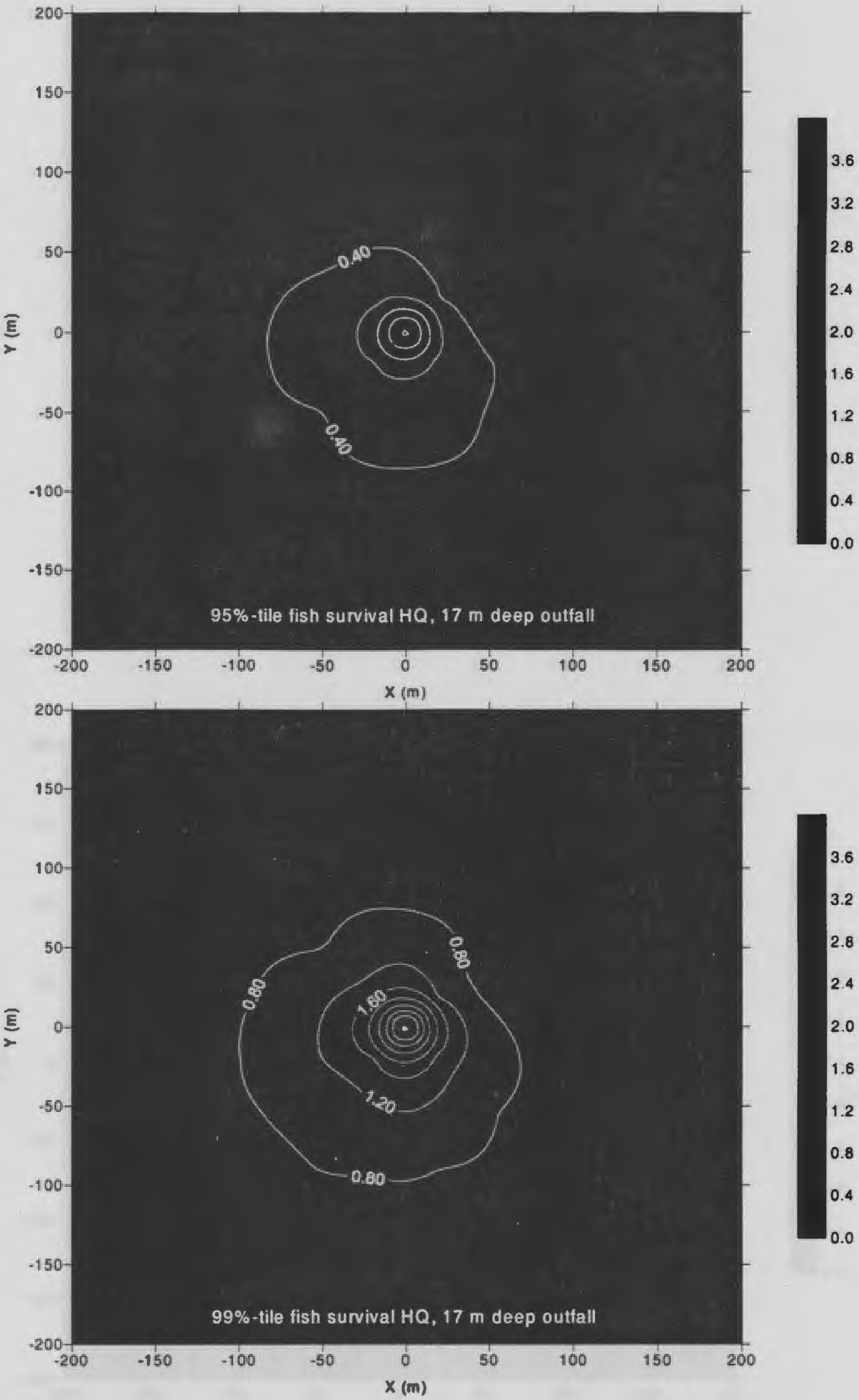
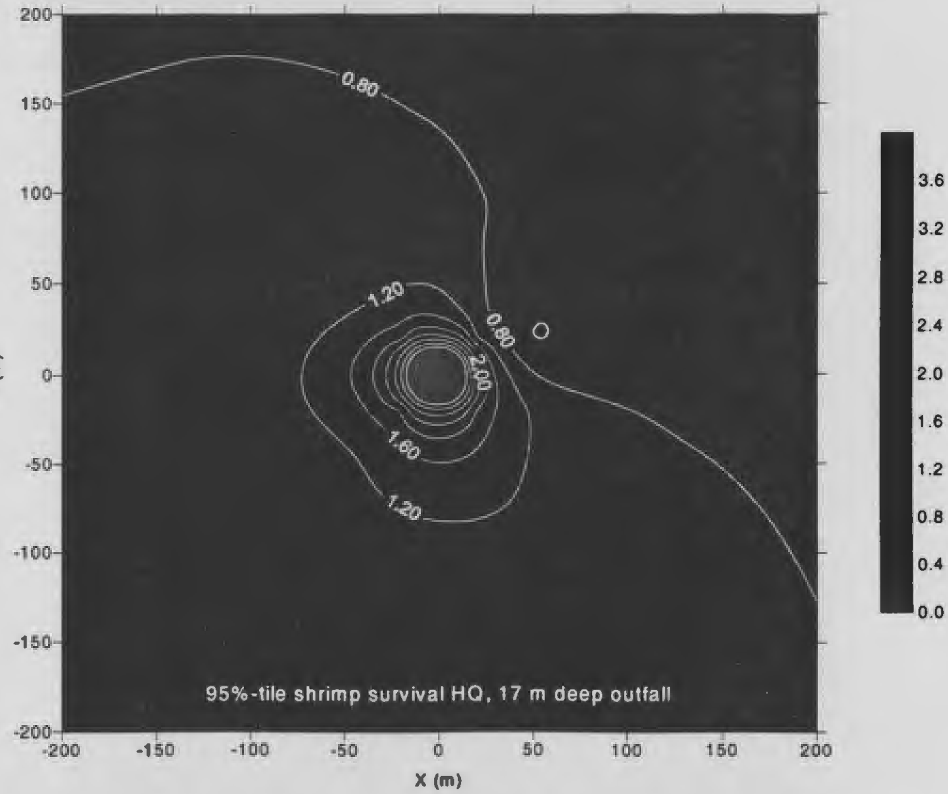


Figure 6.21. Whole effluent chronic hazard quotients, design #5
(Fish survival risks: top is 95%-tile HQ and bottom is 99%-tile HQ)



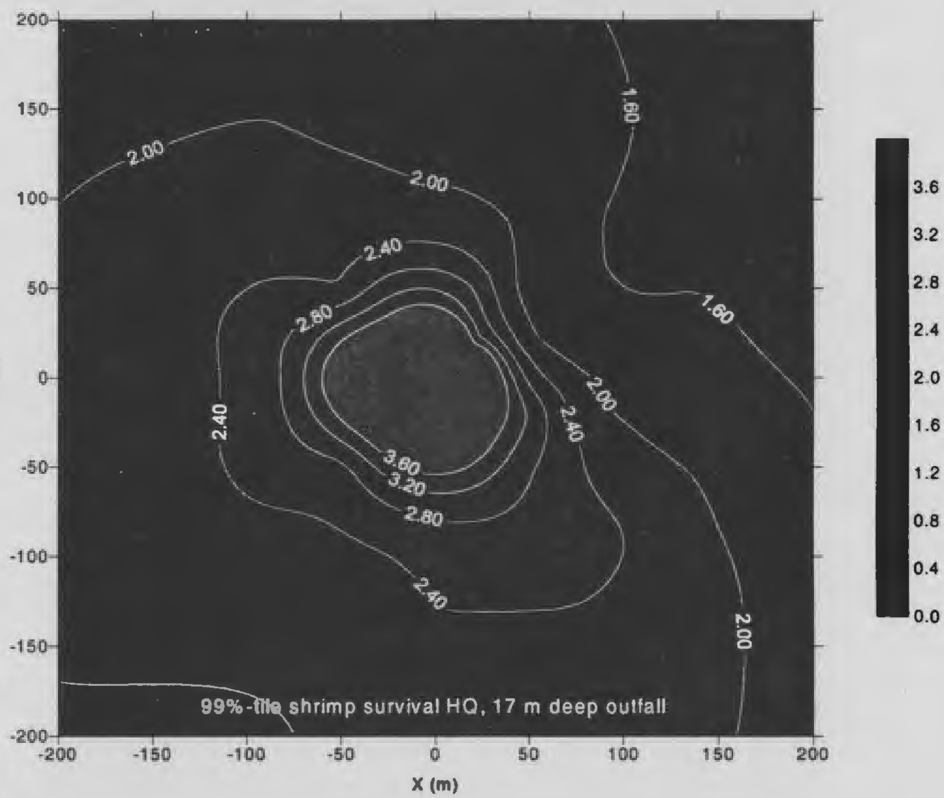


Figure 6.22. Whole effluent chronic hazard quotients, design #5
(Shrimp survival risks: top is 95%-tile HQ and bottom is 99%-tile HQ)

Figure 6.23. Exceedance probability (%) of the whole effluent chronic benchmark (Fish survival risks, design # 3)

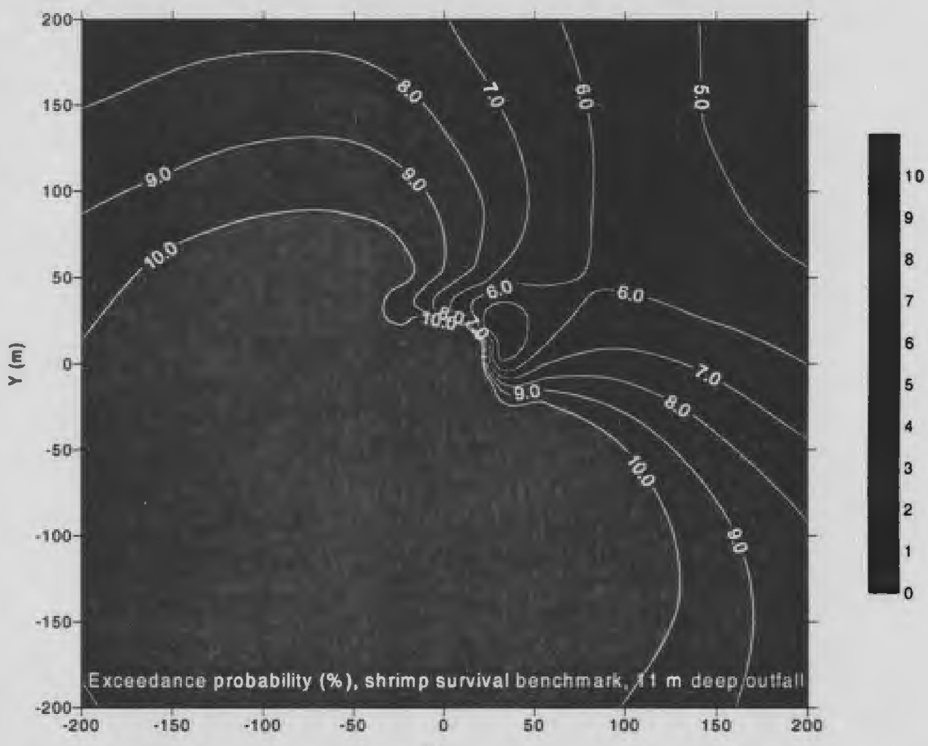


Figure 6.24. Exceedance probability (%) of the whole effluent chronic benchmark (Shrimps survival risks, design # 3)

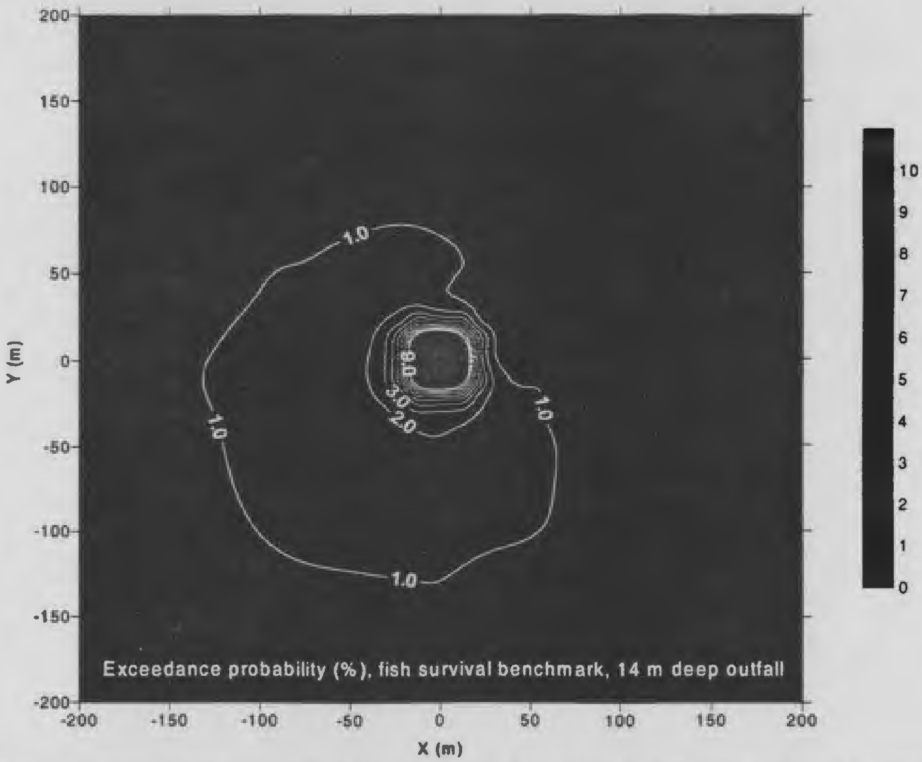


Figure 6.25. Exceedance probability (%) of the whole effluent chronic benchmark (Fish survival risks, design # 4)

