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Human Health Risk Assessment of Naturally Occurring Radioactive Materials in Produced Water – A Case Study

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ABSTRACT

Human health effects from naturally occurring radioactive materials (NORM) in produced water are of concern due to their bioavailability and bioaccumulation characteristics in finfish and shellfish species used for human consumption. Being chemically similar to calcium, radium concentrates mostly in bones, shells and exoskeletons. Previous studies have been based on the whole-body bioaccumulation of radium in fish where the distribution of radium in bone/exoskeleton and the edible parts of fish were not considered separately and thus the predicted risks were relatively high. In this paper, the distribution of radium in the non-edible and edible parts of fish and the probability of exposure to a produced water plume have been studied in order to characterize human health risks. A probabilistic hydrodynamic model has been incorporated in this study. Using the concentration distribution approach, the mean cancer risks to humans were predicted in the range of 8.6×10^{-7} to 9.5×10^{-7} , which were 2.6 to 2.7 times less than the risks predicted by using the whole body concentrations. The exceedence probability of maximum permissible human health cancer risk of 1×10^{-4} is close to zero. At a risk level of 1×10^{-6} , the exceedence probability is 21% while in the whole body concentration approach it is between 45 to 49%. In this study, no effect on fish from exposure to NORM components in produced water was found.

Key words: Produced water, human health risk, NORM, Monte Carlo simulations, dilution model, and fish tissues.

INTRODUCTION

Produced water, which is extracted during oil and gas production, includes formation water, injected water, small volumes of condensed water, and any chemical added down hole or during the oil/water separation process (USEPA 1993). Produced water is the highest volume waste generated in association with oil and gas production operations. Despite treatment before discharge to satisfy regulatory limitations on oil content, produced water contains a certain amount of Naturally Occurring Radioactive Materials (NORM) such as ^{226}Ra and ^{228}Ra . NORM are difficult to remove from produced water, which make the assessment of their effects on human health important to the oil and gas producing industries.

Several studies have been carried out to assess human health risk in terms of the likelihood of cancer as a result of NORM contaminated seafood ingestion (Hamilton *et al.* 1992; Meinhold and Hamilton 1992; Meinhold *et al.* 1996). These previous studies did not consider the concentration distributions in different organs such as the bone/exoskeleton and flesh of fish and thus predicted a relatively high risk from this source; although the predicted risks by the previous studies were below the regulatory permissible limits. Meinhold *et al.* (1996) predicted potential cancer risk from ^{226}Ra and ^{228}Ra in produced water for open bays in Louisiana where the exposure probability for fish and the organ-based distributions of radium were not considered.

Four different radioisotopes of radium (^{223}Ra , ^{224}Ra , ^{226}Ra and ^{228}Ra) exist in nature. The half-lives of ^{223}Ra , ^{224}Ra , ^{226}Ra and ^{228}Ra are 11.4 days, 3.7 days, 1600 years and 5.75 years respectively (USEPA 1999). Due to their short half-lives, ^{223}Ra and ^{224}Ra do not play a significant role in risk assessment. Produced water may contain radium components in addition to metals, polycyclic aromatic hydrocarbons (PAHs), volatile organic carbons (VOCs) and

process chemicals. Two isotopes of radium, ^{226}Ra and ^{228}Ra are of most concern as they are leachable and mobile because of their high solubility in water (Vegueria *et al.* 2002) and they may bioaccumulate in food organisms (Hamilton *et al.* 1992). Radium isotopes probably make up the bulk of nuclide activity in produced water (Snively *et al.* 1989). The isotope ^{226}Ra emits mostly alpha particles at energies of 4.6 and 4.78 MeV and a small amount of gamma rays at energy of 0.186 MeV whereas ^{228}Ra emits beta particles at energy of 0.055 MeV (Neff 2002). The rate of radioactive decay of a radioisotope (activity) is measured in Curies (Ci) or the International System (SI) unit of Bequerels (Bq). The convenient unit of picoCurie ($\text{pCi} = 10^{-12}$ Ci) will be used in this study.

