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Consideration of the Potential Impacts on the Marine Environment Associated with Offshore Petroleum Exploration and Development Activities

Examen des impacts possibles des activités d'exploration et de mise en valeur du pétrole extracôtier sur le milieu marin

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ABSTRACT

Over the past few decades, in addition to global efforts, a significant amount of knowledge from scientific studies and environmental effects monitoring (EEM) programs has been gathered by Fisheries and Oceans Canada (DFO) on the potential impacts of exploration and development from oil and gas activities on marine ecosystems and marine resources. There have also been advances in operational techniques, including mitigation protocols, that have been developed in an effort to minimize the potential impacts of offshore petroleum resource recovery on marine ecosystems and marine resources. This research document is a compilation of DFO's most up-to-date 'state of knowledge' of the potential impacts of typical offshore petroleum activities that may arise in context of marine ecosystems and marine resources. The document has the following objectives under its Terms of Reference:

- description of the potential impacts of discharges and emissions and zones of influence, if any, from offshore vessels/platforms related to petroleum exploration and development, with a focus on the potential impacts on marine species (e.g. finfish, shellfish, invertebrates and mammals);
- overview of lessons learned in Atlantic Canada from offshore petroleum exploration and development activities on the Scotian Shelf and the Grand Banks (including results of EEM programs);
- identification of current offshore petroleum regulatory measures and available mitigation approaches and technologies; and
- where possible, a review of existing knowledge and identification of knowledge gaps in science and technology which, if closed, can lead to a better understanding of the potential environmental risks associated with petroleum-related activities in the Georges Bank area.

The content of this document builds on earlier reports of Gordon (1988) and Boudreau et al. (1999), which summarized and quantified possible environmental impacts of petroleum exploration activities on Georges Bank. The scope of this document has been expanded to include the effects of production operations. The decision to broaden the scope was based on the fact that the overall life of a production field is generally on the order of decades, which far exceeds the duration of seismic surveys or exploratory drilling programs that typically last from tens of days to a few months, respectively. In addition, the potential impacts associated with the offshore petroleum production phase can be considerably different from the potential impacts associated with exploration activities.

The primary intent of the document is to summarize the existing state of knowledge. Some remaining knowledge gaps have been identified, although this is not the major focus of the document and it should not be viewed as a comprehensive review of research needs. The document only considers the state of knowledge of potential impacts on marine environments associated with offshore petroleum exploration prior to April 2010. It does not consider any new knowledge or lessons that may have been learned from the Gulf of Mexico oil spill associated with the April 20, 2010, accident of the Deepwater Horizon. Furthermore, the document is not to be viewed as an Environmental Impact Assessment (EIA) nor will such an assessment be provided with this document. Last, the document complements the document of Kennedy et al. (2011) entitled ‘The Marine Ecosystem of Georges Bank’ (DFO. Can. Sci. Advis. Sec. Res. Doc. 2011/059. xiv + 232pp).

RÉSUMÉ

Au cours des dernières décennies, Pêches et Océans Canada (le MPO) a acquis bien des connaissances sur les impacts possibles des activités d'exploration et de mise en valeur du pétrole et du gaz sur les écosystèmes marins et leurs ressources dans le cadre d'études scientifiques et de programmes de surveillance des effets environnementaux, ainsi que d'initiatives internationales. Des progrès ont été accomplis par ailleurs dans les techniques opérationnelles (notamment dans les protocoles d'atténuation des effets) qui ont été conçues pour réduire les impacts possibles de l'extraction des ressources pétrolières extracôtières sur les écosystèmes marins et sur leurs ressources. Le présent document de recherche dresse l'état récent des connaissances du MPO au sujet des impacts possibles que activités pétrolières courantes entreprises dans les eaux extracôtières peuvent avoir sur les écosystèmes marins et sur leurs ressources. Tel qu'indiqué dans son cadre de référence, le document vise les objectifs suivants :

- décrire les impacts possibles des déversements, émissions et zones d'influence, le cas échéant, associés aux navires et plateformes utilisés pour l'exploration et la mise en valeur du pétrole extracôtier, l'accent étant mis sur les impacts qui pourraient toucher des espèces marines (poissons, crustacés, invertébrés et mammifères);
- donner un aperçu des leçons tirées au Canada atlantique des activités d'exploration et de mise en valeur du pétrole extracôtier sur le plateau néo-écossais et les Grands Bancs (y compris les résultats des programmes de surveillance des effets environnementaux);
- recenser les mesures actuelles de réglementation visant le pétrole extracôtier ainsi que les techniques et méthodes d'atténuation disponibles;
- si possible, examiner les connaissances actuelles et déterminer quelles sont les lacunes dans l'information scientifique et la technologie qui une fois comblées pourraient mener à une meilleure compréhension des éventuels risques environnemnentaux associés aux activités pétrolières dans la région du banc Georges.

Le présent document fait fond sur des rapports antérieurs de Gordon (1998) et de Boudreau et al. (1999), qui résumaient et quantifiaient les impacts environnementaux possibles des activités d'exploration du pétrole sur le banc Georges, mais il englobe les effets des activités de production. La décision d'élargir sa portée à ces activités a été fondée sur le fait que la vie d'un champ de prodution est généralement de l'ordre de plusieurs décennies, ce qui dépasse de beaucoup la durée des relevés sismiques ou des programmes de forage exploratoire, qui est généralement de dix jours et de quelques mois, respectivement. De plus, les impacts possibles de la phase de production du pétrole extracôtier peuvent être très différents de ceux des activités d'exploration.

Le document vise avant tout à établir l'état actuel des connaissances. Bien que cela n'en constitue pas le principal sujet, le document met en évidence certaines lacunes subsistant dans ces connaissances, mais il ne doit pas être considéré comme un inventaire exhaustif des besoins en matière de recherche. Le document ne fait que dresser l'inventaire des connaisances sur les impacts possibles de l'exploration du pétrole extracôtier sur les milieux marins avant avril 2010. Il ne tient donc pas compte de

toute nouvelle connaissance ou leçon ayant pu être tirée du déversement d'hydrocarbures dans le golfe du Mexique ayant découlé de l'explosion du Deepwater Horizon*,* survenue le 20 avril 2010. Il ne doit pas être considéré non plus comme une étude d'impact environnemental (EIE) et pareille étude n'y est pas présentée. Enfin, ce document complète celui de Kennedy et al. (2011) intitulé « The Marine Ecosystem of Georges Bank » (Secr. can. de consult. sci. du MPO, Doc. de rech. 2011/059. xiv + 232 pp).

ACRONYMS AND ABBREVIATIONS

AP	Alkylphenol
APE	Alkylphenol Ethoxylate
AUV	Autonomous Underwater Vehicle
Ba	Barium
BAT	Best Available Technology
BBLT	Bottom Boundary Layer Transport Model
BECPELAG	Biological Effects of Contaminants in Pelagic Ecosystems
BOP	Blowout Prevention System
BTEX	Benzene, Toluene, Ethylbenzene, and Xylenes
CNLOPB	Canada-Newfoundland and Labrador Offshore Petroleum Board
CNSOPB	Canada-Nova Scotia Offshore Petroleum Board
COOGER	Centre for Offshore Oil, Gas and Energy Research
DFO	Fisheries and Oceans Canada
DGGE	Denaturing Gradient Gel Electrophoresis
DNV	Det Norske Veritas
DREAM	Dose-related Risk and Effect Assessment Model
E&P	Exploration and Production
EBM	Ecosystem Based Management
EEM	Environmental Effects Monitoring
EIA	Environmental Impact Assessment
EIF	Environmental Impact Factor
EOR	Enhanced oil recovery
EROD	Ethoxyresorufin-*o*-deethylase
ESRF	Environmental Studies Research Fund
ESSIM	Eastern Scotian Shelf Integrated Management
FPSO	Floating, Production, Storage, and Offloading
FSU	Former Soviet Union
GBS	Gravity Base Structure
GOOMEX	Gulf of Mexico Offshore Operations Monitoring Experiment
GPS	Geographic Positioning System
GSC	Geological Survey of Canada
HI	Hazard Index
$\mathbf{K_{ow}}$	Octanol/Water Partition Coefficient
LTMO	Low Toxicity Mineral Oil
LPS	Lipopolysaccharides
MARPOL	International Convention for the Prevention of Pollution from Ships
MEG	Monoethylene glycol
MFO	Mixed-function Oxidase
MMO	Marine Mammal Observer
MPA	Marine Protected Area
NEB	National Energy Board
NEP	Northeast Peak (of Georges Bank)
NORM	Naturally Occurring Radioactive Material
OBM	Oil-based Mud
OCSG	Offshore Chemical Selection Guidelines
OGP	International Association of Oil and Gas Producers
OMA	Oil-Mineral-Aggregate
OOC	Offshore Operators Committee
PAH	Polycyclic Aromatic Hydrocarbon

PAM	Passive Acoustic Monitoring
PCB	Polychlorinated Biphenyls
PEC	Predicted Environmental Concentration
PERD	Panel of Energy Research and Development
PNEC	Predicted No-Effect Concentration
POM	Princeton Ocean Model
PRAC	Petroleum Research Atlantic Canada
RB	Respiratory Burst
RCR	Risk Characterization Ratios
RMS	Root Mean Square
SARA	Species at Risk Act
SBM	Synthetic-based Mud
SEL	Sound Exposure Level
SIMAP	Spill Impact Model Application Package
SOEP	Sable Offshore Energy Project
TOC	Total Organic Carbon
TPH	Total Petroleum Hydrocarbons
TTS	Temporary Threshold Shift
UM3	Three-dimensional Updated Merge
U.S.	United States of America
U.S. EPA	United States Environmental Protection Agency
VEC	Valued Ecosystem Component
WBM	Water-based Mud
WHMIS	Workplace Hazardous Material Information System
WOR	Produced Water to Oil Ratio

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

1.0 INTRODUCTION

Georges Bank is located in the offshore waters between southwest Nova Scotia and Cape Cod, Massachusetts. It straddles the Canada-United States maritime boundary, with the northeast portion of the Bank in Canadian waters. It is one of the world's richest fishing banks, characterized by a marine ecosystem of high diversity. It has been fished for more than a century by many nations, including Aboriginal peoples and First Nations, and is of major economic and social importance to many coastal communities in Canada and the United States. Other ocean users of the Georges Bank area include maritime transportation, scientific research, telecommunications, and the military. Georges Bank is also an area of interest for oil and gas exploration and development. This followed seismic surveys undertaken by the Geological Survey of Canada more than three decades ago, which identified geological formations typically associated with petroleum reserves.

Offshore petroleum rights exist on Georges Bank, although they have received little notice due to a moratorium on offshore petroleum exploration and development activities in the area. The moratorium was instituted in 1988 pursuant to the *Canada-Nova Scotia Offshore Petroleum Resources Accord Implementation Acts (Accord Acts)*. It was set in place in response to public concern over the potential impacts of offshore oil and gas development on the Georges Bank marine ecosystem and its marine resources. Briefly, in 1988, a moratorium was placed on offshore petroleum activities (i.e. exploration, drilling, and development) on Georges Bank and much of the Northeast Channel until 2000. In December 1999, the federal Minister of Natural Resources and the provincial Minister responsible for the Nova Scotia Petroleum Directorate accepted a Review Panel's recommendation to extend the moratorium until December 31, 2012. On May 13, 2010, the governments of Canada and Nova Scotia again extended the moratorium to December 31, 2015. On December 10, 2010, however, the moratorium was extended indefinitely through Nova Scotia provincial legislation entitled '*Offshore Licensing Policy Act*', unless a resolution is passed in the Nova Scotia House of Assembly accepting a recommendation to allow for offshore petroleum activity on Georges Bank.

Canada has a long-standing history of applying precaution in science-based regulatory programs. The application of precaution is distinctive within science-based risk management and is characterized by three basic tenets: the need for a decision, a risk of serious or irreversible harm, and a lack of full scientific certainty. It is guided by judgment, based on values, and priorities, but its application is complicated by the inherent dynamics of science - even though scientific information may be inconclusive, decisions will still have to be made as society expects risks to be addressed and managed and living standards enhanced. Under the guidance document entitled 'A Framework for the Application of Precaution in Science-based Decision Making About Risk', the application of 'precaution', 'the precautionary principle' or 'the precautionary approach' recognizes that the absence of full scientific certainty shall not be used as a reason for postponing decisions where there is a risk of serious or irreversible harm (Government of Canada, 2003).

In traditional situations of decision making to manage risk, sound scientific evidence is generally interpreted as either definitive and compelling evidence that supports a scientific theory or significant empirical information that clearly establishes the seriousness of a risk. Within the context of precaution, determining what constitutes a sufficiently sound or credible scientific basis is often challenging and can be controversial. The emphasis should be on providing a sound and credible case that a risk of serious or irreversible harm exists. ‘Sufficiently sound’ or ‘credible scientific basis’ should be interpreted as a body of scientific information, whether empirical or theoretical, that can establish reasonable evidence of a theory's validity. This includes its uncertainties, as well as indicates the potential for such a risk. Last, it is noted that regulatory approvals given on the basis of precautionary measures are generally implemented on a provisional basis; that is, they would be subject to review in light of new scientific information or other relevant considerations such as society's chosen level of protection against risk. Follow-up activities, including research and monitoring, are key to reducing scientific uncertainty and allow improved decisions to be made in the future.

Over the past few decades, in addition to global efforts, a significant amount of knowledge from scientific studies and environmental effects monitoring (EEM) programs has been gathered by Fisheries and Oceans Canada (DFO) on the potential impacts of exploration and production from oil and gas activities on marine ecosystems and marine resources. Funding of these research studies was provided by DFO and other sources including the Panel of Energy Research and Development (PERD), the Environmental Studies Research Fund (ESRF), Petroleum Research Atlantic Canada (PRAC), and the offshore oil and gas industry. There have also been major advances in operational techniques, including mitigation protocols, which have been developed in an effort to minimize the potential impacts of offshore petroleum resource recovery on marine ecosystems and marine resources.

This research document is a compilation of Fisheries and Oceans Canada’s most up-to-date ‘state-of-knowledge’ of the potential impacts of typical offshore petroleum activities that may arise in context of marine ecosystems and marine resources. It has been compiled to support discussion that may surround the moratorium on offshore petroleum exploration and development on Georges Bank, while providing readers the information needed to pursue a more detailed review of factors that may surround such discussion.

The document has the following objectives under its Terms of Reference:

- description of the potential impacts of discharges and emissions and zones of influence, if any, from offshore vessels/platforms related to petroleum exploration and development, with a focus on the potential impacts on marine species (e.g. finfish, shellfish, invertebrates and mammals);
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The content of this document builds on earlier reports of Gordon (1988) and Boudreau et al. (1999), which summarized and quantified possible environmental impacts of petroleum exploration activities on Georges Bank. The scope of this document has been expanded to include effects of production operations. The decision to broaden the scope was based on the fact that the overall life of a production field is generally on the order of decades, which far exceeds the duration of a typical seismic surveys or exploratory drilling programs that typically last from tens of days to a few months, respectively. In addition, the potential impacts during the production phase can be considerably different from the potential impacts of exploration activity.

Based on the results of previous studies, the major issues of concern from offshore oil and gas operations include the exposure of marine organisms to operational waste discharges, seismic noise, and accidental oil spills and/or blowouts. Of the waste discharges, spent drilling mud and well cuttings are of primary concern during exploration and development drilling operations, while produced water recovered from the hydrocarbon-bearing strata represents the largest volume of waste generated during production operations. Particular emphasis was given to the review of available information on the potential impacts that may occur in the Georges Bank ecosystem, as a result of future petroleum exploration and production activities.

To capture the full diversity of scientific thought and opinion, the preparation of this report involved a variety of scientific sources, experts from many disciplines, and a formal peer review process. The primary intent of the document is to summarize the existing state of knowledge. Some remaining knowledge gaps have been identified, although this is not the major focus of the document and it should not be viewed as a comprehensive review of research needs. The document only considers the state of knowledge of potential impacts on marine environments associated with offshore petroleum exploration prior to April 2010. It does not consider any new knowledge or lessons that may have been learned from the Gulf of Mexico oil spill associated with the April 20, 2010, accident of the Deepwater Horizon. Furthermore, the document is not to be viewed as an Environmental Impact Assessment (EIA) nor will such an assessment be provided with this document. Last, the document complements the document of Kennedy et al. (2011) entitled ‘The Marine Ecosystem of Georges Bank’ (DFO. Can. Sci. Advis. Sec. Res. Doc. 2011/059. xiv + 232pp).

2.0 PETROLEUM POTENTIAL OF GEORGES BANK

It is difficult to accurately estimate the quantity and types of hydrocarbon resources in the Georges Bank moratorium area. The most recent seismic data was acquired in the early 1980s and is difficult to interpret due to poor resolution, spatial distortion, and displacement (USMMS, 2000). At present, the Nova Scotia Department of Energy (through the Offshore Energy Technical Research Association) is funding research to process the seismic field data that was acquired over Georges Bank in the 1980s; presently this information is not yet available. New seismic data acquisition and processing, both regional two-dimensional (2D) and local three-dimensional (3D), would generate higher resolution images of the geological formations to better map and understand the geology, as well as more accurately assess the hydrocarbon resources in the area (Kidston et al., 2005).

From the 1960s to the early 1980s, approximately 63,000 km of 2D seismic data were collected over the Georges Bank area, of which two thirds were located on the U.S. side of the bank (Kidston et al., 2005). In 1984, JEBCO Seismic Ltd. acquired 6,800 km of data in the area that later became part of the Georges Bank Moratorium area. It remains the most recent seismic dataset available. In contrast, the Canada-Nova Scotia Offshore Petroleum Board has reported that since the early 1960s more than 495,000 km of 2D and 32,000 km^2 of 3D seismic data have been collected in the remaining waters off the coast of Nova Scotia alone.

The Geological Survey of Canada (GSC) undertook resource assessments of the Georges Bank region in 1978 and 1979 based on existing, very limited, and poor quality geophysical data. These reports remain confidential, but a single page summary of the two studies was included in the GSC's 1984 petroleum resource assessment of the Scotian Shelf (Proctor et al., 1984). The authors theorized that the geology and hydrocarbon plays of Georges Bank were similar to those in the similar-sized and gas-rich Sable Subbasin. For the Georges Bank region, the authors' "high confidence" estimate was 62.9 million barrels of oil, the "average confidence" estimate was 1.06 billion barrels (Bbbls) of oil, and the "speculative" estimate was 2.2 Bbbls of oil. For natural gas, the "high confidence" estimate was 1.3 trillion cubic feet (Tcf), the "average confidence" estimate was 5.3 Tcf, and the "speculative" estimate was 10.8 Tcf. It should be noted that these estimates are not considered reliable and should not be compared to more recent estimates of proven reserves in other areas.

Reservoirs of Georges Bank were expected in lower Cretaceous and upper Jurassic age deltaic sandstones located at depths between 2,700-4,900 metres (m), as those below a depth of 4,100 m would be over pressured (USMMS, 2000). Instead of the predicted reservoir-quality, porous carbonates, the ten U.S. wells drilled in the late-1970s to early-1980s encountered low porosity limestones such as micrites, wackestones, and packstones, while high velocity volcanics and halite/anhydrites were responsible for some of the interpreted "bright spots" that in certain cases indicate the presence of hydrocarbon-bearing reservoirs (USMMS, 2000; Kidston et al., 2005). According to USMMS (2000), the source rocks encountered in the wells drilled on the U.S. portion of Georges Bank were poor over the thermal maturity depth range (2,440-6,700 m). No significant hydrocarbon discoveries or shows were encountered in any of the wells. No test wells have been drilled on the Canadian side of the Bank. The existing seismic data, however, suggests that the geology is different in the Canadian portion of the Bank, with a better potential for possible hydrocarbon generation and trapping.

It is important to note that all petroleum resource assessments of Canada's portion of Georges Bank were completed in the 1970s and 1980s, which did not use the then most recent seismic data nor have access to data from the U.S. wells. Up to the imposition of the moratorium, no prospects were identified or mapped by industry and government in water depths greater than 200 m, the-then economic and technological limitations to drilling and production. Therefore, deep water turbidite plays, the Abenaki reef margin, and structures containing ancient shallow water sediments now on the deep water slope, were not considered, though today are targets in the Scotian Basin and elsewhere globally (i.e. up to 3,000 m water depth). Current technology and new petroleum plays and play concepts, however, broadens the area of petroleum potential and exploration in the Georges Bank region. Since imposition of the moratorium, industry has not been permitted to collect any new seismic data and related geoscience information. Lacking new information, government agencies have not undertaken a modern and comprehensive petroleum resource assessment of Georges Bank.

3.0 SEISMIC NOISE

Sound in the ocean is generated by a variety of natural sources (e.g. wind, waves, and the vocalization of marine life) and human activity (e.g. navigation, dredging, pile driving, fishing gear operation, seismic surveys, and military sonar operations). Oceanographic characteristics such as bottom geomorphology, water depth, temperature, salinity, and density can influence the transmission of sound as it travels through the water. For example, sound levels tend to quickly attenuate with range in shallow waters. In deeper waters, sound typically propagates with lower attenuation and, especially so, where acoustic channels exist to conduct and vertically constrain sound energy from lossy interactions with the water surface and bottom (DFO, 2007).

Seismic surveys use sound waves to gather information about geological structures lying beneath the surface of the earth, in order to understand the composition, structure, and deformation history of the earth's crust and/or to locate rock formations that could potentially contain hydrocarbons (Figure 1). The general procedure is for a vessel to transit along straight line transects while towing an array of sound sources (air guns) at a predetermined depth. The sound sources emit a signal capable of penetrating deep into the seabed, with the sound signal reflecting back from the different geological interfaces. The return signals are registered by hydrophones encased in a 'seismic streamer' that is a buoyant cable several kilometres in length and trails behind the seismic vessel. The data that is recovered is processed into an acoustic image of the underlying geological strata from which traps or probable concentration areas of petroleum resources can be identified.

In use since the 1960s, air guns have become the most widely used sound source for seismic surveys. They replaced the use of explosives that entailed both safety and acute environmental concerns. Airguns release a specified volume of air under high pressure, which creates a sound pressure wave from the expansion and contraction of the released air bubble. To yield high intensities, multiple airguns are fired with precise timing to produce a coherent pulse of sound. As described by Greene and Moore (1995), Hildebrand (2005), and Richardson et al. (1995), airguns used by the oil industry are typically arranged in arrays of twelve to forty-eight individual guns of various sizes distributed over a horizontal area approximately 20 m inline by 20 m cross-line, where inline refers to the direction in which the ship is sailing and cross-line is perpendicular to it. In an array, the guns are in 3 to 6 sub-arrays called strings with each string being made up of 6 to 8 guns. The array is towed approximately 200 m behind a vessel, as measured from the vessel's navigation point, and suspended by floats at a depth of 3 to 10 m. The guns operate at pressures of approximately 2,000 pounds per square inch (psi or 137 bar) and fire every 10-15 seconds. To a rough first approximation, the pressure output of an airgun array is proportional to its operating pressure, the number of airguns, and, for a fixed number of guns, the cube root of the total gun volume (Caldwell and Dragoset, 2000).

In order to predict airgun-array sound levels at long range, array source levels are usually expressed as the decibel (dB) pressure level relative to 1 micro Pascal (μPa) observed for an equivalent point acoustic source observed at a 1 m reference range. Typical zero-to-peak source levels for exploration seismic arrays in their primary radiation direction are 245-260 dB relative to 1 μPa at 1 m. While the source level can be used to accurately predict pressures in the 'far field' of the array, an actual seismic array constitutes distributed rather than point source with the result that 'near field' maximum zero-to-peak pressure levels are normally limited to about 190-250 dB relative to 1 μPa. The far-field pressure from an airgun array is focused vertically, with the most important frequency components being enhanced up to 6 dB in the vertical direction by in-phase reflections from the water surface and with broadband vertical sound levels exceeding those radiated in the horizontal direction by typically 8-13 dB depending in part on how these levels are defined and measured (Davis et al., 1998). The peak pressure levels for industry arrays are in the 5-300 hertz (Hz) range. As outlined by the International Association of Geophysical Contractors, industry has adopted a phased sequential approach to the use of seismic noise in their operations: 1) 2D and 3D exploratory operations; 2) more localized geohazard and vertical seismic profiles; and last 3) less frequent 4D operations aimed at determining reservoir changes over time during the production phase (IAGC, 2002).

Figure 1. *A signal source and one or several cables are towed behind a seismic vessel. The sound signals from the air gun are sent into the seabed and reflected by the various stratigraphic layers before being registered by the hydrophones inside the cables (OLF, 2004).*

A modern 3D seismic-reflection survey consists of a series of parallel passes through an area by a vessel towing an airgun array along with six to ten seismic receiving streamers (hydrophone arrays) at a speed of approximately 5 knots (2.5 m s^{-1}). The duration of seismic programs typically range from 14-30 days. In contrast to the 2D methods that use a single air gun array and one seismic cable for mapping widely spaced 2D slices of the sea bottom, 3D methods require the operating vessel to transit along more closely-spaced parallel lines (i.e. 100-500 m apart). They also include multiple hydrophone cables and, in certain instances, several airgun arrays that can produce data sets that can be processed with advanced software to reveal the 3D geometry of the subsurface at high resolution (i.e. fine scale). The offshore petroleum exploration industry has routinely used marine 3D seismic reflection methods for over 30 years to map geological structures down to several kilometers depth and up to resolutions of tens of metres. The aim is to detect hydrocarbon reserves. As noted above, a recent innovation is the use of repeated seismic reflection surveys for 'time lapse' monitoring of producing oil fields, known as four-dimensional (4D) surveys, in which the depletion of producing reservoirs can be directly observed.

The seismic emissions from air guns during survey operations are categorized as pulsed noise, as the sound source is discharged on the order of every ten seconds (at intervals of about 25 m along track). Seismic survey-related noise can also appear in more continuous forms, such as that generated by engines, propellers, motors and pumps, etc. The extreme length of the seismic streamer, which contains receivers, cabling, and oil to maintain its neutral buoyancy, reduces the seismic vessel's maneuverability and, therefore, the vessel requires a large turning radius to transfer its operations from one line to the next. Apart from potential impediments to navigation and small oil leaks, however, the seismic streamer itself is not considered an environmental problem (Figure 2).

Concerns have been raised about the potential impacts of seismic noise on marine species. In response to these concerns, DFO has produced a number of publications that summarize the results of recent noise impact studies, as well as provide an overview of scientific knowledge and recommendations for future research on the impacts of seismic sound on fish, invertebrates, marine turtles, and marine mammals (DFO, 2003a; DFO, 2004; Moriyasu et al., 2004; Payne, 2004; Courtenay et al., 2009). The following sections on the interactions of seismic noise with marine species do not attempt to duplicate these reports, rather they attempt to identify key issues and expand upon new research conducted since their publication.

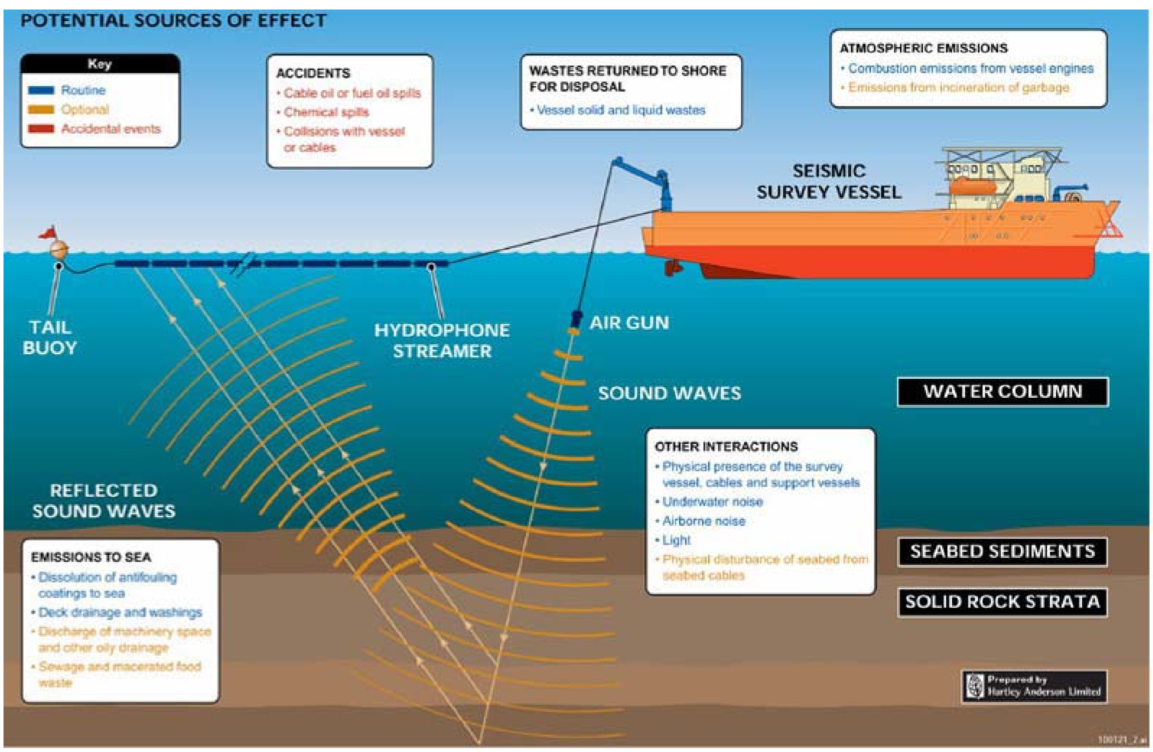


Figure 2. *Potential sources of environmental effects from a seismic survey (Hurley, 2009).*

3.1 IMPACTS ON MARINE ENVIRONMENT

3.1.1 Impacts on Fish and Invertebrates

Numerous studies have shown that organisms can be killed or damaged when exposed to sound pulses that have a rapid rise time that exceed 220 - 230 dB (due to rapidly increasing sound pressure) (reviews by: OLF, 2004; Turnpenny and Nedwed, 1994). Sound pulses from air guns typically have relatively slow rise times, which enable organisms to tolerate higher peak pressure levels than those generated by explosive charges underwater. Sound pulses with a peak pressure of more than 230 db only occur in the immediate vicinity of air guns, within a radius of a few meters.

Studies conducted in Norway (e.g. reviewed by Dalen et al., 2007) and Canada (e.g. Payne, 2004) have conclusively shown that very high levels of seismic noise can adversely impact the eggs and larvae of fish and shellfish. Within the near-field, the sound source generates a compression and decompression wave in the water that impacts certain life stages. Due to three-dimensional dispersion of spreading, the energy decreases quickly with distance from the source. From experiments reported to date, results show that exposure to sound may impact the development of eggs and cause developmental abnormalities in a small proportion of exposed eggs and/or larvae. Results have also shown that injuries and increased mortality from air gun shooting generally occur at distances less than 5 m from the air gun, with the most frequent and serious injuries occurring at distances of less than 1.5 m. Fish in the early stages of life are most vulnerable. In addition, there is also a question of whether chronic low level sound exposures in a given area over a period of 2-3 weeks or more might adversely affect any of a variety of important physiological functions in fish or shellfish.

The potential for population-level impacts as a result of damage to eggs and larvae in the near-field is less clear. Boudreau et al. (1999) postulated that seismic operations in the vicinity of a frontal system or convergent zone that would at certain times of the year have higher densities of eggs and larvae, such as those known to occur on Georges Bank, may cause a significant reduction in year-class size. Following detailed analysis of data, Dalen et al. (1996) concluded that there would be such a small quantity of eggs and fry present within the danger zone that damage caused by air guns would have no consequences for the fish stocks. Thus, according to Dalen et al. (1996), seismic induced mortality is not expected to have a significant negative impact on recruitment. Field studies have not been conducted to test this hypothesis.

There is increasing evidence that significant, but less acute, damage may affect fish from considerably lower sound levels. Damage to the sensory epithelia (i.e. hearing organs) of adult fish has been reported by McCauley et al. (2003), who caged pink snapper and subjected them to the repetitive firing of air guns. Signs of damage to the sensor hair cells in the inner ear were observed as early as 18 hours after exposure to the air gun noise. The damage to the caged animals was severe, with no evidence of repair or replacement of damaged cells even up to 58 days post-exposure. Peak acoustic levels were of the order of 180-190 dB root mean square (RMS) corresponding to predicted acoustic levels less than 500 m from a large exploration seismic array. The fish were caged and unable to swim away from the seismic source as it approached. It is expected that in the wild fish would swim away from the seismic source, which is an anticipated natural avoidance response (at least for some species) that would generally reduce exposure levels. The authors have also observed fish exhibiting disorientation and abnormal swimming behaviour, which might indicate vestibular damage.

There have been no documented cases of large-scale fish mortality due to exposure to seismic sound under field operating conditions. Seismic noise, however, has the potential to have at least short term impacts on certain fish species such as provoking a startle response, changes in swimming patterns (potentially including alterations in swimming speed and directional orientation), and changes in vertical distribution (Worcester, 2006). Such effects have been observed up to a radius of more than 30 km from the sound source (Engås et al., 1996). If fish on route to spawning grounds are exposed to this type of noise or, if they are exposed to seismic noise during actual spawning, the effects may impact their spawning success. For instance, exposed fish may expend more energy on the spawning journey than fish that are not exposed to seismic noise, while spawning itself may be deferred in time or displaced in space (Worcester, 2006).

There have been a number of studies reporting the potential impacts of seismic shooting on the behaviour of fish and catch rates (e.g. Dalen and Raknes, 1985; Pearson et al., 1992; Skalski et al., 1992; Løkkeborg and Soldal, 1993). To collect quantitative data on the spatial and temporal extent of the effects of seismic operations, Engås et al. (1996) conducted a controlled, full-scale experiment to determine if seismic exploration affected abundance or catch rates of cod (*Gadus morhua*) and haddock (*Melanogrammus aeglefinus*). This study, which involved both acoustic mapping and fishing trials with trawls and longlines, was conducted in the central Barents Sea seven days prior, five days during, and five days after seismic shooting with air guns. Results showed that seismic shooting affected fish distribution, local abundance, and catch rates in the entire investigation area of 40 by 40 nautical miles.

Trawl catches of cod and haddock, and longline catches of haddock, declined on average of approximately 50% by mass following the commencement of seismic shooting. This observation was in agreement with the acoustic abundance estimates. The average longline catches of cod decreased by 21% by mass following the commencement of seismic shooting. Reductions in catch rates were observed up to 18 nautical miles from the seismic shooting area (which was 3 by 10 nautical miles in size). The most pronounced reduction in catch rates, however, occurred in the seismic shooting area, where trawl catches of both cod and haddock, and longline catches of haddock, decreased on average by approximately 70% by mass. The longline catches of cod decreased on average approximately 45% by mass. Interestingly, a greater reduction in catch rates by mass was found for larger fish (i.e. greater than 60 cm in length) than for smaller fish (both in trawl catches and acoustic estimates). Last, Engås et al. (1996) found that the abundance and catch rates did not return to preshooting levels during the five day post-seismic shooting period.

Hassel et al. (2004) studied the influence of seismic shooting on a sandeel (*Ammodytes marinus)* in the North Sea. Remote camera systems were used to observe caged animals exposed to full-scale seismic shooting for approximately 2.5 days. Comparisons were made to animals in a control area about 35 km southeast of the seismic shooting area. The results indicated that the seismic shooting had an effect on the behaviour of the sandeel. Changes in the distribution and abundance of the sandeel during daytime in the experimental region could not be detected by acoustic surveys. Repeated grab surveys were also conducted with a van Veen grab at night at predetermined locations during the experimental period, both prior to and after seismic shooting. No differences in lethality were observed in both the caged animals and grab sampled animals. However, analyses of landing data from the Norwegian sandeel trawlers fishing within the area of seismic shooting showed a drop in the sandeel landings for a short period during and after the ensonification period.

As described in an overview by Dalen et al. (2007), scare effects on fish can result in catch reductions that vary by species and by the fishing gear being used. For example, a Norwegian survey demonstrated reduced trawl catches of cod out to approximately 33 km from a seismic sound source, while another study demonstrated reduced line catches out to approximately 8 km from the sound source. Results of a study in Australia demonstrated that scare effects of fish out to distances of 1-2 km from a seismic vessel. The Australian data could not be used to determine if the frightened fish returned to the area that they abandoned or re-appeared in some other manner (e.g. fish migrated off the bottom into echo sounder range).