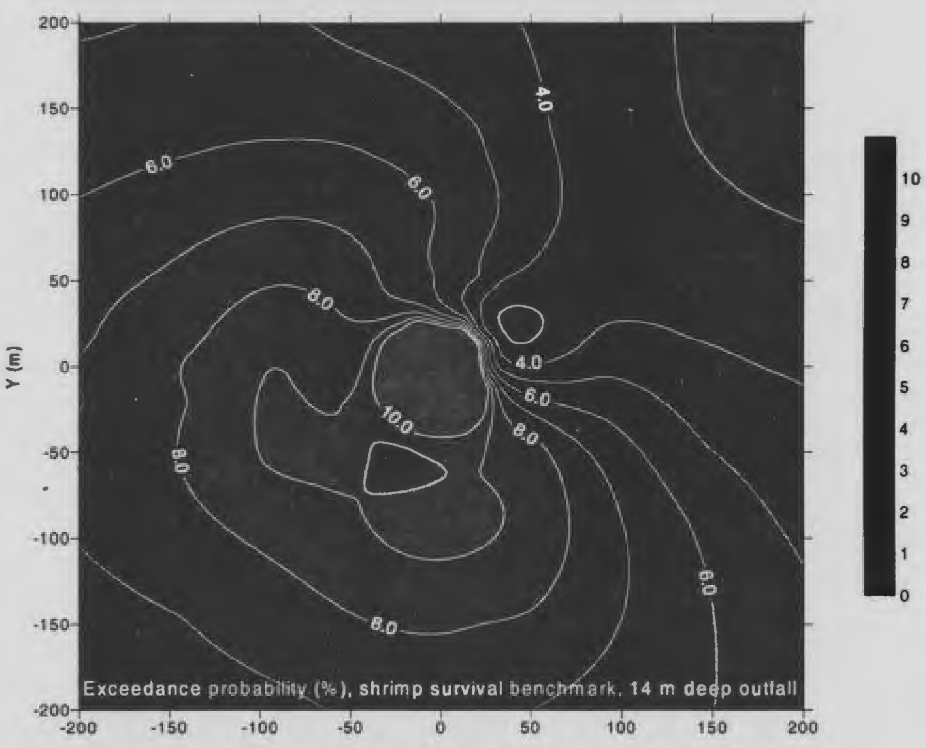


Figure 6.26. Exceedance probability (%) of the whole effluent chronic benchmark (Shrimp survival risks, design # 4)

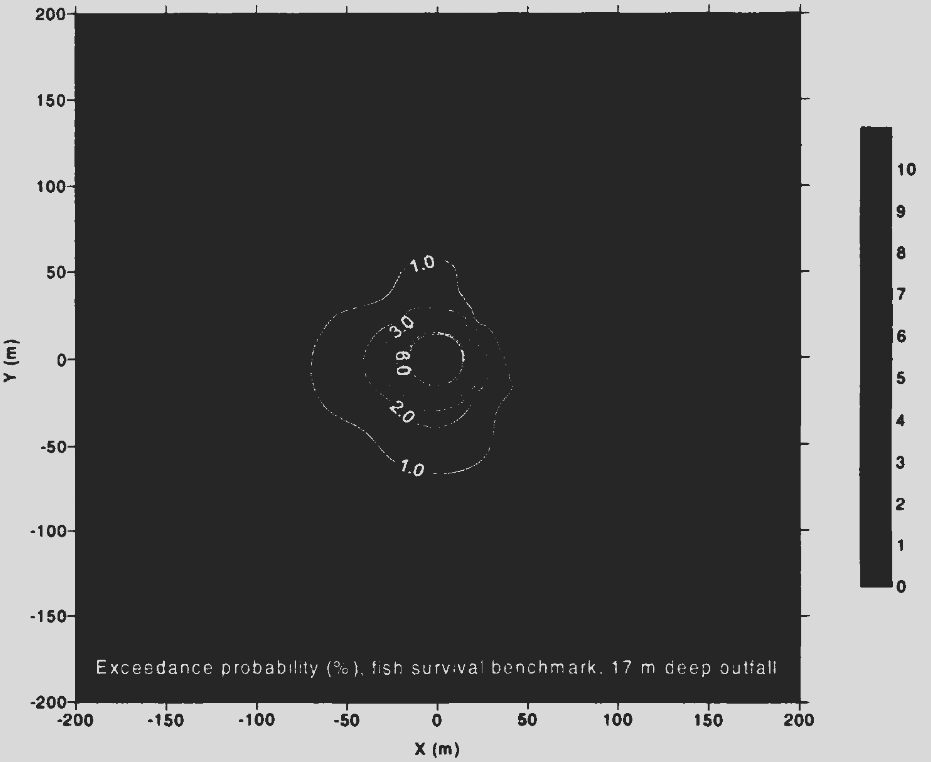


Figure 6.27. Exceedance probability (%) of the whole effluent chronic benchmark (Fish survival risks, design # 5)

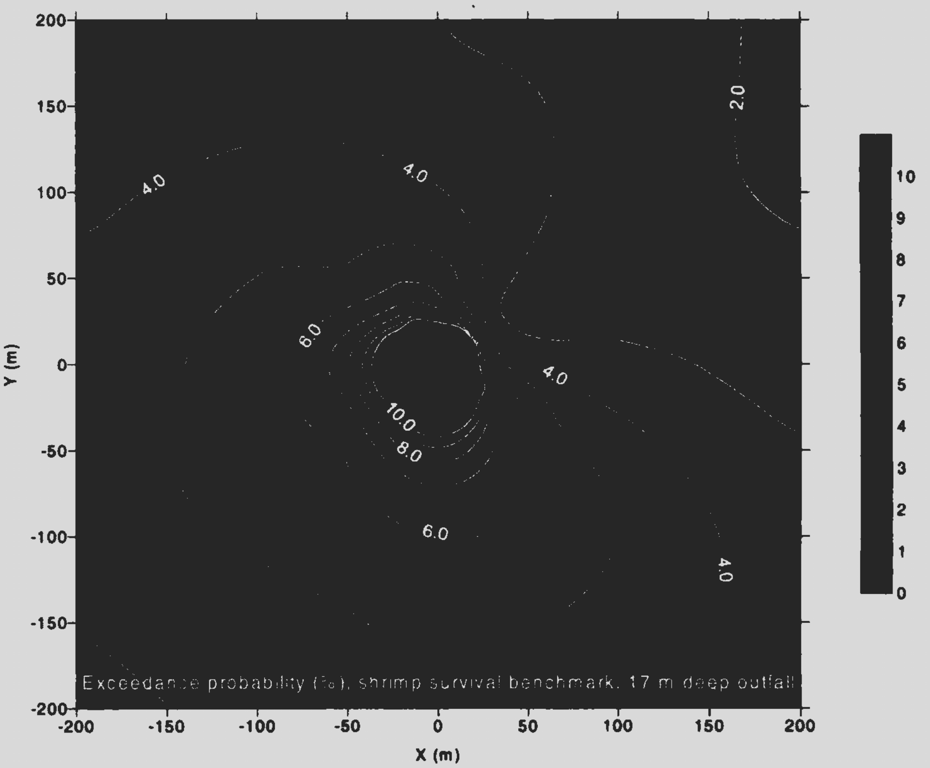
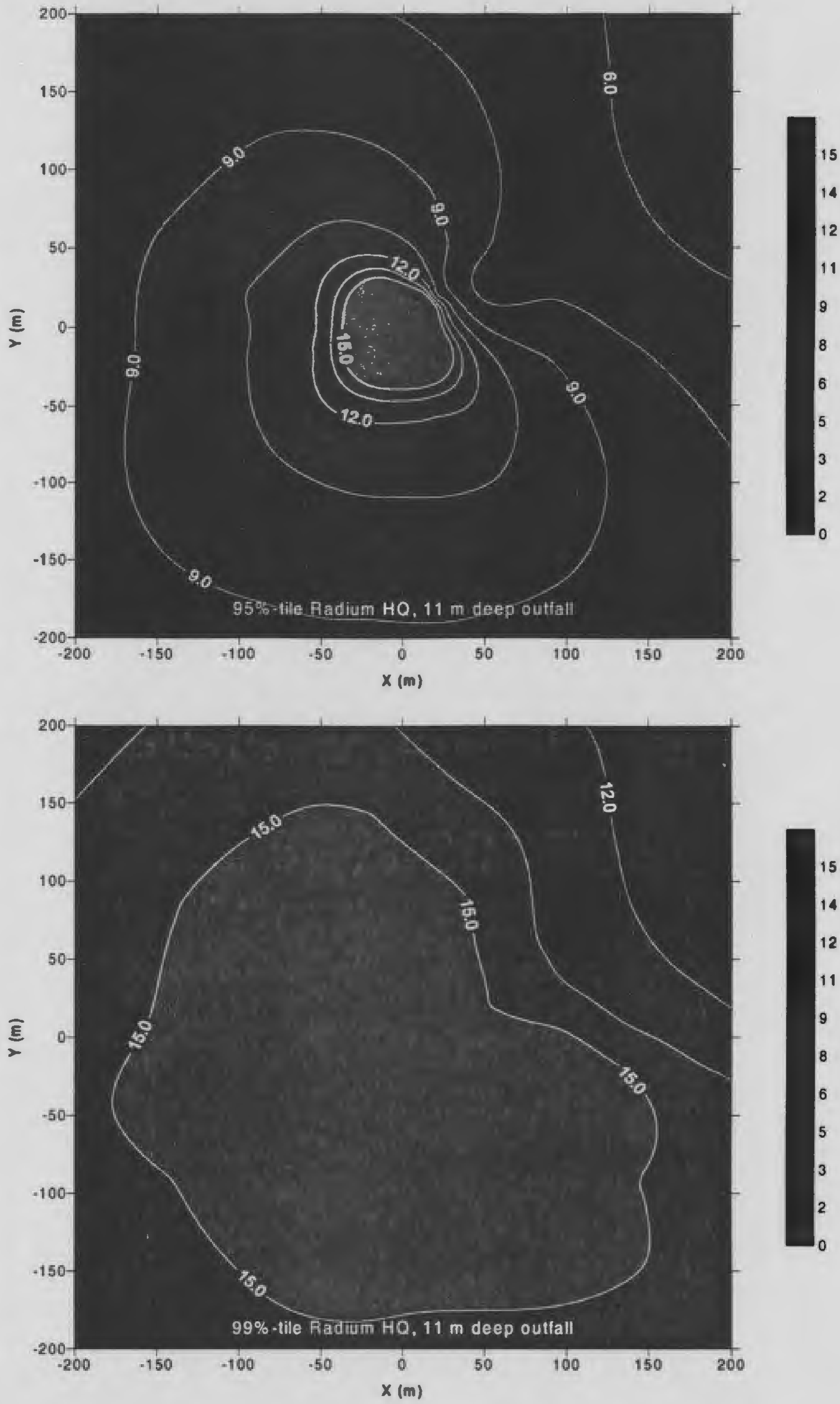


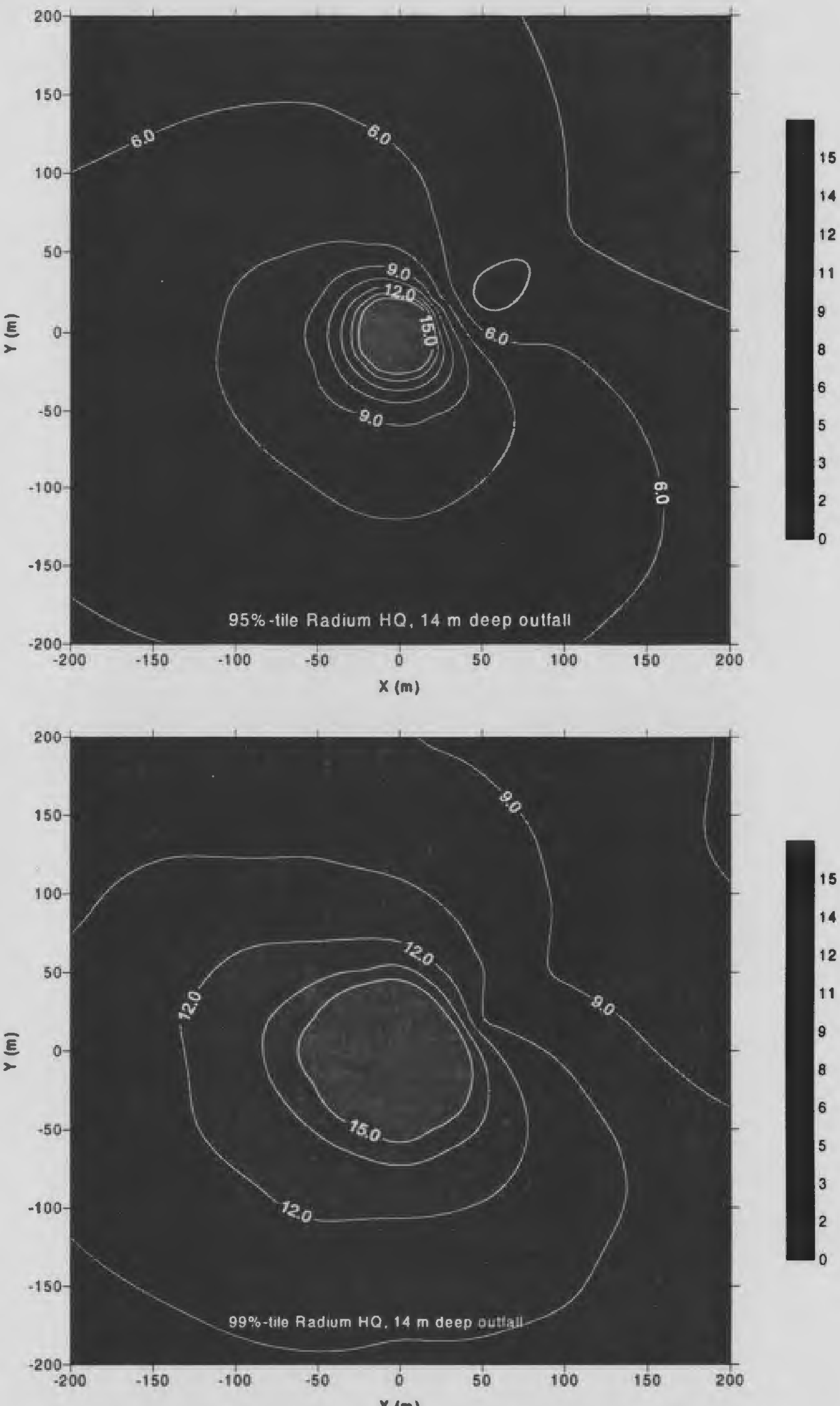
Figure 6.28. Exceedance probability (%) of the whole effluent chronic benchmark (Shrimp survival risks, design # 5)

In the previous approaches of risk characterization, the whole effluent toxicity data were employed by using the chronic fish and shrimp survival NOECs. Ecological risks may also be assessed from chemical specific basis. In the previous subsection, the analysis of ecological effects has identified two specific chemicals, namely radium (^{226}Ra) and cadmium (Cd). In this third approach of risk characterization, the hazard quotients (HQs) associated with ^{226}Ra were evaluated by using the ratio of the exposure concentrations to the ^{226}Ra benchmark concentration. As discussed in the analysis of ecological effects, the ^{226}Ra threshold concentrations for bathypelagic fish are 0.78, 195.65, and 1956.52 pCi/l for no adverse effects, reduced reproductive success, and increased mortality threshold, respectively.