Stephenson and Supernaw (1990) reported ^{226}Ra and ^{228}Ra in the ranges of 4 to 584 pCi/l and 18 to 586 pCi/l for produced water outfalls in the Gulf of Mexico. The Continental Shelf Associates Inc. reported similar ranges of ^{226}Ra and ^{228}Ra in produced water from open bays in Louisiana (CSA 1993). The average activities of ^{226}Ra and ^{228}Ra in the Gulf of Mexico (GOM) were reported as 262 ± 156 and 277 ± 146 pCi/l respectively (Stephenson 1992). Radium activity from two oil fields in Brazil has been found to be less than that of the Gulf of Mexico (Vegueria *et al.* 2002). Produced water in the Gulf of Mexico has a higher radium activity than that present in produced water from open bays in Louisiana (Stephenson 1992; Meinhold *et al.* 1996). The activity of radium in produced waters from other offshore locations in the world is usually much lower than that in produced water from platforms in the Gulf of Mexico.

Radionuclides can accumulate in the soft tissue and bones of marine species (Mulino and Rayle 1992). The concentration factor, which is defined as the ratio of concentration in an organ or organism to the concentration in the media, varies depending on type of fish and organ such as flesh, skin or bone of fish (Meinhold and Hamilton 1992). A study on marine species showed the

concentration factors of ^{226}Ra and ^{228}Ra in bone is higher than that of muscle for Sole, Ray, Sardine, Mackerel, Oil fish, Oyster, Clam, Green mussel and Snail (Iyengar 1984; Iyengar *et al.* 1980; Neff 2002). Radium is chemically similar to calcium and concentrates in bones, shells and exoskeletons (Meinhold and Hamilton 1992). Studies on three fish species suggest that radium is mostly accumulated in bones and least in flesh (Meinhold and Hamilton 1992). Concentration factors of radium content based on the whole organism overestimate the level of radium in the edible portion (Iyengar 1984). A minimum of 40% of the total radium in a fish is accumulated in bones and approximately 6% in edible flesh (Neff 2002). Several species of lake fish also bioaccumulate radium to higher concentrations in bone than in the muscles (Neff 2002).

The distribution of radium in the edible and non-edible part of a fish was not taken into account for human health risk assessment in some previous studies and thus those previous studies have predicted relatively high risks. Meinhold *et al.* (1996) employed Monte Carlo simulations for assessing human health risks from radium and lead in produced water, but radium distribution in different organs (bones, flesh, exoskeleton, shell) of a fish and uncertainty associated with hydrodynamic modeling were not taken into consideration. A methodology for human health risk assessment from NORM components in produced water is discussed in this paper by presenting an example that evaluates discharge scenarios of produced water from an offshore oil platform. The uncertainty associated with model inputs and coefficients, including the uncertainty from an error term, have been considered. This human health risk assessment methodology is discussed in the following sections.

HUMAN HEALTH RISK ASSESSMENT METHODOLOGY

The human health risk assessment methodology for fish consumption consists of three main phases: (1) prediction of exposure concentration for the fish; (2) assessment of radium concentration in the edible part of the fish and (3) characterization of the cancer risk in humans. These phases are discussed in the following sections. The framework for human health risk assessment is presented in Figure 1.

PHASE 1. PREDICTION OF EXPOSURE CONCENTRATION FOR FISH

This study considers a hypothetical discharge scenario for produced water from an offshore platform with the objective of providing a methodology, in which relevant information for potential produced water discharge from an offshore platform was collected from various sources. The hypothetical platform was designed for a capacity of treating $0.212 \text{ m}^3/\text{s}$ of produced water. Assuming the location of the site to be on the Grand Banks, Newfoundland, the ambient environmental conditions were defined on the basis of limited available data (DFO 1999; Petro-Canada 1996; Mukhtasor 2001). The ambient water depth for the location is about 95 m. During winter, the water column is uniformly cold and at other times it is a two-layer system in which the top layer becomes most stratified in August. This top layer has a thickness of 15m (approximately) and can be considered as the depth of the mixing layer. The current speed in that location follows a lognormal $(-3.29, 0.96)$ distribution (Mukhtasor 2001) where the parameters -3.29 and 0.96 represent the natural log of the median and the standard deviation of current speed respectively. The depth above the discharge dominates the dilution and dispersion of produced water plume. The depth of the discharge port for this study was assumed to be 11m below the water surface. The density gradient was assumed to be 0.025.