Løkkeborg et al. (2010) recently completed a study for the Norwegian Petroleum Directorate to evaluate the degree to which fish occupying a designated study area off Vesterålen were affected by seismic shooting activity. Seismic data were collected in an area bounded by 8 by 46 nautical miles during a period of 38 days (June 29 to August 6, 2009), as the vessel *Geo Pacific* conducted a standard 3D seismic data acquisition survey. The seismic survey site overlapped with an area of traditional fishing grounds for Greenland halibut (*Reinhardtius hippoglossoides*), redfish (*Sebastes marinus*), pollock (*Pollachius virens),* and haddock (*Melanogrammus aeglefinus*). The survey was scheduled during a period when few gadoids would be expected in the area and, thus, fishing activity was limited. During the seismic survey, two gillnet boats fished for Greenland halibut, pollock and redfish, while two longline boats fished for Greenland halibut and haddock. Fishing operations began 12 days prior to the seismic survey and continued up to 25 days following completion of the seismic program. The research vessel *Håkon Mosby* and chartered fishing vessel *Eros* carried out an acoustic survey of the distributions of fish and plankton in the area, in order to determine whether changes in abundance could be observed before, during, and after the seismic survey. Sound measurements were also taken at a range of depths and distances from the seismic air-guns.

Results of the Løkkeborg et al. (2010) study revealed that the sound of the air-guns affected the fisheries in the study area off Vesterålen in a number of ways. Specifically, fish catches of individual species increased and decreased by fishing gear type. For example, gillnet catches of Greenland halibut and redfish increased during seismic shooting and remained high following the survey compared to observations made prior to the survey. In contrast, long-line catches of Greenland halibut decreased during the seismic survey, but subsequently increased during the 25-day period following the survey. Results for pollock demonstrated a decrease in gillnet catches both during and after the seismic survey, although the differences were not statistically significant.

The decline in gillnet catches of pollock was in agreement with estimates from the acoustic survey. In general, the acoustic survey demonstrated a decrease in quantity of pollock in the vicinity of the seismic survey and in the adjacent area close to land, during seismic shooting, although the difference was only statistically significant in the shoreward area where the pollock were most abundant. This finding was interpreted as an indication that pollock left the area in response to the seismic survey. Unfortunately, the high degree of variance in the data made it difficult to determine how long the reduction in pollock lasted following the survey. Last, by-catch of ling (*Ophiodon elongatus*) in redfish and pollock nets increased immediately after the start of seismic shooting. After a few days of seismic activity, ling catches subsequently decreased and, during the observation period following the seismic survey, had returned to similar or slightly lower levels than those observed prior to the seismic survey.

There were large day-to-day variations in long-line catches of haddock, although statistically-significant differences in haddock catch rates observed before and during the seismic survey were not observed. The area in which the haddock fishery took place was less affected by the sound of the air-guns than the fishing grounds for other species, since there was no direct overlap between this area and the seismic vessel transects (Løkkeborg et al., 2010). It was only during the final three days of the seismic survey that the survey vessel was less than one nautical mile from the haddock longlines for a short period each day. Nevertheless, there was a decreasing trend in haddock catches towards the end of the period of seismic activity, with a positive correlation between catch rates and distance from the seismic vessel being observed (Løkkeborg et al., 2010). In short, the study demonstrated that catches decreased when the distance to the survey vessel decreased, suggesting that haddock catch rates were affected by the sound of the airguns.

Acoustic survey results on the distribution of demersal fishes in the seismic survey area largely confirm the results of the fishing experiments. During seismic shooting, lower abundances of pollock were measured in the area between the seismic survey area and land, although there was also a tendency for a decrease in pollock abundance in the seismic survey area. No changes in abundance or distribution of other demersal fishes could be ascribed to the seismic campaign, although concentrations of juvenile herring (*Clupea harengus*) were observed to migrate out of the seismic survey area during the experiment. This behaviour, however, was also observed to occur in the control area and was interpreted as the natural feeding migration for juvenile herring.

Plankton observations demonstrated uniform concentrations throughout the study area, with no changes in plankton concentration or distribution being observed throughout the study period. In addition, the analysis of fish stomach contents indicated no change in food uptake as a result of the seismic survey. Decreases in the stomach contents of pollock and haddock of gillnet and longline fishing experiments during the seismic survey, however, were observed, although no decrease in the stomach contents of trawl-caught fish was observed. A decrease in the amount of herring found in the stomach contents of pollock captured by gillnet corresponded to the acoustically-measured decrease in abundance of herring in the experimental area. This was attributed to the natural migration of herring out of the study area.

Results of the Løkkeborg et al. (2010) study indicate that all the fish species studied reacted to sound associated with the seismic air-guns, as catch rates of the various species either increased or decreased during seismic shooting. Sound measurements demonstrated that all of the fish studied were exposed to sound levels above their hearing threshold and in a range where obvious changes in swimming activity can be expected. The findings are supported by fish raising their level of swimming activity, thus, causing Greenland halibut, redfish, and ling more vulnerable to be taken by gillnets, whereas pollock likely migrated out of the study area. The increase in swimming activity of fish may be a symptom of a stress reaction that could lead to reduced longline catch efficiency, either as a result of reduced feeding motivation or because the fish migrated out of the study area. Due to study limitations, it was not possible to determine the maximum distances within which the fish reacted to the sound of the air-guns.

In the Bass Strait of Australia, scallop dredge fishermen expressed concern that seismic testing may increase the mortality of larval scallops, weaken the adductor muscle, and/or kill adult scallops. In response, Esso Australia commissioned a study in 2001 to investigate the impacts of seismic testing. The effects of seismic testing on adult scallops were measured by comparing the mortality and adductor muscle strength of scallops deployed in an area subject to seismic testing with those in a control area distal from the seismic survey area. The effects of seismic testing on plankton, including larval scallops, were measured by comparing plankton communities immediately behind the seismic vessel with those sampled before and 2 km away from the seismic testing. The study found that the mortality rate and adductor muscle strength of scallops suspended 19 m below the surface in the path of the airgun array were not significantly different than those of scallops placed at the control site 20 km away (Exxon Norge AS, 2001).

In the Bay of Biscay, Guerra et al. (2004) noted that two incidents of giant squid (*Architeuthis dux*) strandings appeared to be linked spatially and temporally to geophysical prospecting operations involving air-gun arrays. Evidence of acute tissue damage was observed in the stranded and surface-floating giant squids. The incidence of these cases during periods of geological and geophysical studies using acoustic air guns generating low frequency (<100 Hz), high intensity (200 dB) sounds suggested that acoustic factors caused or contributed to organ and tissue lesions that likely caused the squid mortalities.

In Atlantic Canada, there is evidence from recent studies suggesting that invertebrates are sensitive to acoustic sound (Christian et al., 2003; Payne et al., 2007). In comparison to controls, Christian et al. (2003), as part of a larger preliminary study in which little or no effects were observed, reported delayed embryonic development in fertilized snow crab eggs exposed to 200 seismic shots at 180 dB RMS. To follow up on the preliminary study, a major field study was conducted by DFO, in conjunction with a seismic survey conducted by Corridor Resources off the west coast of Cape Breton, Nova Scotia. A caging study examined short (i.e. 12 days) and medium (i.e. 5 months) term differences in the morphology and physiology of egg-bearing female snow crab at test and control sites. Three definitive conclusions were made following peer-review of the work (DFO, 2004): 1) the seismic survey did not cause any acute or mid-term mortality of the crab nor was there any evidence of changes to feeding in the laboratory; 2) survival of embryos being carried by female crabs, as well as locomotion of the resulting larvae after hatch, were unaffected by the seismic survey; and 3) in the short term, gills, antennules, and statocysts (i.e. balance organs) were soiled in the test group, but they were found to be completely cleaned of sediment when sampled five months later.

Other significantly different observations in the DFO (2004) study between the test and control sites, such as bruising of the hepatopancreas and ovaries (including dilated oocytes with detached chorion), delay of embryo hatch and smaller size of larvae, and turnover time of crabs in the test site, could not be explained due to confounding factors such as temperature differences between sites. This was subsequently addressed in a series of follow-up studies (e.g. Courtenay et al., 2009), which highlighted the need for alterations in the experimental design of field trials. For example, the slower rate of embryo development between the test and control groups could be attributed to temperature differences as well as exposure to seismic sound. Following statistical analysis of the data, Lee and Wright (2009) concluded that no significant correlations could be made with seismic exposure and observed histopathologies in the crab hepatopancreas and gonad tissues. Furthermore, a high degree of correlation was observed between the degree of histological abnormalities and caging time. As a result, it was concluded that chronic damage/recovery was difficult to assess due to confounding events associated with the crowded cage conditions and limited food supply.

Pearson et al. (1992) investigated the effects of airgun discharges on survival and growth of Dungeness crab (*Cancer magister*) larvae. An array of seven airguns discharging as close as 1 m away from the larvae did not affect crab larval survival. Upon completion of a subsequent independent follow-up study on snow crab, Payne et al. (2008) stated:

> *Evidence supporting a hypothesis that the various effects observed were due to normal variability (and not due to seismic) has also recently been obtained from an ESRF supported study in Newfoundland (DFO, NL Region, unpublished). Female crab were exposed to higher sound levels than those measured at the test site in the Cape Breton study and maintained in the lab for several months. No difference was observed with respect to mortality, leg loss, egg loss or hepatopancreas and ovary histopathology. The results support the earlier preliminary study on snow crab carried out by Christian et al. (2003).*

Lobster (*Homarus americanus*) is one of the most commercially-important species in Atlantic Canada. Payne et al. (2007) have conducted pilot studies in the laboratory and field to investigate the potential effects of seismic sound on lobster health. A number of biological endpoints, including sub-lethal responses, were assessed in animals exposed to ‘low level’ (202 dB peak-to-peak) and ‘high level’ (to 227 dB peak-to-peak) sound. The endpoints included: lobster survival; food consumption; turnover rate; serum protein; serum enzymes; and serum calcium. Exposure of lobster to very high and low sound levels had no effect on delayed mortality or damage to mechano-sensory systems associated with animal equilibrium and posture, as assessed by turnover rates (Payne et al., 2007). There was also no evidence for loss of legs or other appendages. Sub-lethal effects, however, were observed with respect to feeding and serum biochemistry, with effects sometimes being observed weeks to months after exposure. A histochemical change was also noted in the hepatopancreas of animals exposed 4 months previous, which may be linked to organ 'stress'. These initial studies were meant to be exploratory in nature and caution is warranted about over interpretation. They do however point to the need for more comprehensive studies regarding the potential for seismic surveys to affect lobster. Studies on moulting and effects on egg development and animal behavior are recommended (Payne et al., 2007).

Exposure to seismic sound is considered unlikely to result in direct invertebrate mortality, but the question of the potential for delayed effects of an adverse nature requires further investigation. Seismic noise has the potential to cause short term impacts on some invertebrate species, including startle responses and changes in swimming/movement patterns (such as changes in swimming/movement speed and directional orientation). Moriyasu et al. (2004) felt that the ecological significance of these effects would be low, except if effects of exposure to seismic sounds were to influence reproduction or growth/moulting activities or lead to a dispersion of spawning aggregations or deflection from migration paths. Parry and Gason (2006) studied catch rates of rock lobster (*Panulirus interruptus*) following seismic surveys and reported that there was no evidence of a relationship between seismic surveys and long-term changes in catch rates. They indicated, however, that seismic-induced mortality would have to be major (i.e. in the 50% range) before it could be resolved from natural and fishing mortality. Parry and Gason (2006) suggest that perhaps because most invertebrates do not contain sound sensitive organs, such as air bladders like those found in fish (Keevin and Hempen, 1997), invertebrates may be less vulnerable to proximal loud sounds/explosions.

3.1.2 Impacts on Marine Mammals

Although seismic airgun arrays are designed to direct the majority of emitted energy downward through the seafloor, their sound emission horizontally is also significant (NRC, 2003a). Six autonomous hydrophones moored near the Mid-Atlantic Ridge (from latitude 15-35°N and longitude 33-50°W) to record spectrograms of whale vocalizations clearly picked up the sound of seismic airgun activity from locations over 3000 km from the array (Nieukirk et al., 2004). The broadband frequency range and repeated firing of these guns make them a major contributor to the low-frequency sound field in the North Atlantic. The effects of seismic noise on whales are not fully known. Possible effects include masking of conspecific sounds, increased stress levels, changing vocalizations, abandonment of important habitat, ear damage, and alteration of reproductive or immune responses (Richardson et al., 1995; Hildebrand, 2005; Weilgart, 2007).

Clark and Gagnon (2006) noted that Mysticete whales produce a wide variety of communication sounds in the very low frequency range (<100 Hz), and possess auditory systems well adapted for hearing low frequency sounds (<1000 Hz). The low-frequency energy associated with seismic operations may cover spatial areas of ecological importance over time periods of biological significance. A comprehensive assessment of the potential impacts linked to such chronic exposures to individuals and populations, either alone or in synergistic combination with other stressors, was recommended to fully document proximate and cumulative exposure levels and the types and scales of responses (e.g. behavioral, endocrinological, physiological, neurophysiological) within the proper ecological context. Di Iorio and Clark (2009) noted that blue whales (*Balaenoptera musculus*) in the St. Lawrence River Estuary changed their vocal behaviour during a seismic survey that deployed a low-medium power technology (sparker with source level of 193 dB relative to 1 µPa peak to peak). The whales were found to consistently call more on seismic exploration days than on non-exploration days, as well as during periods on a seismic survey day when the sparker was operating. This is in line with the prediction from information theory, which indicates an increase in call production to compensate for the masking of information by noise.

The results of a comprehensive study by Stone and Tasker (2006) on the effects of a seismic survey in the United Kingdom on cetaceans demonstrate that cetaceans can be disturbed by seismic exploration. Sighting rates, distance from the airguns, and orientation were compared for periods when airguns were active and when they were silent, both for surveys with airgun arrays of large volume and surveys with smaller volume arrays. Small odontocetes demonstrated the strongest lateral spatial avoidance (extending at least as far as the limit of visual observation) in response to active airguns, while mysticetes and killer whales demonstrated more localized spatial avoidance. Long-finned pilot whales demonstrated only a change in orientation and sperm whales demonstrated no statistically significant effects. Responses to active airguns were greater during those seismic surveys with large volume airgun arrays than those with smaller volumes of airguns.

Stone and Tasker (2006) suggested that the different taxonomic groups of cetaceans may adopt different strategies for responding to acoustic disturbance from seismic surveys; some small odontocetes move out of the immediate area and the slower moving mysticetes orient away from the vessel and increase their distance from the source, but do not move away from the area completely. The study concentrated on examining short-term effects of airgun activity on the occurrence and orientation of cetaceans. Long-term effects, effects on vocalizations, behaviour and physiology, consequences of auditory masking, and the potential for damage to hearing were not considered in the experimental design. The lack of an observed response in some species does not, therefore, imply that the use of seismic airguns has no effect on those species. Furthermore, although the responses that were observed were short-term effects, it is not known whether these may have been biologically-significant effects that persisted beyond the time of disturbance, responses that affected the ability of animals to engage in essential activities (e.g. breeding, feeding, caring for young, migrating, etc.), or effects that had consequences at the population level. Difficulty in determining the biological significance of observed effects is recognized (NRC, 2003a; 2005).

There is evidence that seismic noise may inflict physical impacts to marine mammals. For example, there have been observations, under experimental conditions, of sub-lethal, temporary elevations in hearing thresholds (temporary threshold shift or TTS) in captive marine mammals exposed to pulsed sounds. A panel-based consensus of the best science-based criteria for marine-mammal noise exposure considering, for instance, the varying frequency domain auditory sensitivity of differing categories of marine mammals has been published by Southall et al. (2007) and may well serve as a starting point for refined species-specific exposure limits in the future. There are a few indications that marine mammals suffer injury to internal organs as a consequence of anthropogenic sound. A database was established from the results of studies on three mass strandings of beaked whales that occurred during the same time period as naval exercises involving sonar. There was strong evidence that there were bubble-induced lesions and fat embolisms in vital organs consistent with 'the bends'. The authors concluded that observed damage was unlikely to have been caused directly by acoustic exposure but rather from altered diving patterns induced by the exposures. There are no such observations for seismic surveys (Dalen et al., 2007).

Exposure to seismic sound can result in displacement and/or migratory diversion in some marine mammals, but this effect is species, individual, and contextually related. The ecological significance of such effects is unknown, but there are conditions under which the worst case scenarios could be high, such as: feeding marine mammals being displaced from areas where there are no alternates; marine mammals being displaced from resting areas where there are no alternates; marine mammals being displaced from breeding or nursery areas; or migrating animals being diverted from routes for which alternate routes either do not exist or would incur substantially greater costs to traverse. In addition, exposure to seismic sound can result in changes in dive and respiratory patterns in some marine mammals, but this effect is expected to vary with species, individual, and context. Furthermore, when dealing with species-at-risk (SARA – listed species), detrimental effects suffered by one individual can translate into detrimental effects on the population.

There is evidence that exposure specifically to seismic sounds has sometimes caused changes in vocalization patterns in marine mammals. It has not been possible, however, to measure the functional consequences of these changes (such as loss of contact between individuals or reduced ability to coordinate social behaviours), if any, nor the percent of time with which they would occur (Lawson and McQuinn, 2004). A field study was conducted in the Gully and outer Scotian Shelf of Atlantic Canada in 2003 to determine the impact of a seismic exploration program on marine mammals, including the endangered Northern Bottlenose Whale as listed pursuant to the *Species at Risk Act* (Lee et al., 2005a). This program included studies of seismic sound levels, as well as behaviour, vocalizations, and distributions of marine mammals both in close proximity to the seismic vessels and in adjacent areas including the Gully submarine canyon and shelf edge of the Scotian Shelf during an exploratory 2-D and 3-D seismic program. Marine mammals were observed both visually and acoustically. Marine mammals appear to have avoided very close ranges (<100 m) from the seismic array during seismic acquisition, but the overall number of marine mammals in the observable radius (1-2 km) did not change significantly when the seismic source was ‘on’ or ‘off’. Marine mammals were observed in larger groups and appeared to have become less vocal when the seismic source was active (Potter et al., 2005).

Whale vocalizations were recorded at all stations and analysis of results indicate that they did not abandon the Gully area while 3D seismic exploration was carried out in the survey area, at ranges larger than 30 km (Simard et al, 2005). Changes in the composition, distribution, and abundance of marine mammal species between the spring and summer surveys were most likely attributed to seasonal variation rather than the effect of seismic activity (Gosselin and Lawson, 2005). The validity of any assessment regarding potentially-harmful impacts of seismic sound on marine mammals will depend crucially on the accuracy and applicability of acoustic propagation models and the data used in the process. During the study, measured sound levels were significantly higher than the model predictions at several stations (Cochrane, 2005; McQuinn and Carrier, 2005). The results demonstrated the importance of using sophisticated models, accurate model parameterizations, including proper representations of the source at frequencies above 200 Hz, the necessity of field validation, and the ongoing need for improved instrumentation for monitoring both seismic sounds and the vocalization of marine mammals (Vagle et al., 2005).

Det Norske Veritas conducted a review of reported incidents involving seismic noise and marine mammals (Dalen et al., 2007. They concluded that there has been no documented sea mammal mortality as a consequence of seismic surveys, as studies of individual incidents in which whales have stranded and seismic activity has occurred in the same area during the same time have been unable to document a direct cause and effect link. Nor are there any documented injuries to sea mammals as a result of seismic surveys. The effects observed were typical changes in behavior, such as whales leaving areas where there was seismic activity. While it was confirmed that seismic surveys may have certain negative consequences for marine life in the nearby area, their review concluded that there are no results that indicated serious and long-lasting harm to populations of fish and sea mammals (Dalen et al., 2007).

Many marine mammals use the Georges Bank area on their migratory routes and as a feeding ground because of the high densities of prey available. Although there are no documented cases of marine mammal mortality resulting from exposure to oil and gas exploration seismic surveys, immediate behavioural reactions as a result of exposure to seismic sound have been widely documented for several classes of marine organisms, and especially for marine mammals, in the latter case precipitating behaviours resulting in the animals temporarily avoiding the immediate area of the sound source or reducing their own vocalisations. Despite the research undertaken to date, additional research may be needed to determine the ecological significance of effects attributed to the exposure to seismic sound linked to:

- changes in marine mammal social behaviour;
- reduced communication and echo-location efficiency in marine mammals;
- hampering of acoustic detection of prey by marine mammals;
- increase in the vulnerability of marine mammals to predators;
- hampering of parental care or bonding in marine mammals;
- reduction in the ability of marine mammals to avoid anthropogenic threats;
- chronic effects on marine mammals; or
- indirect effects on marine mammals.

3.1.3 Impacts on Sea Turtles

While there has been an increased interest in sea turtles because of the endangered or threatened status of some species, relatively little is known about the sensitivity of these species to sound, including seismic noise. Studies do indicate that sea turtles are able to detect and respond to sound frequencies in the range generated during seismic surveys. Scientific studies that have been conducted to date (O'Hara and Wilcox, 1990; Moein et al., 1994; McCauley et al., 2000) have provided evidence of short-term physical response (e.g. change in hearing sensitivity), physiological response (e.g. increased levels of creatine phosphokinase, glucose, and white blood cell counts), and behavioural responses (e.g. increased swimming speed and activity) of caged turtles within 500 m of an airgun source.

A few studies have included observations of sea turtles and sea turtle behaviour in the vicinity of seismic surveys. For example, Eckert et al. (1998) attempted a behavioural study of free ranging leatherback turtles in the proximity of a seismic survey, however, limited reporting of experimental detail make results difficult to interpret. Eckert et al. (1998) also attempted to estimate possible broader scale response of sea turtles to seismic noise based on information available from non-seismic related studies. Using the peak pressure level required to obtain temporary threshold shift in a desert tortoise (~120 dB above best hearing threshold with repeated exposure) and the reported sensitivity of the green turtle in air (65-79 dB re 1 μPa), he predicted that repeated exposure to airgun pulses above 185-199 dB re 1 μPa (conservative estimate) could have long-term effects on hearing. It is not clear that this is a valid approach.

Turtles were observed during seismic operations in Brazil between June 2002 and August 2003 (Moreira-de-Gurjao et al. 2005; Parente et al., 2006). Three species of turtle were identified. Most identified turtles were green sea turtles (*Chelonia mydas*); one loggerhead turtle (*Caretta caretta*) and one olive Ridley turtle (*Lepidochelys olivacea*) were also seen. The turtle sightings rate in quiet periods (0.075 turtles per hour) was higher than that during seismic surveying (0.054 turtles per hour). There did not appear to be significant differences in the behaviour of turtles based on whether or not the airguns were active, though swimming velocity and direction was not recorded.

Visual observations for marine turtles have taken place during all 11 Lamont-Doherty Earth Observatory seismic surveys conducted since 2003 (Holst et al. 2006). During use of large seismic sources (11-20 airguns; 3050-8760 cubic inches), the mean closest point of approach for turtles was closer during non-seismic (139 m) than seismic (228 m) periods ($P<0.01$). Using smaller seismic sources (6 airguns; 75-1350 cubic inches), the mean closest point of approach for turtles was also much closer during non-seismic (120 m) than seismic (285 m) periods ($P<0.001$). The conclusion of this report was that sea turtles showed localized avoidance during large and small-source surveys.

Observations of turtles were made during two consecutive 3D seismic surveys off northern Angola between August 2004 and May 2005 (Wier, 2007). Two hundred turtle sightings were recorded, including 33 olive Ridley turtles, three leatherback turtles (Dermochelys coriacea), four loggerhead turtles, and 160 unidentified turtles. The turtle sightings rate in quiet periods (0.43 turtles per hour) was double that during seismic surveying (0.20 turtles per hour). There was no significant difference in the median distance of turtle sightings during active airgun use as compared to quiet periods. While a slightly higher proportion of turtles dived during active airgun use (12.5%) as compared to quiet periods (11%), most turtles (77% during seismic and 83% during quiet) continued to remain at the surface as the vessel passed. Diving reactions were also observed in response to visual detection of the vessel, the towed surface floats, and the inactive airgun array. Observations were only made of turtles at the sea surface (as opposed to turtles swimming below the sea surface) where received sound levels are expected to be reduced.

Based on studies conducted to date, it is considered unlikely that sea turtles are more sensitive to seismic operations than cetaceans or fish. Therefore, mitigation measures designed to reduce risk or severity of exposure of cetaceans to seismic sounds may be informative concerning potential measures to reduce risk or severity of exposure of sea turtles to seismic sounds. There remains however a lack of research on the acoustic sensitivity of sea turtles and on the importance of the acoustic environment for sea turtles. Differences in functional morphology and hearing capabilities among species and life history stages have not been well documented in the literature (Bartol and Musick, 2003).

Investigations on the potential impacts of seismic noise have only been conducted for a limited number of species. Studies on the potential for noise induced hearing damage in turtles, including structural damage or damage to hair cells, are extremely limited. Studies on the responses of free ranging turtles to seismic noise are also limited and are dominated by EEM observations from seismic vessels. In addition to impacts from seismic noise itself, there is also the potential for impacts to sea turtles as a result of direct interaction with seismic vessels and gear. There have been reports of such interactions during seismic surveys in other parts of the world, particularly with regards to interactions with seismic tail buoys. Some work has been done to develop mitigation measures to minimize these types of interactions (e.g. Ketos Ecology, 2007), including development of turtle exclusion devices or 'turtle guards'. The effectiveness of these measures is not well known.

3.2 MITIGATION MEASURES

A number of protocols have been developed in an effort to minimize impacts of seismic surveys, such as: (i) good survey pre-planning to minimize contacts with sensitive marine mammals, endangered marine mammals, as well as to avoid critical fish spawning areas and periods; (ii) "soft starts" of the seismic source to allow mammals (and other sensitive species) the opportunity to vacate the area before commencing the survey; (iii) use of dedicated visual marine mammal trained observers whenever possible with the authority to suspend a survey under pre-defined criteria; and (iv) employment of Passive Acoustic Monitoring (PAM), which is a rapidly developing technology that enables detection of actively vocalizing marine mammals during periods of darkness or reduced visibility (not effective for non-vocalizing animals).

Based on a scientific peer-review of the potential impacts of seismic sound on marine species (DFO, 2004) and an assessment by technical experts of best available and internationally-recognized techniques to mitigate the effects of seismic sound in the marine environment, a group of federal and provincial experts in marine regulatory policy and practice from DFO, Natural Resources Canada, Indian and Northern Affairs Canada, and the provinces of British Columbia, Newfoundland and Labrador, Nova Scotia, and Quebec, developed a 'Statement of Canadian Practice with respect to the Mitigation of Seismic Sound in the Marine Environment' (hereafter referred to as the 'Statement') (Government of Canada, 2007). The Statement sets out minimum standards that are intended to apply to all seismic activities that use air source arrays in Canada's non-ice covered marine waters. It was intended to complement existing environmental impact assessment processes, including those pursuant to settled land claims, as well as existing regulatory requirements that currently govern marine seismic activities. The Statement is considered a 'living' document and is intended to be reviewed and updated every two years.

In the Statement, mitigation requirements are categorized under the following headings:

- Planning Seismic Surveys;
- Safety Zone and Start-up;
- Shut-down of Air Source Array(s);
- Line Changes and Maintenance Shut-downs;
- Operations in Low Visibility; and
- Additional Mitigative Measures and Modifications.

Variations to the mitigation measures set out in the Statement may be allowed if persons wishing to conduct seismic surveys provide an equivalent or greater level of environmental protection. For example, the Canada-Nova Scotia Offshore Petroleum Board (CNSOPB) requires all seismic operators to carry an independent fisheries observer onboard the seismic vessel to minimize interactions between the seismic vessels and their air gun arrays with fishing vessels and their gear. Dedicated observers are also frequently included in programs to record sightings of marine mammals, sea turtles, and sea birds during seismic programs.

Hurley (2009) noted that companies proposing to conduct seismic surveys in Canadian marine waters may be required to have site-specific environmental mitigation measures to further reduce the risk of harm to marine life. Recent examples of enhanced mitigation measures that have been used in the Nova Scotia offshore include:

- avoidance of special marine areas (e.g. whale sanctuaries and marine protected area);

- increased size of the safety zone (minimum radius of 500 m) around air gun arrays, in order to increase the area over which seismic mitigation measures such as ramping up and shut-down are applied during operations;
- use of PAM to detect presence of whales below the sea surface or during times of low visibility;
- acoustical modeling and monitoring to verify seismic noise zone of influence, as predicted in the environmental impact assessment;
- scheduling of survey operations to minimize interactions with fisheries; and
- alteration of vessel speed and orientation of seismic lines to reduce sound energy levels that are required.

Research and development of alternative technologies are also ongoing. These include new sound sources such as marine vibrator systems (e.g. marine Vibroseis), electro-mechanical and petrol-driven acoustic projectors that have reduced sound levels (30-65+ dB) above 100 Hz, and novel electromagnetic surveys methodologies based on the measurement of seabed resistivity that may supplement seismic operations under certain conditions. The evaluation of potential biological effects associated with the use of these new technologies is underway. Regardless of the technology, precautionary guidelines to minimize disturbance associated with seismic surveys should continue to be applied until the biological significance of the observed effects can be determined.

3.3 ANALYSIS OF RISK, KNOWLEDGE GAPS, AND RESEARCH NEEDS

There have been some studies on the impacts of seismic noise from air guns on marine species as the result of physical (i.e. changes in organisms' physical state including death and injury to organs), physiological (i.e. changes in biological functions or body chemistry) and/or, behavioural alterations (i.e. changes in how organisms act). Studies to date on invertebrates, fish, marine mammals, and sea turtles suggest that there can be consequences of seismic noise to individuals or groups of animals in the marine environment, but the potential for this to result in population or ecosystem scale impacts continues to be debated. Observable physical impacts appear to be largely confined to within a few meters of the seismic source arrays. Behavioural impacts have been detected tens of kilometres from a seismic source.

The increasing concerns about the effects of underwater manmade noise on marine organisms calls for a standardized system of how to quantify and mitigate noise exposure with relevant and reproducible measures. The RMS measure specified by most regulatory agencies critically relies upon choosing the size of averaging window for the squared pressures. Derivation of this window is not standardized, which can lead to 2-12 dB differences in RMS sound pressure for the same wave forms. This has resulted in inconsistency between scientific studies as there is inadequate information to reproduce and compare measurements. RMS pressure, being closely related to the average energy of a noise pulse, does not prevent exposure to high peak pressures. Recent research has indicated that the sound exposure level (SEL) or the time-integrated sound energy delivered to an animal over the duration of a seismic pulse may be as or more important damage criteria than the traditionally used RMS and peak pulse pressure levels (Lawson and McQuinn, 2004). Since physical damage and impairment of the auditory system in marine mammals is caused both by high peak pressure and energy flux, Madsen (2005) suggested that safety limits for sound exposure should include both a maximum received energy flux level and maximum received peak–peak pressure level. Such a protocol addresses concerns for physical damage due to short high pressure pulses, as well as the effects of longer, high-energy transients with lower peak pressures.

Fisheries and Oceans Canada coordinated a workshop in 2003 to develop a 'Decision Framework for Seismic Survey Referrals', in order to produce an inventory of ecological factors to be considered when dealing with referrals for seismic surveys in Canadian waters (DFO, 2003a). The workshop discussed sources of uncertainty about effects of seismic sounds on the ecological factors and the methods to address these issues in the context of environmental risk assessments. Results were presented the following year at a National Advisory Process meeting on Seismic Impact Evaluation Framework (DFO, 2004), where it was concluded that seismic sounds in the marine environment are neither completely without consequences nor are they certain to result in serious and irreversible harm to the environment.

The potential for detrimental effects exist and are likely linked to the conditions of the environment (e.g. physical oceanographic factors influencing sound propagation and proximity to and intensity of the sound source arrays) and the organisms being exposed. The documentation of no fish or invertebrate kills during operational surveys, as well as only circumstantial evidence of infrequent strandings of marine mammals and giant squid, however, suggests that seismic surveys with fairly routine mitigation measures in place are unlikely to pose a high risk of mortality of marine organisms. Nevertheless, it was also noted that sublethal or longer-term effects may have occurred and have not been detected by the regulatory monitoring programs that are currently in place. Immediate behavioural reactions (e.g. avoidance of seismic noise and reduced vocalization) have been widely documented in marine organisms, especially marine mammals. The possible long-term effects of these behavioural responses continue to be debated. The effectiveness of mitigation measures, such as a decision not to conduct surveys in specific zones or during critical periods, is influenced by many factors. It was concluded that additional research and development of improved monitoring methodologies are needed to clarify and quantify the unknown risks and uncertain effects of seismic sound on the marine environment.

Payne et al. (2008) concluded (based on a review of scientific literature published between 2003 and 2008) that the primary concern over the impacts of seismic noise on fish or shellfish is at the stock or sub-stock level, such as in a shallow coastal environment (i.e. bay or inlet). There is also some physiological and histopathological evidence suggesting a potential for seismic noise to have sublethal effects at the individual level. Payne (2004) suggests that a few representative studies on distance-effect relationships for keystone fish and shellfish species, of commercial importance, would greatly aid understanding in this area, with the potential for effects stemming from cumulative energy, as well as peak energy, being considered. This includes under conditions of 3D surveys. The results of these studies would help to identify the need, if any, for air-gun-based sound reference levels for fish and shellfish.

Regarding the potential effects of noise in general, Popper (2003) and Payne et al. (2007), among others, have drawn attention to the need for studies on sub-lethal effects, which are recognized as a major knowledge gap. A recent workshop convened in relation to the North Sea also concluded the following (Ainslie et al., 2009):

In the next steps towards an impact assessment, there is insufficient information on physiology and behaviour of the marine fauna of the North Sea. There is also a lack of knowledge on the effects of the various anthropogenic sources of sound on the ecosystem of the North Sea, both individually and cumulatively.

To directly address identified knowledge gaps, the Joint Industry Programme on Sound and Marine Life supported by the International Association of Oil and Gas Producers (OGP) hosted a workshop in 2007 to identify knowledge gaps pertaining to the potential impacts of seismic noise, with participation of the scientific community in both industry and academia. A gap analysis showed that there is a need to develop new approaches to: 1) attach instruments and transfer data; and 2) measure significant behavioural/physiological responses with limited sample sizes.

The Exploration and Production (E&P) Sound and Marine Life Joint Industry Programme has commissioned a broad range of research studies on all sources of sound produced by the offshore oil and gas industries, including seismic airguns, drilling, dredging, pile driving, construction equipment, removal of offshore structures using explosives, shipping, and others. The taxa of concern include: marine mammals, fish (all life stages), turtles, birds, and invertebrates. The primary scope of the programme's research is to describe industry sources, the known or potential effects of these sources on animals, and ways to mitigate these effects. The programme also addresses anthropogenic sounds that go beyond immediate industry needs, such as the global trend in ocean noise from all sources of human sound, the mechanisms that underlie animal injury or death from intense sound exposure, conferences discussing the latest research results, and others. Current funded research studies, including the development of tools described by Hurley (2009), fall into the following categories:

- Sound source characterization and propagation:
  - review of existing data on underwater sounds produced by the oil and gas industry;
  - standardizing methods of measuring underwater sound;
  - measuring 3D acoustic field of seismic air gun arrays; and
  - environmental assessment of marine vibrosis.
- Physical and physiological effects and hearing:
  - marine mammal noise exposure criteria;
  - marine mammal TTS tests;
  - assessing hearing capabilities of mysticete whales;
  - Minke mammal whale hearing and vocalization
  - hearing capabilities of loggerhead sea turtles;
  - blood nitrogen uptake and distribution in diving northern bottlenose dolphins;
  - models for predicting auditory tissue damage in fish; and
  - effects of noise on aquatic life.
- Behavioural reactions and biologically significant effects:
  - effects of sound on behaviour of toothed whales;
  - testing geographic positioning system (GPS)/Depth tags on sperm whales;
  - cetacean stock assessment in relation to exploration and production activities;
  - review of marine mammal population modeling;

    - field studies on seal foraging success; and
    - application of risk assessment for effects of sound from E&P operations.
- Mitigation and monitoring:
    - review of existing mitigation treatments;
    - analysis of marine mammal observer (MMO) data;
    - testing acoustic vector sensor for 3D tracking passive acoustic monitoring (PAM) array;
    - development of open source PAM software for detection and localization of cetaceans; and
    - PAMGUARD system improvements.

In Canada, the ESRF Program has also funded several ongoing projects related to seismic impacts. Ongoing studies include:

- Seismic and Invertebrates – to study the effects of seismic surveys on caged crab in the field and lobsters in a laboratory setting;
- Seismic and Lobster Feeding – to study the effects of airgun exposure on lobster feeding in a laboratory setting;
- Seismic and Monkfish Egg Veils – to study the effects of exposing monkfish eggs collected from the wild to a typical airgun in a laboratory setting;
- Marine Mammal Observation Data Analysis – to analyze selected sets of marine mammal observations collected during seismic surveys on the east coast of Canada to extract information on behavioural responses; and
- Compilation of East Coast Whale Vocalizations - to compile an electronic library of cetacean vocalizations to contribute to a database to be employed by Passive Acoustic Monitoring Software.