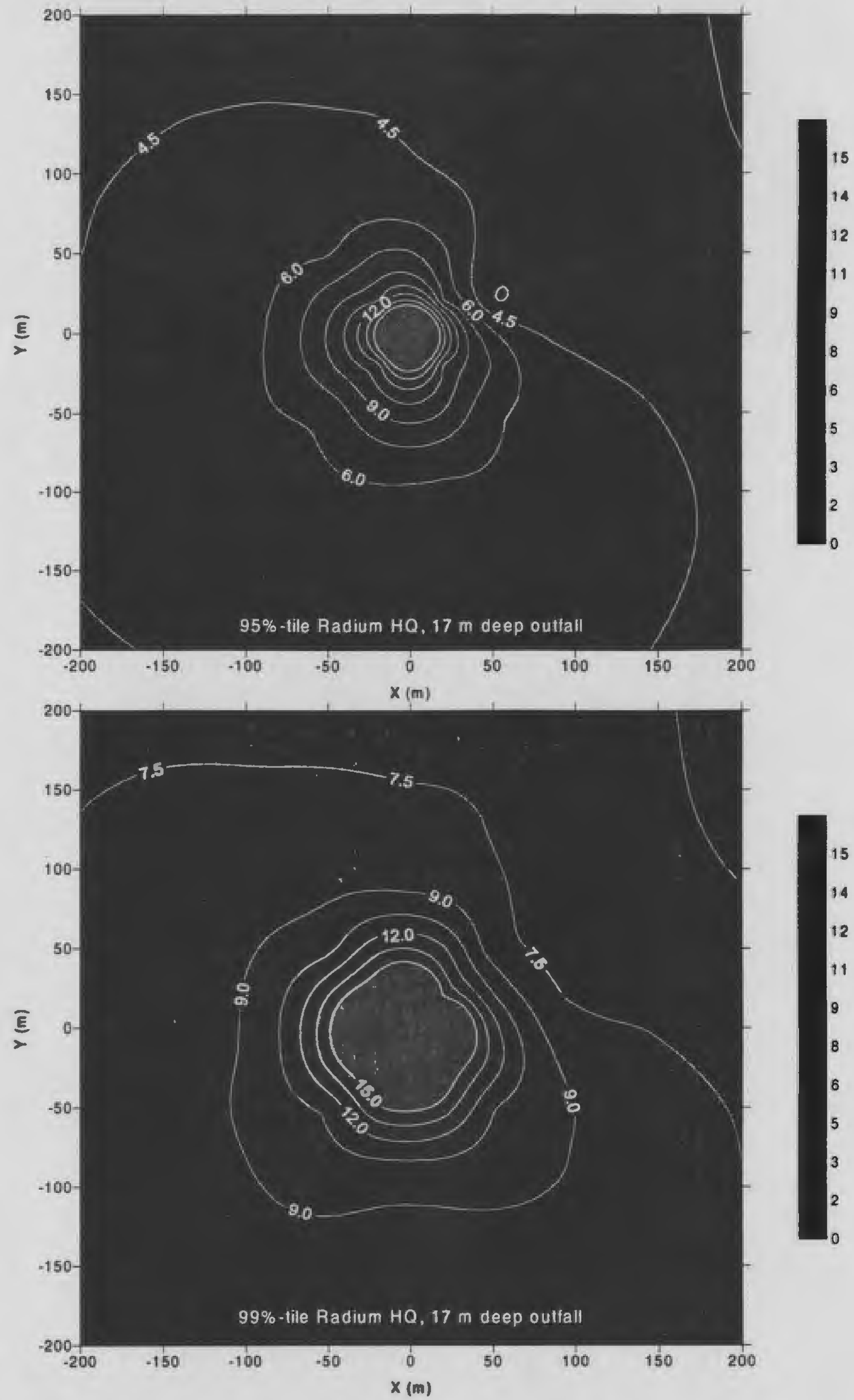
No estimate of radium concentration is available for the Terra Nova project. If radium concentrations from other oil fields are used as estimates, ^{226}Ra concentration may be assumed to be in the range of 4 to 584 pCi/l, with the average of 262 pCi/l, based on produced water from 42 oil production platforms from the Gulf of Mexico (shown in Table 5.2, Chapter Five). With this limited information, radium concentration may be typically assumed to follow a triangular distribution, with a range from 4 to 584 pCi/l and the most likely value of 262 pCi/l. From these ^{226}Ra concentrations, no fish mortality risk may be expected. Results of the MC simulations showed that risks to the fish reproductive success were also negligible since, for example, the maximum concentration at the water surface above discharge point, O(0,0), for the design #3 (11 m deep outfall) is typically about 98 pCi/l or only about half of the reproductive success threshold. If, however, no adverse effect threshold was adopted, potential effects were noticed (Figures 6.29 to 6.34).



6.29. Chemical specific, ^{226}Ra hazard quotients, design #3.
(Risks on fish: top is the 95%-tile HQs and bottom is 99%-tile HQs)



6.30. Chemical specific, ^{226}Ra hazard quotients, design #4.
(Risks on fish: top is the 95%-tile HQs and bottom is 99%-tile HQs)



6.31. Chemical specific, ^{226}Ra hazard quotients, design #5 (Risks on fish: top is the 95%-tile HQs and bottom is 99%-tile HQs)