Dilution model

Upon discharge of a buoyant effluent into the marine environment, it dilutes and disperses so that the pollutant concentration decreases as the effluent moves farther away from the discharge point. Mukhtasor (2001) developed a probabilistic dilution model as follows:

$$\frac{SQ}{uz^2} = (0.13 \pm 0.02) \left(\frac{z}{l_b} \right)^{(-0.31 \pm 0.03)} + (0.46 \pm 0.02) \exp \left(\frac{(-0.22 \pm 0.04)}{z/l_b} \right) + N(0, 0.092) \quad (1)$$

where

S = the initial (centerline) dilution (dimensionless);

Q = the outfall discharge rate;

u = ambient current velocity (m/s);

$N(0, 0.092)$ = mode residual (ε), normally distributed with mean and standard deviation 0 and 0.092 respectively; and

l_b = buoyancy length scale determined as:

$$l_b = \frac{Qg}{u^3} \left(\frac{\rho_a - \rho_o}{\rho_a} \right) \quad (2)$$

where

g = the gravitational acceleration;

ρ_a = ambient seawater density (kg/m³); and

ρ_o = effluent density (kg/m³).

The plume rises in the near field region and surface impingement takes place in the region known as the control volume that connects the near and far fields (Mukhtasor 2001). The dilution model components, such as plume width and thickness at the end of control volume, length of control volume and other related terms can be found elsewhere (Akar and Jirka 1994; Doneker and Jirka 1990; Wright *et al.* 1991; Huang *et al.* 1996; Mukhtasor 2001)

Exposure concentration (EC)

It is unrealistic to assume that a migratory fish stays in the produced water affected area throughout the whole exposure period. The USEPA (1999a) determines the exposure probability as the ratio of the affected zone to the zone under study. An area of 100 m radius around the discharge port was used by USEPA (1999a) for human health risk assessment purposes. As radium is soluble in water, an assumption of 100% bioavailability has been used in this study for a conservative calculation. The exposure concentration has been calculated as (USEPA 1999a):

$$C_{exp} = C_w \times p \times BAF \quad (3)$$

where

C_{exp} = exposure concentration for fish;

C_w = predicted environmental concentration (PEC) using fate and transport models;

p = exposure probability; and

BAF = bioavailable fraction

PHASE 2. PREDICTION OF RADIUM IN THE EDIBLE PART OF FISH

The growth of a fish continues throughout its whole life irrespective of its location, and therefore the growth and other physical changes (e.g. change in lipid and bone content) during the exposure period have an effect on the accumulation of contaminants. Using the whole fish bioconcentration factor, the total radium in fish is predicted as:

$$W_{rad} = C_{exp} \times BCF \times W_t \quad (4)$$

where,

W_{rad} = total radium accumulated in fish (pCi);

W_t = weight of fish (kg);

C_{exp} = calculated using equation (3); and

BCF = whole fish bioconcentration factor for radium.

Meinhold and Hamilton (1992) determined bioconcentration factors for whole fish in the range of 2 to 100. Meinhold *et al.* (1996) used a lognormal distribution for bioconcentration factor in whole fish with mean 30.4 and standard deviation 28 for human health risk assessment from NORM components based on the data provided by Meinhold and Hamilton (1992).

The edible part of fish was calculated as the sum of moisture and lipid content as documented by the USEPA (1996) and shown in Table 1. These data revealed that the edible parts of a fish are in the range of 64-87% with a mean value of 78%, which results a lognormal (4.36, 0.063) distribution for the edible part of fish.

The average concentration of ^{226}Ra in bone is 5.0 to 7.9 times higher than that of the soft tissues in a fish (Iyengar 1984). In lake trout and whitefish, this ratio varies between 1.7 to 9.2 and 3.1 to 34.3 respectively (Clulow *et al.* 1998). The average concentration of ^{226}Ra in the shell of mussels is 9.3 times higher than that of the soft tissues (Neff 2002). The concentration factors for edible and non-edible parts are shown in Table 2. For Crustaceans (marine), the ratio of concentration factors between non-edible and edible parts varies within 0.6 to 22.9 while for White sucker, the ratio is 149.4 (Table 2). The ratio of concentration factors between non-edible and edible parts has a geometric mean of 9.9, minimum ratio of 1 and the maximum of 150, which follow a lognormal (2.3, 1.2) distribution. The concentration factors in the edible and non-edible parts of a fish also follow lognormal distributions (Table 2). The equation for concentration distribution was developed using mass balance assuming no loss of radium in fish as:

$$\begin{aligned}
 x \times W_t \times C_{\text{raded}} + (1-x) \times W_t \times C_{\text{radned}} &= W_{\text{rad}}, \text{ and} \\
 x \times W_t \times C_{\text{raded}} + (1-x) \times W_t \times y \times C_{\text{raded}} &= W_{\text{rad}} \\
 \Rightarrow C_{\text{raded}} &= \frac{W_{\text{rad}}}{[x + (1-x)y] \times W_t} \quad (5)
 \end{aligned}$$

where

C_{raded} = radium concentration in edible part of fish (pCi/kg)