4.0 DRILL MUDS

Exploratory drilling entails drilling into the geological structure to ascertain what hydrocarbon resources it contains. As Hurley (2009) describes in his review of industry practices, drilling can be conducted from a jack-up rig in shallow waters of less than about 100 m or from a semi-submersible rig or drill ship in deeper waters. The duration of exploration drilling programs typically ranges from 30-90 days. The legs of a jack up rig rest on the seabed while the semi-submersible rig/drill ship floats and is anchored in location. The well is drilled in a series of steps of decreasing size with increasing depth (up to several km for some wells). As each step is drilled, the hole is lined with steel pipe, which is cemented in place to prevent collapse or the flow of liquids into or out of the well. As the drill bit penetrates the rock, drilling muds are used by the offshore oil and gas industry to cool and lubricate the drill bit, to balance subsurface hydrostatic pressure, and to carry drill cuttings up to the surface through drilling pipes.

Drilling muds typically contain a base fluid (water-, oil-, or synthetic-based), barite and/or bentonite (weighting agent), and chemical additives (emulsifiers, biocides, lubricants, wetting agents, corrosion inhibitors, surfactants, etc.) to enhance the mud operational properties (GESAMP, 1993). On the drilling-rig, the cuttings and muds are separated to recycle the muds while the cuttings, once treated, may or may not be discharged into the sea. In addition to the use of drilling muds and fluids to control the underground pressures when drilling, offshore oil rigs have a blowout prevention system (BOP) that is used to control the well when there is an influx of pressurized gas or oil during drilling (Hurley 2009). The BOP is a set of hydraulically operated valves and other closure devices (rams) that seal off the well and route the wellbore fluids to specialized controlling equipment. On completion of an exploration well, the well is either removed or left in a safe condition for potential future use. If hydrocarbons are encountered, the next step is to assess the size of the reserves and extent of the reservoir. This is carried out by drilling additional appraisal or delineation wells.

The prime concerns related to discharges of drilling mud and rock cuttings are generally regarded as the burial of, or toxic effects on, seabed fauna, seabird attractions to or collisions with highly illuminated drilling rigs, incineration during flaring/well testing, and, if the well contains hydrocarbons, a possible accidental oil spillage (Figure 3). A number of mitigative measures are in place for exploratory drilling (Hurley, 2009). These include:

- conduct a pre-spud survey to verify characterization of benthic habitat, in particular the presence/absence of coral formations;
- meet or exceed the most recent version of the National Energy Board (NEB)/Canada-Newfoundland and Labrador Offshore Petroleum Board (CNLOPB)/CNSOPB Offshore Waste Treatment Guidelines and Marine Pollution (MARPOL), with regard to waste streams such as drilling muds and cuttings, deck drainage, desalinization brine, sewage and grey water;
- screen chemicals through the most recent version of the NEB/CNLOPB /CNSOPB 'Offshore Chemical Selection Guidelines (OCSG) for Drilling and Production Activities on Frontier Lands' and MARPOL requirements;
- implement well control and drilling procedures as per most recent version of the Nova Scotia Offshore Petroleum Drilling and Production Regulations;
- implement bulk transfer and hose handling procedures as per best available practice;

- minimize flaring and ensure use of high efficiency igniters as per best available practice; and
- establish a 500-m safety zone around the drilling rig at all times.

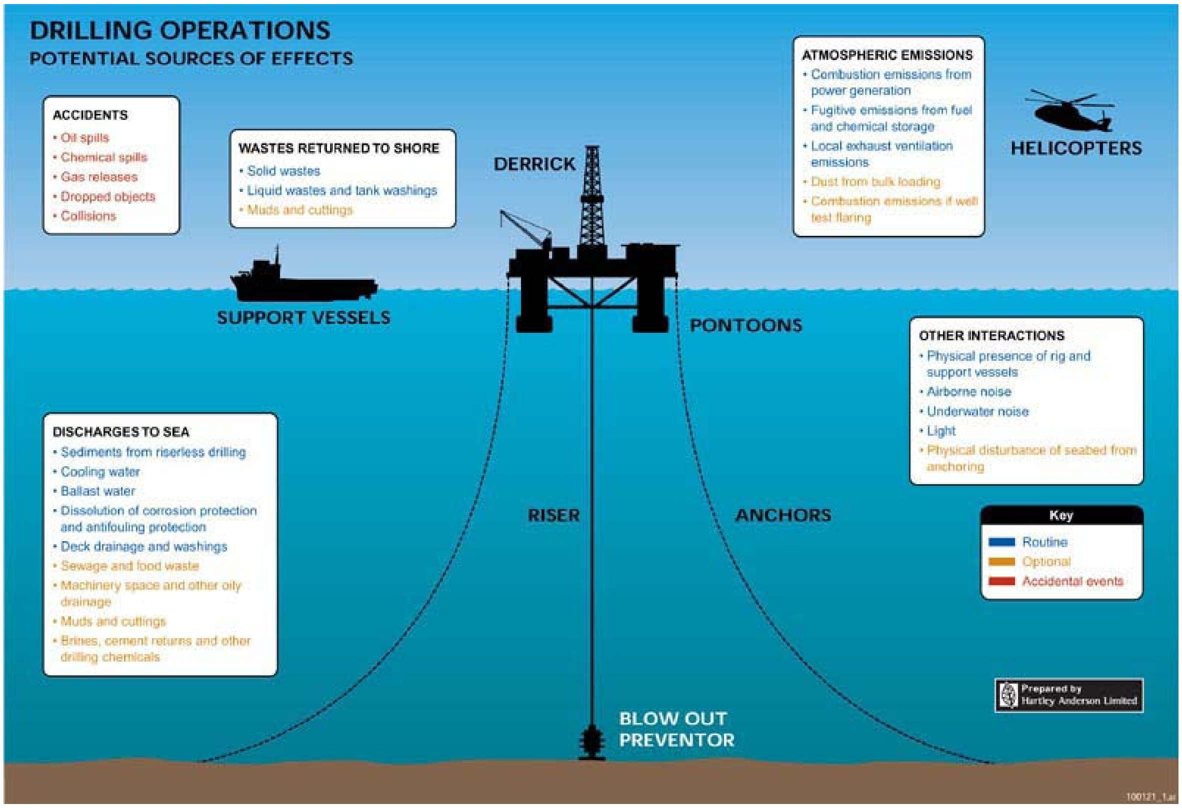


Figure 3. *Potential sources of environmental effects from exploration drilling (Hurley, 2009).*

Companies wishing to conduct exploratory drilling in Canadian marine waters may be required to put in place additional or enhanced environmental mitigation measures to further reduce the risk of harm to marine life (Hurley 2009). For example, the following mitigation measures that have been used in the Nova Scotia offshore:

- focus all area lighting on the work areas of offshore platforms, as well as down-shade as much as practical to minimize marine bird attraction;
- conduct post-drilling survey to verify muds/cuttings plume dispersion modeling estimates of the environmental impact assessment, as well as the predicted zone of influence. Such a survey would include chemical analysis of sediments and benthos;
- use of heavy brine in place of barite as a weighting agent to reduce input of mercury compounds in to the marine environment;
- develop contingency measures and a response plan to address various significant weather scenarios; and
- develop a Code of Conduct for operating in proximity of the Gully Marine Protected Area (MPA) and Sable Island, which specifies the minimum safe working distances for aircraft and vessels nearby these special areas

Hurley (2009) also identified the following research needs to address knowledge gaps on the impact of offshore drilling operations on valued ecosystem components (VECs):

- detection of and potential behavioural effects of animals at night, in foggy conditions, or below the sea surface in the immediate vicinity of the drilling platform;
- delineation of the zone of influence and effects, if any, on marine mammal (particularly at-risk species) from noise associated with floating drilling platforms used for deep-water drilling (e.g. Scotian Slope area);
- improvements in the knowledge of critical periods (i.e. mating and calving) for marine mammals (particularly at-risk species);
- information on deep-water corals; and
- studies on the attraction of seabirds to drilling platforms, particularly during periods of low visibility (e.g. fog and darkness) and flaring.

Most drilling of offshore oil and gas wells in the North Sea, the Gulf of Mexico, and other offshore production areas is achieved with water-based drilling muds (WBM). This is due to strict regulations on discharges of oil-based mud (OBM) and synthetic-based mud (SBM) as a result of their potential environmental impacts. SBM were designed to be less toxic and more environmentally-friendly than OBM. The use of SBM over WBM comes with concerns over the risk of organic enrichment and the potential persistence of synthetic-based fluid biodegradation products. These risks, however, must be weighed against the higher levels of turbidity and trace metal contaminants associated with bulk WBM disposal. Veil et al. (1995) noted that WBM use typically generates between 1100-2000 m^3 of muds and cuttings, depending on the depth and diameter of the well, compared to 300-1300 m^3 for SBM.

The use of SBM first occurred in the North Sea, although they have been curtailed since 2001 when the OSPAR[1] Commission stated that “the discharge into the sea of cuttings contaminated with synthetic fluids shall only be authorized in exceptional circumstances.” Water-based mud, but not OBM or SBM, may be permitted for ocean discharge into European and U.S. offshore waters. Synthetic-based mud, but not OBM, is permitted for discharge into offshore waters of the U.S. Gulf of Mexico, with requirement of the SBM material registering as biodegradable under anaerobic conditions with indigenous micro-organisms.

Current guidelines in Canada stipulate recovery and onshore disposal of cuttings that are generated using OBM (NEB et al., 2002). Synthetic-based mud that remains from a drilling mud change-over or drilling program completion should be recovered and recycled, re-injected down-hole, or transferred to shore. The cuttings produced with WBM are allowed to be discharged into the sea, as are those produced with SBM following treatment with the best available technology (BAT). The international offshore oil and gas community believes that a concentration of 6.9 g per 100 g or less oil on wet solids can be reached through BAT practices. This discharge limit may be modified in individual circumstances where more challenging formations and drilling conditions are encountered or areas of increased environmental risk are identified. Given the relatively shallow water depth of Georges Bank, it is expected that petroleum exploration activities

[1] OSPAR is the mechanism by which fifteen Governments of the western coasts and catchments of Europe, together with the European Community, cooperate to protect the marine environment of the North-East Atlantic region.

in the area would primarily use low toxicity WBM, with potential use of SBM as dictated by the operational requirements. It is extremely unlikely that OBM would be considered under any circumstance as per current guidelines of the CNSOPB.

The biological effects of drilling wastes can generally be thought of as being caused primarily by: (1) chemical toxicity from hazardous pollutants and biodegradation products; (2) organic enrichment of the seabed that may result in anoxic conditions; (3) physical smothering due to accumulation; and (4) physical effects on tissues (causing reduction in growth and reproduction) due to chronic exposure to very low concentrations (>0.05 mg L^{-1}) of the drilling mud compounds bentonite and barite (Cranford, 2006). Early studies on the environmental impacts of drilling wastes suggested that the fine particles of drilling muds would be quickly dissipated in high energy environments, resulting in low concentration of the wastes in areas such as the continental banks along the Canadian Atlantic coast (Neff, 1987a).

Studies in Atlantic Canada (e.g. Muschenheim et al., 1995; Muschenheim and Milligan, 1996) reported that flocculation (the adhesion of smaller particles to form large particles) and surface absorption (the adhesion of smaller particles to larger particles and/or droplets) are important processes in the transport of wastes from drilling sites (Figure 4). Through these processes, the bulk of the drilling muds settles quickly and accumulates on the seabed. Re-suspension and deposition processes in the benthic boundary layer tend to concentrate particulate wastes in suspension near the seabed before being dispersed by the currents (Muschenheim and Milligan, 1996). The consequences of these processes are increased sedimentation rates, oxygen depletion in the sediments, changes in sediment grain size, and an increased concentration of suspended particles in the water column. In addition, Thouzeau et al. (1991) noted that the benthos are highly sensitive to the bioaccumulation of contaminants (i.e. metals and hydrocarbons) discharged in during drilling (Neff, 1987a). These observations support the hypothesis that impacts from drilling operations are most likely to affect benthic communities and that sediment impacts may persist beyond the period at which the discharge is discontinued (Cranford, 2006).

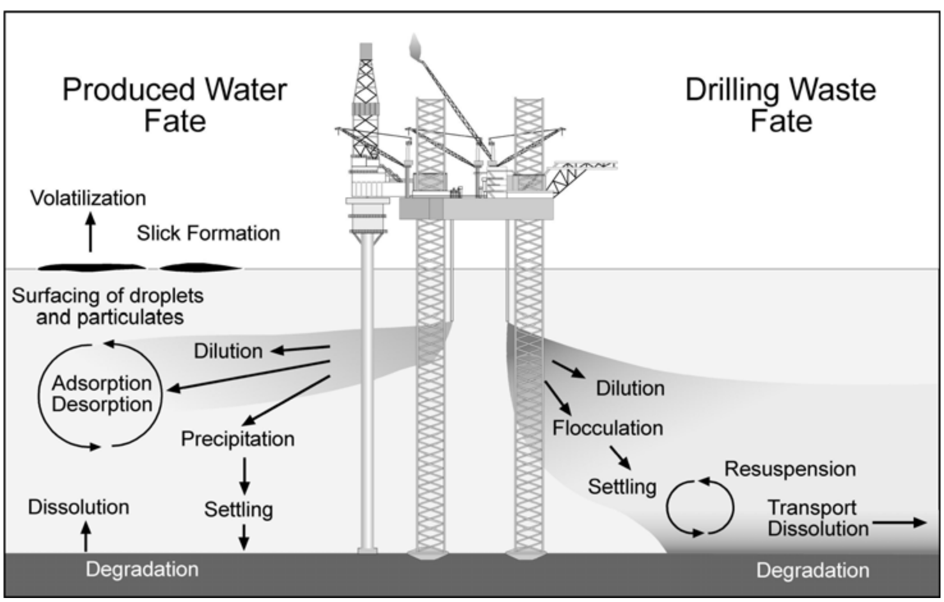


Figure 4. *Major processes controlling the environmental fate of wastes from offshore oil and gas drilling and production activities (Cranford, 2006).*

In terms of environmental persistence of discharged contaminants, recent DFO research studies have revealed that SBM were readily biodegradable in Atlantic marine sediments under ambient environmental conditions (Li et al., 2009a). There can be, however, a significant interference of hydrocarbon degradation by elevated metal ions when the drilling muds are recycled and reused. A trade-off of risks exists between a rapid biodegradation of the drilling muds in the sediments, which may reduce the time-scale of the removal of toxic stress caused by chemicals and the fast consumption of dissolved oxygen. This may result in hypoxia and/or anoxia in the sediment pore water (water that occupies the spaces between sediment grains on the seafloor), which can lead to the suffocation of aerobic organisms that occupy bottom sediments.

Discharged drilling muds and associated cuttings in low energy and/or shallow water systems, however, are expected to accumulate near the point of discharge (i.e. rig) where they can smother slow-moving or sessile benthic organisms. In high energy environments, such as on the top of Georges Bank, drilling muds are quickly spread over a larger area, thereby decreasing the risk to benthic organisms. Results of the American Georges Bank Monitoring Program have indicated that bottom currents on the top of Georges Bank are sufficiently strong to rapidly disperse materials settling from drilling sites (Phillips et al., 1987; Neff et al., 1989a). The Georges Bank Moratorium Area includes the top of Georges Bank and surrounding deeper water areas.

4.1 IMPACTS ON MARINE ENVIRONMENT

4.1.1 Acute Impacts

Mortality may result from direct toxicity of discharged materials (Cranford and Gordon, 1992). Over 70 different water-based drilling mud formulations have been tested in laboratory experiments for their lethal toxicity to a variety of species. Most acute toxicity thresholds for muds and their components are much higher than concentrations expected under field conditions. Because of rapid dispersion rates and the likely use of low-toxicity WBM on Georges Bank, it is predicted that the impacted zone around drilling rigs would be limited to within a few hundred meters, or less, of the discharge pipe. Prolonged exposure (i.e. on the order of a month) to high concentrations (e.g. 10 mg L^{-1}) of bentonite and barite, however, can cause mortality to sea scallops, as well as tissue weight loss and cessation of gonad development (Cranford and Gordon, 1992, Cranford et al., 1999).

Simulations have been made with the Benthic Boundary Layer Transport (BBLT) Model for different regions of Georges Bank to determine the duration that benthic concentrations exceeded 10 mg L^{-1} at various distances along the primary drift line from the drilling rig, assuming typical release concentrations and rates of drill wastes from an exploratory well. The BBLT model was developed by DFO to predict transport and dispersion of suspended particulate drill wastes in the benthic boundary layer (Hannah et al., 1995). The model combines estimates of the current profile, bottom stress, settling velocity of the drill mud, and discharge scenario to generate estimates of drift, effective diffusivity, and concentration. Model results demonstrated that predicted waste concentrations on the bottom were not likely to cause scallop mortality, even at the release point. The longest duration for concentrations exceeding 10 mg L^{-1} occurred in the stratified region of the Bank. Values greater than 10 mg L^{-1} were predicted to occur for a period of nine days at the release site and three days at distances approximately 5 km downstream. In the frontal region, peak durations were approximately one day at the release site. These durations are much shorter than the 30 day period that caused scallop mortalities in laboratory experiments with bentonite concentrations of 10 mg L^{-1}.

A number of acute toxicity tests using aliphatic hydrocarbon-based drilling fluids that have been used on the Grand Banks have been carried out by DFO at the Northwest Atlantic Fisheries Laboratory, St. John's, Newfoundland. Exposure, which involved concentrations well beyond dissolved saturation levels, were as follows: Capelin larvae exposed to 0.05% oil for 72 h; Copepods exposed to 2% oil for 48 h; *Artemia* exposed to 1% oil for 24 h; ctenophores (jellyfish) exposed to 0.2% oil for 120 h; and snowcrab larvae exposed to 0.02% oil for 48 h. It was concluded that aliphatic hydrocarbons in drilling muds posed little or no ecotoxicological risk, since no differences were noted between control and exposed groups in any of the trials (Payne et al., 2001a).

4.1.2 Sublethal Impacts

Sublethal or chronic, long-term interactions are usually associated with the benthos, since the bulk of drilling mud wastes rapidly descend to the seabed where sessile benthic organisms become exposed (Cranford et al., 2005). For example, seabed analyses have shown barium (Ba) levels above background levels out to 2500 m from the Hibernia Gravity Base Structure (GBS), while benthic community analysis indicated a localized impact (i.e. reduced biomass) within a 500 m of the GBS (Cranford et al., 2005). After drilling is completed, however, benthic impacts disappeared rapidly. Of the many benthic species present on Georges Bank, sea scallops are useful organisms to test toxicity effects of drilling mud wastes since they are very sensitive contaminants (Neff, 1987a), and preferentially filter-out and ingest the fine-grained components of the drill mud suspended in the water column (Cranford et al., 2005; Hannah et al., 2006). Cuttings, the largest component of the drilling mud discharge, are not of a size that is ingested by suspension feeders such as scallops (Cranford and Gordon, 1992).

Earlier laboratory studies estimated two kinds of sublethal effects thresholds for scallop exposure to bentonite and barite: (1) zero growth concentration (C_0) is the threshold level where there is no scallop tissue growth at or above the threshold; and (2) no effects concentration (C_1) is the threshold level where there is no significant effect on growth at or below the threshold. With respect to scallops, bentonite has a C_0 of 10 mg L^{-1} and C_1 of 2 mg L^{-1} and barite has a C_0 of 0.5 mg L^{-1} and C_1 of 0.1 mg L^{-1} (Cranford and Gordon, 1992; Cranford et al., 1999; Cranford et al., 2003). The studies demonstrated that chronic exposures resulted in the complete cessation of gonad growth in sea scallops at barite concentrations of 0.5 mg L^{-1}. Scanning electron microscope observations demonstrated evidence of damage to the ctenidia (a comb-like respiratory structure that serves as the gill) of suspension-feeding bivalves after exposure to barite at very high concentrations (Barlow and Kingston, 2001). It has been suggested that barite particles cause nutritional stress by absorption of mucus secretions, which are critical to the normal feeding and digestion processes of scallops (Cranford et al., 1999; Cranford, 2006).

Low toxicity mineral oil (LTMO) based drilling fluids (less than 0.5% aromatic hydrocarbon content) had the greatest impact on sea scallops, causing mortality within 11 days at an exposure concentration of 2 mg L^{-1} (Cranford et al., 1999). Harris (1998) conducted toxicity tests with five drilling muds that have been used and discharged in the offshore of eastern Canada. The drilling fluids include on LTMO (Shellsol DMS), two

SBMs (1A-35 and Neodene 1518), and two WBMs (Silicate mud, Glycol mud). The benthic toxicity tests, using a crustacean (*Corophium volutator*) and mollusk (*Macoma balthica*), indicated that SBM and WBM were less toxic than the LTMO. Low toxicity mineral oil (LTMO) based drilling fluids have not been used since 1989 in Newfoundland waters and the mid-1990s in Nova Scotia waters.

In a subsequent study, Armsworthy et al. (2005) evaluated the effect of different formulations of SBM and their concentrations on sea scallop growth (both somatic and reproductive) and mortality. There was no significant difference in the mortalities of sea scallops exposed to the different laboratory treatments. Mortality in all treatments was less than or equal to 13%. There were differences, however, in tissue growth for SBM treatments at concentrations of 0.2 mg L^{-1} and 0.9 mg L^{-1}, depending on the SBM used. Furthermore, sea scallops exposed to the SBM, at all concentrations tested, exhibited reduced clearance rates compared to the control groups. The results supported conclusions of the previous studies, which indicated that the solid fraction of the drilling fluid was largely responsible for reduced clearance rates. The reduced feeding rates, combined with reductions in food absorption, are thought to cause the observed impacts on somatic and reproductive tissue growth in scallops. That being said, it should be noted that the length and degree of exposures in controlled laboratory studies may not accurately replicate exposures found in natural settings, due to natural dilution and/or dispersion processes caused by tides and currents and variable operational discharge conditions.

Upgrades to the BBLT model since 1999 include the addition of a wave boundary layer, floc break up, and biological impacts (Drozdowski et al., 2004; Hannah et al., 2006). The wave boundary layer incorporates the combined wave-current bottom stress following Grant and Madsen (1986) and Li and Amos (2001). The floc break up capability allows the sediment to inhabit one of three settling velocity states and is bottom stress dependent. A biological impacts module, based on a growth-days-lost formulation determined from laboratory experiments on scallops (Cranford et al., 2003), was used to assess the potential risk of drilling waste discharges to sea scallop stocks on the northeastern part of Georges Bank (Cranford et al., 2003; Cranford, 2006).

Based on BBLT model runs that assumed typical operational discharge rates to be encountered during the development of an exploration well (i.e. a release of approximately 500 tonnes, or t, of drilling mud and 2600 t of cuttings over 59 days at a drilling depth of 4600 m), the number of potential growth-days-lost by scallops was greatest at the release point and decreased with increasing distance along the primary drift line. The model results showed a potential for 0-40 days of growth inhibition over a 93-day drilling scenario (Table 1). Impacts were predicted to be greatest (i.e. 2-48 days of lost growth) in the vertically-stratified region around the side of the Bank (>100 m depth), which supports relatively low numbers of scallops although dense aggregations can be found in some areas.

In the tidal front region (70-100 m depth), where scallop densities are greatest, impacts were predicted to be localized and range from 0-15 days of lost growth. It is important to note, however, that since gonadal growth appears to be affected more than somatic growth, the net effect in the frontal region may be reproductive loss. This could affect the strength of future year classes. In the central, vertically-well mixed area of the Bank (<70 m depth), which supports the lowest scallop densities, growth loss was less than one day over the drilling period. The interpretation of the results from these BBLT applications on Georges Bank depends upon several factors, including the location of the release site, distribution of scallop stocks, and time of year at which the wastes are released. Refer to the appendix in Boudreau et al. (1999) for additional information.

Table 1. *Estimates of days of lost growth of scallops (Cranford et al, 2003). Estimates are given for two different settling velocities, which are averaged by circular areas of three different radii around the discharge point. The relative densities of scallops in the regions are also indicated.*

Radius (km)	Settling Velocity (cm s^{-1})	Region		
		Mixed	Front	Stratified
0.5	0.1	0	0	5
	0.5	2	15	40
2.0	0.1	0	0	3
	0.5	1	6	19
10.0	0.1	0	0	2
	0.5	0	3	11
Relative density of scallops		Low	Medium to High	Low to High

The Georges Bank simulations reported in Boudreau et al., (1999) and Cranford et al., (2003) have not been repeated. The most detailed simulations were carried out for North Triumph near Sable Island. Hannah et al., (2006) simulated the barite concentrations resulting from a production well drilled during October 1999 and compared the simulated and observed barium concentrations 0.5 m above the bottom measured by the Environmental Effects Monitoring program between October 28 and November 5, 1999. The BBLT results provided a reasonable simulation of the barite concentration as a function of distance from the rig. The potential biological impact on scallops was estimated to be a few days of lost growth over scales of a few kilometres.

Hannah and Drozdowski (2005) conducted a comparison of BBLT simulations for North Triumph near Sable Island, Hibernia on the Grand Bank, and a location on the Northeast Peak (NEP) of Georges Bank. All sites are at water depths of 65-85 m. They found that the simulated barite concentrations at the NEP site were much less than at North Triumph or Hibernia. With reference to detailed simulations for North Triumph (Hannah et al, 2006), they estimated that one might expect barite concentrations on the benthos of NEP of Georges Bank of the order of 0.0001 mg L^{-1} a few weeks after discharge. These concentrations are well below the estimated no observed effects threshold limit of 2 mg L^{-1} bentonite and 0.5 mg L^{-1} barite for scallop growth (Cranford et al., 2003). Higher concentrations would be expected in the deep water around the edges of the bank where the currents are weaker and there is less influence from surface waves. Based on these findings, it is expected that re-running the simulations reported by Cranford et al. (2003), using the updated BBLT model, would result in lower near-bottom drill mud concentrations for locations with water depths less than about 100 m.

In the original BBLT, the model's estimated settling velocity was fixed. Boudreau et al. (1999) and Cranford et al. (2003) considered simulations using settling velocities of 0.1 and 0.5 cm s^{-1} (conservative values based on estimates of floc settling velocity) to bracket the anticipated depositional range of drill muds. The highest near bottom concentrations were obtained with the highest settling velocity of 0.5 cm s^{-1}, and those simulations formed the basis for describing potential impacts. The stress dependent settling velocity used in the most current BBLT analysis allows for the initial drill mud macroflocs (that have an assumed settling velocity of 0.5 cm s^{-1}) to break-up when the bottom stress exceeds a certain threshold value. The broken macroflocs then become incorporated into a background population of smaller microflocs that deposit at an average rate of 0.1 cm s^{-1} (refer to Hannah and Drozdowski (2005) for details). Large tidal currents of the NEP result in bottom stresses that frequently exceed the break-up stress of macroflocs. As a result, the drill mud macroflocs do not survive long enough to undergo significant deposition, so are generally characterized by the lower settling velocity. The lower settling velocity results in lower near-bottom concentrations and lower expected impacts of drilling muds on the seabed. The result is expected to hold true on the plateau of the Bank, where the tidal currents are large. This conclusion, however, may not apply to the deeper regions surrounding Georges Bank, where the currents are weaker and the influence of surface waves less.

Cranford et al. (2005), in a summary of chronic impacts of drilling muds on major benthic organisms, indicated that low concentrations (i.e. 0.05-2 mg L^{-1}) of all major types of drilling muds (i.e. water-, low toxicity mineral oil-, synthetic-, and ester-based muds) can significantly impact the growth and reproduction of scallops (Cranford and Gordon, 1992; Cranford, 1995; Cranford et al., 1999; Armsworthy et al., 2005). Such impacts are considered to be the most critical sublethal effects on adult organisms from both ecological and fisheries perspectives (Capuzzo, 1988). According to Armsworthy et al., (2005), fine particulates in water-, synthetic-, and ester-based drilling muds (i.e. bentonite and barite) were the major cause of observed effects on feeding and digestive processes of scallops. Similar physical effects were detected in other bivalve species (Barlow and Kingston, 2001). A modelling study by Cranford et al. (2003) indicated that under certain hydrographic conditions, the routine discharge of water-based mud may impact scallop growth over an area exceeding 200 km^2. Because chemical processes such as degradation and dissolution were not included in BBLT, however the model may over estimate biological effects since the chemical concentrations may be lower in natural settings due to dissolution and degradation.

Sub-lethal effects have been observed in flounder exposed to sediments containing aromatic hydrocarbons at concentrations as low as 1 part per million, or ppm (Payne et al., 1988; Payne and Fancey, 1989). Similar studies with sediment contaminated with aliphatic hydrocarbon based drilling fluid (used in the Arctic), however, have indicated little potential to affect fish health (Payne et al., 1995). The indices studied included organ and body condition, energy reserves, liver and gill histopathology, and mixed-function oxidase (MFO) enzymes. For further evaluation of effects on health condition indices, a flatfish study was carried out to investigate the effects on parasitism in winter flounder chronically exposed for 4 months to graded concentrations of alkanes in drilling mud from the Hibernia field (Khan and Payne, 2004). The abundance of two skin parasites (*Trichodina sp* and *Gyrodactylus pleuronecti*) and a digestive tract parasite (*Steringophorus furciger*) were reduced in fish exposed to sediment-bound hydrocarbons at concentrations of 2000-6000 ppm.

A pilot, chronic toxicity study was carried out on snowcrab (Andrews et al., 2004). Crabs were exposed to elevated concentrations of drilling fluids and subsequently analyzed after one month of exposure. The indices examined included select hepatopancreatic enzymes, haematology, and hepatopancreatic histology. No differences were noted between the control and experimental groups other than a significant induction of palmityl Co-A oxidase found in the exposed animals. Palmityl Co-A oxidase is an enzyme involved in the metabolism of fatty acids. Similar effect levels were also observed in a study with lobsters that were injected with relatively high concentrations of drilling fluid (Hamoutene et al., 2004). Different aspects of lipid and protein metabolism were assessed. No effects were recorded other than an increased amount of protein found in the lobster claw muscle.

A number of dose-response studies were carried out with Microtox®, amphipods, and polychaetes to determine the direct toxicity potential of drilling source aliphatics in sediment (Payne et al., 2001b; Payne et al., 2006). Drilling muds from the Hibernia field were investigated. The study indicated that hydrocarbon concentrations of approximately 6000 ppm were required to produce direct toxicity in the various species that were investigated. Such concentrations would be expected to be confined to a range of tens of metres from cutting piles (typical of certain drilling programs found on the Grand Banks). Hurley and Ellis (2004) found that changes in the diversity and abundance of benthic organisms were detected within 1000 m of drill sites, although most changes were observed within 50-500 m. They observed that benthic communities typically returned to baseline conditions within one year of the completion of drilling. Results of these laboratory and field studies indicated a low potential for toxicity or health effects on a marine bacterium (*Vibrio fischeri*), a marine amphipod (*Rhepoxynius abronius*), Snow Crab larvae, Capelin larvae, planktonic jellyfish, Winter Flounder *(Pleuronectes americanus)*, a marine polychaete (*Neanthes arenaceodentata*) and American plaice.

4.1.3 Tainting Impacts

It is possible that certain metals and organic compounds contained in WBM or released with cuttings may be accumulate in various tissues of exposed organisms, even at relatively low concentrations. An important consideration is whether the accumulated contaminants may be passed along the food web and influence predators or whether they may cause tainting. If tainting was detected under field conditions by a monitoring program, the area around a rig may have to be closed to fishing for a period of time after drilling is completed. It has been shown in laboratory studies that scallops exposed to WBM have the potential to concentrate both barium and chromium in their digestive tract, as well as clay particles. It should be noted that only the scallop adductor muscle is marketed as food for humans.

In the field, measurements made during the U.S. Georges Bank Monitoring Program could not detect any uptake or accumulation of metals or hydrocarbons in ocean quahog found in the wild (Phillips et al., 1987). While the monitoring program effort did not consider scallops, as they were not found in the study area, it is also noted that taint has not been detected for any of the species tested under Canadian offshore petroleum EEM programs to date, where SBM or WBM have been used for drilling (Hurley and Ellis, 2004). Studies on fish and shellfish tainting have been a component of the EEM programs carried out on the Grand Banks and, to date, there has been no instances of tainting in fish collected around the Terra Nova, Hibernia or White Rose offshore facilities. Even worse case Sable Offshore Energy Project (SOEP) spill scenarios for the Scotian Shelf were not anticipated to release sufficient quantities of Scotian Shelf condensate to cause tainting of scallops or cod (S.L. Ross Environmental Research Limited, 1995).

4.2 DISPERSION AND TRANSPORT

Once discharged into the marine environment, drilling muds and cuttings may demonstrate a number of physico-chemical processes: (1) advection; (2) dispersion; (3) aggregation; (4) settling; (5) deposition onto the seabed; (6) consolidation; (7) erosion; (8) re-suspension; (9) re-entrainment; and (10) causes changes to seabed elevation. The relative impacts of these processes on the fate of drilling wastes in marine environment depends on the characteristics of the drilling wastes and physical variables of the receiving marine waters such as depth, current velocity (tidal and residual), waves, and storms. To understand the fate and effect of particles and chemicals associated with drilling wastes discharged into the ocean during drilling, a number of numerical models varying in type and complexity have been developed over the past few decades, in addition to the BBLT model described in the previous section. While some of these models only calculate the fate of drilling waste discharges by considering one or many of the physical-chemical processes listed above, other models predict the potential environmental risks associated with such discharges.

4.2.1 Physical Transport Models

A comprehensive review of sediment transport and drilling waste fate/transport models was conducted by Khondaker (2000). The review noted that while most of the existing models considered advection, dispersion, settling and other prevailing processes, a fully-validated model did not exist. It was concluded that the most significant modeling effort was that of Koh and Chang (1973), which forms the basis of many the subsequent modeling efforts, including the Offshore Operator's Committee (OOC) model that is generally used to model the transport of drilling wastes in the marine environment (Brandsma et al., 1980; Brandsma and Sauer, 1983; O'Reilly et al., 1989).

The OOC model can be used to simulate the behaviour of drill waste discharges from a single, submerged circular port that can be oriented in any direction by assuming a constant discharge rate. The model considers the discharge as a water-miscible fluid phase that can contain less dense particles (e.g. oil droplets) or more dense particles (e.g. drilling fluid solids or drill cuttings) relative to the ambient receiving seawater. The drilling discharge is characterized by bulk density, discharge rate, discharge pipe configuration, and settling velocities assigned to the drill mud and drill cuttings fractions. The model requires natural conditions, such as water depth, temperature, salinity, and current velocity to generate outputs such as the trajectory and shape of the discharge plume, concentrations of soluble and insoluble discharge components in the water column, and the accumulation of discharged solids on the seabed. A major limitation of the OOC model is that it does not account for re-suspension and transport of previously deposited solids. Recent applications and validation of the OOC model have been reported in a number of scientific publications (e.g. Melton et al., 2000; Brandsma, 2003; Nedwed et al., 2004; Smith et al., 2004; and Pivel et al., 2009).

In the past, it was assumed that the fraction of small particles associated with drilling wastes (defined by settling velocities less than 0.01 cm s^{-1}) would readily dissipate to negligible concentrations on the energetic offshore banks of eastern Canada. Drilling wastes, however, have been found to form particle aggregates 0.5-1.5 mm in diameter, which exhibit settling velocities that are much higher than their individual component particles. As a result, it is now thought that the bulk of drilling mud discharges may deposit out of suspension much more rapidly than previously believed, resulting in increased accumulation on the seabed (Muschenheim et al., 1995; Muschenheim and Milligan, 1996; Milligan and Hill, 1998). Muschenheim and Milligan (1996) suggested that the resuspension and deposition processes of drill wastes in the benthic boundary layer could concentrate particulate wastes near the seabed before they are dispersed by currents and waves. Although the aggregation of drill wastes (also termed 'flocculation') has been observed on the Scotian Shelf off Sable Island at the Panuke and Cohasset oilfield drilling sites (Muschenheim and Milligan, 1996), the degree to which it occurs throughout Atlantic Canada remains unknown. During two separate surveys at the Hibernia offshore petroleum production site, flocculation was observed during one survey but not another.