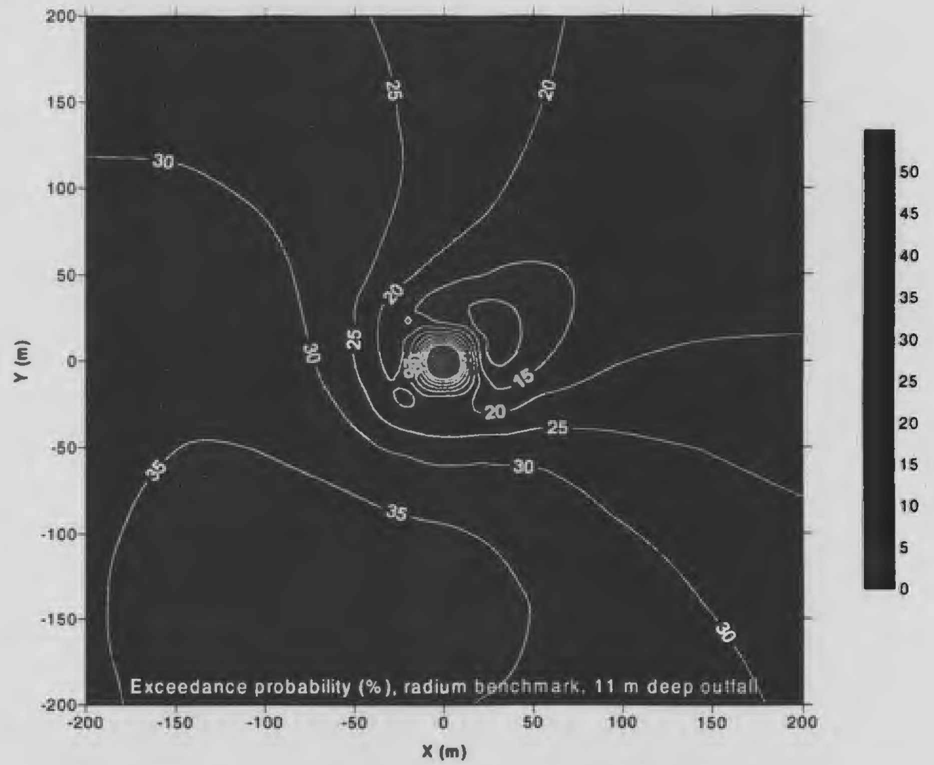


Figure 6.32. Exceedance probability (%) of the ^{226}Ra benchmark, design #3

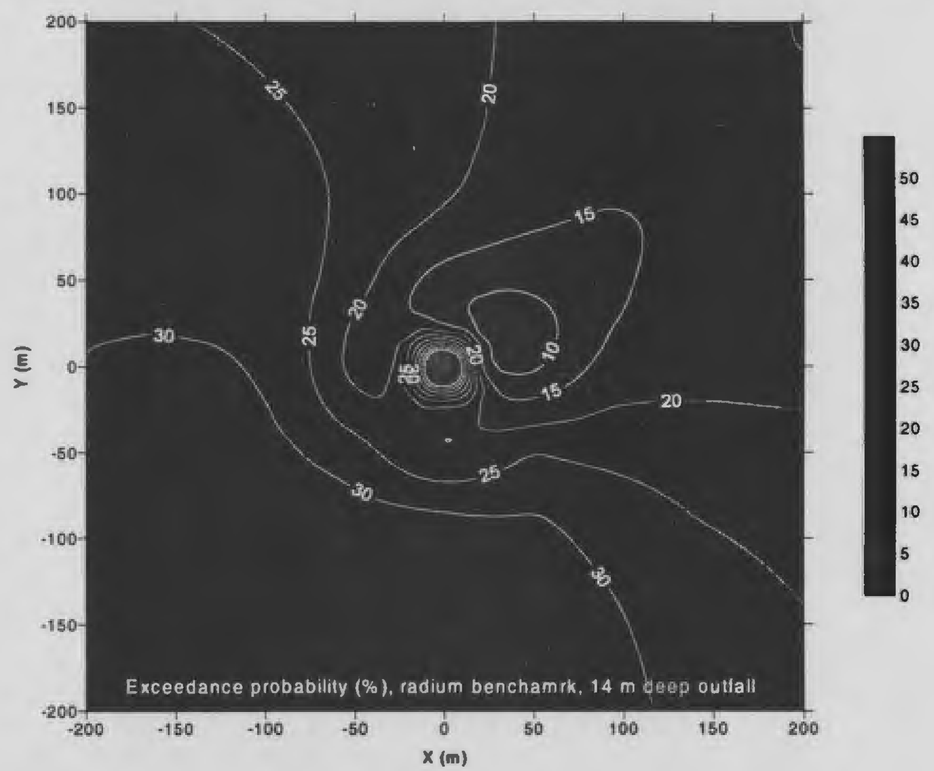


Figure 6.33. Exceedance probability (%) of the ^{226}Ra benchmark, design #4

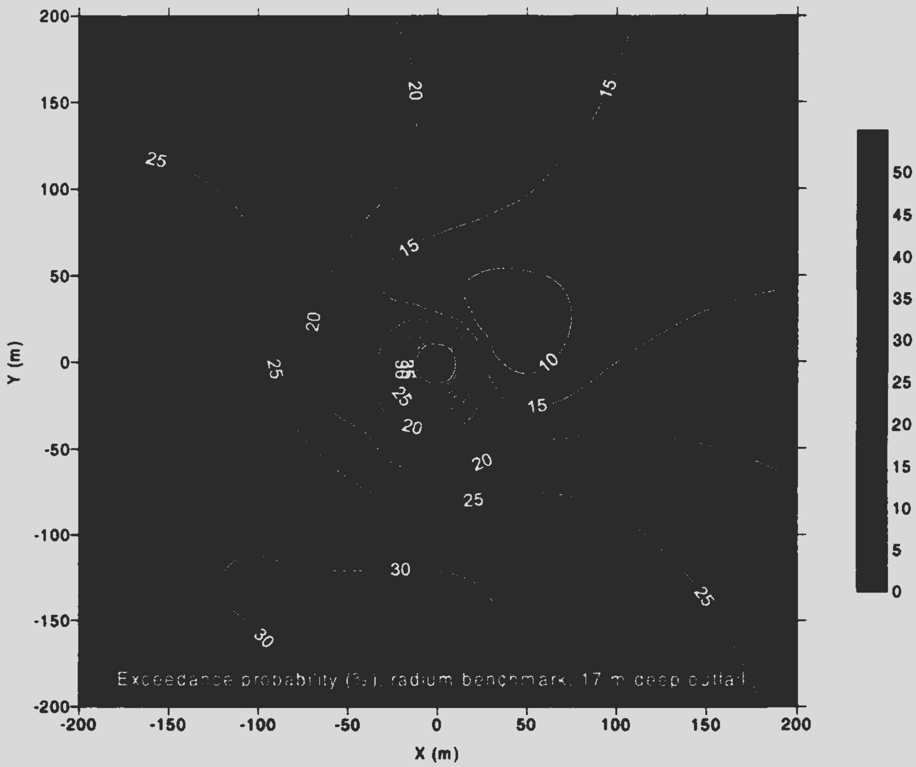
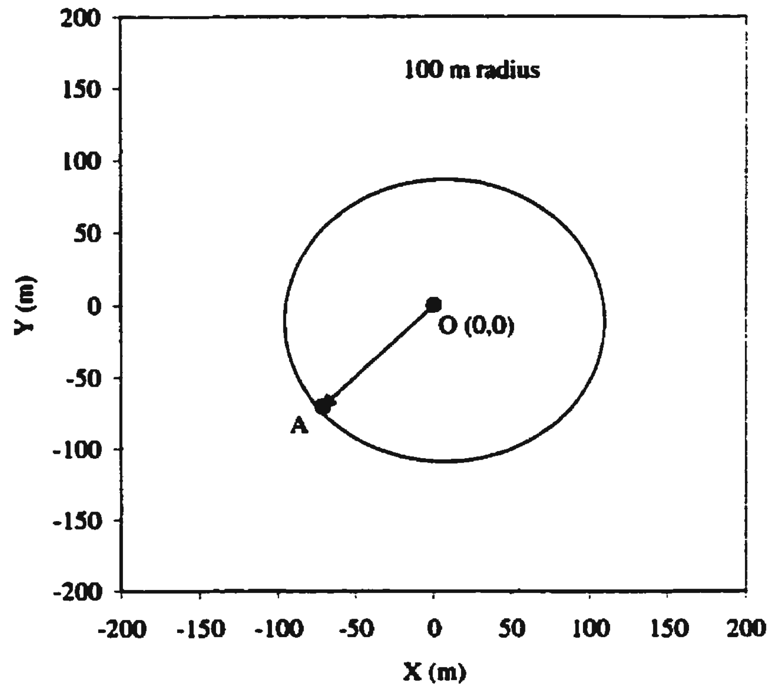


Figure 6.34. Exceedance probability (%) of the ^{226}Ra benchmark, design #5

In the last approach of risk characterization, the protection level in terms of the percentage of ecological species affected as well as associated probability are evaluated for each design scenario. The evaluation is based on data of the chemical specific toxicity data in Table 6.8 presented in the previous subsection. Information on estimates of cadmium concentration for the Terra Nova project is not available for this study. If it is assumed that cadmium concentrations reported from the North Sea based on six discharges from offshore platforms (Table 5.2, Chapter Five) can be used as estimates for the case study, cadmium concentrations may be assumed to follow a triangular distribution with a range from 20 to 10,000 µg/l with the most likely value of 6670 µg/l. As a metal, cadmium may leach so that the concentration in the water column may be expected to be lower than that calculated based on the total concentration. To take leaching into account, a leach factor of 0.11 (U.S. EPA 1999b) was used in this analysis.

To illustrate the procedure of evaluating the protection level, consider a typical point A located at a 100 m radius from the origin O(0,0) as shown in Figure 6.35 (top). For this particular point, exposure concentrations associated with each design scenario can be obtained from MC simulations. Then, the exposure concentrations are presented in terms of a cumulative distribution and are superimposed with the toxicity data associated with cadmium (Figure 6.16, the previous subsection). A typical presentation for evaluating the protection level at the point A is shown in Figure 6.35 (bottom). From Figure 6.35 (bottom), it can be seen that, for example, scenarios #3, #4, and #5 can be associated with the protection from the violation of the 95% of the species toxicity thresholds for 69.8%, 75.5%, and 79.8% probability, respectively. This means that there is an exceedence probability of 30.2%, 24.5%, and 20.2%, respectively.

The procedure presented here is similar to that from Lenwood et al. (1998) and the U.S. EPA (1998). In this case study, the protection level for each point of interest near the discharge was considered and a probability field exceeding a specified protection level was drawn based on results of the MC simulations. If, for example, protection of 95% of the species toxicity thresholds is of concern, the exceedance probability field for whole region of interest can be estimated as shown in Figures 6.36 to 6.38.



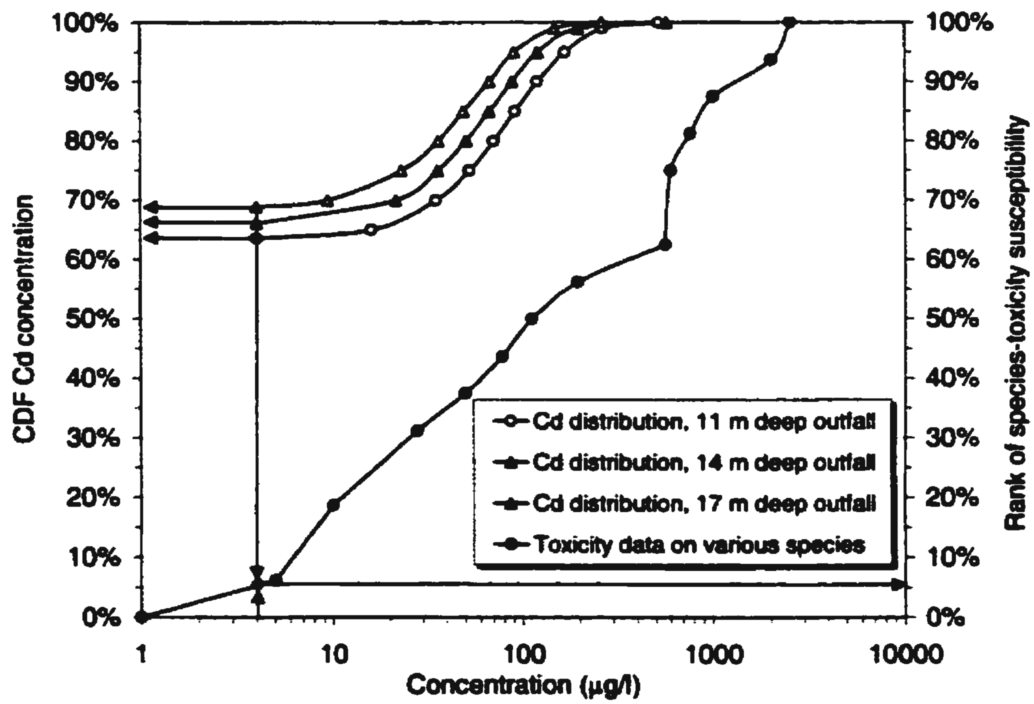


Figure 6.35. Cadmium protection level (the 95% aquatic species toxicity thresholds) and associated probability: Location of a spot A (top) and the protection level (bottom)

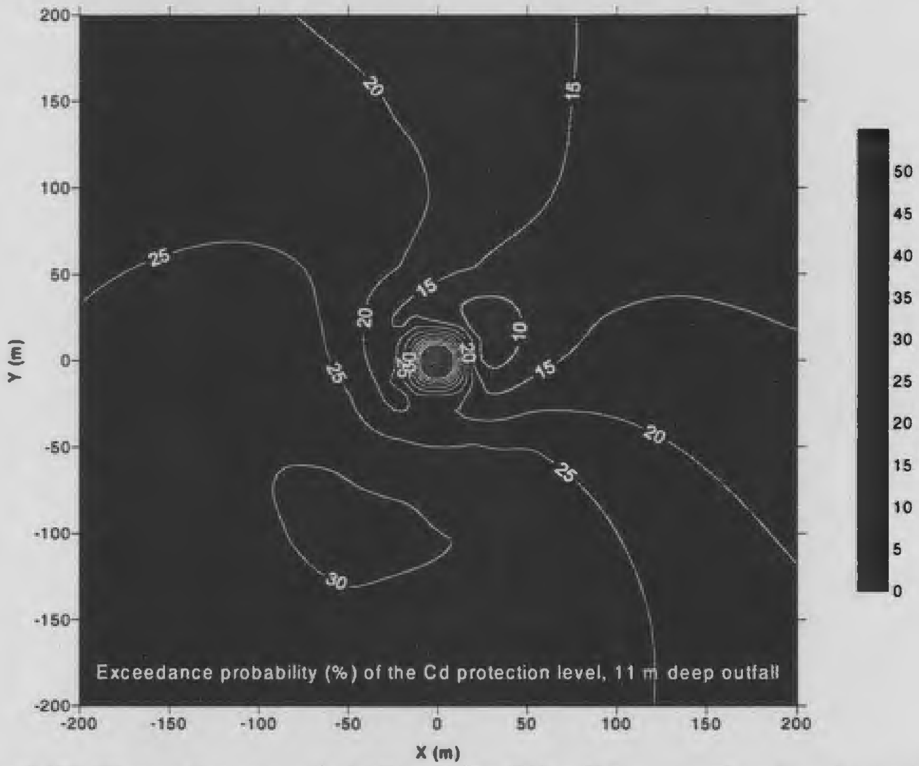


Figure 6.36. Exceedance probability (%) of the Cd protection level, design #3

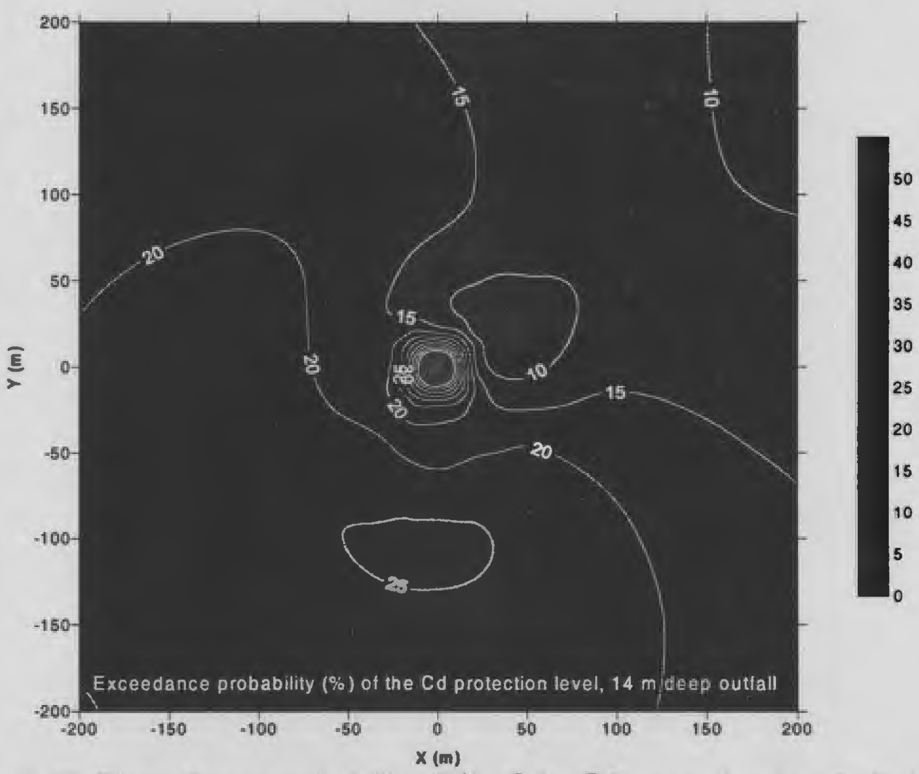


Figure 6.37. Exceedance probability (%) of the Cd protection level, design #4

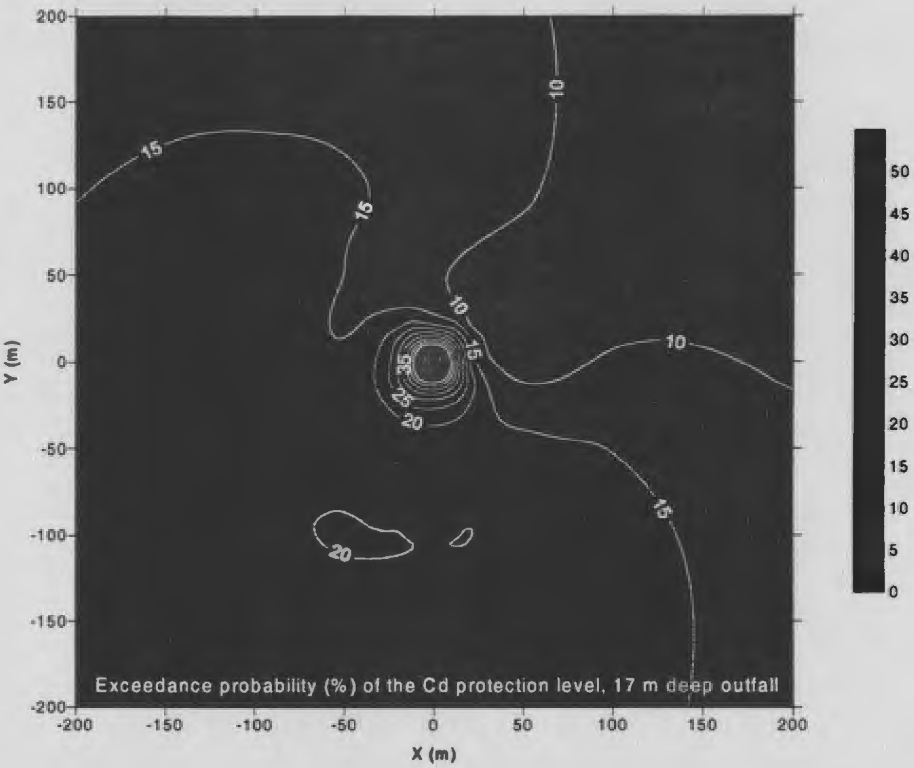


Figure 6.38. Exceedance probability (%) of the Cd protection level, design #5

As discussed, characterization of ecological risks was performed in various ways and the results were shown in Figures 6.17 to 6.38. From these figures, in general, the choice of the design scenario directly affects the extent of the potential ecological risks associated with the produced water discharge. This makes it possible to provide design recommendations on the basis of ecological risks. More detailed discussion of effects of the design scenarios on the potential ecological risks is presented in the next subsection.