C_{radned} = radium concentration in non-edible part of fish (pCi/kg)

W_{rad} = total radium in fish as calculated in equation 4 (pCi)

W_t = weight of fish (kg)

x = edible fraction of a fish

$$y = C_{radned}/C_{raded}$$

Using the value of W_{rad} from equations 4 and 5 can be rewritten as:

$$C_{raded} = \frac{C_{exp} \times BCF}{[x + (1 - x)y]} \quad (6)$$

PHASE 3. CHARACTERIZATION OF CANCER RISK

The total human intake of radium was calculated as:

$$I_T = C_{raded} \times FIR \times EF \times ED \times FR \times GI \times 10^{-3} \quad (7)$$

where

I_T = total intake radium intake (pCi)

C_{raded} = radium concentration in edible part of fish (pCi/kg)

FIR = daily fish ingestion rate (g/day)

EF = exposure frequency (days/yr)

ED = exposure duration (yrs)

FR = fraction of contaminated fish ingested

GI = gastrointestinal absorption factor

10^{-3} = conversion factor for g/kg

Fish ingestion rate (*FIR*)

USEPA (1999a) used 177 g/day as the 99th percentile fish ingestion for human health risk assessment. Meinhold *et al.* (1996) derived a lognormal (3.455, 0.622) distribution for fish caught near an open bay platform in Louisiana. The use of a lognormal distribution was suggested by USEPA (1996) for human health risk assessment in long-term basis.

Fraction of contaminated fish ingested (*FR*)

Throughout the exposure period, it is unrealistic to assume that all the fish ingested are from the contaminated site. A study by USEPA (1997) for the age group 1-20 year old shows that 123 g/day marine fish was ingested from a total of 219g/day fish ingestion. In 1992, a survey of restaurants shows an average of 889 dishes out of 1500 total each week was served with seafood (Schultz *et al.* 1996). The available data (USEPA 1996, 1997; Schultz *et al.* 1996; Dellenbarger *et al.* 1993) suggest that marine fish ingestion is almost 50% of the total fish ingestion. The total fish ingested from marine sources may not be exposed to a produced water plume and thus the use of 50% of the total ingested fish, as the contaminated fish will still provide a conservative prediction of cancer risk.

Exposure frequency (*EF*)

The USEPA (1989, 1991, 1998) recommended the exposure period for all exposure pathways as 350 days/year. In this recommendation, a minimum of 2 wk/yr absence from the exposure has been assumed.

Exposure duration (*ED*)

The US EPA Office of Solid Waste (OSW) recommends the use of a default reasonable maximum exposure (*RME*) for risk assessment. The US EPA OSW recommends the exposure duration for subsistence fisher and adult resident as 30 years (USEPA 1998).

Gastrointestinal absorption factor (*GI*)

The fraction of chemical that is absorbed by blood from the intestinal tracts is known as the gastrointestinal absorption factor (*GI*). The GI factors for NORM components from USEPA (1999) have been considered in this study.

The cancer risk was calculated as:

$$CR_{RAD} = I_T \times SF \quad (8)$$

where

CR_{RAD} = cancer risk from radionuclides

I_T = total radium intake (pCi)

SF = slope factor (pCi)⁻¹

The total cancer risk was calculated using probabilistic summation concept assuming independence in occurrence of individual risks as:

$$Total\ Cancer\ Risk = R_A + R_B - R_A \times R_B \quad (9)$$

where,

R_A = cancer risk from ^{226}Ra

R_B = cancer risk from ^{228}Ra

Various parameters used in this study are tabulated with their respective distributions in Table 3.