Results of physical oceanographic and drilling waste behaviour studies have been used in the application of plume descent models for discharged wastes (e.g. Andrade and Loder, 1997) and in the development and application of a novel model for drilling waste dispersion and drift in the benthic boundary layer. The plume descent model simulations have confirmed rapid initial dilution of discharged wastes, but have also indicated that the fraction reaching the benthic boundary layer can vary greatly by location depending on local oceanographic conditions such as currents, stratification, and water depth. Simulations of Georges Bank predict that regional and temporal variations in physical oceanographic processes have a large influence on the potential zone of influence of any discharged drilling wastes (Gordon et al., 2000).

The BBLT model was developed at the Bedford Institute of Oceanography to predict the transport and dispersion of particulate drilling wastes in the benthic boundary layer. The BBLT model has been widely used in the Atlantic Canadian offshore region to assess the potential impact zones of drilling waste discharged during offshore petroleum drilling activities (Drozdowski et al., 2004). The model assumes that all the discharged materials enter the benthic boundary layer and thus neglects the mechanism of plume surfacing. The primary mechanisms modeled by BBLT are shear dispersion, mud flocculation, and break-up, drift, and vertical mixing. The model has been used to simulate the dispersion of drilling wastes around drilling platforms under typical exploration drilling scenarios on Georges Bank (Hannah et al., 1995; Gordon et al., 2000). Hannah et al. (2003) also used BBLT to predict the drift and dispersion of suspended drilling muds near North Triumph on Sable Island Bank during a drilling program in 1999. The model results were in agreement with the very low concentrations (generally less than 1 $\mu g\ L^{-1}$) of barium observed in the water column during the subsequent EEM iteration. Further evaluation of the sensitivity of the BBLT model to settling velocity data and comparisons to other existing models have been made by Niu et al. (2008) and Niu et al. (2011).

Other modeling efforts include SizeCUT, a numerical model that considers contamination by various agents in drill cuttings released under restricted open water conditions (IESL, 2003). The model considers both the jet of the drilling discharge and individual particles, after a transitional stage. The kinetic energy of the jet gradually decreases during its evolution. Once the kinetic energy is diminished, individual particles are subjected to the gravitational force and to the influence of currents and turbulent motions. Fang et al. (2008) developed a model to predict the formation of underwater cutting piles. Although the model considered the effects of water depth, current velocity, cutting properties (e.g. size distribution, density, and sphericity), and water properties (e.g. density and viscosity), its application in an oceanographic environment was limited by the need to use constant current conditions. ASA (2009) developed the MUDMAP model based on a random-walk particle tracking approach to simulate the dispersion of drilling wastes. As with the OOC model, MUDMAP also accounts for three stages of movement, based on the Koh and Chang (1973) approach, to predict the transport and dilution of drilling fluid discharges.

The application of circulation and dispersion models has provided means to assess the potential spatial scale of dispersion of drilling waste discharges. It should be noted, however, that models are simulations, open to assumptions and simplifications, and caution is warranted in their application to decision making. Special care must be taken, since parameters related to the settling velocity of different sized particles under various conditions are not known, and this can lead to major variability in predictions. Regional and temporal variations in physical oceanographic processes, which determine the degree of dilution, suspension, dispersion, and drift of drill wastes in the benthic boundary layer have a large affect on their potential zone of influence.

4.2.2 Physico-chemical Models

Two drilling waste transport models have been developed to predict the fate of chemicals associated with drilling waste discharges: PROTEUS and DREAM. Sabeur and Tyler (2000) described PROTEUS, which considers the dispersion of drill muds and cuttings as a distribution of solid particles with given size, density, chemical content, and settling velocities through three three-dimensional hydrodynamic flow, turbulent advection, and passive diffusion models. PROTEUS also models the chemical transformations that occur following discharge using a decoupled geochemical model. This is achieved by associating chemical mass with particles, representative of the solid or dissolved phase. The chemical mass carried by each particle is updated in time by taking into account mass loss through various processes such as degradation, volatilization, and adsorption/dissolution between solid and dissolved phases.

The second comprehensive model that has been extensively used to study both the physical transport of drilling wastes and corresponding chemical transformations is the Dose-related Risk and Effect Assessment Model (DREAM) described in Rye et al. (2006a), Rye et al. (2006b), and Rye et al. (2008). The DREAM model uses a Lagrangian particle tracking approach. The model generates particles with associated properties (such as mass of various compounds, densities, and settling velocities) at the discharge points which are transported with currents and turbulence in the sea. The particles can also represent state variables, such as gas bubbles, droplets, and dissolved and/or solid matter. The DREAM model incorporates a near-field plume model in order to account for the descent of the cutting/mud plume. The model determines the ‘depth of trapping’ or ‘sinking’ of a plume by considering governing factors such as current velocity and stratification. In water-based mud, most of the added chemicals are mainly assumed to dissolve in the water column, while the dissolution rate of chemicals may be lower for oil-based and synthetic-based muds. The chemicals may have a high capacity for adsorption to organic matter found in the sediment or water column.

For chemicals in the drilling wastes, the DREAM model calculates the water column concentrations as a function of time and space. The model considers the degradation but not the adsorption to ambient suspended matter. The particulate matter in the water column is calculated similar to the chemicals except the degradation is omitted. The model can also estimate the water column concentration of heavy metals originating from particulate matter based on equilibrium partitioning. For the sediments, the DREAM model can calculate four stressors: the deposit of drill cuttings/mud that may cause burial effects; the change of grain size distribution that may favour other species; chemical concentration in the sediment layer that may cause toxic effects; and biodegradation of chemicals that may cause oxygen depletion. The DREAM model is used extensively in the North Sea for environmental risk assessments. A sample DREAM model prediction of two of the four sediment stressors is shown in Figure 5.

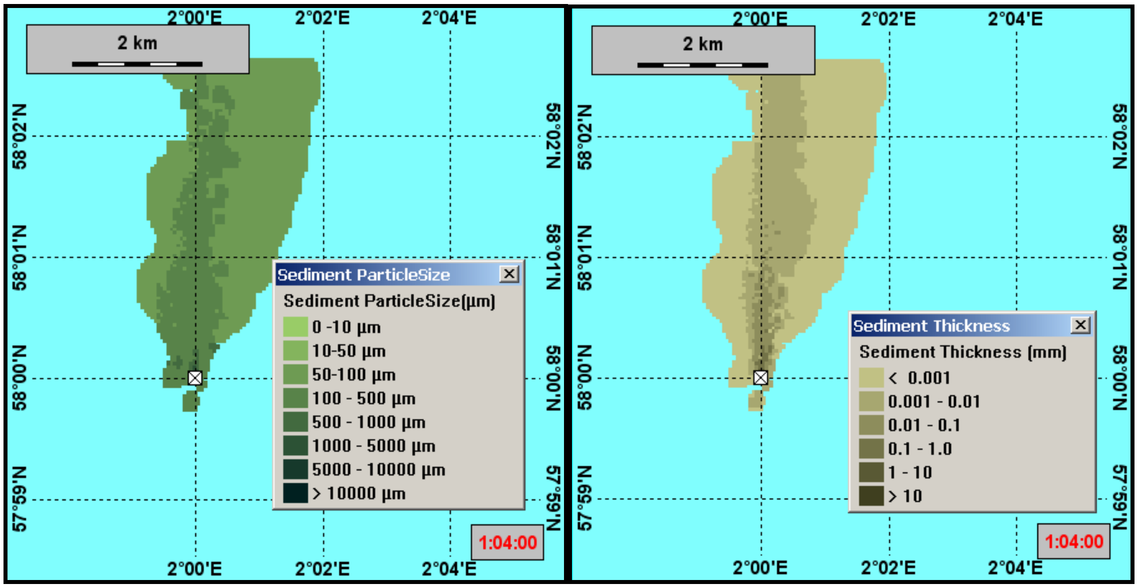


Figure 5. *Example of a DREAM model simulation of two sediment stressors from a discharge in the North Sea: grain size (left panel) and depositional layer thickness (right).*

4.2.3 Risk Assessment Models

The integration of risk assessment modules into physical transport/fate and effect models will enable the calculation of potential risks to the Georges Bank from drilling waste discharges. For example, the BBLT model incorporates a component to predict biological effects on sea scallops based on laboratory toxicity tests with different concentrations of drilling wastes (Cranford et al., 2003). Chemical processes such as degradation and dissolution have not been included in BBLT, and chemical concentrations may be lower in field conditions due to dissolution and degradation. Since drilling waste may induce impacts in both the water column and sediments, the DREAM model calculates the effects from six potential stressors.

For the water column, the two stressors are the toxicity of chemical substances and physical effects of suspended clay particles. The sediment stressors include toxicity of chemical substances, burial of organisms, oxygen depletion, and change in sediment structure. The model uses the ratio of predicted environmental concentration (PEC) and predicted no-effect concentration (PNEC) to first evaluate the risk associated with individual stressors and then determine the overall risk from all stressors expressed as an environmental impact factor (EIF) (Altin et al., 2008; Neff, 2008; Rye et al., 2008; Singsaas et al., 2008; Smit et al., 2008). A sample DREAM model prediction of risks from two of four sediment stressors is shown in Figure 6. The contribution of individual sediment stressors to total EIF for a test case is illustrated in Figure 7.

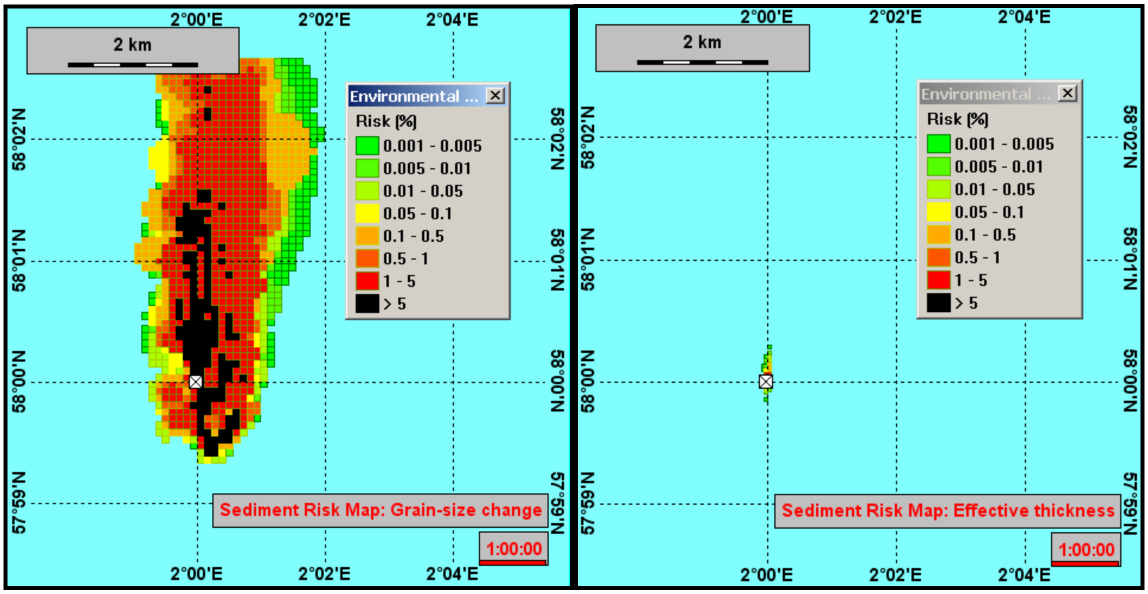


Figure 6. *Example of a DREAM model simulation of risk from a discharge in the North Sea: effects of grain size change (left panel) and effects of burial (right panel).*

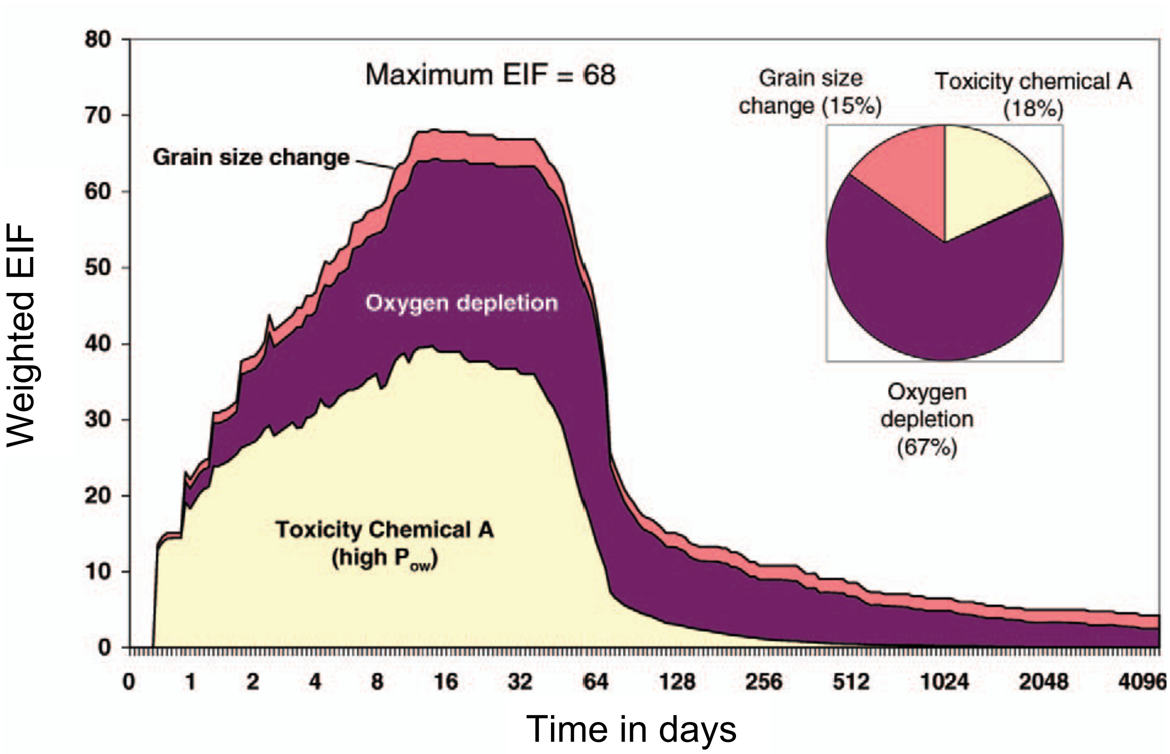


Figure 7. *Contribution of four sediments stressors in the North Sea to total environmental impact factor (EIF) (from Singsaas et al., 2008).*

4.3 ENVIRONMENTAL EFFECTS MONITORING

Environmental Effects Monitoring (EEM) programs are a requirement of long-term offshore petroleum development projects approved by the National Energy Board, Canada-Nova Scotia Petroleum Board, and Canada-Newfoundland and Labrador

Offshore Petroleum Board (e.g. Cohasset-Panuke, Sable Offshore Energy, Hibernia, and Terra Nova). They are used as: 1) follow-up to comprehensive studies and screenings pursuant to the *Canadian Environmental Assessment Act*, 2) verify the accuracy of project EIAs, 3) determine the effectiveness of mitigation measures, and 4) identify unforeseen environmental problems, as they emerge, to enable a timely management response. Hurley and Ellis (2004) reviewed EEM programs on the east coast of Canada, which have been largely focused on the effects of drilling waste discharged to the marine environment (i.e. drilling mud and drill cuttings), as this was considered the primary environmental concern during exploration and development drilling operations. Results were compared to those from foreign offshore sites (e.g. Gulf of Mexico and North Sea).

4.3.1 Detection Zones

Most of the early knowledge regarding the dispersion and effects of muds/cuttings discharged into the marine environment was gained through seabed studies conducted at North Sea offshore petroleum sites that began operations in the 1970s using diesel based OBMs. Davies et al. (1988) concluded that chemical tracers associated with OBMs could be documented out to 800-4000 m distances from the drilling centre. They also noted a transition zone in benthic diversity and community structure that occurred between 200-2000 m from the drilling centre. Subsequent observations over an extended period indicated that the spatial extent and magnitude of effects of OBM cuttings discharge on benthic communities was highly variable.

Olsgard and Gray (1995) documented, over a period of six to nine years after conclusion of drill cuttings discharge, the spread of sediment contamination so that nearly all sample stations 2000-6000 m from the drill site showed evidence of elevated hydrocarbon metals. Olsgard and Gray (1995) also noted that effects on benthos persisted longer than the elevated hydrocarbons, suggesting that metals or other components in the drill cuttings contributed to long term biological effects. For three different oil fields, Olsgard and Gray (1995) documented the spatial extent of sediment that exhibited elevated hydrocarbon and metal (primarily barium) content, which ranged from 10 to more than 100 km^2. In another study, Daan et al. (1996) indicated that localized hydrocarbon contamination and biological effects were detectable up to eight years after completion of drilling programs. Consequently, in light of its environmental persistence and toxicity, the discharge of OBM cuttings has largely been prohibited by offshore regulatory agencies, in favour of WBM, SBM, and LTMO formulations.

Based on chemical indicators of drilling muds, such as barium in association with total petroleum hydrocarbons (TPH), large development projects (e.g. Hibernia and Terra Nova) with several wells at the same location exhibit larger zones of detection (out to 8000 m along the major current axis) than single well developments (maximum 1000 m) at similar water depths (80 m) on the Grand Banks (e.g. White Rose H-20 and N-30) and the Scotian Shelf (e.g. SOEP North Triumph #1). The spatial and temporal extent of detectable discharged drilling wastes was influenced by differences in the number of well, volume of discharges, mud types, current speed and direction, water depth, or sediment mobility at the drilling location. For example, strong physical transport processes, such as the highly-mobile sandy bottom sediments on shallow areas of Sable Bank (less than 40 m), could account for the relatively small contaminant zone around the multiple-well Cohasset-Panuke and SOEP Venture and Thebaud sites (Ross et al., 2003). Two different classes of SBMs have been used for drilling in eastern Canada; that is, the paraffin-based Pure Drill IA-35 used on the Grand Banks and internal olefin-based NovaPlus used on the Scotian Shelf. Discharges of the paraffin-based SBM may have been distributed farther in ocean currents because of differences in oil-on-cuttings concentration and/or the floatation properties of paraffin compared to the olefin-based SBM that tends to settle out of suspension much more quickly as cohesive, well-defined mounds (Hurley, 2000).

The EEM observations made in Atlantic Canada are similar to those found in other offshore petroleum regions. A review of 18 studies of drilling wells that discharged WBM indicated that barium was observed up to 1000-3000 m around the discharge location and up to 8000 m along the major flow axis (Houghton et al., 1980; Mariani et al., 1980; Meek and Ray, 1980; Menzie et al., 1980; Ray and Meek, 1980; Continental Shelf Associates, 1986; Boothe and Presley, 1989; Jenkins et al., 1989). One notable exception, based on a study of eight exploration wells discharging WBM, documented that drilling fluid solids can be transported over long distances, up to 35-65 km, to regional depozones, albeit at low concentrations (Bothner et al., 1985; Neff et al., 1989a). Synthetic-based muds found in sediments documented from nineteen case studies were detected over a more localized area than for WBM. The detection limits for background values ranged from 100-2000 m about the discharge location. Increases in other metals known to be associated with drilling fluids (e.g. arsenic, cadmium, chromium, copper, mercury, lead, and zinc) have been observed with distance from single well sites. Elevated concentrations are generally found over a more spatially-limited range of 250-500 m about the drilling centre, in comparison to that for barium. Exceptions, however, have been reported. For example, chromium was detected up to 1000-2000 m from sites on the continental shelf (Continental Shelf Associates, 1986) and at deeper locations greater than 80 m water depth in the Gulf of Mexico, while concentrations of cadmium and mercury above biological effect threshold limits were found to exist several years after drilling had completed (Kennicutt et al., 1996).

4.3.2 Biodiversity and Abundance

Changes in the diversity and abundance of benthic organisms have been detected within 1000 m of drill sites, most commonly within the 50-500 m range. Results are consistent for both literature review case studies and for the Canadian EEM data. This scale of effects apply to wells discharging SBM or WBM and for multiple or single wells drilled at the same site. Beyond the bottom area covered by the cuttings pile, benthic communities generally returned to baseline conditions within one year after completion of drilling discharges.

With the disposal of WBMs, biological effects have routinely been detected at distances within 1000 m of drill sites (DOI, 1977; Lees and Houghton, 1980; Menzie et al., 1980; Montagna and Harper, 1996; Hurley and Ellis, 2004). Effects include alterations to benthic community structure, including changes in abundance, species richness (number of species), and diversity. Impacted taxa include annelids, molluscs, echinoderms, and crustaceans. Changes have been attributed to purely physical alterations in sediment texture and to platform-associated effects more frequently than to toxic effects. Studies of benthic community change around single exploration wells suggest that communities generally returned to baseline conditions one year after the completion of drilling. A report of effects beyond 1000-2000 m around single exploration wells, as a reduction in the epibenthic coverage of suspension feeding bryozoan communities on hard bottom substrates (Continental Shelf Associates, 1989), has been questioned due to the difficulty in distinguishing effects related to natural variability over time over that from changes in community composition with respect to drilling activities.

The area of detection and scale of biological effects resulting from SBM cuttings discharged were smaller than that resulting from the release of WBM. Biological impacts associated with the release of SBM cuttings were generally detected at distances of 50-500 m from well sites (Bakke et al, 1990; Smith and May, 1991; Candler et al., 1995; Daan et al., 1996; Terrens et al., 1998; Orentas, 2000; Hurley and Ellis, 2004). Patin (1999) also noted that biological effects in the form of structural changes in benthic communities can be found up to 1000 m from platforms. These changes lead to either depressed or enhanced species diversity in the near field (MMS, 1995). While recovery of benthic communities were generally documented to occur within one year of completion, one case study documented that benthic species richness and abundance were reduced at a distance of 50 m two years after exploratory drilling was completed (Candler et al., 1995).

4.3.3 Toxicity and Biological Effects

Heavy particles within drilling wastes tend to settle near the discharge site and can form a pile on the seafloor. There is potential for biological effects of these cutting piles that include smothering of benthic communities and artificial reef effects as the piles attract marine organisms and provide substrate for epifaunal animals such as crabs (Forteath et al., 1982; Hurley and Ellis, 2004). Beyond the cutting pile, observed impacts of drilling wastes have generally been attributed to chemical toxicity or organic enrichment, although there was evidence in the EEM studies that fine particles in drilling wastes contributed to the effects observed around drilling platforms. This observation is consistent with laboratory studies conducted on filter feeding sea scallops, which indicated that physical interference by bentonite and barite particles in drilling wastes could affect growth and reproduction at environmentally-relevant concentrations (Cranford et al., 2005).

Results of EEM studies conducted in Atlantic Canada showed that the effects of drill waste on the benthic community varied with the volume of muds and cuttings discharged and proximity to the point of discharge. In areas closest to the platform, benthic organisms can be physically smothered under cuttings piles (e.g. Newburn-23). Beyond the cuttings piles, species diversity and abundance of infauna can either be depressed (e.g. Terra Nova- within 200 m of discharges from multiple wells), or enriched (e.g. SOEP North Triumph No.1- within 1000 m of discharges from a single well), but appear to return to baseline levels within one year of the completion of drilling discharges. In one case, seasonal variations may have masked potential effects of drilling discharges on infauna (e.g. Cohasset-Panuke 1993). Naturally low diversity in some locations (e.g. EnCana H-08) also makes it difficult to discern changes in the benthic community.

A number of early warning bioindicators including fish condition indices, skin and organ lesions, liver and gill histopathology, and levels of MFO enzymes, were studied in American plaice at the Terra Nova site on the Grand Banks (Mathieu et al., 2005). Chronic toxicity studies showed limited bioaccumulation of polycyclic aromatic hydrocarbons (PAH) in flounder exposed to sediments containing high levels of petroleum hydrocarbons. Under Canadian east coast EEM programs in the field, no early warning health effects have been observed with biochemical and/or histopathological indicators of chemical stress in American plaice and a variety of shellfish species (Payne et al., 1995; Cranford et al., 2001; Hurley and Ellis, 2004). Mixed-function oxygenase induction was not observed in fish collected around some platforms in the Gulf of Mexico (McDonald et al., 1996), although it is noted that the reference fish also contained high levels of PAH metabolites suggesting extensive background contamination by hydrocarbons at the reference site.

In contrast, enzyme induction has been observed in a number of adult fish species, as well as fish larvae around rigs in the North Sea, compared to nearby reference sites (Davies et al., 1984a; Stagg et al., 1995; Stagg and McIntosh, 1996). Histopathological effects have also been observed in fish and shellfish in association with some oil developments in the Gulf of Mexico (Galloway et al., 1981; Grizzle, 1986; Wilson-Ormond et al., 1994). The effects observed in fish in both the North Sea and the Gulf of Mexico may be related to production waters. Peterson et al. (1996) documented readily detectable changes in benthic meiofauna and macrofauna, but failed to detect evidence of exposure or sublethal impacts on demersal fish species. They concluded that the mobility of fish species, the relevant scales of environmental change, and their negligible exposure to hydrocarbons and other contaminants were the primary reasons for a lack of documented effects in the fish species studied.

4.3.4 Bioaccumulation and Taint

Body burden concentrations of TPH in sea scallops extended as far as 2600 m about Terra Nova, but were not detected in sea scallops at SOEP NT#1. Total petroleum hydrocarbon was also detected in blue mussels about COPAN and SOEP NT#1, but not in snow crabs about SOEP NT#1 or American plaice about Terra Nova and Hibernia. Bioavailability and bioaccumulation of hydrocarbons and other oil components depend upon the balance and ratios between the rate of their input into the organisms, the efficiency of biochemical transformation in organs and tissues, and the rate of excretion (Patin, 1999). Benthic invertebrates, especially bivalves, usually have an increased ability to accumulate oil due to their high filtration rates, contact with bottom sediments, and less-developed and less-active enzyme and metabolic systems when compared with fish systems (Patin, 1999).

Taint was not detected for any of the species tested in the Canadian EEM programs, except for blue mussels at COPAN, which discharged more toxic LTMO muds. Mathieu (2002) noted that although valuable for assessing fish health, quality, and marketability, chemical analyses of tissues can be inadequate as a measure of fish health because: 1) many chemicals do not accumulate in body tissues to any degree yet they can be quite damaging; 2) only a limited number of toxic chemicals in complex nature can be measured; 3) the toxicity of many chemicals may not be due to the chemicals measured but by degradation products which are not readily measured; and 4) there are few dose-response experimental studies linking body burdens of chemicals to effects.

Biodegradation processes may influence the level of impact from SBM discharges. Synthetic fluids are likely to produce substantial sediment oxygen demand when discharged in the amounts typical of offshore drilling operations. At present, there is disagreement among the scientific community as to whether slow or rapid degradation of synthetic base fluids is preferable. Highly-biodegradable materials will deplete oxygen more rapidly than slow degrading materials. Rapid biodegradation, however, reduces the exposure period of aquatic organisms to materials that may bioaccumulate or have toxic effects. Currently, the United States Environmental Protection Agency (U.S. EPA) believes rapid degradation is preferable because seafloor recovery has been correlated with disappearance of the SBM base fluid.

5.0 PRODUCED WATER

Produced water is the wastewater that is generated during the production and processing of offshore oil and gas. Formation water is sea water or fresh water that has been trapped with oil and natural gas in porous sedimentary rock formations between layers of impermeable rock within the earth's crust for millions of years (Collins, 1975). When a hydrocarbon reservoir is penetrated by a well, produced fluids may contain significant volumes of this formation water, in addition to the oil, natural gas, and/or gas liquids. In some oil fields, fresh water, brine/seawater and 'treatment' chemicals may also be injected into a reservoir to enhance both recovery rates and the safety of operations. These chemicals may be recovered along with oil and gas during production (Neff, 2002; Veil et al., 2004). Produced water (formation and injected water and process chemicals that are recovered) represents the largest volume waste stream in oil and gas production operations on most offshore platforms (Stephenson, 1991; Krause, 1995). For instance, produced water may account for 80% of the wastes and residuals produced from natural gas production operations (McCormack et al., 2001).

The ratio of produced water to oil equivalents (the WOR) from a well varies widely from essentially zero to more than 10 (91% water and 9% oil). The average world-wide WOR is about 2 to 3. In general, the volume of produced water usually increases with the age of the well (Henderson et al., 1999). In nearly depleted fields, production may be 98% produced water and 2% fossil fuel (Stephenson, 1992; Shaw et al., 1999). On the Canadian East Coast, the produced water discharged by the Cohasset field on the Scotian Shelf outnumbered oil by a ratio of 2:1 during its life. The Hibernia field on the Grand Banks is expected to follow a similar trend (Ayers and Parker, 2001). In 2003, an estimated 667 million metric tonnes of produced water were discharged offshore throughout the world, including 21.1 million tonnes to offshore waters of North America (mostly the U.S. Gulf of Mexico) and 358-419 million tonnes to offshore waters of Europe (mostly the North Sea) (OGP, 2004; Garland, 2005).

These are underestimates of actual discharges because reporting of production to OGP (2004) ranged from 11-99% in the seven regions of the world being monitored. In terms of discharge capacity, it is generally reported that the total amount of water produced from gas fields is much smaller than that from oil production fields. While many gas fields discharge less than 10 m^3 of produced water per day, most oil fields discharge hundreds or even thousands of cubic metres of produced water per day (OGP, 2002). In 2009, the Venture field on the Scotian Shelf was discharging 100-600 m^3 of produced water per day (personal communication, ExxonMobil). A considerable amount of concern has recently been raised over the disposal of produced water from production operations as its release is continuous, in comparison to the episodic release of contaminants associated with the disposal of drill muds and fluids during the drilling of exploration and production wells.

5.1 CHEMICAL COMPOSITION

Produced water is a complex mixture of organic and inorganic chemicals in both dissolved and particulate phases. The physical and chemical properties of produced water vary widely depending on the age, depth, and geochemistry of the hydrocarbon-bearing formation, as well as the chemical composition of the oil and gas phases found in the reservoir. Typical chemicals found in produced water include inorganic salts, metals, radioisotopes, and a wide variety of organic chemicals (Table 2). Because no two produced waters are alike, region specific studies should be conducted to address the environmental risks that may be associated with its discharge.

Table 2. *Concentrations (mg L^{-1}) of several classes of naturally-occurring metals and organic chemicals in produced water found world-wide (Neff, 2002).*

Chemical Type	Concentration* (mg L^{-1})
Organic Chemicals:	
Total benzene, toluene, ethylbenzene, and xylenes (BTEX)	0.068-578
Total organic acids	≤0.001-10,000
Total organic carbon	≤0.1->11,000
Total phenols (primarily C_0-C_5-phenols)	0.4-23
Total polycyclic aromatic hydrocarbons (PAH)	0.04-3.0
Total saturated hydrocarbons	17-30
Total steranes/triterpanes	0.14-0.175
Inorganic Chemicals:	
Ammonia	14-246
Arsenic	0.000004-0.32
Barium	≤0.001-2,000
Cadmium	0.0000005-0.49
Chromium	≤0.000001-0.39
Copper	≤0.000001-55
Iron	≤0.0001-465
Lead	≤0.000001-18
Manganese	0.0002-7.0
Mercury	≤0.000001-0.075
Nickel	≤0.000001-1.67
Nitrate	0.6-15.8
Orthophosphate	0.1-6.6
Salinity (mostly sodium and chloride)	<2000-> 300,000
Sulfate	≤1.0-8,000
Sulfide	0 - 140
Total radium (pCi L^{-1})	0-5,150
Zinc	0.000005-200

*Note: Many of the high concentrations reported for metals may be anomalous due to matrix interferences from high concentrations of dissolved salts that may be found in the produced water (Neff, 1987b). The highest concentrations are extremely rare. Concentrations of most chemical constituents listed above, measured by modern, accurate, analytical methods, reside in the lower part of the concentration ranges listed above

5.1.1 Total Organic Carbon

The concentration of total organic carbon (TOC) in produced water ranges from less than 0.1 to more than 11,000 mg L^{-1}. It is highly variable from one location to another (Table 2). Produced water from Hibernia has a TOC content of approximately 300 mg L^{-1} (Ayers and Parker, 2001). Produced water from wells off Louisiana contains 67-620 mg L^{-1} of dissolved TOC and 5-127 mg L^{-1} of suspended TOC (Veil et al., 2005). A large fraction of the dissolved TOC may be found in colloidal suspension (Means et al., 1989).

5.1.2 Organic Acids

Much of the TOC in produced water consists of a mixture of low molecular weight carboxylic acids, such as acetic acid, propionic acid, butyric acid, and valeric acid (Means and Hubbard, 1987; Somerville et al., 1987; Barth, 1991). These low molecular weight organic acids are readily synthesized, as well as biodegraded, by bacteria, fungi, and plants. They represent nutrients for phyto- and zooplankton growth. The most abundant organic acid typically found in produced water is acetic acid. The abundance of the various organic acids typically decreases with increasing molecular weight (Fisher, 1987). Many heavy crude oils contain high concentrations of naphthenic acids and cycloalkane carboxylic acids (with one or more saturated 5- or 6-ring carbon structures). They are slightly water soluble and, when present in a heavy crude oil, also are present in the associated produced water. Organic acids are produced by hydrous pyrolysis or microbial degradation of hydrocarbons in the hydrocarbon-bearing formation (Borgund and Barth, 1994; Tomczyk et al., 2001).

5.1.3 Petroleum Hydrocarbons

Petroleum hydrocarbons are the organic components of greatest environmental concern in produced water. Petroleum hydrocarbons are classified into two groups: saturated hydrocarbons and aromatic hydrocarbons. The solubility of petroleum hydrocarbons in water decreases as their size (i.e. molecular weight) increases. Aromatic hydrocarbons are more water-soluble than saturated hydrocarbons of the same molecular weight. The hydrocarbons in produced water appear in both dissolved and dispersed (i.e. oil droplets) forms. Current regulatory guidelines for produced water discharge in Canada are based on petroleum hydrocarbon content. In Canada, the hydrocarbon content of produced water must be reduced to acceptable levels pursuant to the 'Offshore Waste Treatment Guidelines (2002)' prior to discharge into the ocean (NEB et al., 2002). The minimum regulated standard for the treatment and/or disposal of wastes associated with the routine operations of drilling and production installations offshore Canada is a 30-day weighted average of oil in discharged produced water of 30 mg L^{-1}, which is coupled with a 24-hour arithmetic average of oil in produced water that shall not exceed 60 mg L^{-1}.

Existing oil/water separators, such as hydrocyclones, are quite efficient in removing oil droplets from produced water. As a result, petroleum hydrocarbons discharged to the ocean in produced water are typically low molecular weight aromatic hydrocarbons, as well as smaller amounts of saturated hydrocarbons found in the dissolved phase. Since there are no clean-up procedures that are completely effective, treated produced water still contains some dispersed oil droplets in the size range of 1-10 μm in diameter (Johnsen et al., 2004). The droplets contain most of the higher molecular weight, less soluble saturated and aromatic hydrocarbons. The most abundant hydrocarbons in produced water are the one-ring aromatic hydrocarbons, benzene, toluene, ethylbenzene, xylenes (benzene, toluene, ethylbenzene, and xylenes are known as BTEX), and low molecular weight saturated hydrocarbons. Xylenes may be present in produced water from different sources and at concentrations as high as 600 mg L^{-1} (Table 2). Benzene often is the most abundant BTEX compound in produced water, followed by toluene. Because BTEX is extremely volatile, it is lost rapidly during produced water treatment, by air stripping and during initial mixing of the plume in the ocean (Terrens and Tait, 1996).

Polycyclic aromatic hydrocarbons (PAH), are defined as hydrocarbons containing two or more fused aromatic rings. These are the petroleum hydrocarbons of greatest environmental concern in produced water because of their toxicity and persistence in the marine environment (Neff, 1987b; Neff, 2002). Concentrations of total PAH in produced water typically range from approximately 0.04-3.0 mg L^{-1} and primarily consist of 2- and 3-ring PAH compounds such as naphthalene, phenanthrene, and their alkyl homologues (Table 3). Higher molecular weight, 4- through 6-ring PAH compounds rarely are detected in properly-treated produced water. When present, they are typically associated with dispersed oil droplets (Johnsen et al., 2004).

5.1.4 Phenols

Concentrations of total phenols in produced water usually range from 0.4-23 mg L^{-1} (Table 2). Measured concentrations of total phenols in produced waters from the Louisiana Gulf coast and the Norwegian Sector of the North Sea range from 2.1-4.5 mg L^{-1} and 0.36-16.8 mg L^{-1}, respectively (Neff, 2002: Johnsen et al., 2004). The most abundant phenols in these produced waters are phenol, methylphenols, and dimethylphenols. Alkylphenol ethoxylate (APE) surfactants, containing octylphenols and nonylphenols, are sometimes used in the production system to facilitate the pumping of viscous or waxy crude oils. If the surfactant degrades some alkylphenols may dissolve in the produced water. Because of the toxicity of the more highly-alkylated phenols as endocrine disruptors, alkylphenol APE surfactants have been replaced in applications where the surfactant or, its degradation products, may reach the environment in significant amounts (Getliff and James, 1996).