It should be noted here that ecological risks associated with each design scenario described above are subject to uncertainty. Uncertainty associated with hydrodynamic modeling has been quantitatively taken into account as discussed in Chapter Four. Another uncertainty is associated with assumption used in the modeling. The assumption needs to be considered with care so that they are reasonably suitable for the case under investigation. The case study used several assumptions that directly affect the results of the characterized ecological risks as presented in Figures 6.17 to 6.38 above. These assumptions include the flow rate of the produced water discharge, Cd and ^{226}Ra concentrations in the produced water prior to discharge, relative difference of the produced water and ambient water densities, and the toxicity benchmarks such as fish and shrimp survival benchmark concentrations, ^{226}Ra no adverse effects concentration, and Cd protection level. Use of the results of the analysis may not be possible until these assumptions are accepted.

6.4.6 Discussion and design recommendations

As discussed previously, the problem in an ecological risk-based design it to find design scenarios that may be associated with the least ecological risks. The problem formulation has determined that fish and shrimp survival NOECs may be used as measurement endpoints in the characterization of ecological risks. The potential fish and shrimp risks were evaluated based on the whole effluent toxicity approach. For comparative and discussion purposes, other potential ecological risks were also evaluated to look at potential effects of the radium and the cadmium protection level.

It appears from the risk characterization above that the extent of the ecological effects may be reduced by selecting an appropriate design scenario. In general, the deeper the water above discharge, the less the area potentially associated with ecological effects. For fish, although design #5 is associated with the least survival risks among other designs, HQs for all design scenarios are generally low except in the area very close to the discharge point. For example, Figures 6.17, 6.19 and 6.21 show that the 95%-tile fish survival HQs exceeding unity are observed only in the area less than 500 m^2 (about 40 m in radius). Even the 99%-tile fish survival HQs are reasonably low and are within a typical regulatory mixing zone of radius of about 400 m (Huang et al. 1996). Exceedance probabilities for the fish survival benchmark presented in Figures 6.23, 6.25 and 6.27 reveal the same results.

For shrimp, survival risks are expected to be higher than those for fish because benchmark concentrations for shrimp are lower than those for fish (Table 6.6). These are observed in Figures 6.17 to 6.28 above. Shrimp survival HQs are generally low, particularly those associated with designs #4 and #5. For all design scenarios, the 95%-tile shrimp survival HQs of 2 are within an area of approximately within 31,500 m^2 (about 100 m in radius) as shown in Figures 6.18, 6.20 and 6.22. For design #3, the 99%-tile HQs seem to be higher (Figure 6.18) and the exceedance probabilities of 10% are within an area with a radius of about 175 m (Figure 6.24). In design #5, the exceedance probability of 5% is within an area of about 70,000 m^2.

Considering the measurement endpoints defined in the problem formulation, the acceptability of the design scenarios may be evaluated from Figures 6.17 to 6.28 above. However, the acceptability has also to be evaluated based on an allowable area of specified ecological effects and associated exceedance probability. Traditionally, criteria associated with wastewater discharges have been defined by legislation which, in many jurisdictions, leans towards the mixing zone concepts. For example, a maximum allowable surface area of 502,655 m^2 (or equivalent circular area with a radius of approximately 400 m) was defined as a mixing zone in the state of Florida, U.S.A. (Huang et al. 1996). The state of Michigan determined a mixing zone with a radius of approximately 300 m (Doneker and Jirka 1990). On the other hand, engineering design of an ocean outfall is currently not explicitly based on ecological risk criteria (although they could be in the future), which are an acceptable way of comparing one outfall to a number of alternatives from the ecological point of view. For example, if an exceedance probability of 10% for fish and shrimp survival HQs at a maximum mixing zone of 150 m in radius are specified as the criteria, then design #4 is acceptably good. The challenge is now to define the criteria for different purposes of ecological protection in different environments. This may be recommended as a potential new research direction and is beyond the scope of this study.

As indicated, design recommendations depend on the ecological protection, which is reflected by use of measurement endpoints as discussed in the problem formulation. For comparative purposes, the risk characterization was also performed using measurement endpoints other than those defined in the problem formulation. These were based on the chemical specific approach, i.e. the ^{226}Ra no adverse ecological effect and Cd protection level. As discussed, no fish mortality risk associated with ^{226}Ra may be expected in this case study, and risks to the fish reproductive success were also negligible. However, if the ^{226}Ra no adverse ecological effect was defined as the measurement endpoint, Figures 6.29 to 6.34 show that none of the design scenarios are acceptable based on the exceedance probability of 10% at a maximum mixing zone of 150 m in radius in the above example. In this case, the design may be modified by, for example, adopting a diffuser type of outfall.

The following case shows another example how recommendations may be given using the Cd protection level based on available toxicity data. If the protection of 95% of the species toxicity thresholds is of concern, it appears from Figures 6.36 to 6.38 that the exceedance probability of the specified protection level may typically be considered to be high and no design scenario satisfies the exceedance probability of 10% at a maximum mixing zone of 150 m in radius in the above example.

As shown in Figure 6.35, the protection level in this typical analysis was associated with a threshold Cd concentration of about 4 μg/l, at which only the toxicity benchmark of J (reduced ^{14}C fixation of phytoplankton) is exceeded (Table 6.8). MC simulations were also carried out to evaluate the exceedance probability when the protection level was lowered to the 90% toxicity benchmarks. At this protection level, the toxicity benchmarks of D, E and J (reduced fertilization of *Spring-spawning herring*, reduced growth of *Pleuronectes platessa*, and reduced ^{14}C fixation of phytoplankton) are allowed to be exceeded. With this change, the exceedance probability for designs #3 and #5 are shown in Figure 6.39 and 6.40 below. It can be seen that at this protection level design #5 is acceptable to satisfy the exceedance probability of 10% at a mixing zone of 150 m in radius in the above example.

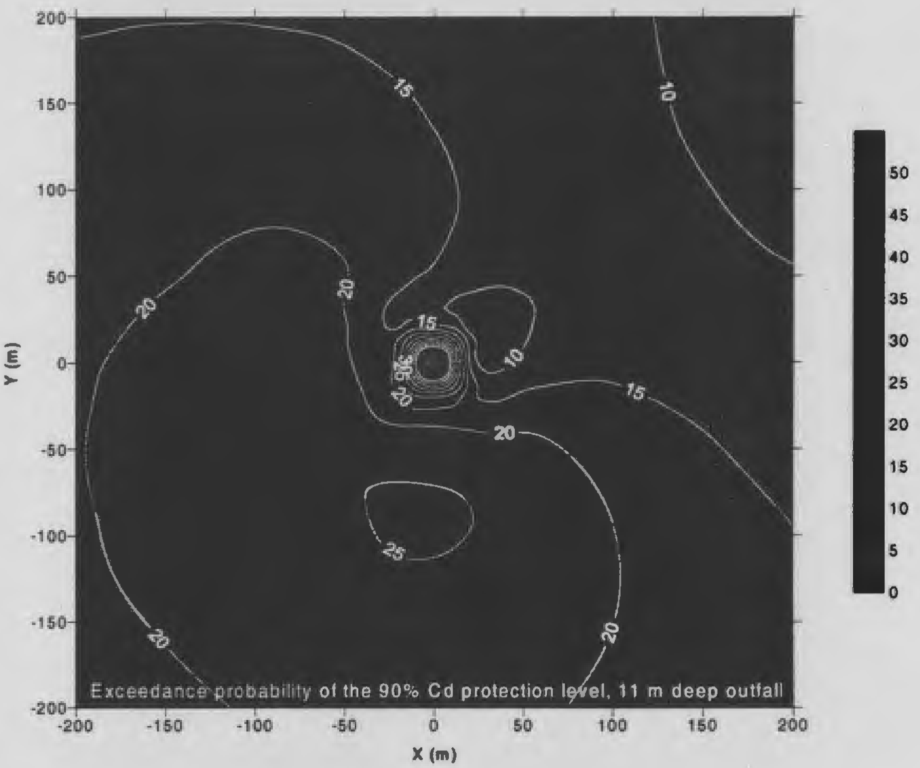


Figure 6.39. Execeedance probability of the 90% Cd protection level (design # 3)

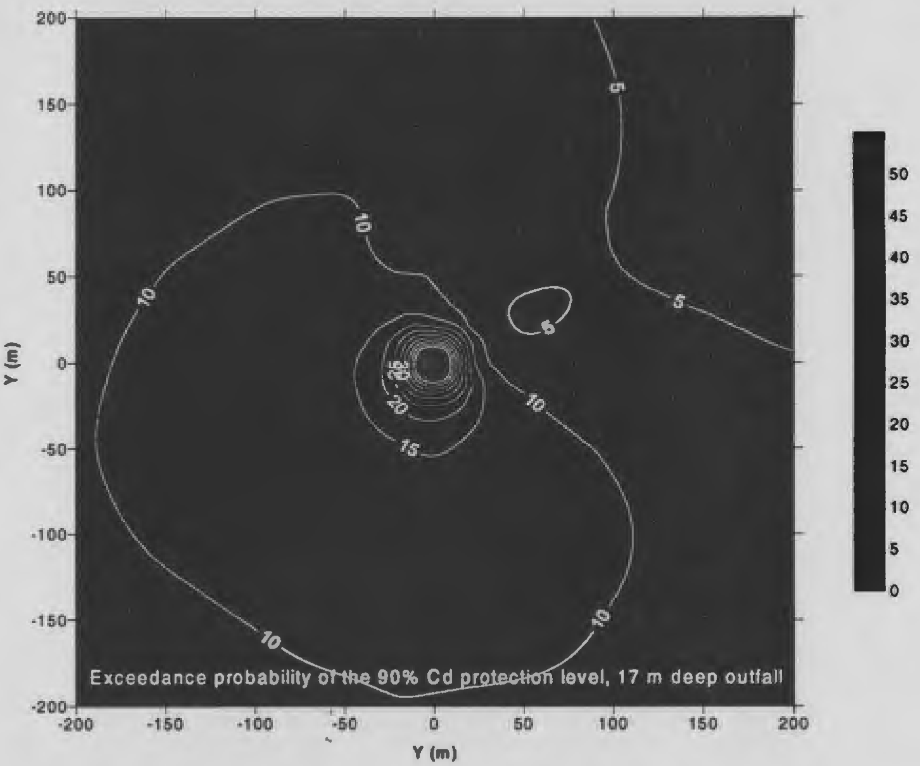


Figure 6.40. Execeedance probability of the 90% Cd protection level (design # 5)

6.5. Summary

A framework for the design on the basis of potential ecological risks is presented in this chapter. The relevance of the framework is highlighted by reviewing traditional outfall design approaches, which are conventionally directed at compliance with relevant water quality standards. The standards are in turn commonly specified upon an epidemiological and ecological viewpoint. As a complementary tool, the framework suggests a possibility that the design of produced water outfalls could itself be looked at from the point of view of the environmental risk from exposure to produced water or specific pollutants associated with the produced water.

The framework of ecological risk-based design is developed by integrating the methodology of hydrodynamic modeling and ecological risk assessment, which has been discussed in Chapters Four and Five. The framework is directed at providing design recommendations on the basis of ecological risk perspectives. The framework is straightforward and consisted of six steps, and is discussed systematically within this chapter by evaluating scenarios of produced water discharge relevant to an offshore oil production platform located on the Grand Banks, southeast of St. John's, Newfoundland, Canada. Instead of providing a solution for a particular problem of an existing oil production platform, the emphasis of the case study is to show how the risk-based design of produced water discharge could be undertaken.

Chapter 7

Conclusions and Recommendations

7.1. Conclusions

In this section, conclusions are presented in the context of the scope and purpose of the research, in which the general objective was to develop a methodology for ecological risk-based design of produced water discharge from an offshore oil production platform. The study was carried out through integrating a probabilistic hydrodynamic model with an ecological risk assessment model, and consisted of six parts: (1) developing an initial dilution model; (2) integrating the developed initial dilution model with far field dilution models; (3) developing a methodology for probabilistic hydrodynamic modeling; (4) identifying methodologies for ERA of produced water discharge; (5) developing a framework for ecological risk-based design of produced water outfall; and (6) applying the framework for a case study dealing with an outfall design of potential discharge from an oil offshore platform.

Keeping in perspective these objectives, it can be concluded that:

1. An initial dilution model was developed after conducting a critical review of presently used initial dilution models, with emphasis on their conceptual and numerical problems as discussed in Chapter Two. A new model is proposed in Chapter Three as an alternative initial dilution model, which is more elegant and more justifiable. The model was derived based on the hypothesis of additive shear and forced entrainment combined with nonlinear regression. It gives a unique, continuous, solution of centerline dilution, which can be presented in either a deterministic or probabilistic form. Comparison of the proposed model with other available models shows that the proposed model is better in a number of ways: (1) it does not assume that the current has no effect in the BDNF, which asymptotic solutions do; (2) in the BDFF region the model has one parameter fewer than the Huang et al. (1998) model yet it is no less accurate; (3) in the transition region it gives a unique solution which the asymptotic models do not; (4) unlike the Huang et al. (1998) model, the proposed model has approximately the same precision for all regions, i.e. the BDNF, the BDFF, and the transition; and (5) the proposed model can also be presented in a probabilistic form that permits calculation of failure probability for specified model inputs and a threshold dilution.