RESULTS AND DISCUSSION

The data of NORM components in produced water is sparse. The mean and standard deviations of ^{226}Ra and ^{228}Ra concentrations in produced water from different platforms were documented by Stephenson (1992); Mulino *et al.* (1992); Hart *et al.* (1996); Vegueria *et al.* (2002); Meinhold *et al.* (1996) and Wiedeman (1996). No statistical distribution could be established for ^{226}Ra and ^{228}Ra concentrations in produced water based on the data from the Gulf of Mexico. In this study, three cases were considered for ^{226}Ra and ^{228}Ra concentrations in the Gulf of Mexico (GOM) as the risk from NORM components would be the highest in this region because the produced water in the Gulf of Mexico has the highest concentrations of NORM components than any other known offshore oil field (Neff 2002).

In Case I, the mean concentrations of ^{226}Ra and ^{228}Ra were used to analyze the cancer risk scenario (Table 3). The probability of risk exceedence for case I is plotted in Figure 2. Case II represents uniform distributions of ^{226}Ra and ^{228}Ra as quoted in Table 3 and the probability of risk exceedence for case II is plotted in Figure 3. Lognormal distributions of ^{226}Ra and ^{228}Ra were used for Case III and Figure 4 shows the probability of risk exceedence for case III. In all the three figures, the probability of cancer risk exceedence is more in the whole body approach. The results of all cases and their risk ratios are shown in Table 4. The mean risks in the whole

body approach are 2.62-2.68 times the risk in the edible part approach. The 95th percentile risks for the whole body approach have been predicted as 2.46-2.57 times the risks in the edible parts approach (Table 4). The USEPA accepts an individual lifetime cancer mortality risks less than 1×10^{-4} to 1×10^{-6} (Federal Register 1991). The exceedence probability of maximum acceptable risk (1×10^{-4}) was close to zero for all the cases. The exceedence probability of the risk level 1×10^{-5} and 1×10^{-6} is presented in Table 5. At a risk level of 1×10^{-5} , the whole body approach predicts 3.6-4.1% more exceedence probability than those of the edible part approach (Table 5; Figures 2-4). The whole body approach predicts 24-27% higher exceedence probability than those of the edible part approach at a risk level of 1×10^{-6} (Table 5). The mean and 95th percentile risks for uniform distributions of NORM components in produced water were highest in both the edible part and whole body approaches compared to the other two cases (Table 4). If the data for NORM components is not available, uniform distributions of NORM components can be employed for a conservative risk calculation. The concentrations of ^{226}Ra and ^{228}Ra in the produced waters from Gulf of Mexico are the highest (Stephenson 1992; Neff 2002) and the human health cancer risk from those concentrations has been predicted as minimal through fish ingestion pathway (Figures 2-4); therefore no or negligible risk to human health is expected from NORM components in produced water. The predicted mean risks for whole body approach were lower than those of Meinhold *et al.* (1996). This may be due to the following causes:

- ❑ higher dilution factor for higher discharge depth and different ambient conditions
- ❑ the probability of exposure (p) for fish to the produced water plume was considered in this study like USEPA (1999)
- ❑ modified risk factors were used by Meinhold *et al.* (1996)

- the total risk was calculated using probabilistic summation concepts.

The frequency distributions of typical risk (Case I) are presented in Figures 5 and 6 for edible part and whole body approaches respectively. In Figure 5, approximately 88% of the cumulative frequency lies within a risk level of 3.4×10^{-6} while Figure 6 shows 64% frequency below a risk level of 3.1×10^{-6} . The figures indicate lower risk in the edible part approach than those of the whole body approach. In the case of canned food ingestion, where both bones and muscles are packed together after processing (includes Tuna, Sardines), the whole fish ingestion concept may be used. In such cases, radium intake is comparatively more and thus higher risk prevails, although the predicted risk in the whole body approach is less than the regulatory limitations.

The skewness of the lognormally distributed concentration data (Case III) is more than Case I and Case II and the use of the largest values of exposure concentrations would provide the most conservative estimate. The maximum exposure concentrations to the continuously exposed marine organisms have been predicted as 24 and 15 pCi/l for ^{226}Ra and ^{228}Ra respectively. Continuous exposure of a migratory marine organism to produced water plume is an extremely rare case. A study on salmon embryos shows that the exposure to 821pCi/l irradiation for approximately 20 days has no impact on mortality (IAEA 1988). At open bays in Louisiana, Meinhold et al. (1996) predicted no effects to aquatic organisms from ^{226}Ra and ^{228}Ra discharged with produced water. In this study, the predicted exposure concentrations of ^{226}Ra and ^{228}Ra are well below the concentration that causes effects to the selected species. It is unlikely that any harmful effect to the marine fish will occur from exposure to produced water NORM components.