5.1.5 Salinity and Inorganic Ions

The salt concentration (i.e. salinity) of produced water can range from a few parts per thousand (‰) to that of a saturated brine of approximately 300‰, compared to the typical salinity of seawater of 32-36‰ (Rittenhouse et al., 1969; Large, 1990) (Table 2). Most produced water has salinities greater than that of seawater and, are therefore, more dense than seawater (Collins, 1975). Produced water contains the same salts as seawater, with sodium and chloride the most abundant ions. Sulfate and sulfide concentrations usually are low, allowing barium and other elements that form insoluble sulfates and sulfides to be present in solution at high concentrations. However, if seawater, which naturally contains a high concentration of sulfate (i.e. approximately 29 millimolar, or mM), is injected into the formation for production purposes, barium and calcium may be precipitate onto the inner surface of production pipes as a scale coating. Ammonium ions may be present in some produced waters at highly elevated concentrations, which may induce inhibitory (i.e. toxic) and/or stimulatory (e.g. eutrophication) responses from resident biota (Anderson et al., 2000).

Table 3. *Concentrations (in µg L^{-1}) of individual polycyclic aromatic hydrocarbons (PAH)s found in produced waters of the Scotian Shelf and Grand Banks, compared to the U.S. Gulf of Mexico (DOE, 1997; OOC, 1997), Indonesia (Neff and Foster, 1997), and Thailand (Battelle, 1994).*

Compound	Gulf of Mexico	Indonesia	Thailand	Scotian Shelf[1]	Grand Banks[2]
Acenaphthene	ND-0.10	ND-1.3	ND	ND	ND
Acenaphthylene	ND-1.1	ND	ND	1.3	2.3
Anthracene	ND-0.45	0.17-0.63	ND	0.26	ND
Benz(a)anthracene	ND-0.20	0.06-0.43	ND	0.32	0.60
Benzo(a)pyrene	ND- 0.09	ND	ND	ND	0.38
Benzo(b)fluoranthene	ND-0.03	ND-0.31	ND	ND	0.61
Benzo(e)pyrene	ND-0.10	0.05-0.81	ND	ND	0.83
Benzo(ghi)perylene	ND-0.03	ND	ND	ND	0.17
Benzo(k)fluoranthene	ND-0.07	ND	ND	ND	ND
Biphenyl	0.36-10.6	2.3-6.9	NA	ND	ND
Chrysene	ND-0.85	0.10-1.4	ND	ND	3.6
C_1-Chrysenes	ND-2.4	0.41-5.6	0.16	ND	6.3
C_2-Chrysenes	ND-3.5	0.69-8.8	ND	ND	18.8
C_3-Chrysenes	ND-3.3	0.52-6.4	ND	ND	6.7
C_4-Chrysenes	ND-2.6	ND-2.4	ND	ND	4.2
C_1-Fluorenes	0.09-8.7	3.3-9.2	3.55	3	23.7
C_2-Fluorenes	0.20-15.5	2.2-13.3	4.03	0.35	4.8
C_3-Fluorenes	0.27-17.6	1.9-15.6	2.24	ND	ND
C_1-Fluoranthenes/ Pyrenes	ND-2.4	1.0-4.3	0.34	0.43	5.8
C_2-Fluoranthenes/ Pyrenes	ND-4.4	0.93-7.3	NA	ND	9.1
C_1-Naphthalenes	4.2-73.2	63.5-100	207	499	186
C_2-Naphthalenes	4.4-88.2	43.4-126	166	92	163
C_3-Naphthalenes	2.8-82.6	19.3-81.3	71.1	17	97.2
C_4-Naphthalenes	1.0-52.4	6.4-36.2	21.0	3.0	54.1
C_1-Phenanthrenes	0.24-25.1	4.2-29.5	4.06	1.30	45.0
C_2-Phenanthrenes	0.25-31.2	3.8-31.5	3.43	0.55	37.1
C_3-Phenanthrenes	ND-22.5	3.0-25.4	1.91	0.37	24.4
C_4-Phenanthrenes	ND-11.3	1.3-11.9	ND	ND	13.2
Dibenz(a,h)anthracene	ND-0.02	ND	ND	ND	0.21
Fluoranthene	ND-0.12	0.13-0.24	ND	0.39	0.51
Indeno(1,2,3-cd)pyrene	ND-0.01	ND	ND	ND	ND
Naphthalene	5.3-90.2	45.6-156	395	1512	131
Perylene	0.04-2.0	ND-0.09	ND	ND	ND
Phenanthrene	0.11-8.8	9.3-29.5	3.12	4.0	29.3
Pyrene	0.01-0.29	0.12-0.49	ND	0.36	0.94
Total PAHs	**40-600**	**230-745**	**888**	**2148**	**845**

ND: Not detected
NA: Not analyzed
[1] Thebaud (DFO-Centre for Offshore Oil, Gas and Energy Research (COOGER), Unpublished Data)
[2] Hibernia (DFO-COOGER, Unpublished Data)

5.1.6 Metals

Produced water may contain several metals in solution. The type and concentration of metals present in produced waters from different sources is variable, depending on the age and geology of the formations from which the oil and gas are produced (Collins, 1975). A few metals may be present in produced waters from different sources at concentrations substantially higher (i.e. 1000-fold or more) than their concentrations in clean natural seawater. The metals most frequently present in produced water at elevated concentrations, relative to those in seawater, include barium, iron, manganese, mercury, and zinc (Neff et al., 1987) (Table 4). Typically, only a few of these metals are present at elevated concentrations in a particular produced water sample. On the east coast of Canada, elevated concentrations of barium, iron and manganese have been reported in close proximity to the Hibernia platform located on the Grand Banks (Yeats et al., 2011).

Table 4. *Concentrations (µg L^{-1}) of metals found in produced water from the Scotian Shelf and the Grand Banks, compared to the northwestern Gulf of Mexico and Norwegian Sector of the North Sea (Neff, 2002). Concentrations found in seawater are also included.*

Metal	Scotian Shelf[1]	Grand Banks[2]	Gulf of Mexico[3]	North Sea[4]	Seawater
Arsenic (As)	90	<10	0.5-31	0.96-1.0	1-3
Barium (Ba)	13,500	301-354	81,000-342,000	107,000-228,000	3-34
Cadmium (Cd)	<10	<0.02-0.04	<0.05-1.0	0.45-1.0	0.001-0.1
Chromium (Cr)	<1-10	<1	<0.1-1.4	5-34	0.1-0.55
Copper (Cu)	137	<5	<0.2	12-60	0.03-0.35
Iron (Fe)	12,000-28,000	1,910-3,440	10,000-37,000	4,200-11,300	0.008-2.0
Lead (Pb)	<0.1-45	0.09-0.62	<0.1-28	0.4-10.2	0.001-0.1
Manganese (Mn)	1,300-2,300	81-565	1,000-7,000	NA	0.03-1.0
Mercury (Hg)	<10	NA	<0.01-0.2	0.017-2.74	0.00007-0.006
Molybdenum (Mo)	NA	<1	0.3-2.2	NA	8-13
Nickel (Ni)	<0.1-420	1.7-18	<1.0-7.0	22-176	0.1-1.0
Vanadium (V)	NA	<0.1-0.6	<1.2	NA	1.9
Zinc (Zn)	10-26,000	<1-27	10-3,600	10-340	0.006-0.12

[1] SOEP/DFO
[2] Combined results from Hibernia and Terra Nova (DFO-COOGER, Unpublished Data)
[3] Combined results from seven platforms
[4] Combined results from 12 platforms

Injecting seawater into a well during production may increase the concentration of sulfate, which is present in seawater at a concentration of approximately 900 mg L^{-1}, in the formation water causing barium to precipitate as barite ($BaSO_4$). This lowers the concentration of free barium found in the produced water (Stephenson et al., 1994). Several other metals in produced water, particularly radium isotopes, may co-precipitate with barium and reduce their concentrations. Formation water is anoxic and iron and manganese may be present in solution at high concentrations. When these formation waters are brought to the surface and exposed to the atmosphere, the iron and manganese precipitates out as iron and manganese oxyhydroxides. Several other metals in produced water may co-precipitate with iron and manganese and be dispersed, adsorbed to, or complexed with very fine, solid hydrous iron and manganese oxides in the receiving waters (Lee et al., 2005b; Azetsu-Scott et al., 2007). Zinc and, possibly lead, may be derived in part from galvanized steel structures in contact with the produced water or with other waste streams that may be treated in the oil/water separator system.

5.1.7 Radioisotopes

Naturally occurring radioactive material (NORM) is present in produced water in many parts of the world. The most abundant NORM radioisotopes in produced water are the natural radioactive elements radium-226 (^{226}Ra) and radium-228 (^{228}Ra). Radium is derived from the radioactive decay of uranium and thorium associated with certain rocks and clays found in the hydrocarbon reservoir (Reid, 1983; Kraemer and Reid, 1984; Michel, 1990). ^{226}Ra, which has a half-life of 1601 years, is an α-emitting daughter of uranium-238 and uranium-234. ^{228}Ra, which has a half-life 5.7 years, is a β-emitting daughter of thorium-232. Surface waters of the ocean have ^{226}Ra and ^{228}Ra concentration values of 0.027-0.040 picocuries per liter (pCi L^{-1}) and approximately 0.005 pCi L^{-1}, respectively (Santschi and Honeyman, 1989; Nozaki, 1991).

Concentrations of the combined total ^{226}Ra and ^{228}Ra found in produced water from oil, gas, and geothermal wells along the Gulf of Mexico coast range from less than 0.2 pCi L^{-1} to 13,808 pCi L^{-1} (Kraemer and Reid, 1984; Neff et al., 1989b) (Table 5). There is no correlation between the concentrations of the two radium isotopes in produced water, because of their different origins in the geologic formation. Concentrations of radium and other NORM found in produced water from elsewhere in the world, other than the northern Gulf of Mexico, are generally low, with the activity of Ra^{226} and Ra^{228} usually less than 200 pCi L^{-1} (Table 5). Preliminary studies on produced water samples collected in Atlantic Canada indicate radium isotope levels several orders of magnitude above normal seawater concentrations. Due to the level of natural dispersion, however, only ambient background concentrations can be detected in seawater samples collected in the area of production platforms located on the Grand Banks and the Scotian Shelf (Nelson, 2009).

***Table 5.** Activities (pCi L^{-1}) of radium-226 (^{226}Ra) and radium-228 (^{228}Ra) found in produced water from different locations (Neff, 2002). Activities found in seawater are also included.*

Offshore Location	Radium-226 (^{226}Ra)	Radium-228 (^{228}Ra)	Reference
Scotian Shelf	1.2	9.2	Nelson, 2009
Grand Banks	33.0	229.7	Nelson, 2009
Cook Inlet, Alaska	<0.4-9.7	NA	Neff, 1991
Louisiana Gulf Coast	ND-1,565	ND-1,509	Kraemer and Reid, 1984
North Sea	44.8	105	Stephenson et al., 1994
Offshore U.S. Gulf of Mexico	91.2-1,494	162-600	Hart et al., 1995
Santa Barbara Channel, California	165	137	Neff, 1997
S. Java Sea, Indonesia	7.6-56.5	0.6-17.7	Neff and Foster, 1997
Texas	0.1-5,150	NA	Fisher, 1987
Seawater	0.027-0.04	0.005	Santschi and Honeyman, 1989; Nozaki, 1991

5.1.8 Production Chemicals

Large numbers of specialty additives, or treatment chemicals, are available for use in the production system of a well that aid in the recovery and pumping of hydrocarbons, protect the production system from corrosion, and facilitate the separation of oil, gas, and water (Table 6). These include biocides, scale inhibitors, emulsion-breakers, and gas-treating chemicals. Many of these chemicals are more soluble in oil than in produced water, so remain in the oil phase. Others are water-soluble, concentrate in produced water, and are disposed with it. The point in the production stream where the chemical is added influences the amount that may be discharged to the ocean or re-injected into the reservoir with the produced water. Treatment chemicals are used in direct response to a problem and are not added where there is no demonstrated need. The use of treatment chemicals is managed through the use of best management practices such as the Offshore Chemical Selection Guidelines (OCSG) and regulatory compliance effluent toxicity testing protocols.

***Table 6.** Typical use concentration (ppm) of production chemicals used on North Sea oil and gas platforms and the estimated amounts (t y^{-1}) discharged to the ocean (Johnsen et al., 2004).*

Chemical	Concentration (ppm)	Phase	Discharge (t y^{-1})
Biocide	10 – 200	Water	81
Coagulants and flocculants	<3	Water	197
Corrosion inhibitor	25 – 100	Oil	216
Emulsion Breaker	10 – 200	Oil	9
Gas treatment chemicals	Variable	Water	2846
H_2O/O_2 scavenger	5 – 15	Water	22
Scale inhibitor	3 – 10	Water	1143

5.2 TREATMENT

Most environmental regulatory agencies in countries that have significant offshore oil and gas production place limits on the concentrations of petroleum, usually measured as total oil and grease, that can be present in produced water discharged to the ocean. Table 7 gives examples of limits on oil and grease set by various countries. Produced water intended for ocean disposal is usually treated on the platform or at a shore treatment facility to meet regulatory limits. The objective of oil-water-gas treatment on an offshore platform is to produce stabilized crude oil and gas for pipeline or tanker transport to shore facilities and to generate a produced water that meets discharge requirements (if discharged to the ocean) or is suitable for reinjection into the producing formation or other geologic formation (Bothamley, 2004).

Table 7. *Monthly average and daily maximum concentrations (mg L^{-1}) of total oil and grease allowed in produced water discharged to the ocean, as permitted by various offshore petroleum producing countries (Veil, 2006).*

Country	Monthly Average (mg L^{-1})	Daily Maximum (mg L^{-1})
Canada	30	60
Brazil	---	20
Mediterranean Sea	40	100
Nigeria	40	72
OSPAR (NE Atlantic)	30	---
U.S.	29	42
Western Australia	30	50

It is necessary to treat produced water before ocean discharge to avoid the harmful effects that the chemicals in waste waters may have on the receiving environment. Treatment removes solids and non-aqueous liquids from the waste water, including dispersed oil, suspended solids, scales, and bacterial particles, as well as corrosive gases such as carbon dioxide (CO_2) and hydrogen sulphide (H_2S). Experience with produced water treatment for ocean disposal by the offshore oil industry has shown that, if dispersed oil is removed, concentrations of dissolved hydrocarbons are reduced to acceptable levels (Ayers and Parker, 2001). If the treated waste water is intended for disposal in freshwater, recycled for steam generation for the various thermal enhanced oil recovery (EOR) technologies, or for reinjection into the formation, most of the dissolved salts also should be removed. Salt removal is not necessary if discharge is into the ocean.

The produced water oil/gas/water mixture may be processed through devices to separate the three phases from each other. The types of equipment used on many platforms to remove oil and grease from produced water include mechanical and hydraulic gas floatation units, skimmers, coalescers, hydrocyclones, and filters (Otto and Arnold, 1996). Chemicals may be added to the process stream to improve the efficiency of separation. The combination of mechanical and chemical treatments is effective in removing volatile compounds and dispersed oil from the produced water, but they are ineffective in removing dissolved organics and inorganics. Even with the most advanced separation equipment, the oil/water separation is not completely efficient.

Produced water from offshore oil and gas wells is treated to remove volatile hydrocarbons, dispersed petroleum, and suspended solids to the extent afforded by current waste water treatment technology. The global average concentration of total petroleum hydrocarbons in produced water discharged offshore in 2003 was 21 mg L^{-1}, with a range for different geographic regions of 14-39 mg L^{-1} (OGP, 2004) (Figure 8). The quality of produced water discharge is primarily a function of the strictness and degree of enforcement of environmental discharge regulations. In Canada, the formulation of regulatory guideline values for produced water discharge is an adaptive process that promotes the development of improved environmental effects monitoring (EEM) programs and takes into consideration the level of environmental risk, Best Available Technology (BAT) for mitigative measures, and social-economic benefits.

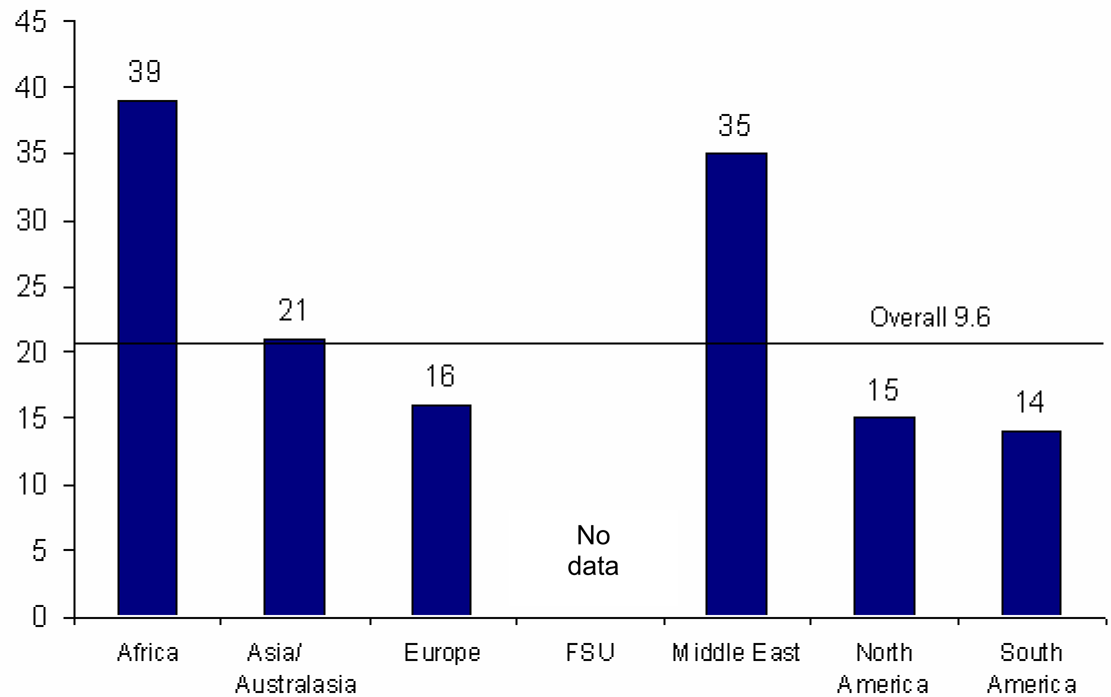


Figure 8. *Concentration (mg L^{-1}) of total petroleum hydrocarbons (TPH) in produced water discharged offshore in several parts of the world. The overall average TPH concentration in produced water from both onshore and offshore petroleum production is 9.6 mg L^{-1} (solid line). Data are not available for the former Soviet Union (FSU) (OGP, 2004).*

5.3 FATE OF DISCHARGE IN THE OCEAN

Treated produced waters on offshore platforms are typically discharged above or below the sea surface once regulatory compliance concentrations are achieved. The location of subsurface discharge pipes may range at depths of 10-100 m. Saline produced waters are usually as dense as or denser than seawater and disperse below the sea surface, diluting rapidly upon discharge into well-mixed marine waters. Low salinity produced water may form a plume on the sea surface and dilute more slowly (Nedwed et al., 2004). Dispersion modeling studies of the fate of produced water differ in specific details, but all predict a rapid initial dilution of discharges by 30- to 100-fold in the first few tens of meters of the outfall. This is followed by a slower rate of dilution at greater distances (Terrens and Tait, 1993; Strømgren et al., 1995; Brandsma and Smith, 1996; Smith et al, 2004). When discharge volumes of buoyant or neutral-density produced water are very high, dilution may be slower. Factors that affect the rate of dilution of produced water include discharge rate, height above or below the sea surface, ambient current speed, turbulent mixing regime, water column stratification, water depth, differences in density (as determined by temperature and total dissolved solids concentration), and chemical composition between the produced water and receiving seawater.

5.3.1 Plume Dispersion Models

Brandsma and Smith (1996) modeled the fate of produced water discharged under typical Gulf of Mexico conditions. They used two discharge rates: 115.7 m^3 per day, which is the median flow rate for offshore discharges to the Gulf of Mexico, and 3,975 m^3 per day, which is the maximum allowable discharge rate from a single discharge pipe to the Gulf of Mexico. The effluent was a hypersaline brine that was discharged at a temperature of approximately 29ºC. As a result, it was denser than the ambient seawater and tended to sink. For a median produced water discharge rate of 115.7 m^3 per day, the predicted concentration of produced water in the plume 100 m down-current from the discharge point ranged from 0.043-0.097% produced water depending on ambient current speed. At the higher discharge rate, dilutions at 100 m down-current from the discharge ranged from 0.18-0.32 percent produced water depending on current speed. For large produced water discharges, such as the discharge from the platform *Irene* off Santa Barbara, California, which has a discharge rate of 6,359 m^3 per day, the modeling work by Brandsma (2001) estimated a rapid decrease of plume centerline concentration to approximately 2% produced water at 100 m from the discharge point.

High dilution rates for produced water appear to be typical. In a modeling study by Terrens and Tait (1994), using the Offshore Operators Committee (OOC) model to predict the behaviour of a 14,000 m^3 per day discharge from the *Halibut* platform in Bass Strait, Australia, the produced water was reported to be dynamically-indistinguishable due to a high degree of initial dilution. Dilutions of seawater to produced water ranged from 100:1 to 252:1). At 6 km, the dilution factors were approximately 13,000:1 for suspended oil and 18,000:1 for dissolved oil. Skåtun (1996) used a BJET model to study the near-field mixing of warm (i.e. 32°C) produced water of high salinity (84 ppt) released from a platform in the Gulf of Mexico. At 103 m downstream of the release point, a dilution factor of 400:1 was reported in the presence of a 0.15 m s^{-1} current. The physical dispersion models of projected discharges from the Sable Offshore Energy Project (SOEP) wells located on the Scotian Shelf of Canada also indicated a rapid dilution of discharge to non-acute toxic levels within short distances of the discharge points (SOEP, 1996).

The OOC model is used for predicting the short term fate of produce water discharge and the other models described above only consider near field mixing. It is important, however, to also study the long-term, far-field transport of produced water. Hodgins and Hodgins (1998) studied the dispersion of produced water from the Terra Nova floating production, storage, and offloading (FPSO) vessel off the East Coast of Canada using the Three-dimensional Updated Merge (UM3) model coupled with a particle tracking based far-field model. For a maximum discharge of 18,000 m^3 per day, the estimated worst-case initial dilution was 5:1. As the pooled effluent near the hull of the FPSO is carried away by the ambient currents, the far field model predicted a minimum secondary dilution of 5:1, which yields a combined total dilution of 25:1 after the plume has been dispersed a few hundred meters from the FPSO.

The same modeling concept was also applied to the White Rose development off the East Coast of Canada (Hodgins and Hodgins, 2000). For the discharge of produced water with a density of 728 kg m^{-3} at a maximum discharge rate of 30,000 m^3 d^{-1} from a

14 inch diameter pipe at 5 m below sea surface, the near-field model of UM3 estimated an initial bulk dilution of 35:1 (Hodgins and Hodgins, 2000). The far-field dispersion simulation showed that the 1% impact line (concentration greater than 0.1 mg L^{-1} or dilution less than 400:1 for at least 1% of the time) extend from 1.8-3.2 km. Similar results were also described by AMEC (2006), who predicted near-, intermediate-, and far-field dilution rates using the U.S. EPA Visual Plumes model (Baumgartner et al., 1994), USACE CDFate model (Chase, 1994) and an advection/diffusion model. Assuming a maximum flow rate of 6400 m^3 d^{-1} of produced water from the Deep Panuke facility (not yet in production) to be located in the offshore of Nova Scotia, the model estimated a dilution of 70:1 at 500 m from the proposed discharge centre and 400:1 at 2 km from the discharge centre.

The ASA MUDMAP^{TM} model (refer to Spaulding, 1994) was run by Burns et al. (1999) to study the dispersion of produced water from the Harriet oil field off the western Australian coast. Based on an average discharge of 8000 m^3 d^{-1} and total oil concentration of 5.9-16 mg L^{-1}, the model predicted that the produced water plume would exhibit oil concentrations of 0.006 mg L^{-1} near the discharge centre to 0.00016 mg L^{-1} at approximately 8 km downstream. Zhao et al. (2008) integrated a random walk based particle tracking model with the Princeton Ocean Model (POM) for fast prediction of future dispersion and risks of produced water discharges. For a produced water discharge at 882 m^3 h^{-1} from the Hibernia platform off the East Coast of Canada, the model predicted a lead concentration of 0.002 µg L^{-1} at approximately 5.3 km south of the platform. An overestimation of dilution and underestimation of pollutant concentration may result from the Zhao et al. (2008) method due to an omission of near-field buoyant jet behaviors. Accuracy of the model could be improved by coupling a near-field model with the particle tracking model using the method described by Zhang (1995) and Niu et al. (2011).

For a hypothetical study of the produced water discharged from the Terra Nova FPSO, Mukhtasor et al (2004) estimated a mean concentration of 0.5% of the initial concentration at approximately 225 m from the discharge. The 95%-tile concentration from their analysis at the same location was approximately 2.25% the initial concentration. Similar approaches were also used by Niu et al. (2009a) to study the effects of surface waves on the dispersion of produced water. Since the models of Mukhtasor (2004) and Niu et al. (2009a) can only be used for a limited number of discharge conditions, Niu et al. (2009b) expanded the same approach into a more general model (PROMISE) which can be used over a wider range of discharge and environmental conditions. A validation study of the PROMISE model has been reported by Niu et al. (2009c) and good agreement was found between model predictions and laboratory measurements.

As is evident in this overview, during the last two decades a significant amount of effort has been put forward to model the dispersion of produced water plumes in the marine environment. Researchers from different disciplines have approached the problem from different perspectives and developed models of varying degrees of sophistication. Produced water plume dispersion models have evolved from the simple steady state near-field, short term dilution models to comprehensive coupled hydrodynamic/dispersion models that predict both the near- and far-field dispersion processes in 3D non-steady state conditions. Although the dispersion models are now able to simulate the near-field mixing process very well and, predict the far-field mixing process reasonably well, they are still limited in their ability to predict the fate of the various chemical components in produced water.

5.3.2 Chemical Fate and Transport Models

In the majority of physical transport models the chemicals in the produced water stream are treated as passive tracers. Dye injection tracer studies of produced water plumes (e.g. DeBlois et al., 2011) are based on the same assumptions. The drawback of using dyes is that they may not become fully integrated into the produced water plume, resulting in an inaccurate prediction of transport of the various contaminants that react chemically and separate from the plume. Based on recent studies by Lee et al. (2005b) and Azetsu-Scott et al. (2007), consideration must also be given to the chemical reactivity that may influence the subsequent fate, transport, and effects of the contaminants of concern in the produced water following its discharge.

To study the fate of benzene and naphthalene in produced water discharges from the SOEP Thebaud platform on the Scotian Shelf, Berry and Wells (2004a; 2004b) used the CORMIX model (refer to Doneker and Jirka, 2007) to determine the exposure pathways and potential compartment interactions and then a Level III fugacity model (refer to Mackay, 1991) to study the distribution of benzene and naphthalene among environmental medias, such as water, suspended particles, fish, and sediments. The study indicated that the average water column bulk concentration of benzene and naphthalene over a 1 km by 1 km area were 5.28×10^{-6} μg g^{-1} and 8.49×10^{-7} μg g^{-1}, respectively, for a maximum discharge rate of 211.7 m^3 d^{-1}. Since the CORMIX model predicted that the produced water plume may not be fully mixed in the selected compartments, the fugacity model may underestimate the concentration in the produced water plume and overestimate the concentration outside of the plume.

In a study by Smith et al. (1996), a coupled model was used to simulate the transport of produced water from the Pertamina/Maxus petroleum operation area in the Java Sea, Indonesia. The CORMIX model was first used to predict effluent dispersion and the results were then integrated into a PISCES model (refer to Turner et al., 1995) to study partitioning, degradation, and volatilization. The model predicted that mercury at 500 m away from the discharge centre was approximately 5.25-522 ppt and arsenic was approximately 5-12 ppb. At 3000 m downstream, the mercury concentration decreased to 65 ppb and arsenic to 3 ppb. The use of separate dispersion and chemical models cannot account for the dynamics that change the chemical concentrations following discharge. Murray-Smith et al. (1996) applied the particle-based model TRK to the Clyde offshore petroleum platform in the United Kingdom sector of the North Sea. Besides physical dispersion, the model allowed a first order degradation to simulate biodegradation or other removal processes. The study found that the initial dilution was rapid and a minimum dilution factor with seawater of 300 to 3000 times within 100 m of discharge can be achieved. Further away, the physical dilution was less rapid and other removal processes such as biodegradation may become important. The model predicted an overall dilution (including biodegradation) of 1000-16000 at 1 km from the discharge centre.

Reed et al. (1996) described a PROVANN model for the simulation of 3D transport, dilution, and degradation of chemicals associated with produced water discharges from one or more simultaneous sources. The model included various transport processes such as adsorption/dissolution kinetics, entrainment, and dissolution of oil droplets, volatilization, degradation, and deposition from the water column. For two platforms off of Trondheim, Norway, the simulation showed that the naphthalene concentration at the edge of the plume 40 km downstream after 50 days was extremely low (approximately 0.00006 ppb). The simulation also found that the inclusion of degradation is clearly an important factor in modeling long term exposure. Reed et al. (2001) described a Dose-related Risk and Exposure Assessment Model (DREAM) for simulation of produced water discharges (Figure 9). The model has been recently used by Durell et al. (2006) to model the fate of various PAHs in produced water discharged from the Ekofisk and Tampen fields in the Norwegian sector of the North Sea. For Ekofisk region stations, the model predicted naphthalene concentrations in the range of 5-29.6 nanograms per litre (ng L^{-1}). The total PAHs for this region were 5.55-32.3 ng L^{-1}. The predicted concentrations in the Tampen region were higher than the Ekofisk region, with naphthalene ranging from 5-311 ng L^{-1} and total PAHs ranging from 5.55-344 ng L^{-1}.

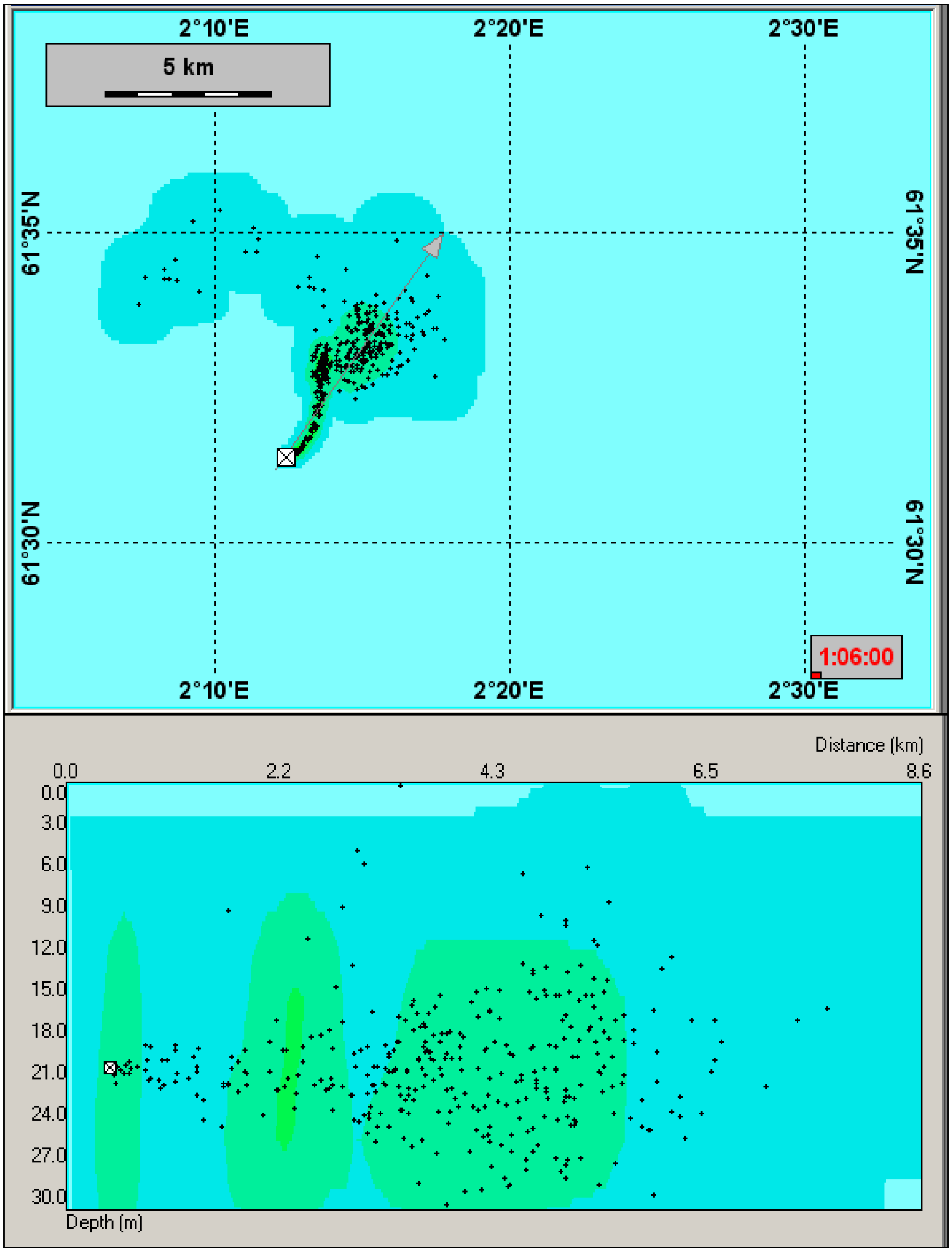


Figure 9. *Example of a DREAM simulation of a produced water plume: aerial view (top) and cross-sectional profile (bottom).*

5.3.3 Field Measurements of Produced Water Dilutions

Field measurements of produced water dilutions are highly variable but confirm the predictions of modeling studies that dilution is typically rapid. In 1993, Continental Shelf Associates reported that radium-226 from a 1070 m^3 d^{-1} produced water discharge at depth of 18 m in the Gulf of Mexico was diluted by a factor of 426 at 5 m from the discharge centre and by a factor of 1065 at 50 m from the discharge centre (Continental Shelf Associates, 1993). The current speed at the time of measurement was 15 cm s^{-1} with little vertical stratification of the water column. The OOC model, but not the

CORMIX model, accurately predicted the dilutions measured in the field. Produced waters of the Gulf of Mexico contain high concentrations of dissolved barium. It is probable that the radium in the produced water co-precipitated rapidly with barium sulfate in the sulfate-rich receiving waters.

The comparison of field measurements of hydrocarbons and various other organic components from both fixed stations sampling and continuous towed fluorometer with modeled data showed that measured dilutions were generally much higher than predicted (Murray-Smith et al., 1996). The measured dilutions with distance for alkanes and methylnapthalenes at 100-1000 m were extremely low. The measured dilution factors ranging from 5000-50,000 were much greater than predicted. The study also found that the measured concentrations of other organics such as phenol and methylphenol in the water column were below detection limits. Similar results were also reported for the analysis of five heavy metals in produced water samples collected from the Java Sea, where a high dilution ratio was observed as predicted in the models (Smith et al., 1996).

Terrens and Tait (1996) measured concentrations of BTEX and several PAHs in ambient seawater 20 m from an 11,000 m^3 d^{-1} produced water discharge from a platform in the Bass Strait, southeastern Australia (Table 8). They observed an inverse relationship between molecular weight (and volatility) and the dilution of individual aromatic hydrocarbons. The produced water discharge contained an average BTEX concentration of 6,410 µg L^{-1}, although the average concentration at 20 m from the discharge centre was 0.43 µg L^{-1}, marking a dilution of 14,900 fold. The PAH concentrations were diluted by 11,000 fold for naphthalene and 2,000 fold for pyrene. Concentrations of higher molecular weight PAHs at the sample location were below the detection limit of 0.0002 µg L^{-1}. The inverse relationship between molecular weight of the aromatic hydrocarbons and their rates of dilution probably was due in large part to the high temperature of 95ºC of the discharged produced water, which may have favoured evaporation of the lighter aromatic hydrocarbons.