2. An integrated hydrodynamic model is presented in Chapter Four. A mixing process occurring in two separate regions, near and far fields, were discussed and integrated. Modeling of the intermediate region connecting the near- and far-fields was provided using a control volume approach. An application example of the integrated model was discussed using a comparison with a presently available model, i.e. the CORMIX

model. The comparison showed that the proposed model and the CORMIX model are generally in good agreement, particularly in estimating average effluent concentrations. However, the proposed model also provides a concentration field in the X-Y directions so that it may be applicable for analysis of the mixing zone, which in some cases is defined in terms of horizontal area around the discharge location. The proposed model is also readily modified into a probabilistic analysis to take into account uncertainty associated with model inputs and model coefficients.

3. A methodology for probabilistic analysis was developed in Chapter Four for hydrodynamic modeling using Monte Carlo (MC) simulations. Concern regarding the "excessive" number of simulations was addressed by comparing two methods of sampling in the simulations, i.e. random sampling and Latin Hypercube Sampling (LHS) methods. A comparison between random sampling and LHS for MC simulations of a case of hydrodynamic modeling shows that LHS-based MC simulations are typically about 15% more efficient that the random sampling MS simulations. This chapter also shows that probabilistic analysis not only provides distribution shapes, but also takes into account the uncertainty factors simultaneously.

4. In Chapter Five, methodologies for ERA of produced water discharge were presented and problems associated with presently used approaches were discussed. Substantial effort has been devoted in the past to assessing ecological risks of produced water discharges; however, ERA was usually directed at monitoring purposes, making no consideration to the integration between ERA and engineering design of the produced

water outfalls. An approach is identified to deal with specific problems relevant to design of produced water discharges into the marine environment, and consists of three phases: problem formulation, analysis, and risk characterization.

5. A framework of ecological risk-based design was presented in Chapter Six. The traditional outfall design approaches are conventionally directed at compliance with relevant water quality standards, which are in turn commonly specified upon an epidemiological and ecological viewpoint. As a complementary tool, the framework suggests a possibility that the design of produced water outfalls could itself be looked at from the point of view of the environmental risk from exposure to produced water or specific pollutants associated with it. The framework was based on the integration of the methodology of hydrodynamic modeling and ERA. It consists of six steps, namely (1) formulating a problem of ecological risk-based design of produced water discharge; (2) identifying and evaluating preliminary design scenarios; (3) screening the preliminary design scenarios, and if potentially acceptable scenarios are not identified in the screening, returning to step 2; (4) performing analysis of exposures and ecological effects associated with potentially acceptable scenarios; (5) characterizing ecological risks associated with potentially acceptable scenarios; and (6) providing discussions and design recommendations on the basis of ecological risks.

6. The framework of ecological risk-based design was described systematically in Chapter Six by evaluating scenarios of produced water discharge as a case study. Produced water potentially discharged from an offshore oil production platform located on the

Grand Banks, southeast of St. John's, Newfoundland, Canada, was considered. Instead of providing a solution for a particular problem of an existing oil production platform, the emphasis of the case study was to show how the risk-based design of produced water discharge could be potentially undertaken.

7.2. Recommendations

Recommendations, which may be useful for further research, are given here based on the limitations or problems faced during the study. These include:

1. The initial dilution model proposed in this study is based on experimental data from Lee and Cheung (1991). Applicability of the approach, i.e. the length scale analysis combined with nonlinear modeling, has not been evaluated using other data sets from other laboratory or field experiments. It may be useful to validate the proposed model using other data sets, if available, and to investigate the applicability of the approach for different discharge characteristics.

2. The integrated hydrodynamic model presented in this study is only valid for a short distance from the discharge port where the buoyant spreading is more important than the turbulent diffusion. It may be useful to develop a methodology that considers a case where the turbulent diffusion is more important that the buoyant spreading, or a case where both processes are equally important.

3. This study uses Monte Carlo (MC) simulations for the probabilistic analysis. Care must be taken in performing the MC simulations to deal with cases in which probability distributions of the input parameters and correlation among the parameters are not known or weakly defined. There are methods that do not require assumptions on the probability distribution and the correlation, for example, interval analysis and probability bound analysis. However, these methods, which are also available in the form of software (e.g. Risk Calc, Applied Biomathematic 1999), are at present only capable of handling relatively simple mathematical formulations. In this research, efforts were made to apply these methods for comparative purposes, but the methods did not work because of complexities of the functional form in the hydrodynamic models (Mukhtasor et al. 2001d). It may be useful to develop a methodology that makes it possible to apply these methods to the case under investigation, and to compare their results to those from MC simulations.

4. The methodology for ERA was integrated with the principles of outfall design. The methodology was based on a simplified model of biological characteristics as discussed in Chapter Five. It may be worth performing a more detailed study on modeling of biological characteristics for ERA and its integration with the outfall design concept. This, for example, includes modeling contact between ecological entities of interest and the produced water plume.

5. As emphasized Chapter Six, the framework of ecological risk-based design of produced water discharge is meant as a complementary tool in addition to the traditional approach

to the design. However, regulatory criteria, which are specifically meant for ecological risk-based design, have not been established. This may require inputs from regulatory bodies and other interested communities. Further study on outfall design criteria from the ecological risk viewpoint may be useful.

Chapter 8

Statement of Originality

Originality of the work presented in this thesis can be viewed from different aspects:

1. In initial dilution modeling, a new approach was proposed in this research. It is based on the hypothesis of additive shear and forced entrainment combined with nonlinear least squares regression analysis. Unlike the presently available modeling approach (Huang et al. 1998), which is "trial and error", the proposed approach is systematic and provides an objective means of evaluating the models.

2. A new model of initial dilution of buoyant jet in moving water is developed in this research. The proposed initial dilution model differs from presently available models in that it provides a unique, continuous, solution for the whole range of BDNF, transition and BDFF, without suffering from "structured bias". Compared with presently available models, the proposed model is more robust and justifiable conceptually and numerically.

3. The proposed initial dilution model is presented in deterministic and probabilistic forms and has uncertainty measures associated with the model formulation, which is reflected by the model coefficients and error term. These make it possible to perform a probabilistic analysis considering both input and model uncertainties. The approach of providing uncertainty measures to empirical models is not new. For example it has been applied for the probabilistic riprap model (Tung 1994). However, no application of the approach to initial dilution in the whole range of BDNF, transition and BDFF has been found in the literature.

4. The deterministic approach to hydrodynamic modeling, particularly for the intermediate and far fields, is not new. For example, it has been used for modeling sewage discharges (Doneker and Jirka 1990, Huang et al. 1996). In this study, this approach was used to combine the proposed initial dilution model and modified into a probabilistic analysis. No similar probabilistic modification of these models and its application to an ecological risk assessment of produced water discharge has been found in the literature.

5. A probabilistic analysis was performed in this study using Latin Hypercube Sampling (LHS)-based Monte Carlo (MC) simulations. Application of MC simulations is not new. For example these have been applied for ocean outfall analyses (Bale et al. 1990, Orlob and Tumeo 1986, Webb 1987). These applications, however, consider only uncertainty associated with model inputs and that associated with the model itself is left unaccounted for. This study employed LHS-based MC simulations considering

uncertainty from model inputs and model coefficients and an error term for the analysis of hydrodynamic modeling. No similar work has been found in the literature.

6. The framework of ecological risk-based design of produced water discharge proposed in this study is meant as a complementary tool in addition to the traditional approach to engineering design. It consists of six steps described in Chapter Six. As a complementary tool, it suggests a possibility that the design of produced water outfalls could itself be looked at from the point of view of the ecological risk. No similar framework has been found in the literature.

7. The methodology for ecological risk assessment used in this study is not new. It has been used in other areas of research (CCME 1997, U.S. EPA 1998). In this study, however, the ERA methodology was applied for assessing potential risk associated produced water discharge, based on chemical specific and whole effluent toxicity approaches. No such application has been found in the literature.

References

Akar, P.J. and Jirka, G.G 1994a, Buoyant Spreading Processes in Pollutant Transport and Mixing, Part 1: Lateral Spreading with Ambient Current Advection, *Journal of Hydraulic Research*, **32(6)**: 815-831.

Akar, P.J. and Jirka, G.G 1994b, Buoyant Spreading Processes in Pollutant Transport and Mixing, Part 2: Upstream Spreading in Weak Ambient Current, *Journal of Hydraulic Research*, **33(1)**: 87-100.

Andrade, Y. and Loder, J.W. 1997, *Convective Descent Simulations of Drilling Discharges on Georges and Sable Island Banks*, Canadian Technical Report of Hydrography and Ocean Sciences 185, 83 pp.

Ang, A. H-S. and Tang, W. H. 1975, *Probability Concepts in Engineering Planning and Design, Volume I: Basic Principles*, John Wiley & Sons.

Ang, A. H-S. and Tang, W.H. 1984, *Probability Concepts in Engineering Planning and Design, Volume II: Decesion, Risk and Reliability*, John Wiley & Sons.

Australia and New Zealand Environment and Conservation Council (ANZECC) and Agriculture and Resource Management Council of Australia and New Zealand (ARMCANZ) 1999, *National Water Quality Management Strategy: Australian and New Zealand Guidelines for Fresh and Marine Water Quality, Overview, Draft*, July.

Bale, A.E. Meyer, G.K., and Orlob, G.T. 1990, Probabilistic Simulation of Coastline Contamination, *Proceedings of International Symposium on Water Resource Systems Applications*, University of Manitoba, Winnipeg, pp. 492-501.

Bennet 1981, Initial Dilution: a Practical Study on the Hastings Long Sea Outfall, *Proceedings of Institution of Civil Engineering*, Part 1, **70**:113-122.

Bettess and Munro 1980, *Field Measurements of Initial Dilutions at a Sea Outfall*, Report No. IT 198, Hydraulics Research Station, Wallingford, England.

Bishop, C.M. 1994, *Neural Networks and Their Applications: Review Article*, Rev. Sci. Instrum. 65(6): 1803-1831.

Brandsma, M.G. and Smith, J. P 1996, Dispersion Modeling Perspectives on the Environmental Fate of Produced Water Discharges, in *Produced Water2: Environmental Issues and Mitigation Technologies*, Edited by M. Reed and S. Johnsen, Plennum Press, New York, pp. 215-224.

Brandsma, M.G., Smith, J. P., O'Reilly, J.E., Ayers, R.C., Jr. and Holmquist, A.L. 1992, Modeling Offshore Discharges of Produced Water, in *Produced Water: Technological/Environmental Issues and Solutions*, Edited by J.P. Ray and F.R. Engelhart, Plennum Press, New York, pp. 59-71.

Brandsma, M.G., Davis, L.R., Ayers, R.C., Jr. and Sauer, T.C., Jr., 1980, A Computer Model to Predict the Short Term Fate of Drilling Discharges in *the Marine Environment, Symposium/Research on Environmental Fate and Effects of Drilling Fluid and Cuttings*, January 21-24, Lake Buena Vista, Florida, pp. 588-610.

Brendehaug, J., Johnsen, S., Bryne, K.H., Gjose, A.L., Eide, T.H. and Aamot, E. 1992, Toxicity Testing and Chemical Characterization of Produced water – A Prelimnary Study, in *Produced Water: Technological/Environmental Issues and Solutions*, Edited by J.P. Ray and F.R. Engelhart, Plennum Press, New York, pp. 245-256.

Brooks, N.H. 1960, Diffusion of Sewage Effluent in an Ocean Current, in *Proceedings of the First International Conference on Waste Disposal in the Marine Environment*, Edited by Pearson, E.A., Pergamon, pp. 246-267.

Brown, J. S., Sauer, Jr., T.C., Wade, M.J. and Neff, J.M. 1992, Chemical and Toxicological Characterization of Produced Water Freon Extracts, in *Produced Water: Technological/Environmental Issues and Solutions*, Edited by J.P. Ray and F.R. Engelhart, Plennum Press, New York, pp. 113-131.

Canadian Council of Ministers of the Environment (CCME) 1999, *Canadian Environmental Quality Guidelines, Summary Table.*

Canadian Council of Ministers of the Environment (CCME), 1997, *A Framework for Ecological Risk Assessment: Technical Appendices*, The national Contaminated Sites Remediation Program, Winnipeg, Manitoba.

Canadian Council of Ministers of the Environment (CCME), 1996a, *Introduction to Environmental Risk Assessment Studies: Environmental Technology*, Canadian Standards Association, Z763-96.

Canadian Council of Ministers of the Environment (CCME), 1996b, *A Framework for Ecological Risk Assessment: General Guidance*, The national Contaminated Sites Remediation Program, Winnipeg, Manitoba, PN 1195.

Canadian-Newfoundland Petroleum Board (CNOPB) 2001, http://www.cnopb.nfnet.com

Cederwall, K.1968, *Hydraulics of Marine Waste Disposal*, Rep.42, Hydraulics Division, Chlamers Institute of Technology, Goteburg, Sweden.

Christoulas and Andreakadis 1994, Application of the EU Bathing Water Directive to the Design of Marine Sewage Disposal Systems. *Water Science and Technology*, **32(2)**: 53-60.

Department Fisheries and Ocean (DFO) 1999, http://www.mar.dfo-mpo.gc.ca/science/ocean/welcome.html

Doneker, R. L., and Jirka, G.H. 1990, *Expert System for Hydrodynamic Mixing Zone Analysis of Conventional and Toxic Submerged Single Port Discharges (CORMIX1)*, Rep. No. EPA/600/3-90/012, U.S. Environmental Protection Agency, Washington, D.C.