SUMMARY AND CONCLUSION

Several models are available for human health risk assessment purposes due to the substantial efforts devoted in the past to assess human health risk from NORM components in produced water. Those efforts usually employed whole body bioconcentration factors and unrealistic worst-case scenarios such as fish limited to the contaminated zone throughout their life. Inter compartmental distribution of radium in bone/skeleton and edible parts of a fish were not taken into consideration and thus the cancer risk from NORM components was overestimated. An approach is presented here for human health risk assessment based on a discharge of produced water from an offshore platform. The approach consists of three main phases: prediction of exposure concentration for fish, estimation of radium concentration in the edible part of fish, and characterization of cancer risk. No or minimal effect to the marine fish is expected from NORM components in produced water. The approach can be applied to other waste effluent discharges like streams with radioactive materials and industrial effluent. A probabilistic analysis using Monte Carlo simulation was used to deal with uncertainty associated with the models and other parameters used in this study. The discussion reveals that the risk associated with NORM components in produced water is well below the allowable limit.

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Table1. Moisture and lipid content in selected species (USEPA 1996)

Species	Moisture Content (%)	Total Lipid Content (%)	Category	Comments
Anchovy, European	73.37	4.101	Finfish	Raw
Bass, Stripped	79.22	1.951	Finfish	Raw
Carp	76.31	4.842	Finfish	Raw
Haddock	79.92	0.489	Finfish	Raw
Halibut, Atlantic and Pacific	77.92	1.812	Finfish	Raw
Halibut, Greenland	70.27	12.164	Finfish	Raw
Herring, Atlantic & Turbot	72.05	7.909	Finfish	Raw
Herring, Pacific	71.52	12.552	Finfish	Raw
Mackerel, Atlantic	63.55	9.076	Finfish	Raw
Ocean Perch, Atlantic	78.80	1.296	Finfish	Raw
Pike, Northern	78.92	0.477	Finfish	Raw
Salmon, Atlantic	68.5	5.625	Finfish	Raw
Salmon, Chinook	73.17	9.061	Finfish	Raw
Salmon, Coho	72.63	4.908	Finfish	Raw
Salmon, Pink	76.35	2.845	Finfish	Raw
Sardine, Atlantic	59.61	10.545	Finfish	Canned in oil
Seatrout, mixed species	78.09	2.618	Finfish	Raw
Trout, Rainbow	71.48	2.883	Finfish	Raw
Trout, mixed species	71.42	5.901	Finfish	Raw
Crab, Alaska King	79.57	N/A	Shellfish	Raw
Lobster, Northern	76.03	0.358	Shellfish	Raw
Shrimp, mixed species	52.86	10.984	Shellfish	Cooked, breaded and fried
	72.56	1.421		Canned
Mussel, Blue	80.58	1.538	Shellfish	Raw
Oyster, Eastern	85.14	1.620	Shellfish	Raw
Squid	78.55	0.989	Shellfish	Raw
	64.54	6.763		Cooked, Fried

Table 2. Concentration factors for edible and non-edible parts in a fish
(adapted from Swanson 1983; Iyengar 1984; Iyengar *et al.* 1980; Hamilton *et al.* 1992;
Meinhold and Hamilton 1992)

Species (1)	Radium Component (2)	CF (flesh) (3)	CF (bone, shell, skeleton) (4)	Ratio (4)/(3)
White sucker	Ra	12	1793	149.4
Lake whitefish	Ra	3	360	120
Lake trout	Ra	1	10	10
Oyster	Ra	50	500	10
Green mussel	Ra	46	419	9.1
Snail (<i>Petalla radiata</i>)	Ra	44	256	5.8
Sole	²²⁶ Ra	55	370	6.7
	²²⁸ Ra	21	160	7.6
Oil sardine	²²⁶ Ra	130	610	4.7
	²²⁸ Ra	44	180	4.1
Ray	²²⁶ Ra	60	65	1.1
	²²⁸ Ra	--	98	
Prawn	²²⁶ Ra	80	360	4.5
	²²⁸ Ra	56	300	--
Moluscs- marine	²²⁶ Ra	44-63	156-500	2.5-11.4
Crustaceans- marine	²²⁸ Ra	35-360	230-800	0.6-22.9
Crab	²²⁸ Ra	35	800	22.9
Clam	Ra	--	200	--
Moluscs	Ra	2-240	--	--