Table 8. *Concentrations (µg L^{-1}) of four PAHs in produced water discharge, seawater collected 20 m down-current of the discharge centre, and its dilution from the Kingfish B Platform in the Bass Strait, southeast Australia (Terrens and Tait, 1996).*

PAH	Produced Water (µg L^{-1})	Down-current (µg L^{-1})	Dilution
Fluoranthene	0.8	0.0002	4,000
Naphthalene	440	0.04	11,000
Phenanthrene	18	0.003	6,000
Toluene	3,000	0.18	16,700

Field measurements are important for both understanding of the fate of produced water and for model validation. Traditionally, this was achieved by either collecting water samples at pre-determined stations or by continuous towing of a fluorometer (Smith et al., 1994; Murray-Smith et al., 1996; Smith et al., 1996; Terrens and Tait, 1996; Smith et al., 2004). Traditional ship-based sampling methods are often expensive and time consuming and, as a result, only limited information can be collected. Increased water depth also increases the level of sampling error. Recently, new and innovative means of conducting field measurements using Autonomous Underwater Vehicles (AUV) have been proposed and may be used in the future study of produced water fate and to collect data to validate mathematical fate and transport models (Niu et al., 2007; Niu et al., 2009d).

To date, the majority of plume dispersion and chemical fate and transport models that have been developed focus on the process of dispersion and treat produced water as a single conservative contaminant. Only a few models have attempted to include other transformation processes, such as biodegradation, evaporation, and adsorption. Among these models, the most comprehensive appears to be the DREAM model which is capable of handling a multitude of complex processes and data, including discharge volumes, physical, chemical, and biological fates of discharged substances, and biological uptake and effects (Reed et al., 2001). Capable of predicting the fate of individual contaminants associated with produced water, the model is currently used extensively by North Sea operators to achieve the regional regulatory goal of ‘zero harmful effect discharges’. The model can also be used to study the environmental effects of produced water using two approaches: the environmental impacts factor and a body burden-related risk assessment model.

5.4 IMPACTS ON MARINE ENVIRONMENT

Based on the concentrations of toxic chemicals found in most produced waters and their predicted dispersion rates, it is envisioned that there would be only limited potential for acute toxicity beyond the immediate vicinity of rig sites in Atlantic Canada. This is attributed to the sensitivity of the biotests, primarily regulatory acute toxicity assays, and the rapid dispersion of the process stream (Lee et al., 2005b). Holdway (2002), however, noted that to fully assess the potential impact of produced water discharges, the chronic impacts associated with long-term exposures must be quantified. Continual long-term chronic exposure may cause sub-lethal changes in organisms including decreased community and genetic diversity, lower reproductive success, decreased growth and fecundity, respiratory problems, behavioural and physiological disorders, decreased developmental success, and endocrine disruption. Chronic toxicity studies are ongoing by DFO to support the development of cost-effective and sensitive monitoring protocols for regulatory use.

5.4.1 Impacts on Pelagic Organisms

Biological effects in water column biological communities near open-ocean produced water discharges are expected to be localized because of the rapid dilution and dispersion rates of most produced water discharges into the ocean. Some produced waters, however, contain chemicals that are toxic to sensitive marine species, even at low concentrations. In particular, for produced water discharges in areas of limited water turbulence and low current speeds, concentrations of chemicals may remain high for enough time to cause ecological harm (Neff, 2002). The chemicals of greatest environmental concern in produced water, because of their potential for bioaccumulation and toxicity, are metals and hydrocarbons. Highly-alkylated phenols (i.e. octyl- and nonyl-phenols), though well-known to be endocrine disruptors, have not been detected in produced water at high enough concentrations to cause harm to water column animals following initial dilution. Nutrients, such as nitrate, phosphate, ammonia, and organic acids, may stimulate microbial and phytoplankton growth in the receiving waters (Rivkin et al., 2000; Khelifa et al., 2003). Some production treatment chemicals are toxic and, if they are discharged at high enough concentrations in produced water, may cause localized harm. Salts, such as sodium, potassium, and chloride, are not of concern in produced water discharges to the ocean, but they are of concern when the treated water is discharged to land or in fresh, surface waters or brackish waters.

5.4.2 Accumulation and Impacts in Sediments

If water depths are shallow, some metals and higher molecular weight aromatic and saturated hydrocarbons may accumulate in bottom sediments near produced water discharge centres and possibly harming bottom living biological communities (Neff et al., 1989b; Means et al., 1990; Rabalais et al., 1991). In well-mixed estuarine and offshore waters, elevated concentrations of saturated hydrocarbons and PAHs in surficial sediments may be observed out to a few hundred metres from a high-volume produced water discharge. The concentrations of PAHs in sediments near offshore produced water discharges are related to the volume and density of the produced water discharged, the PAH concentration in it, and the local water depth and mixing regime. Polycyclic aromatic hydrocarbons in sediments near offshore platforms may also come from drilling discharges, particularly if oil based drilling muds are used (Neff, 2005).

Barium, iron, and manganese are the metals most enriched in produced waters compared to their concentrations in natural seawater. A phase transition typically occurs following the release of produced water where the metals tend to rapidly precipitate rapidly when produced water is discharged to well-oxygenated surface waters high in sulfate. These particulate metals tend to settle slowly out of the water column and accumulate to slightly elevated concentrations in surficial sediments over a large area around the produced water discharge centre (Neff, 2002; Lee et al., 2005b). In addition, the transport and concentration of inorganic constituents in produced water (e.g. metals) that reaches the surface microlayer may be promoted by the interaction between residual oil droplets and metal precipitates (Lee et al., 2005b). Toxicity assessment using the Microtox Test®, a regulatory bioassay protocol based on inhibition of a primary metabolic function of a bioluminescent bacterium, demonstrated that unfiltered samples containing metal precipitates generally had higher toxicity levels than filtered samples (Azetsu-Scott et al., 2007). Results from regulatory EEM programs generally indicate that natural dispersion processes control the concentrations of toxic metals in the water column and sediments, where they have been observed to be slightly above natural background concentrations.

5.4.3 Toxicity

Most treated produced water has a low to moderate toxicity. A typical distribution of produced water toxicities can be seen in the data of produced waters discharged to the Gulf of Mexico off the Louisiana coast (Table 9). A small number of produced waters are moderately toxic to mysids, a small shrimp-like crustacean, and sheepshead minnows, with acute and chronic toxicities less than 0.1 percent produced water (1,000 mg L^{-1}). A few produced waters are largely nontoxic, as there is no evidence of acute and chronic effects in test waters containing less than 35-40% produced water. Most produced waters, however, have moderate toxicities with acute and chronic toxicities approximately 2-10% for mysids and 5- 20% for sheepshead minnows. Based on earlier toxicity studies of produced waters from the Gulf of Mexico, Neff (1987b) reported that nearly 52% of all median lethal concentrations (LC_{50}) were greater than 10% produced water, 37% were between 1-9.9%, and 11% were less than 1%. These toxicity threshold limits are consistent with those reported for Atlantic Canada.

Table 9. *Toxicity of over 400 produced water (PW) samples from the Gulf of Mexico (offshore of Louisiana) on mysids (Mysidopsis bahia) and sheepshead minnows (Cyprinodon variegatus). Exposure concentrations are percent produced water (modified from Neff, 2002, p.30).*

Toxicity Test	Number of Samples Tested	Concentration (% PW)	Standard Deviation
Mysidopsis bahia:			
Acute Toxicity (96 h LC50)	412	10.8	10.4
Chronic Survival (NOEC)	407	3.4	5.8
Chronic Growth (NOEC)	391	2.4	3.6
Chronic Fecundity (NOEC)	274	2.7	3.2
Cyprinodon variegates:			
Acute Toxicity (96 h LC50)	359	19.2	14.8
Chronic Survival (NOEC)	401	6.3	9.0
Chronic Growth (NOEC)	395	5.2	8.1

NOEC: No observed effects concentration.

Mixed-function oxidase enzyme activity was highly induced in the liver, gills, and heart of juvenile cod exposed to 5% produced water for 72 hours (Andrews et al., 2009). Histological changes were observed in the gills of fish exposed to relatively high levels of produced water. In studies with *Calanus finmarchicus,* a major prey species for fish in the Northwest Atlantic Ocean, no differences in mortality were found between control and experimental animals exposed to 5% produced water for 48 hours (Payne et al., 2001a). In a comprehensive study on the acute effects of produced water recovered from a Scotian Shelf offshore well on the early life stages of haddock, lobster, and sea scallop, in terms of survival, growth, and fertilization success, Querbach et al. (2005) noted that fed, stage I lobster larvae were the most sensitive with an observed LC_{50} of 0.9%. Feeding stage haddock larvae and scallop veligers were the least sensitive with LC_{50} of 20% and 21%, respectively. In terms of chronic responses, the average size of scallop veligers was significantly reduced after exposure to produced water concentrations greater than 10%.

There are poorly characterized species differences in the toxicity of produced waters to marine organisms. When bioassays were performed with two or more marine taxa and the same sample of produced water, crustaceans were generally more sensitive than fish (Neff, 1987b; LDEQ, 1990; Jacobs and Marquenie, 1991; Terrens and Tait, 1993). Gamble et al. (1987) introduced produced water at a concentration equivalent to a 400-500 fold dilution of produced water (expected in 0.5-1.0 km from the Auk and Forties platforms in the North Sea) into 300 m^3 mesocosm tanks containing natural assemblages of phytoplankton, zooplankton, and larval fish. Bacterial biomass increased, although phytoplankton production and larval fish survival were unaffected in the produced water-dosed containers. Early life stages of copepods, however, were sensitive to the produced water and exhibited a high mortality. Decreases in zooplankton abundance resulted in an increase in the standing stock of phytoplankton and a reduction in the growth rates of fish larvae. In other mesocosm studies summarized by Stephenson et al. (1994), larval mollusks and polychaete worms were adversely affected by produced water exposure. The mesocosm studies demonstrate that low concentrations of produced water may have subtle effects on marine planktonic communities. It is noted, however, that mesocosm studies represent conservative, worst-case exposure scenarios because produced water chemicals in the mesocosm enclosures do not degrade and disappear as rapidly as they would in well-mixed ocean environments.

5.4.4 Bioaccumulation, Biomarkers, and Exposure

Bioaccumulation is the uptake and retention of a bioavailable chemical from any one of or all possible external sources (e.g. water, food, substrate, and air) (Neff, 2002). Marine animals near a produced water discharge may bioaccumulate metals, phenols, and hydrocarbons from the ambient water or from their food. Biomarkers are biochemical, physiological, or histological changes in an organism that result from exposure to specific chemicals in their water or food (Forbes et al., 2006). Biomarkers typically are not direct indicators of harmful effects caused by exposure, but they can be used as early warnings of possible risk to an exposed organism, with the understanding that histopathological changes can be linked to harmful effects to a greater degree than for instance a slight biochemical or immune response. The most useful biomarkers respond to a single or small group of chemical contaminants and, so, can be used as evidence of exposure to a particular class of chemicals. For example, any of several measures of the induction (i.e. increase in activity) of the enzyme system, cytochrome P450 mixed-function oxygenase, can be used as evidence of exposure to PAHs, polychlorinated biphenyls (PCBs), and any of several other chlorinated hydrocarbon pesticides.

Produced water can be a source of PAHs in waters and sediments around oil development sites. The effect of PAHs has been determined in numerous studies conducted in the laboratory and field studies using endpoints based on biochemical, histopathological, immunological, genetic, reproductive, and developmental parameters (Payne et al., 2003). Concentrations of PAH were measured in the water column and in blue mussels (*Mytilus edulis*) deployed at different distances from production platforms in the Norwegian sector of the North Sea (Johnsen et al., 1998; Røe Utvik et al,. 1999; Durell et al., 2006 ; Neff et al., 2006). Direct measurements of PAH in the water gave inconsistent results, because concentrations were too low and variable. The mussels, however, did bioaccumulate PAHs from the water, even though concentrations decreased with distance down-current from the platforms.

Polycyclic aromatic hydrocarbon residues in mussel tissues were used to estimate PAH concentrations in surface waters (Neff and Burns, 1996). Surface water total PAH concentrations ranged from 0.025-0.35 μg L^{-1} at 1 km from the platform discharge centre, reaching background levels of 0.004-0.008 μg L^{-1} at 5-10 km from the discharge centre. This was equivalent to approximately a 100,000 fold dilution of the PAH concentration. Dilution modeling exhibited that most of the produced water plume was restricted to the upper 15-20 m of the water column and that dilution was very rapid. The DREAM model predicted that the concentrations of PAH and other chemicals in the produced water plume exhibited wide cyclic concentration variations due to tidal and wind-driven current flows. Because of rapid dilution and fluctuating water-column concentrations, the model predicted that potentially toxic concentrations and contact times of PAH would not occur even in the near-field.

Børseth and Tollefsen (2004) monitored bioaccumulation and biomarker responses in mussels and Atlantic cod held in cages in the vicinity of the Troll B Platform, Norway. Cages were deployed for six weeks both inside (500 and 1,000 m from the discharge centre) and outside the zone of expected influence of the produced water plume. The investigators found no significant difference in levels of plasma vitellogenin, an indicator of exposure to endocrine-disrupting chemicals, in male cod from exposed and control sites. No significant differences were detected in ethoxyresorufin-*o*-deethylase (EROD) activity, a biomarker of exposure to chemicals, including PAHs, which induced the cytochrome P450 mixed function oxygenase enzyme system, in livers of fish of exposed and control sites. This indicated little or no exposure to PAHs. Levels of PAH metabolites in cod bile were also low, confirming the low level exposure to PAHs. Furthermore, concentrations of naphthalene metabolites in cod bile decreased with distance from the platform, indicating that the low-level exposure to PAHs was likely the result of the produced water discharge. Other biomarkers showed little or no evidence that the cod were affected by exposure to chemicals from the produced water plume.

Børseth and Tollefsen (2004) found that concentrations of metals and PAHs in soft tissues of caged mussels correlated well with distance from the produced water discharge centre, with the highest body burdens in mussels observed closest to the platform. The PAH assemblage in mussel tissues was dominated by alkyl homologues of naphthalene, phenanthrene, and dibenzothiophene, which suggests exposure to PAHs from the produced water discharge. Biomarker responses in the mussels provided equivocal evidence of exposure to produced water chemicals. The authors concluded that mussels and cod deployed near a produced water discharge likely were exposed to low concentrations of produced water chemicals, below levels that might represent a health risk to water-column organisms.

All three developers on the Grand Banks (i.e. Petro-Canada, Husky, and Hibernia) are carrying out health effect studies, such as condition index, gross pathology, liver and gill histopathology, haematology, and induction of MFO enzymes, on American plaice (*Hippoclossoides platessoides*) collected in the vicinity of their development sites. American plaice, an important commercial fish species on the Grand Banks, is presently undergoing recovery and was identified as the species of primary interest for studies on fish health early on in the development of EEM programs (DeBlois et al., 2005; Husky Energy, 2005). American plaice undergoes feeding related movements into the water column in addition to its tendency for sediment contact (Beamish, 1966; Pitt, 1967). Among other favourable characteristics, this enhances its potential for ‘integrating’ contaminant exposure via the water column through both feeding and water contact, as well as through interaction with sediments.

Studies carried out on American plaice at the Terra-Nova development site for a number of years, both before and after release of produced water, have noted that the over all health of American plaice collected in the vicinity of the development site is similar to the health of American plaice collected at distal control sites (Mathieu et al., 2005; Mathieu et al., 2011). Although studies have been of shorter duration, similar results have been observed at the Husky and Hibernia development sites through their EEM programs. Elevated MFO enzyme levels, however, have been noted in fish larvae collected downstream of the Hibernia field (Payne et al., 2003). Induction may only be occurring near the rig site with the induced larvae being transported downstream with currents. Nevertheless, the observation on MFO induction in fish larvae is of interest since it has been correlated with larval mortality (Carls et al., 2005).

Sturve et al. (2006) exposed juvenile Atlantic cod to North Sea oil, nonylphenol, and a combination of the North Sea oil and an alkylphenol mixture in a flow-through system. A suite of hepatic biomarkers were monitored. While exposure to North Sea oil resulted in strong induction of CYP1A protein levels and EROD activities, the corresponding exposure to nonylphenol resulted in decreased CYP1A levels and EROD activities. As a result, nonylphenol appeared to down-regulate CYP1A expression in Atlantic cod. Meier et al. (2007) described the effects of alkylphenols (APs) on the reproductive potential of first-time spawning Atlantic cod. In laboratory studies, cod were fed with feed paste containing a mixture of four reference alkylphenols at a range of concentrations for 1 or 5 weeks. Results demonstrated that AP-exposed female fish had impaired oocyte development, reduced estrogen levels, and a delayed estimated time of spawning of 17-28 days. Male AP-exposed fish had impaired testicular development, with an increase in the amount of spermatogonia and reduction in the amount of spermatozoa present. From the results of the laboratory studies, the Meier et al. (2007) concluded that APs associated with produced water discharge may have a negative impact on the overall reproductive fitness of cod populations. As toxic effects are directly linked to dosage and exposure time, the ecological significance of laboratory biomarker studies remains uncertain, since factors such as fish movement and contaminant uptake and elimination are not considered, as well, alkylphenol concentrations in samples of sea water near platforms are typically below the limits of detection.

Andrews et al. (2009) recently conducted chronic toxicity studies on cunners and juvenile cod exposed to produced water from the Hibernia field on the Grand Banks. The health effect indicators studied included fish and organ condition, visible skin and organ lesions, levels of MFO enzymes, haematology (differential white blood cell counts), a variety of histological indices in liver and gills, and vitellogenin (a biomarker for ‘oestrogenic’ endocrine disruption). The fish were dosed every 2-3 days for 6-8 hours with 1-2 ppt of produced water. Exposures were carried out for a 3 month period. No changes in various indices were noted with the exception of red and white pulp in the spleen, which is associated with red and white blood cells respectively. Lymphocyte levels were also depressed in the blood (Payne et al., 2005). Similar results were obtained with juvenile codfish with the exception of haematological effects, which unlike cunner were, not observed in cod. Elevated levels of MFO and vitellogenin, however, were recorded in the exposed fish (Payne et al., 2005). A chronic toxicity study was carried out with cunner exposed to barite, which can form from the interaction of barium in produced water with seawater. Barite is also a major constituent in drilling muds. Cunners were exposed on a weekly basis for 40 weeks to 200 g clouds of insoluble barite in a 1800 L tank. Barite, which accumulated on the bottom of the tank, was not removed. Fish survival and indices of fish health, as assessed by fish and organ condition and detailed histological studies on liver, gill and kidney tissue, did not differ between the control and experimental groups. Elevated levels of MFO enzyme activity, however, was found in the exposed fish (Andrews et al., 2007).

As part of the Biological Effects of Contaminants in Pelagic Ecosystems (BECPELAG) Program, bioaccumulation and several biomarkers were measured in wild and caged marine animals along a transect away from the Statfjord Platform in the North Sea (Hylland et al., 2006). Produced water discharge was 74,100 m^3 d^{-1} from three platforms in the Statfjord field (Durell et al., 2006), which is among the highest discharge rates of any offshore field in the world (Figure 10). Førlin and Hylland (2006) measured EROD activity and bile metabolites in juvenile cod caged at several distances down-current from one of the discharges. There were no significant trends in EROD activity in male and female cod with distance from the discharge, though there was a trend for EROD activity in female cod to increase with distance from the discharge (Figure 11a). Concentrations of alkyl naphthalene metabolites, abundant in produced water, in fish bile were highest in cod near the platform and decreased with distance from the platform

(Figure 11b). There were no distance trends in concentrations of other PAH metabolites in the cod bile. The authors concluded that the cod were exposed to low levels of PAHs from the produced water discharges, although exposure levels were well below those that would pose a health risk to fish living proximal to the platforms. In addition to the Førlin and Hylland (2006) study, a number of other studies have noted induction in wild fish and caged fish: North Sea (e.g. Davies et al., 1984b; Stagg et al., 1995; Abrahamson et al., 2008) and coastal Australia (e.g. King et al., 2005; Zhu et al., 2008). Elevated levels of MFO enzymes have been observed in fish larvae collected around some platforms in the North Sea (Stagg and MacIntosh, 1996). A slight elevation of MFO enzymes in American Plaice has been observed on occasion around some sites on the Grand Banks (Mathieu et al., 2011).

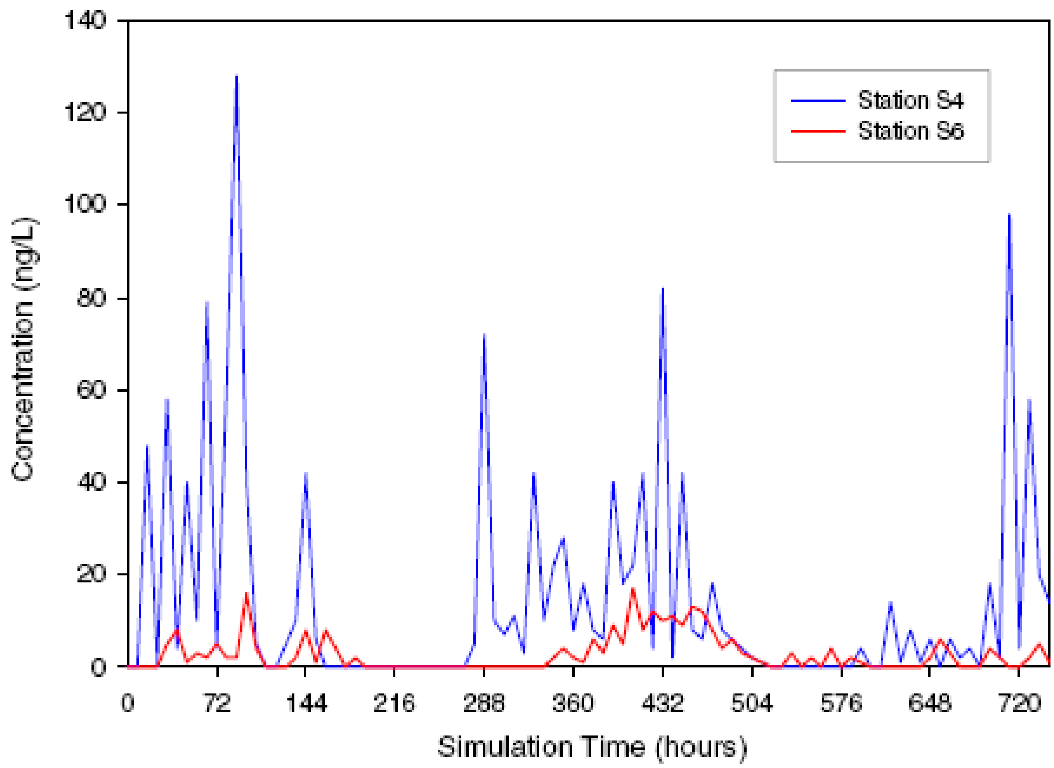
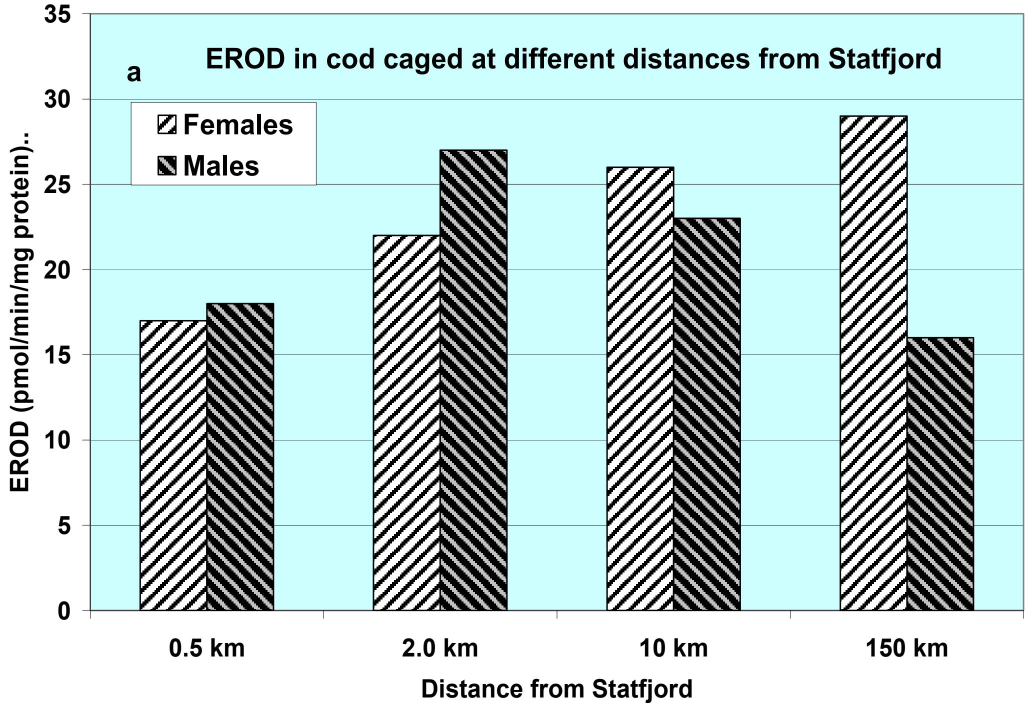


Figure 10. *Time series of modeled total PAH concentrations (naphthalenes through 5-ring PAH) at Station S4 and Station S6 down current from a produced water discharge centre, Ekofisk Field (Durell et al., 2006).*



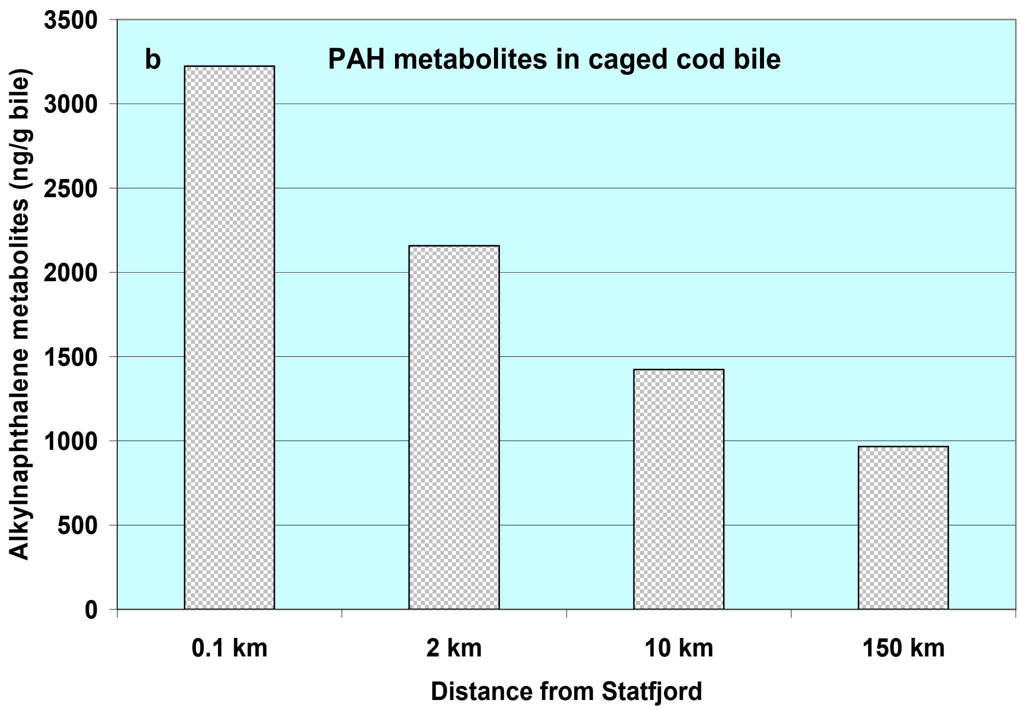


Figure 11. *EROD activity (a) and PAH metabolites (b) in caged cod at different distances from the Statfjord Oil Field (Førlin and Hylland, 2006).*

A study by Hamoutene et al. (2011) investigated the effects of produced water cod immunity, feeding, and general metabolism by exposing fish to diluted produced water at concentrations of 0, 100, and 200 mg L^{-1} for 76 days. No significant differences were observed in weight gain or food intake. Similarly, serum metabolites, whole blood fatty acid percentages, and mRNA expression of a brain appetite-regulating factor (i.e. cocaine and amphetamine regulated transcript) remained unchanged between groups. Other than an irritant-induced alteration in gill cells found in treated cod, resting immunity and stress response were not affected by produced water. Catalase and lactate dehydrogenase changes in activities were recorded in livers but not in gills, suggesting an effect on oxidative metabolism subsequent to hepatic detoxification processes. To evaluate potential effects of produced water discharges on cod immunity, fish from the three groups were challenged by injection of *Aeromonas salmonicida* lipopolysaccharides (LPS) at the end of exposure. The LPS injection affected respiratory burst activity of head-kidney cells and circulating white blood cell ratios, which increased serum cortisol in all groups. The most pronounced changes were seen in the group exposed to the highest dose of produced water of 200 mg L^{-1}.

Pérez-Casanova et al. (2010) investigated the effects of chronic exposure to produced water on some aspects of juvenile Atlantic cod immunity, stress response, and growth by intermittently exposing fish to 0, 100 or 200 mg L^{-1} of produced water for 22 weeks. The respiratory burst (RB) of circulating leukocytes was significantly elevated in the 100 mg L^{-1} treatment, while the RB of head-kidney leukocytes was significantly decreased in both the 100 and 200 mg L^{-1} treatments. Significant up-regulation of the mRNA expression of β-2-microglobulin, immunoglobulin-M light chain, and interleukins-1-β and -8 was observed in the 200 mg L^{-1} treatment and the down-regulation of interferon stimulated gene 15 was obvious for both the 100 and 200 mg L^{-1} treatments. No significant effects of produced water were observed on growth, hepatosomatic index, condition factor, or plasma cortisol. With most immune parameters being stimulated, the results support the need for further studies to determine if chronic exposure to environmentally-relevant concentrations of produced water could cause modulations of the immune system of juvenile Atlantic cod, which results in an energetic cost to the fish that may be detrimental. In terms of benthic organisms, a preliminary study was carried out with scallops in which they were exposed to 2 ppt of produced water from the Hibernia field every 2 days over a period of approximately four months. No differences in mortality or condition indices were observed between the control and exposed treatment groups. A similar long-term study using mussels also found no effect of produced water on mortality (pers comm. J. Payne, DFO).

5.5 ECOLOGICAL RISK OF PRODUCED WATER DISCHARGES

The DREAM model has been adopted for determination of contaminant effects of produced water discharges and the level of risk it may impose on the marine ecosystem. Risk assessments can be conducted under the DREAM model using two approaches: 1) a body burden-related risk assessment model, and 2) the determination of an environmental impact factor. The first case is based on the ratio of a predicted environmental concentration (PEC) to a predicted no-effect concentration (PNEC), known as the PEC/PNEC ratio (Karman and Reerink, 1998). This can be followed up with the calculation of an environmental impact factor (EIF), which can be used for produced water impact reduction, management, and regulation (Johnsen et al., 2000).

The Norwegian oil and gas industry advocates ecological risk assessment as the basis for managing produced water discharges to the North Sea. Neff et al. (2006) compared estimates of ecological risks of PAHs in produced water to water-column communities, based on data of hydrocarbon residues in soft tissues of blue mussels deployed for a month near offshore platforms and based on predictions of DREAM. The study was performed near produced water discharges of the Tampen and Ekofisk Regions, Norwegian Sector of the North Sea. Because PAHs are considered the most important contributors to the ecological hazard posed by produced water discharges, comparisons focused on this group of compounds. The mussel approach is based on PECs of individual PAHs, estimated from PAH residues in mussels following deployment for a month near several produced water discharges, as well as on PNECs based on the log of the geometric mean of the acute toxicity value for a PAH regressed against its octanol/water coefficient (K_{ow}). The octanol/water partition coefficient is the concentration of a chemical in octanol (an organic solvent that is used as a surrogate for natural organic matter) divided by its concentration in water. A substance with a high K_{ow} is of environmental concern because it is readily absorbed by sediment and the fatty tissue of living organisms.

In the DREAM method, PECs for three PAH fractions were estimated in the 3D area around the produced water discharge centre. Predicted no-effect concentrations for each fraction were based on the chronic toxicity of a representative PAH from each fraction divided by an application factor to account for uncertainty in the chronic value.

The mussel method gives much lower estimates of ecological risk than the DREAM method (Figure 12). The differences are caused by the much lower PNECs used in DREAM than derived from the regression model approach, as well as by the lower concentrations of aqueous PAHs predicted by DREAM than estimated from PAH residues in mussel tissues. The two methods, however, ranked stations at different distances from produced water discharges in the same order and both identified two- and three-ring PAHs as the main contributors to the ecological risk of produced water discharges (Neff et al., 2006). Neither method identified a significant ecological risk of PAHs in the upper water column of the oil fields. The DREAM model may produce an overly conservative estimate of ecological risk of produced water discharges to the North Sea, because of the extremely conservative PNEC values for the various PAH fractions.

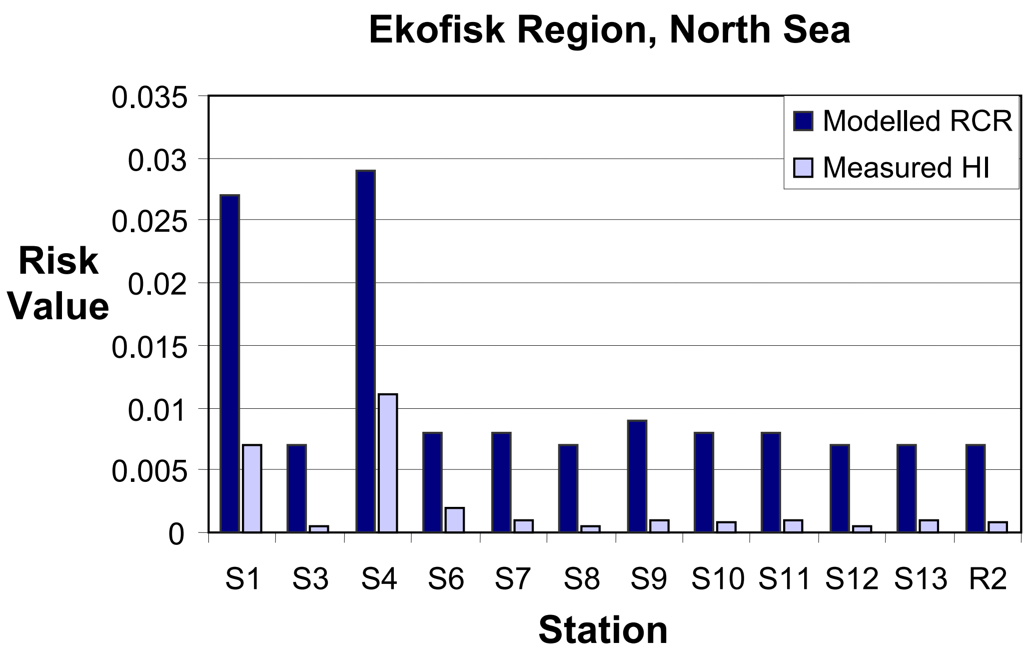


Figure 12. *Risk characterization ratios (RCRs) and hazard indices (HIs) for total PAHs. The RCR is the ratio of predicted environmental concentration (PEC) from DREAM to the predicted no effect concentration (PNEC) for select PAHs. The HI is the sum of ratios of measured PAH concentration in water to a modeled PNEC value. A risk value of 1 or higher indicates a possible risk to the health of marine organisms from the site (Neff et al., 2006).*

Myhre et al. (2004) studied the reproductive effects of alkylphenols (APs) in produced waters on fish stocks of the North Sea using the DREAM model. The fish stock distributions (i.e. based on cod, pollock, and haddock data of the international bottom trawl surveys IBTS database) and a PNEC for APs of 4 ng L^{-1} were used as base data for the calculations of effects from the combined produced water discharges of three major Norwegian oil fields (Tampen, Ekofisk, and Sleipner). The total amount of >C4 APs discharged from all the oil production units was estimated to be 25.6 kg d^{-1}, dissolved in 364.300 m^3 d^{-1} produced water. The conclusion of the risk assessment was that "the overall results of the simulations with DREAM show that there is no significant risk potential" (Myhre et al., 2004). In the computer simulation, fish were represented mathematically as 'fish particles' which moved in and out of the effluent plume. In conclusion, there were no fish particles that accumulated APs above the critical body burden of 2 μg kg^{-1} in any of the simulations. The highest accumulated body burden in any of the fish particles was 0.09 μg kg^{-1}.