Efroymson, R.A., Jackson, B.L., Jones, D.S., Sample, B.E., Suter II, G.W. and Welsh, C.J.E. 1996, *Waste Area Grouping 2, Phase I Task data Report: Ecological Risk Assessment and White Oak Creek Watershed Screening Ecological Risk Assessment*, Prepared for the U.S. Department of Energy, ORNL/ER-366.

Ferson, S., Root, W., and Kuhn, R. 1999, *RAMAS Risk Calc: Risk Assessment with Uncertain Numbers*, Applied Biomathematics, Setauket, NY.

Fischer, H. B., List, E. J., Koh, R. C. H., Imberger, J, and Brooks, N. H., 1979, *Mixing in Inland and Coastal waters*, Academic Press, New York.

Finley, B., and Paustenbach, D., 1994, The Benefits of Probabilistic Exposure Assessment: Three Case Studies Involving Contaminated Air, Water, and Soil, *Risk Analysis*, **14(1)**: 53-73.

French, D. P., Applied Science Associates, Inc. 1999, personal communication, email on July 07.

Frey H. C. and Burmaster D. E. 1999, Methods for Characterizing Variability and Uncertainty: Comparison of Bootstrap Simulation and Likehood-based Approaches, *Risk Analysis*, **19(1)**: 109-130.

Frost, T.K., Johnsen, S. and Utvik, T.R. 1998, *Produced Water Discharges to the North Sea, Fate and Effects in the Water Column, Summary Report*, December.

Furuholt, E. 1996, Environmental Effects of Discharge and Reinjection of Produced water, in *Produced Water2: Environmental Issues and Mitigation Technologies*, Edited by M. Reed and S. Johnsen, Plennum Press, New York, pp. 275-288.

Girling, A.E. 1989, Acute and Chronic Toxicity of Produced Water from a North Sea Oil Production Platform to the Calanoid Capepod *Acartia tonsa*, *Bulletin of Environmental Contaminant Toxicology*, **43**: 280-286.

Gowda, R. N., 1992, *Field and Laboratory Studies of Mixing Tubes for Marine Outfalls*, M.Eng. Thesis, Memorial University of Newfoundland, St. John's, Newfoundland, Canada.

Grace, R.A. 1978, *Marine Outfall Systems: Planning, design and construction*, Prentice-Hall, Inc., Englewood Cliffs, New Jersey.

Hamdy, Y. 1981, *Dispersion of Effluent Plumes from Diffusers on Near-shore Regions of the Great Lakes, Vol. 1: Initial Mixing Processes*, Working Group II of the Water Management Steering Committee, Ministry of the Environment, 47 p.

Hazen and Sawyer 1994, *Southeast Florida Outfall Experiment II*: Final Report, June.

Hibernia Management and Development Company, Ltd. 1996, *Hibernia Development Project Production Phase: Environmental Effects, Monitoring Plant, April 1996*, prepared by Jacques Whitford Environment.

Huang, H., Fergen, R.E., Proni, J. R., and Tsai, J. J. 1998, Initial Dilution Equations for Buoyancy-dominated Jets in Current, *Journal of Hydraulic Engineering*, **124(1)**:105-108.

Huang, H. and Fergen, R.E. 1997, A Model for Surface Plume Dispersion in An Ocean Current, *Proceedings of Theme B, Water for A Changing Global Community*, The 27 th Congress of the International Association for Hydraulic Research, 370-375.

Huang, H., Proni, J. R., and Tsai, J. J. 1996, Probabilistic Analysis of Ocean Outfall Mixing Zones, *Journal of Environmental Engineering*, **122(5)**:359-367.

Huang, H., Proni, J. R., and Tsai, J. J., 1994, Probabilistic Approach to Initial Dilution of Ocean Outfalls, *Water Environment Research*, **66(6)**: 787-793.

Hodgins, D.O. and Hodgins, S.L.M. 2000, *Modelled Predictions of Well Cuttings Deposition and Produced Water Dispersion for the Proposed White Rose Development*, Seaconsult Marine Research Ltd. Vancouver, BC.

Jacobs, R.P.W.M. and Marquenie, J.M. 1991, Produced water Discharges from Gas/Condensate Platforms, Environmental Considerations, in *Proceedings First Int. Conf. HSE in Oil and Gas Expl. And Prod.*, SPE, Richardson, Texas, 2:89-96.

Jirka, G.H., Doneker, R.L., Hinton, S.W. 1996, *User's Manual for Cormix: A Hydrodynamic Mixing Zone Model and Decision Support System for Pollutant Discharges into Surface Waters*, U.S. Environmental Protection Agency, Office Science and Technology, Washington, D.C.

Jirka, G.H. and Lee, J.H.W. 1994, Waste Disposal in the Ocean, Water Quality and Its Control, in *Hydraulic Structures Design Manual*, Edited by Mikio Hino, IAHR, A.A. Balkema, pp. 193-242.

Karman, C.C. Johnsen, S., Schobben, H.P.M. and Scholten, M.C.Th. 1996, Ecotoxicological Risk of Produced water Discharged from Oil Production Platforms in the Statfjord and Gullfaks Field, in *Produced Water2: Environmental Issues and Mitigation Technologies*, Edited by M. Reed and S. Johnsen, Plennum Press, New York, pp. 127-134.

Karman, C.C. and Reerink, H.G. 1998, Dynamic Assessment of the Ecological Risk Assessment of the Discharge of Produced Water from Oil and Gas Producing Platforms, *Journal of Hazardous Materials*, **61**: 43-51.

Kennish, M.J. 1997, *Practical Handbook of Estuarine and Marine Pollution*, Boca Raton: CRC Press.

Klemm, D.J., Morrison, G.E., Norbrg-King, T.J., Peltier, W.H. and Heber, M.A. 1994, *Short-term Methods for Estimating the Chronic Toxicity of Effluents and Receiving Water to Marine and estuarine Organisms*, Second Edition, United States Environmental Protection Agency, Research and Development (MD-591), EPA-600-4-91-003.

Lee, J.H.W. and Cheung, V. 1991, Mixing of Buoyancy-dominated Jets in a Weak Current, *Proceedings of Institution of Civil Engineering*, Part 2, **91**:113-129.

Lee, J. H. W. and Koening, A., 1995, Discussion of Probabilistic Approach to Initial Dilution of Ocean Outfalls, *Water Environment Research*, **67(5)**: 878-880.

Lee, J.H.W. and Neville-Jones, P. 1989, Initial Dilution of Horizontal Jet in Crossflow: Closure, *Journal of Hydraulic Engineering*, **115(2)**: 279-284.

Lee, J.H.W. and Neville-Jones, P. 1987a, Initial Dilution of Horizontal Jet in Crossflow, *Journal of Hydraulic Engineering*, **133(5)**: 615-629.

Lee, J. H. W and Neville-Jones, P.1987b, Sea Outfall Design --prediction of initial dilution, *Proceedings of Institution of Civil Engineers*, Part 1. 82: 981-994.

Lenwood, W.H., Jr., Scott, M.C., and Killen, W.D. 1998, Ecological Risk Assessment of Copper and Cadmium in Surface Waters of Chesapeake bay Watershed, *Environmental Toxicology and Chemistry*, 17(6):1172-1189.

Malik, O.P. 1993, An Introduction to Artificial Neural Networks, A Tutorial Presentation at the Canadian Electrical Association, *Electricity '93 Conference*, Montreal, Quebec, 17 p.

Markham, S. 1993, Modelling of Sewage Outfalls in the Marine Environment, in *An Introduction to Water Quality Modelling*, Edited by James, A., Second Edition, John Wiley and Sons, 293-308.

Meinhold, A.F., Holtzman, S. and DePhillips, M.P. 1996a, Risk Assessment for Produced Water Discharges to Open Bays in Louisiana, in *Produced Water2: Environmental Issues and Mitigation Technologies*, Edited by M. Reed and S. Johnsen, Plennum Press, New York, pp. 395-409.

Meinhold, A.F., DePhillips, M.P. and Holtzman, S. 1996b, *Final Report: Risk Assessment for Produced Water Discharges to Louisiana Open Bays*, Brookhaven National Labolatory, BNL-62975 (Rev. 6/96), 146 p.

Meinhold, A.F., Holtzman, S., and DePhillips, M.P. 1996c, *Produced Water Discharges to the Gulf of Mexico: Background Information for Ecological Risk Assessments*, Brookhaven National Labolatory, BNL-63331, 156 p.

Memorial University of Newfoundland (MUN) 2001, http://www.mun.ca

Melching, C.S. 1995, Reliability Estimation, in *Computer Models of Watershed Hydrology*, edited by Singh, V.P., Chapter 3, Water Resources Publications.

Middleditch, B.S. 1984, Ecological Effects of Produced Water Effluents from Offshore Oil and Gas Production Platforms, *Ocean Management*, 9:191-316.

Moffit, C.M. Rhea, M.R. Dorn, P.B., Hall, J.F. and Bruney, J.M. 1992, Short-term Chronic Toxicity of Produced Water and Its Variability as a Function of Sample Time and Discharge Rate, in *Produced Water: Technological/Environmental Issues and Solutions*, Edited by J.P. Ray and F.R. Engelhart, Plennum Press, New York, pp. 235-244.

Montgomery, D.C. 1976, *Design and Analysis of Experiments*, John Wiley & Sons, 418 p.

Muellenhoff, W.P., Soldate, Jr, A.M., Baumgartner, D.J., Schuldt, M.D., Davis, L.R. and Frick, W.E. 1985, *Initial Mixing Characteristics of Municipal Ocean Discharges*: Vol. 1., Procedures and Applications, Report No. EPA/600/3-85/073a, U.S. Environmental Protection Agencies.

Mukhtasor 2000, Risk-based Design of Produced Water Discharge from Offshore Oil Production Platform, *Applications of Risk Analysis in Industry and Government, Society for Risk Analysis (SRA) Annual Meeting*, Crystal Gateway Marriott, Virginia, December 3-6.

Mukhtasor, Lye, L.M. and Sharp, J.J. 2001a, Initial Dilution Models for Buoyancy-dominated Jet in Moving Water, *Journal of Hydraulic Research*, (Submitted).

Mukhtasor, Lye, L.M. and Sharp, J.J. 2001b, Methods of Compliance Evaluation for Ocean Outfall Design and Analysis, *Environmental Management* (Submitted).

Mukhtasor, Husain, T., Lye, L.M. and Sharp, J.J. 2001c, Human Health Risk-based Design of Ocean Outfall, *The Croucher Foundation Advanced Study Institute on Recent Developments in Coastal Eutrophication Research: Prediction, Decision Support Systems, and Management*, The University of Hong Kong, 5-12 February.

Mukhtasor, Lye, L.M. and Husain, T. 2001d, Far Field Hydrodynamic Modeling of Ocean Discharges: Methods of Dealing with Uncertainty, *Conference of Canadian Society of Civil Engineering* (CSCE), June, Victoria (Accepted).

Mukhtasor, Lye, L.M. and Sharp, J.J. 2001e, Mixing Zone Analysis of Produced Water Discharge from Offshore Oil Producing Platform, accepted for presentation at 2nd *World Engineering Congress*, Sarawak, Malaysia, 22-25 July 2002.

Mukhtasor, Sadiq R., Husain, T., Veitch, B. and Bose, N. 2000a, Acute Ecological Risk Associated with Soot Deposition: A Persian Gulf Case Study, *Journal of Ocean Engineering* (Accepted).

Mukhtasor, Husain, T., Lye, L.M. and Sharp J.J. 2000b, Human Health Risk-based Design of Ocean Outfall, *Journal of Water, Maritime and Energy, The Institution of Civil Engineers*, London, UK (Accepted).

Mukhtasor, Husain, T., Bose, N. and Veitch, B. 2000c, Produced Water Discharge into the Marine Environment: Water Quality Criteria and Environmental Risk Assessment, *Proceedings of Third Specialty Conference on Environmental Progress in the Petroleum & Petrochemical Industries*, 1-3 May, Bahrain, paper #23.

Mukhtasor, Lye, L.M. and Husain, T. 2000d, Problems of Asymptotic Solution-based Initial Dilution Models and New Research Directions, *Proceedings of the Conference of Canadian Society of Civil Engineering* (CSCE), June, London, ON.

Mukhtasor and Husain, T. 1999, Environmental Risk Assessment and Ocean Outfall Design, *Proceedings of the 1999 Canadian Coastal Conference*, 19-22 May, Victoria, BC, **2:** 517-530.

Mukhtasor, Sharp, J.J. and Lye, L.M. 1999a, Uncertainty Analysis of Ocean Outfalls, *Canadian Journal of Civil Engineering*, **26**: 434-444.

Mukhtasor, Lye, L.M. and Husain, T. 1999b, Probabilistic Initial Dilution Model for Ocean Outfalls, *Proceedings of the Conference of Canadian Society of Civil Engineering* (CSCE), 2-5 June, Regina, **2:** 139-147.

Mukhtasor, Sharp, J.J. and Lye, L.M. 1998, Applicability of First Order Second Moment and Monte Carlo Simulations in Ocean Outfall Design and Analysis, in *Proceedings of the 11 th Congress of the IAHR-APD*, September 8-10, Yogyakarta, Indonesia, **2**: 331-340.

Mukhtasor 1998, *Probabilistic Ocean Outfall Design*, M.Eng. thesis, Memorial University of Newfoundland.