Table 3. Input parameters and their characterization

Model parameters	Characterization	Reference	Comments
Q (Effluent discharge rate m^3/sec)	0.212	Petro-Canada (1996)	
u (Ambient current speed m/sec)	LN (-3.29,0.96)	Mukhtasor (2001)	
z (Depth above discharge m)	11	Mukhtasor (2001)	
BCF (Whole fish bioconcentration factor)	LN (2.99,0.92)	Meinhold and Hamilton (1992); Meinhold <i>et al.</i> (1996)	
x (Edible part of fish %)	LN (4.36,0.063)	US EPA (1996)	
y , (Concentration factor ratio)	LN (2.29, 1.2)	Swanson (1983); Iyengar (1984); Iyengar <i>et al.</i> (1980), Hamilton <i>et al.</i> (1992); Meinhold and Hamilton (1992)	
FIR (Fish ingestion rate g/day)	LN (3.46, 0.62)	Meinhold <i>et al.</i> (1996)	
FR (Fraction of contaminated fish ingested %)	50	US EPA (1997), Schultz <i>et al.</i> (1996); Dellenbarger <i>et al.</i> (1993).	
EF (Exposure frequency $days/yr$)	350	US EPA (1996)	
ED (Exposure duration, yr)	30	US EPA (1996)	
^{226}Ra input (pCi/l)	262	Stephenson (1992)	Case I
^{228}Ra input (pCi/l)	277	Stephenson (1992)	
^{226}Ra input (pCi/l)	U (4,584)	Stephenson (1992)	Case II (Distribution was assumed for comparative purpose)
^{228}Ra input (pCi/l)	U (18, 586)	Stephenson (1992)	
^{226}Ra input (pCi/l)	LN (5.39, 0.6)	Stephenson (1992)	Case III (Distribution was assumed for comparative purpose)
^{228}Ra input (pCi/l)	LN (5.49, 0.53)	Stephenson (1992)	

LN: Lognormal distribution; U: Uniform distribution

Table 4. Risk results

Risk	Case I (Mean concentrations)			Case II (Uniform distribution)			Case III (Lognormal distribution)		
	Whole body approach	Edible part approach	Risk ratio	Whole body approach	Edible part approach	Risk ratio	Whole body approach	Edible part approach	Risk ratio
Mean	2.32E-06	8.83E-07	2.62	2.51E-06	9.51E-07	2.64	2.30E-06	8.60E-07	2.68
Median	9.46E-07	3.02E-07	3.14	9.61E-07	2.88E-07	3.34	8.19E-07	2.88E-07	2.84
Min.	4.66E-09	6.03E-10	7.72	4.20E-09	7.37E-10	5.69	3.47E-09	3.99E-10	8.71
Max.	4.72E-05	5.34E-05	0.88	8.51E-05	8.28E-05	1.03	7.49E-05	4.63E-05	1.62
95 th %	2.60E-06	1.05E-06	2.47	2.86E-06	1.1605E-06	2.46	2.61E-06	1.01E-06	2.57
Std. Dev.	4.56E-06	2.73E-06	1.67	5.56E-06	3.38E-06	1.64	4.92E-06	3.38E-06	1.46

Table 5. Probability of exceeding the maximum acceptable risk of 1×10^{-5} and 1×10^{-6}

Case	Risk level 1×10^{-5}			Risk level 1×10^{-6}		
	Whole body approach (2)	Edible part approach (3)	Col (2)- (3)	Whole body approach (5)	Edible part approach (6)	Col (5)- (6)
Case I	0.043	0.005	0.038	0.48	0.22	0.26
Case II	0.05	0.009	0.041	0.49	0.22	0.27
Case III	0.045	0.009	0.036	0.45	0.21	0.24

Figure 1. Human health risk assessment framework for NORM in produced water

Figure 2. Probability of exceedence to risk level (Case I)

Figure 3. Probability of exceedence to risk level (Case II)

Figure 4. Probability of exceedence to risk level (Case III)

Figure 5. Frequency distribution of risk for edible part approach (Case I)