The accuracy of contaminant risk assessment models is dependent on the identification and quantification of the various chemicals that induce toxic effects. Unfortunately, the causative agents of toxicity in the most toxic produced waters are not known. Toxic responses may be linked to the extremely high total dissolved solids (salinity) concentrations, altered ratios of major seawater ions, and elevated concentrations of ammonia in some Gulf of Mexico produced waters (Moffitt et al., 1992). Salinity and ion ratios quickly return to those in sea water following ocean discharge of produced water, while ammonia evaporates or degrades rapidly. As such, the contaminants of concern found in produced water discharge streams rarely cause acute toxicity responses in the field.

Chemical kinetic reactions that occur following the release of produced water in an anoxic state into the open ocean have been found to alter the toxicity of produced water over time following its discharge (Lee et al., 2005b). The significance of this process is clearly illustrated in controlled dose-response experiments using natural microbial populations as the test organisms (Figure 13). A typical toxicity dose-response curve, with initial increase in productivity at low concentrations of produced water due to addition of nutrients followed by inhibition above a threshold value, is observed with fresh produced water. Following aeration for 44 hours, to simulate equilibration in the ocean following discharge, additions of the produced water over the same concentration gradient elicited a stimulatory response. The difference is attributed to the loss of low molecular weight hydrocarbons and precipitation of hydrolysis metals that may have sequestered toxic metals associated with sample aeration. The results imply that accurate comparisons of toxicological studies with similar end-points (e.g. LC_{50}) can not be made unless sample collection and handling protocols are standardized prior to toxicity testing.

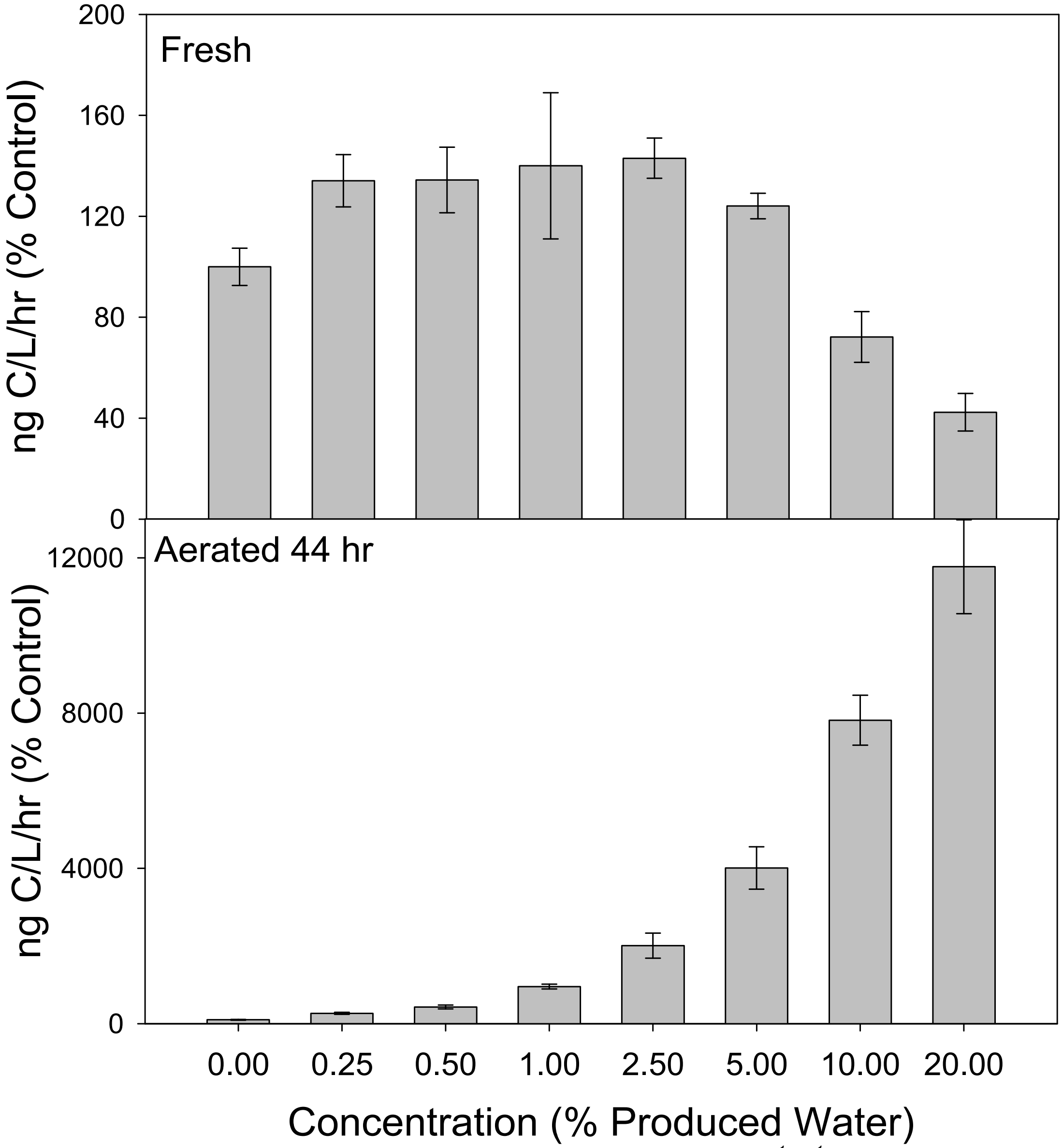


Figure 13. *Dose-response of microbial carbon fixation rate (ng C L^{-1} h^{-1}) versus concentration of produced water. Produced water recovered from the Terra Nova FPSO on the Grand Banks was added to natural seawater and microbial productivity was measured by ^{3}H-thymidine uptake into DNA. Identical produced water samples were evaluated under identical conditions: immediately after collection (fresh) and after being aerated 44 hours.*

In a modeling study to assess potential perturbations in food web structure and energy flow due to the discharge of produced water, Rivkin et al. (2000) predicted significant increases in productivity and sedimentation fluxes over large spatial domains, in response to produced water derived ammonia and dissolved organic carbon. At typical discharge rates, however, the effects of produced water discharges may be limited. Yeung et al. (2010) monitored changes in indigenous microbial community structures in response to produced water discharges from an offshore platform on the Grand Banks by denaturing gradient gel electrophoresis (DGGE). Results demonstrated that the production water did not have a detectable effect on microbial community structure in the surrounding water. Cluster analysis further showed a greater than 90% similarity for all near surface water (2 m) samples (at 2 m), 86% similarity for all the 50 m and near bottom samples, and 78% similarity for the whole water column from top to bottom across a 50 km range. The results were based on two consecutive yearly sampling events.

There were clear differences in the composition of the bacterial communities in the produced water compared to seawater near the production platform (approximately 50% similarity), indicating that the effect from produced water may be restricted to the region immediately adjacent to the platform. Members of the bacteria genus *Thermoanaerobacter* and of the Archaea genera *Thermococcus* and *Archaeoglobus* were identified as significant components of the produced water. These particular signature microorganisms could become useful markers to monitor the dispersion of produced water into the surrounding ocean. In general, bacteria have very short generation times and respond rapidly to environmental changes. Bacteria use in EEM programs has been recommended, since they are involved in primary processes including the production of carbon, nutrient cycling, and the biodegradation and/or biotransformation of contaminants (Lee and Tay, 1998; Wells et al., 1998). Studies by Anderson et al. (2000), with naturally occurring bacteria, indicated the potential for produced water to both inhibit (i.e. short term exposure at high concentrations) and enhance bacterial growth (i.e. lower concentrations over an extended period).

5.6. FURTHER RESEARCH NEEDS

There have been many advances in scientific knowledge during the past decade regarding the composition, environmental fate, and biological effects of produced water discharges into the ocean. The general consensus of the 2007 International Produced Water Conference was that any effects of produced water on individual development sites in the open ocean are likely to be minor (Lee and Neff, 2009). The toxicity threshold limits for acute effects are not likely to occur beyond the immediate vicinity of the discharge pipe due to the effectiveness of natural dispersion processes driven by tides and currents. Unresolved questions, however, regarding aspects of produced water composition, its fate, and potential effect on the ecosystem remain. The effects of chronic toxicity may only become evident after monitoring several life stages or generations. It is important to acknowledge the consequences of long-term effects from offshore oil and gas facilities that may have a 15-20 year life-cycle. Furthermore, cumulative effects linked to future expansion of production operations must be considered.

It is evident that additional information is needed to improve the accuracy of existing risk assessment models for produced water discharge. Multidisciplinary scientific studies are needed under an ecosystem based management (EBM) approach to provide information on the environmental fates (i.e. dispersion, precipitation, and biological and abiotic transformations) and effects of chronic, low-level exposures to the different chemicals found in produced waters. Numerical models also need to be improved to better predict the fate and effects of chemical constituents found in produced water plumes that are rapidly dispersed. There is a need, however, to develop improved sample recovery and analytical techniques to support model validation needs. At present, many of the potential contaminants of concern in produced water cannot be detected in the open ocean environment with standard analytical protocols. The future development of high efficiency, cost-effective produced water treatment technologies is dependent on the identification and monitoring of the primary target constituents of environmental concern found in produced water (e.g. PAH, phenols, metals).

Interpretation of ecological risk from biological effect studies based on biomarker techniques remains a challenge. Biomarkers may be used to indicate that: 1) an organism has been exposed to a specific chemical or group of chemicals; 2) an organism is affected by a contaminant and responding to it; or 3) the organism has been damaged. According to Gray (2002) in an editorial comment entitled ‘Perceived and Real Risks: Produced Water from Oil Extraction”, it remains uncertain of the risk of produced water discharge to populations found in the field. Regarding discussion on the use of biomarkers and their ‘translation’ into population level effects, Payne et al. (1987) noted over 20 years ago that there is little or no conceptual basis for carrying out such ‘translations.’ It is now commonly accepted, however, that biomarkers are especially valuable for surveillance monitoring, diagnosing unanticipated health effects, and providing information on their geographical reach (Payne, 2007). As such, they are a valuable tool for the risk-management toolbox vis-à-vis scoping potential areas of concern or the dimension of the problem and assessing impacts on fish. It is also important to note that all bioindicators cannot be treated equally in carrying out risk analysis. For instance, a variety of histopathological lesions appearing in the liver of fish would not be afforded the same adverse health status risk factor, as a small change in a sensitive enzyme activity. Even a sensitive response such as MFO enzyme induction has now been associated with a variety of metabolic, cellular, organ, and developmental disturbances (Mathieu et al., 2011 and reference therein) and, as such is, more than an indicator of contaminant exposure.

For a comprehensive protection plan, there is a need to support the development of improved monitoring protocols to provide early warning of any potential problems related to sediment and water quality (e.g. primary productivity), fish quality and fish health. Development of real-time monitoring systems (i.e. contaminant specific sensors and data-transfer technologies) may enhance our capacity to manage the ocean and its living resources. In consideration of natural perturbations currently occurring in the ocean (e.g. climate change) and the impacts potentially associated with other marine users (e.g. marine transport, fisheries, etc.), an ecosystem based integrated management approach must be taken to fully evaluate the risks of produced water discharge into the ocean. In addition, alternative approaches to produced water management may also be considered. As an example, an alternative to ocean discharge of treated produced water is underground injection. The feasibility of this practice, however, at offshore installations is dependent on a number of site-specific factors, including access to a suitable disposal formation, chemical interactions that may result in precipitates that may plug the receiving formation, and cost. Last, the net environmental benefit of reinjection must be considered as, on the basis of energy requirements for injection, it is estimated that 2.6-4.3 g of carbon dioxide is emitted per liter of produced water reinjected into a sub-surface well (Shaw et al., 1999).

6.0 NATURAL SEEPAGE, SPILLS, BLOWOUTS, AND MALFUNCTIONS

6.1 NATURAL SEEPAGE

A significant amount of crude oil is discharged each year from 'natural seeps'; areas from which liquid and gaseous hydrocarbons leak out of the ground into the marine environment. The observation of oil on the sea surface that has leaked from sub-marine oil reservoirs has been recorded throughout history. The locations of natural seeps have been used by geologists in their quest to locate unique geological structures and petroleum hydrocarbon reservoirs. In the North Atlantic Ocean, the occurrence of natural oil seeps has been well documented, from the Canadian Arctic (e.g. Grant et al., 1986) to the Gulf of Mexico (e.g. MacDonald, 1998). Levy and Lee (1988) reported the natural seepage of hydrocarbons in the Georges Bank region and postulated its potential ecological significance, as a carbon source, to the fisheries. The two most recent international reviews on the global input of oil into the sea (NRC, 2003b; GESAMP, 2007) still base their estimates of 4.2-14 million bbl annually of natural seepage on the values provided by Wilson et al. (1974) and Kvenvolden and Harbaugh (1983). Kvenvolden and Cooper (2003) stated "natural oil seeps may be the single most important source of oil that enters the ocean, exceeding each of the various sources of crude oil that enters the ocean through its exploitation by humankind". The most recent analysis of U.S. oil spillage conducted by the American Petroleum Institute, it was reported that from 1998 to 2007 production-related petroleum spillage was less than 0.9% of the amount discharged from natural seeps.

6.2 SPILLS AND BLOWOUTS

In addition to the potential impacts of routine exploration and production activities, there are some events (e.g. oil spills and blowouts) that, although having a low probability of occurrence, have a much greater risk to the Georges Bank ecosystem. Spills, blowouts, and malfunctions are types of detrimental accidents to the environment that may occur during any offshore oil and gas exploration activities. As defined by the U.S. National Oceanic and Atmospheric Administration, a blowout occurs when operators of a drilling rig are unable to control the flow of oil, gas, or other fluids from the well and it is released into the underground formation, marine environment, and/or atmosphere. The U.S. Minerals Management Service also includes flow through a diverter or uncontrolled flow resulting from failure of surface equipment or procedures under its definition of a blowout (API, 2009). Blowouts can involve continuous discharge of petroleum gas into the atmosphere and/or crude oil into surrounding waters. Blowouts may not lead to a significant loss of hydrocarbons, as they often seal naturally and cease flowing within a matter of hours or days. In addition to the routine use of drilling fluids to control the underground pressures when drilling, offshore oil rigs have blowout prevention (BOP) systems to control the well when there is an influx of pressurized gas or oil during drilling. The BOP seals off the well and routes the wellbore fluids to specialized controlling equipment.

The rate of blowouts is very low in the oil exploration and development industry. For example, in the Gulf of Mexico from 1979 through to 1998 there were 19,821 wells drilled, with 118 wells resulting in uncontrolled flows or blowouts. This results in a 0.6% occurrence rate of blowouts for the region during this time. Furthermore, the majority of reported events involved the diversion of gas. Due to differences in defining a blowout, API (2009) states that there has only been 17 marine well blowouts in the U.S. since

1964, which resulted in a total of 249,000 bbl spilled. The largest of these blowouts occurred from the Alpha Well 21 off Santa Barbara, California, which spilled 100,000 bbl. In general, spill volumes have been small, with 50% of the well blowouts involving 400 bbl of oil or less.

During offshore petroleum exploration, there is usually no bulk storage or transfer of oil or gas, thus the risks and impacts of an oil spill are no more than that associated with marine shipping. With an increase in exploration and production operations, the incidence of marine oil spills would be expected to arise due to the escalation of marine traffic associated with construction and supply operations, including the potential use of sub-surface pipelines for oil and gas transportation. With advances in technology and improved operational safety measures, data from U.S. operations has indicated a decrease in combined spill volumes from offshore supply vessels, pipelines, and platforms, from 30,400 bbl y^{-1} (from 1969-1977) to 3,900 bbl y^{-1} (from 1998-2007), resulting in an 87% reduction in the average annual spill volume since the 1970s (API, 2009). Although the potential for harmful effects from an accidental petroleum discharge, such as a large oil spill, may be large, the probability of such an event occurring is low. Historical spill and blowout events support the prediction of probability of spill and blowout occurrence for new oil and gas drilling exploration and production activities (Table 10). Studies on marine oil pollution indicate that accidental petroleum discharges from platforms contribute 0.07% of the total petroleum input to the world's oceans (NRC, 2003b).

Table 10. *Historical large oil spills in barrels (bbl) from offshore well blowouts (Oil Spill Intelligence Report database).*

Offshore Region	Spill Size (bbl)	Year	Type of Activity
Mexico (Ixtoc 1)	3,000,000	1979	Exploratory Drilling
Dubai	2,000,000	1973	Development Drilling
Mexico	247,000	1986	Work over
Nigeria	200,000	1980	Development Drilling
North Sea/Norway	158,000	1977	Work over
Iran	100,000	1980	Development Drilling
U.S.A., Santa Barbara	77,000	1969	Production
Saudi Arabia	60,000	1980	Exploratory Drilling
Mexico	56,000	1987	Exploratory Drilling
U.S.A., S. Timbalier 26	53,000	1970	Wireline
U.S.A., Main Pass 41	30,000	1970	Production
U.S.A., Timbalier Bay/Greenhill	11,500	1992	Production
Trinidad Development	10,000	1973	Drilling

Worldwide, there have been five oil spills greater than 150,000 bbl in size in the history of offshore drilling, two of which occurred during development drilling and two of that occurred during production activities. The fifth, the Ixtoc 1 oil well blowout in the Bay of Campeche, Mexico, 1979, occurred during exploration drilling. The CNSOPB has predicted a 1-in-1800 chance per year of having any sort of deep water blowout off the Continental Shelf during exploratory drilling, with the probability of shallow water gas blowouts without a release of oil having a three to four times higher possibility of occurrence (Hurley and Ellis, 2004). For operations in the offshore of Nova Scotia and Newfoundland and Labrador, one exploration gas well blowout, Uniake G-72, occurred off Sable Island in 1984 (Shell Canada Resources, 1984; Boudreau et al., 1999). Hurley and Ellis (2004) provided expected frequencies of exploration well blowouts based on the historical frequency of blowouts per wells drilled. They calculated that the predicted frequency of an extremely large blowout (>150,000 bbl) was one every 17,500 years. The frequency of a very large spill (>10,000 bbl) was one every 5800 years, and of a large spill (>1000 bbl) was one every 4400 years.

Oil spills other than from blowouts can occur during drilling and production activities due to an increased amount of marine traffic in the region. These include spills of diesel oil or lubricating oil on the platforms, spills from transfer operations, and spills from similar accidents involving the handling of oil that is needed to run operations. Based on statistical data from the offshore of Nova Scotia, presented by Hurley and Ellis (2004), the highest frequencies of oil spills are for the smaller, platform-based spills. Spills that are less than one barrel in size may occur once every two years. Oil spills during exploration, which are larger than one barrel but less than 50 bbl, have an approximately 1-in-10 or 1-in-20 chance of occurrence per year. From January, 1994, to August, 2002, five oil spills, including spills of synthetic oil-based drilling mud, occurred during exploration drilling on the Scotia Shelf that resulted from exploration and production activities (CNSOPB, 2010). In the Newfoundland and Labrador offshore petroleum region, in 2004, a 1000 bbl crude oil spill occurred from the Terra Nova FPSO due to equipment failure. While there was 296 spill events associated with exploration and production activities in the Newfoundland and Labrador region from 1997 and 2007, the total volume of hydrocarbons released was less than 1200 bbl (CNLOPB, 2010).

Concern has been raised in Atlantic Canada regarding the accuracy of predictions generated under the existing EIA process by Fraser and Ellis (2008), following the comparison of oil spill frequency predictions of small batch spills of hydrocarbons and synthetic hydrocarbons less than 50 bbl in volume to observed data for three projects. The authors noted that three projects exceeded their predicted frequencies, with the ratio for actual to predicted values greater than six for two of the projects. The implementation of a feed-back management process that responds to the results of EIA follow-up monitoring pursuant to the *Canadian Environmental Assessment Act* were recommended. In response, the CNSOPB concluded that their original EIA prediction for the Sable Offshore Energy Project was largely accurate, of 0.5 spills per year, when compared to the observed spill occurrence of 0.57 spills per year. For the other three sites in Atlantic Canada, the CNLOPB noted that the forecasts of spill occurrence for the Terra Nova and White Rose Projects were developed using the best data available at the time. In both cases the regulatory boards noted that they follow-up on all spills and that local spill occurrence information is used in the preparation of future EIAs.

Scientific studies on the transport and effects of petroleum hydrocarbons from oil spills and blowouts have been ongoing for decades. When crude oils and their refined products (including heavy fuel oils) are released to the marine environment, oil weathering processes occur. These processes include spreading, evaporation, dissolution, and dispersion of whole-oil droplets into the water column, water-in-oil emulsification, photooxidation, microbial degradation, uptake by organisms, adsorption onto suspended particulate material, sinking, and sedimentation. The rate and extent of oil weathering processes are dependent on many factors, including: oil types; mixing energy; temperature; and salinity of the marine environment. As a general rule, most fresh oils released to the sea surface spread in a few hours to reach an average thickness of 0.1 mm (ITOPF, 2005). The speed of oil spreading is determined by many physicochemical factors, for example, light crude oil spreads much faster than heavy fuel oil. Wind can significantly increase the spreading rate of oil. A rough sea with high mixing energy will significantly enhance the rate of spreading.

Volatile organic compounds in the spilled oil are evaporated and dissolved into the atmosphere and surrounding waters immediately after the oil is released into the environment. In wave tank studies, Payne et al. (1991) observed that all compounds with vapour pressures greater than n-C11 were lost, likely by evaporation, in 9 days under spring and summer conditions. The evaporation of oil under winter conditions is typically slower, but eventually the oil will evaporate to approximately the same degree as it would if spilled on the water in summer. Evaporation/dissolution loss can be a significant weathering process for many light crude oils, which can lead up to 50-60% loss of the spilled oil mass. Since evaporation is a surface phenomenon, the evaporation rate can be significantly influenced by the oil film thickness as a result of spreading. The rate of evaporation is also increased by strong winds, rough seas, and higher air temperature. Under turbulent hydrodynamic regimes (e.g. breaking wave conditions), a surface oil slick can be entrained as small oil droplets that are dissipated and diluted into the water column (Delvigne and Sweeney, 1988; Fraser and Wicks, 1995; Lee and Stoffyn-Egli, 2001; Lee et al., 2003). This is a natural process that is beneficial to oil spill clean up, because oil in the form of small droplets has a much higher surface-to-volume ratio than an oil slick and, as a result, undergoes biodegradation at a much higher rate. Furthermore, the formation of oil droplets enhances interaction with suspended particulate material and other natural attenuation processes.

Water-in-oil emulsion, formed by dispersion inversion or by water drop entrainment, is a competing process with dispersion of oil. Emulsification increases the volume of the contaminant oil and also decreases its rate of biodegradation. The formation of stable water-in-oil emulsions is an important factor influencing the success of various oil spill cleanup processes, including containment, in situ burning (due to lowered combustibility of the emulsions), and chemical dispersion (with increased difficulty). Rapid emulsion formation diminishes rates of evaporation and dissolution and, therefore, more toxic lower and intermediate molecular aromatic compounds will be retained in the residual oil for a longer period. The rate and extent of the formation of water-in-oil emulsions are dependent on oil properties, turbulence energy level, and natural environmental conditions. Photo-oxidation may be an important transformation processes of petroleum products released into the marine environment (Garrett et al., 1998; Dutta and Harayama, 2000; Prince et al., 2003a). Aromatic compounds are particularly sensitive to photo-oxidation, whereas saturated compounds are more resistant (Garrett et al., 1998); large size and increasing alkyl substitution increase the sensitivity of aromatic compounds to photo-oxidation. In contrast, larger and more substituted compounds are more resistant to biodegradation (Prince, 1993; Prince and Clark, 2004). It has been demonstrated that, while natural microbial populations in seawater partially biodegraded crude oil when sufficient nutrients were supplied, pre-treatment with photo-oxidation increased the amount of crude-oil components susceptible to biodegradation. This lead to significantly increased biodegradation of crude oil (Dutta and Harayama, 2000). The photo-oxidation of petroleum products may increase toxicity (Shemer and Linden, 2007).

Biodegradation by microbial communities is the major process controlling the 'weathering' and eventual removal of oil entering the marine environment (Leahy and Colwell, 1990; Atlas and Bartha, 1992; Atlas, 1995). A large number of hydrocarbon degrading microorganisms have been isolated from a variety of marine environments

(ZoBell, 1973; Atlas, 1984; Leahy and Colwell, 1990). Recently, Prince et al. (2010) published a list of 181 genera of bacteria, 163 genera of filamentous fungi and yeast, and 22 genera of algae that are able to degrade hydrocarbons by growing them as carbon source. This is not surprising considering that marine microorganisms have long been exposed to petroleum hydrocarbons from natural seepages. Most crude oils are buoyant. When low molecular weight components have been removed by weathering or following interaction with suspended particulate material, however, residual fractions may sink. The relative amounts entering the various environmental compartments and the behaviour of the residual hydrocarbons depends on the type of event (e.g. platform blowout, seafloor blowout, or sea surface oil spill), composition and physical and chemical properties of the source hydrocarbon released, and environmental (e.g. wind, temperature, and salinity) and oceanographic conditions (e.g. waves, tides, and currents).

Spills of production chemicals used to ensure safety and to maintain production operations, such as monoethylene glycol (MEG), lubricating oil, diesel fuel, and hydraulic oil may also be spilled into the marine environment during offshore petroleum activities. Monoethylene glycol is an industrial anti-freeze that is injected into natural gas pipelines to prevent the formation of hydrates, or ‘ice plugs’, which can cause blockages (CNSOPB, 2008). As it is classified to be a low toxicity substance, the discharge of MEG has been approved in various jurisdictions. In 2006, a cracked pipe at the Sable Offshore Energy Project resulted in over 150 m^3 of MEG being released to the marine environment. No measurable impact was identified (CNSOPB, 2008).

6.3 OIL SPILL MODELING

A large number of oil spill models, which cover a range of capabilities from simple trajectory modeling to three-dimensional trajectory and fate modeling, are used globally. Reviews of some of the most widely used models have been reported by Huang (1983), Spaulding (1988), Fingas (1995), ASCE Task (1996), and Reed et al. (1999). Advances in oil spill models have improved the ability to predict oil spill trajectory and impacts in the event of an accidental release of petroleum hydrocarbons in the marine environment. The application of predictive numerical models can be used to support risk assessment studies and to provide critical guidance for clean-up operations.

Oil spill trajectory modelling for summer conditions on the Northeast Peak of Georges Bank suggest that oil would travel in one of two principal directions. If winds are light then trajectories would likely be influenced by the residual current and slicks would generally move to the south and southeast. In contrast, under storm conditions, surface water movement would be driven by the winds and, as such, oil would move in the direction of the prevailing wind. The distance between the Georges Bank moratorium area and the Canadian shoreline, as well as the residual current, greatly reduce the probability of an oil slick encountering the shorelines of Atlantic Canada. It would be expected that a large portion of the crude oil in a slick would evaporate and disperse during transit, thus, the probability of shoreline fouling is considered low.

6.3.1 Fate and Transport Models

The accuracy of oil spill fate and transport models is continuously being improved. For example, while historical oil spill models have focused on surface trajectory and weathering, recent research has begun to evaluate the fate and impacts of spills that are largely entrained in the water column, as a result of high mixing energy due to the application of a dispersant (French-McCay and Payne, 2001; Reed et al., 2004; Nazir et al., 2008). To provide support for spill response and evaluation of habitat damage, Nazir et al. (2008) studied the fate of spilled oil in the water column and sediments using a Fugacity-based multimedia model to determine the extent of the area of impact and to define the size of the water and sediment compartments. They found that the water compartment exhibits a rapid decrease of oil concentration after input of to the sea is stopped, but that the sediment is slower to respond. Varlamov et al. (1999) simulated the movement of spilled oil, after an incident involving the Russian tanker *Nakhodka* in the Sea of Japan, using a particle-based oil spill model coupled with a regional ocean circulation model. The simulation demonstrated that 91.5% of the simulated oil volume and 62% of the modeled oil droplets were found in the upper 30 m layer of the water column. The study also found that the estimations of volume of oil redistributed both at the sea surface and in deep water is sensitive to the parameters used for weathering processes and the size distribution of the initial oil droplets.

French-McCay and Payne (2001) have modeled fate of spilled oil, specifically focusing on water column concentrations, resulting from oil spills with and without the application of oil-spill dispersant. For a hypothetical spill of 1500 t of Louisiana light crude on the water surface 25 km from the entrance of Galveston Bay, the results demonstrated that the application of dispersant at 2-3 hours after the spill greatly increased the percentage of the aromatics that dissolved into the water. It was also found that increasing the amount of dispersant and applying it sooner reduced the surface area of the shoreline that is ultimately oiled. Much of the simulations of oil spills have been conducted on large scales and the effects of waves have been accounted for using empirical formulations (Lehr, 2001). Since waves play an important role in the dispersion of oil slicks in the water column, Boufadel et al. (2007) simulated the oil droplet transport associated with regular waves (waves of constant frequency and height). It was found that Stokes' drift was the major mechanism for horizontal transport. The light oil droplets were found to propagate faster, but spread less, than heavier oils. Tkalich and Chan (2002) developed a kinetic model that considered the dominant forces affecting droplet formation and vertical distribution to describe the vertical mixing of oil droplets in breaking waves. They combined properties of oil, waves, and the water column into a single mixing factor to describe the droplet mass kinetics in the upper part of the water column. Because the wave dissipation rate that determines oil droplet size and affects the subsequent kinetics was not examined in the Tkalich and Chan (2002) approach, Chen et al. (2008) subsequently proposed a method that addressed turbulent dissipation.

Traditional stochastic oil spill models have focused on the production of statistics for oil on the sea surface and along shorelines. Environmental risk assessment models, therefore, have tended to neglect potential risks to organisms in the water column and sediments. A stochastic model has been described by Skognes and Johansen (2004), with modules for statistical analysis of oil drift, spreading, and weathering with respect to sea surface impacts and eventual shoreline impacts, as well as with respect to subsurface oil concentration. A model application for a spill off the coast of Norway demonstrated that oil contamination probabilities are higher for subsurface than for surface in areas surrounding the spill site. The simulation indicated that the average total subsurface hydrocarbon concentration in the impacted area ranged from 0.001-10 ppb.

6.3.2 Biological Effect and Risk Assessment Models

The vast majority of oil spill models developed to date are only focused on the prediction or hind-casting of the trajectory and fate of oil on the surface of water, in the water column, and sediments for the purpose of providing information for spill response, spill contingency planning, and evaluating a spill's mass balance. A few models, however, have been designed to evaluate the impacts of oil on aquatic organisms and habitats. Xiong et al. (2000) described an oil spill impact analysis software system that evaluated potential effects to exposed organisms based on results from a physico-chemical fate component, including the extent and characteristics of the surface slick, and dissolved and total concentrations of hydrocarbons in the water column. The model considered exposure to migratory adult fish populations, spawning planktonic organisms (eggs and larvae), and wildlife species (sea birds or marine mammals). Similar goals were also included in the OSCAR model described by Reed et al. (2000). Furthermore, Price et al. (2003) described an Oil Spill Risk Analysis model that estimated the probabilities of oil spill occurrence and contact to biological and economic resources using historical records of oil spills, winds, and ocean currents.

In terms of biological effects, the Spill Impact Model Application Package (SIMAP) proposed by French-McCay (2004a) integrated both physical fate and biological effect models. The biological exposure model of SIMAP estimates the area, volume, or portion of a stock or population affected by oil. The model calculates the losses resulting from acute exposure in terms of mortality and loss production because of direct exposure or loss of food resources from the food web. The application of SIMAP for the North Cape oil spill has been reported in French-McCay (2003; 2004a), with the predicted number of birds oiled and lobsters killed in agreement with field observations. In another application to the Exxon Valdez oil spill, SIMAP estimated that 3555 otters and 26 seals may be oiled. Further applications of SIMAP in the evaluation of environmental impacts are described by French-McCay (2004b) and French-McCay et al. (2002; 2003).

Although the formation of Oil-Mineral-Aggregates (OMAs) enhances the dispersion of marine oil spills, the potential impacts of settled OMAs to benthic organisms is not well known. A comprehensive numerical approach has been undertaken by Niu et al. (2010) to model the transport of OMAs and assesses their potential risks. The predicted environmental concentrations of settled oil in OMAs was calculated using a random walk particle tracking model and a benchmark concentration of individual hydrocarbon groups were computed based on a equilibrium partitioning approach. The risks in terms of a Hazard Quotient were then determined using a Monte-Carlo-Simulation method. The predicted results from a case study based on a spill of 1000 t of South Louisiana crude oil in the Gulf of St. Lawrence, with a water depth of 80 m, demonstrated that the possibility of OMAs to cause sediment impacts depends on sediment type and physical environmental conditions.

6.4 IMPACTS ON MARINE ENVIRONMENT

If a blowout occurs, it usually results in the release of a mixture of gas, gas condensate, and/or oil. The three products behave differently in the water column and have different potential impacts. In the event of an accidental release of gas, it is anticipated that much of it would be rapidly dissipated into the atmosphere with the assistance of the wind. This is often true of the lighter components in condensate and crude oils. In the first hours and days after release, the lighter fractions evaporate. Gas condensate is a low density mixture of hydrocarbons present in natural gas that condense out of solution forming a liquid under standard temperature and pressure conditions. It is below the dew point of the gas. A large fraction of the hydrocarbon components in condensate may be soluble in water. Furthermore, many of the hydrocarbons found in gas condensate are highly toxic and, as such, the release of large quantities of condensate at the seafloor may cause local mortalities. It is expected that the impacts would be short-lived following the stoppage of flow, although depending on the duration, timing, and location there could be significant mortalities.

Two blow-outs occurred offshore Eastern Canada in 1984 near Sable Island; one at the West Venture N-91 site where no gas, oil, or condensate was released (Booth, 1990); and the other at the Uniacke G-72 well, which resulted in a spill of 240 cubic meters of condensate. With respect to the latter, a total of seven oiled marine birds were observed during surveys over an extensive area around the well the day after capping - 11 days after the blow-out (Carter et al., 1984). Cod and haddock fillet and liver were analyzed for hydrocarbons by spectrofluorometry and by gas chromatography-mass spectrometry. Fluorescence emission, di-, tri-, and tetramethylbenzenes, naphthalene and its methyl- and dimethyl homologues, were used to detect contamination of gas condensate. Increased fluorescence and dimethyl benzenes were detected in the livers. It appeared that fish were exposed to the gas condensate, but the exposure was very low and did not affect the fish. The condensate was readily accumulated by juvenile Atlantic salmon in the laboratory. The excretion half-life was of the order of days (Zitko et al., 1984).

Oil associated with surface slicks may become mixed into the water column and/or become incorporated into sediments. The relative amounts entering the various environmental compartments, and their subsequent behaviour, depends on the type of event (i.e. platform blowout, seafloor blowout, or sea surface oil spill), composition and physical-chemical characteristics of the oil, atmospheric conditions (e.g. wind and temperature) and oceanographic conditions. An overview on the fate of oil spills and their impacts has been published by GESAMP (1993). This report included samples of case studies of blowouts and major oil spills under different environmental conditions, in order to summarize the general understanding of the behaviour, fate, and effects of oil released into the sea. It is expected that the bulk of any oil released in the marine environment would initially concentrate at the sea surface to form a slick, which would immediately be subjected to evaporation. Based on previous spills (e.g. Bunker C oil spill on Nantucket Shoals from the *Argo Merchant*) and model predictions (e.g. for Hibernia), it is expected that evaporation would remove 40-50% of the oil in the first 24 hours. The resultant oil would be broken up over a few days by processes such as dispersion, dissolution into the water column, photo-oxidation, and biodegradation. Under most conditions, surface slicks of unrefined oil should disappear after one to two weeks. The presence of an oil slick on the surface will have the most serious biological impacts on birds and marine mammals in the area. The amount of spilled oil that enters the water by dispersion and dissolution varies considerably with composition and environmental conditions, but is generally on the order of 5-15%. Dissolution is considerably less than dispersion because of the low solubility of most oil components. Oil in the water may have a higher potential toxicity than surface slicks due to the reduced potential for evaporation of the lighter toxic components.

Oil products enter the water column primarily through downward mixing. The depth to which oil penetrates depends on the wind, mixing, currents, and water column structure. High rates of vertical mixing may increase the amount of petroleum product entrained in the water column compared to other areas. Short-term concentrations that can be expected in the water column under blowout or spill conditions on Georges Bank are on the order of 10 to 200 ppb with an upper maximum of about 300 ppb (Boudreau et al., 1999). In stratified regions around the perimeter of the Bank, concentrations in deeper water should be substantially lower. Any high concentrations should be short-lived and return to background levels within a week or two (Boudreau et al., 1999).