Murray-Smith, R., Gore, D., Flynn, S.A., Vance, I. and Stagg, R. 1996, Development and Appraisal of a Particle Tracking Model for the Dispersion of Produced water Discharged from an Oil Production Platform in the North Sea, in *Produced Water2: Environmental Issues and Mitigation Technologies*, Edited by M. Reed and S. Johnsen, Plennum Press, New York, pp. 225-245.

Myer, R.H. and Montgomery, D.C. 1995, *Response Surface Methodology: Process and Product Optimisation using Designed Experiments*, John Wiley & Sons, 700 p.

Myers, R. H. 1990, *Classical and Modern Regression with Applications*, Second Edition, PWS-KENT Publishing Company, Chapter 9.

National Energy Board, C-NOPB, C-NSOPB. 1996, *Offshore Waste Treatment Guidelines*, September 1995.

Neff 1997, *Metals and Organic Chemicals Associated with Oil and Gas Well Produced Water: Bioaccumulation, Fates, and Effects in the Marine Environment, Gulf of Mexico Produced Water Bioaccumulation Study*, Continental Shelf Associates, Inc. for Offshore Operators Committee, April.

Neff, J.M. and Sauer, Jr., T.C 1996, Aromatic Hydrocarbons in Produced Water, in *Produced Water2: Environmental Issues and Mitigation Technologies*, Edited by M. Reed and S. Johnsen, Plennum Press, New York, pp. 163-175.

Ofjord, G.D., Bakke, S., and Vik, E.A. 1996, The CHARM Model Used in Environmental Risk Management of Produced Water on Ula, in *Produced Water2: Environmental Issues and Mitigation Technologies*, Edited by M. Reed and S. Johnsen, Plennum Press, New York, pp. 303-316.

Orlob, G.T. and Tumeo, M.A. 1986, Monte Carlo Simulation of Coastline Contamination by an Ocean Discharge of Wastewater, in *Proceedings of International Sysmposium on Buoyant Flows*, Athens, Greece, pp. 469-480.

Osenberg, C.W., Schmitt, R.J., Holbrook, S.J., and Canestro, D. 1992, Spatial Scale of Ecological Effects Associated with an Open Coast Discharge of Produced Water, in *Produced Water: Technological/Environmental Issues and Solutions*, Edited by J.P. Ray and F.R. Engelhart, Plennum Press, New York, pp. 387-402.

Palesade 1997, *@Risk, Advanced Risk Analysis for Spreedsheets*, Guide to Using @Risk: Risk Analysis and Simulations, Add-In for Microsoft Excell or Lotus 1-2-3, Palisade Corporation, Newfield, NY.

Petro-Canada 1996, *Development Application, Terra Nova Development: Environmental Impact Statement*, Prepared by Petro-Canada on behalf of the Terra Nova Proponents.

Pillard, D.A., Tietge, J.E. and Evans, J.M.1996, Estimating the Acute Toxicity of Produced Waters to Marine Organisms using Predictive Toxicity Models, in *Produced Water2: Environmental Issues and Mitigation Technologies*, Edited by M. Reed and S. Johnsen, Plennum Press, New York, pp. 49-59.

Proni, J.R., Huang, H., Dammann, W.P. 1994, Initial Dilution of Southeast Florida Ocean Outfalls, *Journal of Hydraulic Engineering*, **120(12)**:1409-1425.

Ray, J.P., 1996, Session Summary, in *Produced Water 2: Environmental Issues and Mitigation Technologies*, Edited by M. Reed and S. Johnsen, Plennum Press, New York, 1996, pp. 3-5.

Reed, M. 1996, Session Summary for Section V: Environmental Modelling, in *Produced Water2: Environmental Issues and Mitigation Technologies*, Edited by M. Reed and S. Johnsen, Plennum Press, New York.

Reed, M. SINTEF Applied Chemistry, Trondheim, N. 1999, personal communication, emails on July 07-08.

Reed, M., Johnsen, S., Melbye, A. and Rye, H. 1996, PROVANN: a Model System for Assessing Potential Chronic Effects of Produced Water, in *Produced Water2: Environmental Issues and Mitigation Technologies*, Edited by M. Reed and S. Johnsen, Plennum Press, New York, pp. 317-330.

Roberts, P.J.W. and Snyder, W.H. 1993, Hydraulic Model Study for Boston Outfal, I: Riser Configuration, II: Environmental Performance, *Journal of Hydraulic Engineering* **119**: 970-1003.

Roberts, P.J.W., Snyder, W.H., Baumgartner, D.J. 1989a, Ocean Outfalls. I: Submerged Wastefield Formation, *Journal of Hydraulic Engineering* **115**:1-25.

Roberts, P.J.W., Snyder, W.H., Baumgartner, D.J. 1989b, Ocean Outfalls. II: Spatial Evolution of Submerged Wastefield, *Journal of Hydraulic Engineering* **115**: 26-48.

Roberts, P.J.W., Snyder, W.H., Baumgartner, D.J. 1989c, Ocean Outfalls. III: Effect of Diffuser Design on the Submerged Wastefield, *Journal of Hydraulic Engineering* **115**: 49-70.

Roe, T.I., Johnsen, S. and the Norwegian Oil Industry Association 1996, Discharges of Produced Water to the North Sea, in *Produced Water2: Environmental Issues and Mitigation Technologies*, Edited by M. Reed and S. Johnsen, Plennum Press, New York, pp. 13-25.

Rye, H., Reed, M., Melbye, A. and Johnsen, S. 1996, An Intercomparison Between Field Measurements and Three Different Calculation Models for Estimates of Dilution Factors, in *Produced Water2: Environmental Issues and Mitigation Technologies*, Edited by M. Reed and S. Johnsen, Plennum Press, New York, pp. 257-269.

Sauer, T.C., Costa, H.J., Brown, J.S. and Ward, T.J. 1997, Produced Water Series, Toxicity Identification Evaluations of Produced Water Effluent, *Environmental Toxicity and Chemistry*, **16(10)**: 2020-2028.

Schiff, K.C., Reish, D.J., Anderson, J.W. and Bay, S.M. 1992, A Comparative Evaluation of Produced Water Toxicity, in *Produced Water: Technological/Environmental Issues and Solutions*, Edited by J.P. Ray and F.R. Engelhart, Plennum Press, New York, pp. 199-207.

Schobben, H.P.M., Vik, E.A., Hutjes, G.G., Karman, C.C. and Ofjord, G.D. 1996, An Introduction to the CHARM model, *in Produced Water2: Environmental Issues and Mitigation Technologies*, Edited by M. Reed and S. Johnsen, Plennum Press, New York, pp. 295-301.

Sharp, J. J. 1989a, Hydraulic Design of Marine Outfalls, in *Marine Outfall Design and Siting*, Seminar/Workshop, November 20-21, Continuing Education Division, Technical University of Nova Scotia

Sharp, J. J. 1989b, Discussion on Initial Dilution of Horizontal Jet in Crossflow, *Journal of Hydraulic Division, Proceeding ASCE*, **115 (2)**: 282-283.

Sharp, J.J. and Moore, E. 1989, Marine Outfall Design - Computer Models for Initial Dilution in a Current, *Proceedings of Institution of Civil Engineering*, Part 1, **86**:953-961.

Sharp, J.J. and Moore, E. 1987, Estimation of Dilution in Buoyant Effluents Discharged into a Current, *Proceedings of Institution of Civil Engineering*, Part 2, **83**:181-196.

Skatun, H. M. 1996, A Buoyant Jet/Plume Model for Subsea Releases, in *Produced Water2: Environmental Issues and Mitigation Technologies*, Edited by M. Reed and S. Johnsen, Plennum Press, New York, pp. 247-255.

Smith, J.P., SPE, Tyler A.O., Rymell, M.C., Shidharta, H. 1996, Environmental Impact of Produced Water in the Java Sea, Indonesia, in *Proceedings of the 1996 SPE Asia pacific Oil and Gas Conference*, Australia, 28-31 October.

Smith, M. 1993, *Neural Networks for Statistical Modeling*, Van Nostrand Reinhold, New York, 235 p.

Somerville, H.J., Bennet, D., Davenport, J.N., Holt, M.S., Lynes, A., Mahieu, A., McCourt, B., Parker, J.G., Stephenson, R.R., Watkinson, R.J., and Wilkinson, T.G. 1987, Environmental Effect of Produced Water from North Sea Oil Operations, *Marine Pollution Bulletin*, **18(10)**: 549-558.

Stephenson, M.T. 1992, A Survey of Produced Water Studies, in *Produced Water: Technological/Environmental Issues and Solutions*, Edited by J.P. Ray and F.R. Engelhart, Plennum Press, New York, pp. 1-11.

Stromgren, T. Sorstrom, S.E., Schou, L., Kaarstad, I., Aunaas, T. Brakstad, O.G. and Johansen, O. 1995, Acute Toxic Effects of Produced Water in Relation to Chemical Composition and Dispersion, *Marine Environmental Research*, **40(2)**:147-169.

Sullivan, P. K. and Vithanage, D., 1994, Impacts on the Nearshore Marine Environment from the Sand Island Municipal Outfall in Honolulu, Hawaii, in *Proceedings of the MTS94, Challenges and Opportunities in Marine Environment*, 663-668.

Suter II, G.W. 2000, Generic Assessment Endpoints are Needed for Ecological Risk Assessment, *Risk Analysis: an International Journal*, 20(2): 173-178.

Suter II, G.W. 1993. *Ecological Risk Assessment*, Lewis Publishers.

Terra Nova Project 2000, Website: http://www.terranovaproject.com, 14 December.

Thatcher, M., Robson, M., Henriquez, L.R. and Karman, C.C. 1999, *A CIN Revised CHARM III Report, An User Guide for the Evaluation of Chemicals Used and Discharged Offshore*, Version 1.0, 3 August, Tung, Y.K. 1994, Probabilistic Hydraulic Design: a Next Step to Experimental Hydraulics, *Journal of Hydraulic Research*, 32(3): 323-336.

U.S. Environmental Protection Agency (EPA) 1999a, *National Recommended Water Quality Criteria -Correction*, Office of Water, Washington, D.C., EPA 822-Z-99-001.

U.S. Environmental Protection Agency (EPA) 1999b, *Environmental Assessment of Proposed Effluent Limitations Guidelines and Standards for Synthetic-based Drilling Fluids and Other Non-aqueous Drilling Fluids in the Oil and Gas Extraction Point Source Category*, Office of Water, Washington, D.C., EPA 821-B-98-019.

U.S. Environmental Protection Agency (EPA) 1998, *Guidelines for Ecological Risk Assessment*, Risk Assessment Forum, US EPA, Washington, D.C., EPA/630/R-95/002F.

U.S. Environmental Protection Agency (EPA) 1997, *Fact Sheet And Supplemental Information for The Proposed Reissuance Of the NPDES General Permit for New And Existing Sources In The Offshore Subcategory Of The Oil And Gas Extraction Point Source Category For The Western Portion Of The Outer Continental Shelf Of The Gulf Of Mexico*, US EP Region 6, Dallas, TX.

U.S. Environmental Protection Agency (EPA) 1996a, *Summary Report for the Workshop on Monte Carlo Analysis*, Risk Assessment Forum, Office Research and Development, Washington DC, EPA/630/R-96/010.

U.S. Environmental Protection Agency (EPA) 1996b, *Water Quality Benefits Analysis of Final Effluent Limitations Guidelines and Standards for the Coastal Subcategory of the Oil and Gas Extraction Point Source Category*, Office of Water, EPA-821-R-96-024.

U.S. Environmental Protection Agency (EPA) 1993, *Development Document for Effluent Limitations Guidelines and New Source Performance Standards for the offshore Subcategory of the Oil and Gas Extraction Point Source Category: final*, Office of Water, EPA-821-R-93-003.

U.S. Environmental Protection Agency (EPA) 1991, *Technical Support Document for Water Quality-based Toxics Control*, US EPA, Office of Water, Washington, DC., EPA/505/2-90-001 PB91-127415.

Washburn, L., Stone, S., MacIntyre, S. 1999, Dispersion of Produced Water in a Coastal Environment and Its Biological Implications, *Continental Shelf Research*, **19**: 57-78.

Webb, A.T. 1987, Ocean Outfalls – a Simulation Design Approach, *Conference on Hydraulics in Civil Engineering*, Melbourne, 12-14 October.

Wiedeman, A. 1996, Regulation of Produced Water by the U.S. Environmental Protection Agency, in *Produced Water2: Environmental Issues and Mitigation Technologies*, Edited by M. Reed and S. Johnsen, Plennum Press, New York, pp. 27-41.

Williams, B.L., 1985, *Ocean Outfall Handbook*, Water and Soil Miscellaneous Publication No. 76, Wellington, New Zealand.

Wolfe, D. A., 1988, Urban wastes in coastal waters: assimilative capacity and management, in *Oceanic Processes in Marine Pollution*, Volume 5: Urban Wastes in Coastal Marine Environments, Wolfe, D.A. and O'connor, T. P. (Editors), Chapter 1, Robert E. Krieger Publishing Company.

Wood, I.R., Bell, R.G. Willkinson, D.L. 1993, *Ocean Disposal of Wastewater*, World Scientific, Singapore

Wright, S.J., Roberts, P.J.W., Zhongmin, Y. and Bradley, N.E. 1991, Surface Dilution of Round Submerged Buoyant Jets, *Journal of Hydraulic Research*, **29(1)**: 67-89.

Wright, S.J. 1977a, *Effects of Ambient Crossflows and Density Stratification on the Characteristic Behavior of Round Turbulent Buoyant Jets*, Report No. KH-R-36, California Institute of Technology, Pasadena.

Wright 1977b, Mean Behavior of Buoyant Jets in a Crossflow, *Journal of the Hydraulic Division, Proceedings of the American Society of Civil Engineers*, **103(HY5)**: 499-513.