Oil hydrocarbons concentrate in the surface microlayer (i.e. water-air interface) and at the water-seafloor boundary. Alterations in fish larvae mortality have been documented with increasing concentrations of oil contaminants in the surface microlayer (NERC, 1994). Sublethal effects, which are more difficult to measure, include changes in biochemical responses of enzyme systems in fish and invertebrates (Addison, 1992; Stebbing et al., 1992; ICES, 1994; GESAMP, 1995), increased frequency of histopathological changes and diseases in bottom fish and invertebrates (ICES, 1994), and degradation of ichthyoplankton communities in response to oil contaminants (Cameron et al., 1992; Shagaeva et al., 1993). Effects of large spill events are predicted to be greater in coastal areas and enclosed seas compared to open ocean pelagic areas (Patin, 1999). Oil concentrations on the order of 100 ppb have been demonstrated to cause both lethal and sublethal effects on planktonic organisms. Despite many studies, it is difficult to detect whether major spills or chronic oil input have any irreversible impacts on the marine planktonic communities. Ecosystem level impacts are often low since, (i) the volume of water contaminated with high oil concentrations is often limited in both space and time because of rapid dispersion and weathering; (ii) planktonic organisms generally have rapid rates of regeneration on the order of days to months and can therefore quickly compensate for any loss, and (iii) replacement phytoplankton and zooplankton can be readily mixed in from surrounding waters. In addition, natural bacteria have the capacity to metabolize a large fraction of the components in condensates and crude oil (Swannell et al., 1996; Lee, 2000). For example, Braddock et al. (1995) noted increased abundance of hydrocarbon-degrading microorganisms and the restructuring of microbial communities in bottom sediments following large spill events (e.g. Exxon Valdez).

Since spawning events of fish and invertebrates are generally restricted in time and place, there can be impacts on the year class recruitment if an oil spill coincides with such a spawning event. Most major commercial species on Georges Bank have pelagic eggs and/or larvae and, therefore, are potentially vulnerable (Hodson et al., 2007; Billiard et al., 2008). In addition, there is a potential for convergence zones on the Bank to concentrate both oil and early life stages together in near surface waters thereby magnifying deleterious effects even further. Impact 'windows' can be defined which extend from the first day of spawning until such time that larvae or juveniles have sufficient mobility to avoid contaminated areas. While it is difficult to show the impacts of oil-induced mortality on early life stages because of large and variable rates of natural mortality, a number of previous laboratory studies have shown the presence of both lethal and sublethal effects (such as reduced growth and abnormal development) in eggs, larvae and juveniles of various species exposed to oil. Since at least two commercial species are spawning every month of the year on Georges Bank, any hydrocarbon release has the potential to affect fisheries resources. Previous modeling studies have predicted the type and level of impacts that various spill scenarios on Georges Bank could have on cod, haddock, and herring stocks. Some scenarios predicted cumulative losses in excess of 20% for both cod and herring (Spaulding et al., 1983; Reed and Spaulding, 1984; Spaulding et al., 1985).

The effects of oil on adult fish in the field are difficult to study and, therefore, knowledge is incomplete. Fish have the ability to avoid contaminated areas, providing these areas are small enough. In addition, it is difficult to determine cause and effect relationships between reduced population size and oil contamination due to high levels of natural variability. A number of studies, however, have documented MFO enzyme induction in association with oil spills (Kurelec et al., 1977; Payne et al., 1984; George et al., 1995; Woodin et al., 1997; Martinez-Gomez et al., 2009). Results have demonstrated the value of the MFO biomarker in delineating areas of potential impact and whether, for instance, fisheries closures or more detailed biological effect studies may be justified. Some populations of fish and marine mammals that are under pressure, already at historic lows, or designated under *SARA* may have a more pronounced vulnerability. In terms of impacts on benthic organisms, the natural interaction of oil and suspended particulate matter may facilitate the transport of oil to the sea bed (Muschenheim and Lee, 2003). Concentrations of hydrocarbons in the range of 10-100 ppm may be expected to occur in bottom sediments as a result of a blowout or spill.

6.5 OIL SPILL COUNTERMEASURES

While containment by booms and recovery of bulk oil by the use of devices such as skimmers following a spill is the first line of defence, the effectiveness of physical recovery processes for oil spilled in the open-sea is limited by logistical constraints such as high spreading rates of the spilled oil and the operational limitations of containment booms (e.g. most booms are of little or no use when wind speeds exceed 20 knots). Under ideal conditions, due to the efficiency of skimmers being linked to encounter rate, physical recovery of spilled oil can be expected to achieve a maximum of 30% (NRC, 1989). Because of this, greater emphasis is now being placed on in situ methodologies such as in situ burning, dispersion, and bioremediation methods, which require fewer response personnel, can cover large impacted areas, and do not require the storage, transport, and disposal of waste.

In situ burning involves controlled burning of oil that has spilled from a vessel or a facility. It greatly reduces the need for storage and disposal of the collected oil and the waste it generates. Oil can be successfully burned on water if the oil layer is of sufficient thickness and an adequate ignition system is available (Fingas et al., 1994). In the last decade, introduction of improved fire-resistant booms has enabled the performance of contained and controlled burns with increased burning efficiencies (Buist et al., 2005; Potter and Buist, 2008). The use of ignition promoters, emulsion breakers, and chemical herding agents can also increase the ignitability of a slick (Buist et al., 2008). Although this method is capable of rapidly removing bulk oil from the sea surface, public acceptance of the technique has not been widely received due to concerns over environmental impacts and human health and safety issues linked to the potential toxicity of burn residues and smoke. The fate of burn residues has been investigated. Guénette et al. (1995) reported that most of the burn residue of a weathered emulsified Statfjord crude oil remained buoyant. The remaining fraction was found submerged in the water column around 30 cm below the sea surface. Subsequent experiments with eight different oil types revealed that the quantity of the burn residue to remain buoyant and the portion of it that sinks were dependent on oil type (Buist et al., 1995). While residues of thicker, heavy oil slicks and weathered crude may sink when they cool down to ambient temperatures, residues of the lighter oil may not sink.

Fingas et al. (2005) analyzed the soot resulting from the burning of heavy oils and an emulsion of bitumen. It was shown that the residues are mainly resins and asphaltenes not burned in the fire and the enriched pyrogenic PAH derived from the petrogenic PAH of the oil. The PAH concentrations correlated negatively with burning efficiency. As such, attaining higher burning efficiencies reduces the amount of pyrogenic PAH in the soot.

The application of chemical oil dispersants has re-emerged as one of the major response options for oil spills, due to the availability of a new generation of products that are lower in toxicity and higher in efficiency for the treatment of more viscous oils up to 10,000 cSt (Fiocco et al., 1999; Colcomb et al., 2005). Dispersants offer many advantages over mechanical recovery, including the ability to treat large areas quickly since they can be applied from aircraft and proven effectiveness under moderate sea states. As the efficiency of dispersants is diminished by oil weathering processes, they are typically applied within 2-5 days of spill occurrence. Dispersants are not generally able to dissipate emulsified oil in the form of a mousse. Approved dispersant products have been designed to promote the transfer of oil in surface slicks into the water column by promoting the generation of oil droplets by reducing the oil-water interfacial tension (NRC, 1989). Oil is thus removed from the surface to protect seabirds and the suspension of dispersed oil droplets and/or oil-suspended particle aggregates in the water column minimizes the adverse risk of petroleum hydrocarbon contamination to the benthic community. This countermeasure technique is based on the premise of promoting the dilution of oil to concentrations below toxicity threshold limits for water-borne organisms and has been demonstrated under operational conditions (Lunel et al., 1997). Furthermore, sufficient dilution of petroleum hydrocarbons in the seawater has the advantage of facilitating un-inhibited biodegradation of hydrocarbon compounds. Biodegradation can be compromised either due to self-inhibition at high concentrations, restriction caused by limited nutrient supply at high oil loading, or reduced bioavailability because of limited oil-water interfacial area.

A recent study that has been conducted by DFO's Center for Offshore Oil, Gas and Energy Research (COOGER), in collaboration with the Unites States Environmental Protection Agency (U.S. EPA), has demonstrated the chemical dispersant effectiveness for a variety of crude oils under various sea mixing energy conditions (Lee et al., 2009). The study found that the use of chemical dispersants significantly increases the amount of oil dispersed in the water column and changes the dispersed the droplet size distribution (Li et al., 2009a; Li et al., 2009b; Li et al., 2009c). Another alternative oil spill response measure for smaller spills in the open-sea is based on the enhancement of physical dispersion of oil through the formation of stable OMAs that stimulate oil removal in the water column via microbial biodegradation (Lee, 2002, Lee et al., 2002; Lee et al., 2003; Lee et al., 2008). Studies on the toxicity of dispersed oil are currently underway to develop operational guidelines for the application of these oil spill countermeasures. For the remediation and restoration of petroleum hydrocarbon contaminated shorelines, DFO scientists have been involved in research, development, and technology transfer of bioremediation as one of the most cost-effective and eco-friendly decontamination technologies (Venosa et al., 1996; Lee and De Mora, 1999; Owens et al., 2003; Sergy et al., 2003).

Bioremediation, a strategy based on the enhancement of oil degradation rates by living organisms (Leahy and Colwell, 1990; Margesin et al., 2003), has been considered to be a potential secondary treatment option in oil spill cleanup operations, as well as a primary response strategy for the cleanup of environmentally sensitive areas that are not amenable to conventional cleanup techniques and/or with a low-level petroleum hydrocarbon contamination (Venosa and Zhu, 2003). This remediation strategy has several potential advantages over conventional technologies: it is less costly; less intrusive to the contaminated site; and, when applied in situ, generates no wastes with the end products being benign to the environment. Existing bioremediation strategies are essentially linked to biostimulation and/or bioaugmentation treatments. Marine waters typically have low concentrations of nitrogen and phosphorus, which limit the optimal rates of oil degradation following a spill. Biostimulation by the addition of fertilizers has been reported to enhance the rate and extent of crude oil degradation under field conditions (Lee et al., 1993; Prince, 1993; Rosenberg et al., 1993; Bragg et al., 1994; Lee et al., 1995, Swannell et al., 1996; Venosa et al., 2002; Prince et al., 2003b). Bioaugmentation consists of the addition of known oil-degrading bacteria that are non-native to the contaminated sites, in order to supplement the existing microbial population of clean up wastes (Azarowicz, 1973; Bartha, 1986; Forsyth et al., 1995).

Fisheries and Oceans Canada scientists have been involved in research, development, and technology transfer of bioremediation as one of the most cost-effective and eco-friendly decontamination technologies (Venosa et al., 1996; Lee and De Mora, 1999; Owens et al., 2003; Sergy et al., 2003). Bioremediation efforts have been concentrated in shoreline and wetland environments. This is due to the fact that effective biodegradation treatment of an oil spill is likely to take longer than the period for an oil slick to reach the coast (NRT, 2000). It is also difficult to maintain elevated nutrient concentrations in an open system where rapid dilution processes occur (Leahy and Colwell, 1990). To date, bioremediation field trials have consistently shown that bioaugmentation seems to have little benefit for the treatment of spilled oil in an open environment (Nichols and Venosa, 2008). In Canada, a screening protocol has been designed to evaluate the hydrocarbon degradation efficacy of oil spill bioremediation agents (Blenkinsopp et al., 1995). Products that pass the test of efficacy and toxicity are assumed to have a good potential application in spill cleanup. This does not ensure a successful outcome when applied in a real scenario, however, since the commercial culture still has to survive in the foreign and possible hostile environment, effectively compete with the indigenous microorganisms, and overcome any limitation of mass transfer.

In terms of impacts of oil spills and blowouts, there is always a trade-off of risks of oil contact or toxicity to different marine species. When oil is diverted from the surface slick into the water column risks to the avian species are diminished. Oil in the water column, however, may be more toxic to pelagic and benthic species due to the increased percentage of dissolution of lighter and more toxic components that would otherwise evaporate to the atmosphere if left on the surface of the slick. Dissolution of less soluble components is also facilitated by dispersion in the water column, which increases the oil-water interfacial area. Use of dispersants to enhance dispersion of oil to the water column may also incur increased bioavailability of toxic components to the pelagic organisms (Ramachandran et al., 2004; Couillard et al., 2005; Schein et al., 2009).

7.0 INFRASTRUCTURE CONSTRUCTION, OPERATIONS, AND DECOMISSIONING

7.1 MARINE NOISE

In addition to the noise of airguns used for seismic exploration, a significant amount of noise is generated during the exploration, construction, production, and decommissioning phases of offshore oil and gas development. The sound levels and frequency emitted is dependent on the source, such as the operation of hull mounted machinery, dredging operations linked to construction, and the use of explosives for decommissioning, to name a few The biological impact of sound levels is linked to the hearing threshold (i.e. detection level) of the organism in question, alterations in behavioural response such as avoidance, habituation (diminished sensitivity with exposure) and sensitization (increased sensitivity with repeated exposure), and temporary or permanent hearing impairment. It should be noted that the ocean is not quiet. Sources of constant ambient sound include breaking ice, waves, tidal currents, shipping traffic, and earthquakes, to name a few.

Biological sound sources can be significant. For example, Stafford et al. (1998) recorded blue whale calls over a distance of 600 km. Both natural and anthropogenic sounds can mask (i.e. drown out) weak sound signals from distant sources. Desharnais and Collins (2001) suggested that oil and gas development activities had a smaller effect on the background noise field on Sable Bank than fishing, shipping, and fin whales. Of the various sources of sound in the ocean, seismic surveys associated with oil and gas exploration has attracted the most attention. Payne (2004), however, has raised concern over other sources of chronic sound exposure such as those associated with ship traffic lanes. While the noise levels from these sources may not induce physical injury to marine life, they may interfere with normal communication and behavioural functions.

7.2 INFLUENCE OF LIGHT

Seismic vessels, offshore drilling/production platforms, floating production storage and offloading units, and service vessels are all equipped with navigation and work lights for safety of operations at night. Sea birds are typically attracted to sources of light. Some plankton and pelagic species, such as squid, may also be attracted to the lights and subject to higher predation at the surface due to their aggregation. Drill rigs and production platforms may also have flares that produce high levels of light and heat during periods of operation.

7.3 DISPOSAL OF WASTES

There are a variety of liquid and solid wastes produced during exploration and production operations, in addition to drilling muds and fluids, drill cuttings, and produced water, as described in previous sections. Additional waste materials may include sanitary/domestic waste water, cooling water, bilge water, ballast water, and garbage. Sewage and food wastes are typically macerated and treated to some degree, before disposal at sea with other liquid wastes. Considering the volume discharged and the level of dilution with tides and currents, the discharge of liquid domestic wastes and sewage is expected to cause negligible effects on fish species. In Nova Scotia's offshore, under the Offshore Waste Treatment Guidelines and the Nova Scotia Offshore

Petroleum Drilling and Production Regulations, all solid wastes are to be brought to shore for treatment and disposal.

Excess chemicals or chemicals in damaged containers are also returned to shore on supply vessels. The Transportation of Dangerous Goods and Workplace Hazardous Materials Information System (WHMIS) regulations govern the handling, use, storage, transport, and disposal of hazardous materials and substances. The Guidelines also encourage operators to minimize volumes of wastes generated by their operations and to minimize the quantity of substances of potential environmental concern. In addition, offshore operators must adhere to the Offshore Chemical Selection Guidelines for Drilling and Production Activities on Frontier Lands, as well as other relevant sections pursuant to the International Convention for the Prevention of Pollution from Ships with respect to marine drilling discharges and emissions. Prior to entering Nova Scotia's offshore, offshore petroleum operators are required to evaluate chemical substances used in their operations to ensure that those used are the most environmentally appropriate. In addition, all drilling operators must work within the framework of a corporate management system that includes an Environmental Protection Plan and Waste Management Plan, which outline corporate and regulatory requirements and procedures for the handling, treatment, and disposal of discharges and wastes. Cement slurry and blowout preventer fluids may also be discharged during drilling operations. Mitigation of impacts is largely based on the selection of low toxicity products, as described in the Offshore Chemical Selection Guidelines and limited discharge volumes.

Last, solid operational debris, such as anchors and chains from offshore activities, has been a problem in the North Sea and the Gulf of Mexico. Canadian drilling regulations, however, require that the sea floor be cleared of any material that could interfere with other commercial uses of the area when a well is abandoned, unless otherwise determined by the regulator. Typically, the well casing itself must be sealed at least 1 m below the sea floor to prevent damage to fishing gear. In a year of the cessation of the U.S. exploration activities on Georges Bank, only four large items remained unrecovered in the area. None of them exhibited sufficient interference with commercial fishing activities (Danenberger 1983).

7.4 ATMOSPHERIC EMISSIONS

Atmospheric emissions are associated with all stages of oil and gas activities. The main sources atmospheric emissions include:

- operation of oil or gas fired generators, pumps, turbines, and engines on the platform, ships, and onshore facilities;
- evaporation or venting of hydrocarbons during different operations of their production, treatment, transportation, and storage; and
- burning of gas and hydrocarbons during well testing and development, as well as flaring to eliminate gas from storage tanks and pressure-control systems.

Flaring is typically the major source of atmospheric emissions from offshore oil and gas operations. Numerous pressure relief valves on pressure vessels would occasionally vent natural gas, but instead of being released to the environment, vented material goes up the flare stack and is burned producing carbon dioxide. Flares are typically 80-98% efficient at converting hydrocarbons to carbon dioxide. Flaring serves two functions:

- in the event of an accident or leak, operators may need to depressurize the facility whereby natural gas is sent into the flare stack and burned off; and
- under normal operating conditions, the flare serves to control and reduce emissions.

From an ecological perspective, the most hazardous components associated with flaring are nitrogen and sulfur oxides, carbon monoxide, and the products of the incomplete burning of hydrocarbons. These interact with atmospheric moisture, transform under the influence of solar radiation, and can precipitate back to the Earth's surface where they may increase local and regional pollution. To date, atmospheric emissions from offshore installations in the offshore of Atlantic Canada have not been of major concern to regulatory agencies due to the scale of emissions, distance from populated areas, and dispersion by offshore winds.

7.5 DECOMMISSIONING

Typically, offshore exploratory wells are abandoned and decommissioned by the removal of the wellhead and mechanical severance below the sediment surface. Detonation of explosives below the mud line may be used if mechanical severance proves difficult. In general, all infra-structure installed for production are removed once the site is abandoned, unless otherwise permitted by the regulator.

8.0 OTHER OFFSHORE PETROLEUM REGIONS

8.1 SCOTIAN SHELF AND GRAND BANKS

The Scotian Shelf and Grand Banks, which are to the east of Georges Bank in the Northwest Atlantic, currently support a number of petroleum activities. The Scotian Shelf is a broad continental shelf off the coast of Nova Scotia. It has a sand, gravel, and cobble bottom on the top of the banks with gravel and finer sediments in the deeper basin areas. There are proven hydrocarbon resources (e.g. the Cohasset-Panuke oil field, the Sable Project gas field, and the Deep Panuke gas field) all on the shallow Scotian Shelf (Kidston et al., 2007). In addition, exploration activities are targeting the deep water areas (e.g. the Scotian Slope), since advancing technologies have enabled hydrocarbon discoveries and high success rates in deep water of other circum-Atlantic basins such as the Gulf of Mexico, offshore Brazil and West Africa, and recently Northwest Africa (e.g. Mauritania) (Kidston et al., 2007). The Scotian Shelf basins and Scotian Slope remain virtually unexplored, with the potential for significant hydrocarbon discoveries.

Operators are required to meet certain regulatory requirements before the CNSOPB can approve any offshore petroleum activities. The regulatory framework which governs offshore petroleum operations consists of legislation, regulations, guidelines, and policies, as overseen by the CNSOPB. The CNSOPB's resource conservation approach is designed to ensure that operators are committed to resource conservation, to audit activity authorization applications from a resource conservation perspective, and to perform necessary studies and surveillance in order to develop and support an independent understanding of the resource. On the Scotian Shelf, much of the fisheries yield is directed toward demersal fish stocks, and the protection of fisheries resources has been a priority. Commercial scallop stocks are found on the Scotian Shelf and these are recognised as major valued ecosystem components in the SOEP EEM Program.

In an ecosystem assessment report for the Eastern Scotian Shelf Integrated Management (ESSIM) area, Zwanenburg et al. (2006) noted that oil and gas activities would likely not have caused significant overall loss of biodiversity due to the relatively small proportion of the ESSIM area affected by related activities and the modest scope of exploration (and development) having been conducted to date. Several ecosystem status reports for the Scotian Shelf (e.g. Breeze, 2002; DFO, 2003b; Zwanenburg et al. 2006), however, have suggested that the cumulative effects of human use activities (e.g. fishing, commercial shipping, naval operations, and offshore oil and gas activities) may act additively and/or synergistically to impact the marine ecosystem. Hurley (2009) noted that two assessment reports concluded that the eastern Scotian Shelf ecosystem has been profoundly altered over the past few decades mainly by fishing, which occurred against a backdrop of a dynamic biophysical environment. The result has been a significant decrease in the biomass of demersal groundfish species and a significant increase in biomass of grey seals, small pelagic species, commercial crustaceans, and phytoplankton.

The Grand Banks is a large bank system off eastern Newfoundland with significant biodiversity, as well as hydrocarbon resources. Much of the biological production is demersal fish species. Although there are some Icelandic scallops, there are no significant scallop or lobster fisheries in the vicinity of the petroleum development sites (Boudreau et al, 1999). There are five major petroleum fields on the Grand Banks: Hibernia, Terra Nova, Hebron, White Rose, and Ben Nevis fields. Only Hibernia, Terra Nova, and White Rose have been developed with 67, 34, and 24 development wells drilled, respectively (CNLOPB website). Similar to the projects in the offshore of Nova Scotia, these projects are required to meet certain regulatory requirements as contained in the Statutes and Regulations governing the CNLOPB.

8.2 NORTH SEA

The North Sea is a relatively contained shallow sea on the European continental shelf. It has a mean depth of approximately 90 m, with the exception of the Norwegian trench that has a maximum depth of 725 m. It has low tidal energy and lower average physical forcing, which results in more stratified waters compared to Georges Bank. In addition, the environment is less dispersive than Georges Bank due to weaker currents. This results in discharges from drilling platforms accumulating close to the site and may cause more severe local ecosystem impacts (Kingston, 1992; Olsgard and Gray 1995; Daan et al, 1996; Boudreau et al, 1999). Much of the petroleum activity is concentrated in the southern North Sea and takes place within 150 km to land. This places the potential impacts much closer to human populations that live around the North Sea. The population in the drainage basin of the North Sea is on the order of 150 million people. This larger population, shorter distances, and weaker circulation system results in higher background levels of pollution than are observed on the Georges Bank area (Boudreau et al, 1999).

8.3 GULF OF MEXICO

The Gulf of Mexico is an ocean basin largely surrounded by the North American continent (Gulf coast of the United States and northern Mexico), with Cuba on its eastern boundary. It includes both tropical and temperate climate regimes with some coral reefs. Its mean sea surface temperatures range from 28-30^{0}C and 20-24^{0}C in the summer and winter, respectively (Boudreau et al, 1999). The tidal range in the Gulf of Mexico is approximately 0.3 m, with semi-diurnal tides in the east and diurnal in the west. The Gulf provides habitat for a large range of tropical species of fish, warm water invertebrates, and marine turtles. Sea turtles have associated nesting beaches in the Gulf. There are fewer large whales compared to Georges Bank, although offshore in deeper waters there is a resident population of sperm whales (Boudreau et al, 1999). The Gulf of Mexico is not as pristine an environment as Georges Bank, as it receives large amounts of freshwater, sediment, and nutrient inputs, particularly from the Mississippi River. The Gulf of Mexico supports a large number of petroleum production projects (Table 11).

Table 11. *Number of wells in the Gulf of Mexico by water depth. Data as of June 8, 2009 (USBEM, 2010).*

Water Depth (m)	Approved Drilling Permits	Active Leases	Active Platforms
0 - 200	33423	2740	3676
201 - 400	1095	186	21
401 - 800	827	-	9
801 - 1000	486	409	7
> 1000	1522	3343	23

The Gulf of Mexico Offshore Operations Monitoring Experiment (GOOMEX) was undertaken from 1992 to 1995 to develop and recommend sensitive and appropriate techniques for monitoring activities of offshore oil and gas production. To accomplish this goal, a broad range of biological, biochemical, and chemical methodologies were tested to detect and assess potential chronic, sublethal, and long-term effects of offshore oil and gas production. The GOOMEX study components included measurements of abiotic characteristics to indicate environmental state (e.g. chemical patterns in sediments and water, geological patterns, and physical patterns) and biotic responses (e.g. tissue body burdens, detoxification response by fish and invertebrates to contaminant exposure, sediment toxicity to invertebrates, meiofaunal, macrofaunal and megaepifaunal community structure, harpacticoid reproduction and population genetic structure, and megaepifaunal reproduction).

In general, the biological patterns observed around platforms in the Gulf of Mexico have been the result of the complex interactions of variations in sediment grain size, organic matter enrichments, and toxic response to contaminant exposure. Contaminant levels only exceeded levels thought to produce deleterious biological effects at a few stations close to the platform, with the effects being limited to 100 m from platforms. Relative to background (i.e. 200 m), the zone near platforms had sediments with higher levels of contaminants and toxicity, reduced levels of abundance, species diversity, genetic diversity, and reproductive success, and feeding guilds with more deposit feeders. While further study is needed, the results of GOOMEX suggest that benthic environments around platforms have been disturbed as a result of the presence of the platform and discharges derived from exploration and production activities (Kennicutt 1995).

9.0 CONCLUSION

Concern over the potential environmental impacts from offshore petroleum development and production is largely linked to the possible exposure of marine organisms to seismic noise, operational waste discharges (e.g. drill wastes, produced water, and other associated wastes), and accidental oil spills and/or blowouts. The impacts of seismic noise to fish and invertebrates include mortality of eggs and larvae, sub-lethal effects, and changes in fish swimming patterns. Mortalities likely occur within a few metres of the sound source, although little is known about whether acute high level sound exposures or chronic low level exposures adversely affect marine organisms at some depth in the water column. There is now conclusive evidence demonstrating the impact of seismic noise on fish behaviour and fisheries catch success. For marine mammals, although there are no definitive case studies linking mortality resulting from exposure to oil and gas exploration seismic surveys, there is evidence that exposure to seismic sound can result in dispersion and/or temporary displacement of animals. Since behavioral observations of marine mammals are typically variable, some findings are contradictory, and the biological importance of observed effects has not been measured. While impacts to individuals or groups of marine organisms have been observed, there is ongoing debate/investigation of the potential for population or ecosystem scale impacts. In addition to sound, a drilling rig and its associated supply vessels generate light during routine operations. While marine mammals may avoid the immediate area around a rig due to the unusual and/or increased lights, marine birds and pelagic species, such as squid, may be attracted to the lights of the offshore installation or service vessel.

Drilling wastes (i.e. spent drilling mud and well cuttings) are of primary concern during exploration and development drilling operations. In recognition of the results from toxicity studies, current regulatory guidelines in Canada require the recovery and onshore disposal of drill cuttings generated with the use of OBM, when possible. If re-injection of drill solids associated with SBM or Enhanced Mineral Oil Based Muds is not possible, the solids may only be discharged at the site following treatment with best available technology. In contrast, low toxicity WBM is permitted to be discharged. Drilling muds can have both acute (i.e. lethal) and sublethal impacts on marine organisms. Acute effects are predominantly associated with the smothering of slow-moving or sessile benthic organisms near the point of discharge. Sublethal or chronic, long-term impacts such as reduced growth or reproduction may also be associated with sessile benthic organisms if they become exposed to drilling mud components that have settled on the bottom. The magnitude of potential toxic effects is directly linked to exposure time and concentrations (i.e. dosage). Operational models predicting the fate and transport of drilling waste have been coupled with biological effects data to enable the risk assessment of drilling waste disposal.

Under field conditions, drilling muds and cuttings undergo a number of physico-chemical processes in the marine environment following discharge including: (1) advection, (2) dispersion, (3) aggregation (4) settling, (5) deposition, (6) consolidation, (7) erosion, (8) re-suspension, (9) re-entrainment, and (10) change in bed elevation. Documented results from numerous case studies on water based muds suggest that the degree of impact of drilling fluids and cuttings on benthic and demersal species is highly dependent on a number of local environmental variables (e.g. depth, current and wave regimes, and substrate type), as well as on the nature and volume of the discharges, including cuttings size and location of the outfall in the water column. Numerical models have been developed to improve our knowledge on the probable fate and effect of particles and chemicals associated with drilling wastes discharged into the sea. Integration of risk assessment models with physical transport/fate and effect models, will help to evaluate site-specific risks from drilling waste discharges.

There is consistency in the results of environmental effects monitoring (EEM) studies from various Canadian East Coast programs despite differences associated with the volumes and type of drilling waste discharges, scale and location of drilling, and variations in EEM sampling programs. The highest levels of drill waste (sediment and body burden concentrations) were associated with multiple-well, production-based fields that had been operating for several years in the offshore Newfoundland area. While sampling stations have extended out from the immediate vicinity of platforms to tens of kilometers away for reference sites, study results have demonstrated that changes in the diversity and abundance of benthic organisms have been generally limited to within 1000 m of the drill site and returned to baseline conditions within 12 months of cessation of drilling discharges. The spatial area for observed biological effects was generally smaller than the area over which drilling muds were detected.

Produced water, a complex mixture of inorganic salts, metals, radioisotopes, production chemicals (e.g., biocides and emulsion breakers), and a wide variety of organic chemicals (e.g., organic acids, petroleum hydrocarbons, and phenols) in both dissolved and particulate phases, represents the largest volume (up to 80%) waste stream in oil and gas production operations. The physical and chemical properties of produced water vary widely depending on the age, depth, and geochemistry of the hydrocarbon-bearing formation, as well as the chemical composition of the oil and gas phases in the reservoir. The chemicals of greatest environmental concern in produced water, because of their potential for bioaccumulation and toxicity, are metals and hydrocarbons. While their concentration in the produced water stream is generally below the effects threshold, in some regions there has been concern that highly-alkylated phenols (octyl- and nonyl-phenols), which are well-known endocrine disruptors and some toxic production treatment chemicals, may cause localized effects. If water depths are shallow, some metals (e.g., barium, iron, and manganese) and higher molecular weight aromatic and saturated hydrocarbons may accumulate in sediments near produced water discharges, possibly harming bottom living biological communities.

In Canada, the formulation of regulatory guideline values for produced water discharge is an adaptive process that promotes the development of improved EEM programs that takes into consideration the level of environmental risk, Best Available Technology for mitigative measures, and socio-economic benefits. The current regulatory guidelines for produced water discharge in Canada (i.e. Offshore Waste Treatment Guidelines) are based on petroleum hydrocarbon content. Treatment of produced water for regulatory compliance before ocean discharge removes solids and non-aqueous liquids from the waste water, including dispersed oil, suspended solids, scales, and bacterial particles, as well as corrosive gases, such as carbon dioxide and hydrogen sulphide. Existing clean-up protocols for produced water are effective in removing volatile compounds and dispersed oil from the produced water, but not the dissolved organic and inorganic fractions. Treated produced waters on offshore platforms are typically discharged below the sea surface at a depth from 10-100 m.

The various models used to predict the fate of produced water (e.g. plume dispersion models and chemical fate/transport models) differ in specific details, but all predict a rapid initial dilution of discharges by 30- to 100-fold in the first few tens of meters of the discharge, followed by a slower rate of dilution at greater distances. Factors that affect the rate of dilution of produced water include discharge rate and height above or below the sea surface, ambient current speed, turbulent mixing regime, water column stratification, water depth, and differences in density (as determined by temperature and total dissolved solids concentration) and chemical composition between the produced water and ambient seawater. Based on the chemical composition of treated produced water and predicted dispersion rates for potential sites of concern, it is expected that toxicity threshold limits for acute effects are not likely to occur beyond the immediate vicinity of the discharge pipe for offshore platforms located on the East Coast of Canada, due to natural dispersion processes driven by tides and currents. Over the 15 to 20 year life-span of a typical offshore platform, however, there is concern that continual long-term chronic exposures to produced water discharges may cause sub-lethal changes in marine organisms.

It has recently been suggested that the toxicity of produced water may diminish over time following its discharge, due to natural chemical kinetic reactions that occur following its release (in an anoxic state) into the open ocean. Due to the differing rates of degradation of the multiple toxic components in produced water and the variation of ecosystem effects resulting from differences in toxicity, modeling efforts have been employed to try to account for the variations and determine the general ecological risks associated with produced water discharge overtime. The accuracy of contaminant risk assessment models is dependent on the identification and quantification of the various chemicals that induce the toxic effects. Additional information on the toxicity of produced water components will improve the accuracy of the Environmental Impact Factor model currently used for risk assessment of produced water discharges. In regard to other wastes, the *Offshore Waste* Treatment Guidelines, Nova Scotia Offshore Petroleum Drilling and Production Regulations, and domestic waste treatment technologies recommended by the Department of the Environment are in place to mitigate the potential risks associated with the disposal of domestic waste from offshore facilities. Furthermore, under Canadian drilling regulations, solid operational debris (e.g. anchors, chains, etc.) on the sea floor is to be removed when a well is abandoned.

For offshore oil and gas exploration and production activities, the risk and consequence of oil spills is of the greatest environmental concern. From a socioeconomic perspective however, tainting and contamination of fisheries resources leading to closures would be of greatest concern. Although there is a low probability of occurrence, spills, blowouts, and malfunctions can result in the release of liquid and gaseous hydrocarbons. Blowouts occur when operators are unable to control the flow of oil or gas and it is released into the atmosphere or surrounding water. Oil spills other than from blowouts can occur during drilling and production activities. These include spills of diesel oil or lubricating oil for operations on the platforms, as well as spills from transfer and transport operations. In general, spills are usually small and have minimal/negligible environmental impact. Larger spills, however, can occur. In addition to accidental releases, a significant amount of crude oil, estimated between 4.2-14 million barrels, is discharged each year from natural seepage. Natural seepage accounts for the largest, cumulative source of oil that enters the ocean. In the North Atlantic, the occurrence of natural oil seeps has been well documented from the Canadian Arctic to the Gulf of Mexico, including the Georges Bank region.

Scientific studies on the transport and effects of petroleum hydrocarbons (i.e. gas, gas condensate, and/or oil) from spills and blowouts have been ongoing for decades. In the event of an accidental release of gas, it is anticipated that much of it would be rapidly dissipated into the atmosphere with the assistance of the wind. While a large fraction of gas condensate would evaporate, some components such as benzenes and naphthalenes may elicit an acute toxic effect near the point of release depending on duration, timing, and location of the event. The release of crude oil may result in a surface slick that eventually becomes entrained in the water column and/or become incorporated into sediments. Oil concentrations on the order of 100 ppb have been demonstrated to cause both lethal and sublethal effects on planktonic organisms in laboratory studies. It has been difficult, however, to demonstrate that either major spills or chronic oil input have any irreversible impacts on the marine planktonic communities. For fish and marine invertebrates, the major concern is a spill incident that coincides in the time and place of a fish spawning event. Benthic organisms, such as scallops, are also susceptible to exposure. Residual hydrocarbons may not persist in surficial sediments over an extended period, due strong currents that could continually disperse and transport fine sediment particles over a wider area and into deep waters. To predict spill trajectories, a number of oil spill models and operational tools (e.g. tracker buoys) covering a range of capabilities, from simple trajectory models to 3D trajectory and fate models, are in use in the world today. The application of predictive models is also used to support risk assessment studies and operations to provide critical guidance for contingency planning as well as clean-up operations.

In the offshore of Nova Scotia, individual operators have the responsibility to implement a corporate Emergency and Oil Spill Response Plan approved by the CNSOPB that includes routine spill response exercises. Physical containment and recovery of bulk oil is generally considered the first line of defense following a spill. The effectiveness of physical recovery processes for oil spilled in the open-sea is limited by logistical constraints (e.g. the spreading rate of oil spills and operational limitations for containment booms), but mostly by weather (e.g. sea state) which is the largest limiting factor to containment and recovery. As a result, there is a renewed interest regarding the application of chemical dispersants, which promote the transfer of oil on the surface into the water column by promoting the generation of oil droplets. This countermeasure technique is based on the premise of promoting the dilution of oil to concentrations below toxicity threshold limits for water-borne organisms. Furthermore, the dispersion of oil into the water column will enhance the biodegradation rates of residual hydrocarbon compounds. Decisions on chemical oil dispersant use should include a net benefit analysis. While the application of dispersants would reduce the contact between seabirds and surface oil slicks, and the probability of oil impacting shoreline environments, there is concern that they may also increase the bioavailability of toxic components to pelagic organisms. With advances in scientific knowledge, improvements are continually being made in the areas of prevention, environmental effects monitoring, risk assessment, mitigation, and remediation of potential impacts to the marine environment associated with offshore petroleum development and production activities.

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