5. Assessment of Risks to Marine Aquatic Populations Resulting from Exposures to Radionuclides in Arctic Seas

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The RAIG's primary goal in assessing the risks to aquatic populations in the Arctic is to determine whether the concentrations of radionuclides in Arctic waters resulting from potential releases of radionuclides from the FSU are expected to exceed threshold levels of observable effects on the reproductive success of marine organisms. Indications of decreased reproductive success of species are important to decision makers because such decreases may reduce biodiversity, impact endangered species, and change food webs.

A useful way to assess risk is with a tiered approach, each tier progressing to less conservative assumptions and requiring more extensive databases. If, in worst-case scenarios, which include very conservative assumptions (Tier-I assessment), the radionuclide concentrations produce doses or dose rates that are below no-observable-effects levels (NOELs) in the most radiosensitive organisms known, no losses or significant decreases in indigenous species populations are expected in the ecosystems of concern. If they are about equal to or exceed the NOELs, then further assessment is required, and less conservative assumptions are used (Tier-II assessment). If, under Tier-II assumptions, they are about equal to or exceed the NOELs, then site-specific information and population models in the assessment (Tier III) may be needed. The advantage of a tiered approach is that no expenditures of time, effort, and funds are made until the need is documented.

An aquatic population risk assessment requires (1) knowledge of the basic structure and composition of the ecosystems in the areas of concern, (2) information on the responses of organisms to radiation, (3) data on the quantities of radionuclides that potentially may be present in the organisms and their habitat, and (4) understanding of the impact of internal- and external-emitting radionuclides on their reproductive success. In its consideration, the RAIG will address aquatic species that are important ecologically for maintaining the environment's stability and those that may be endangered. It also considers species that are important economically as well as those that are important in Native cultures. First, the RAIG provides a brief overview of the areas of primary interest in the Arctic and briefly describes the different types of ecosystems (including information on some of the major types of food webs and important species), and ecologically and economically important marine populations at risk. Next, it performs a Tier-I assessment of potential doses to biota. This includes the development of standards for protecting aquatic life. The RAIG defines for radiosensitive species the doses and dose rates resulting in no observable effects on mortality, sterility, or fertility (NOELs). The RAIG then compares these doses and dose rates to those that may occur from the FSU radionuclide contamination of the Arctic seas, to those from radionuclides occurring naturally, and to those from fallout radionuclides in the 1960s and 1990s. These comparisons allow the RAIG to draw conclusions about potential detrimental effects of exposures that may occur and those that have occurred in the past.

5.1 Arctic Marine Ecological Systems

Arctic marine ecosystems near Alaska and the FSU are very diverse with seasonally fluctuating physical conditions and, in some areas, massive influx of freshwater. An in-depth review of Arctic marine ecosystems is beyond the scope of this report. An overview of some characteristic marine ecosystems is included to provide the reader with an indication of the biological and geographical diversity of the area and of the incompleteness of some of the databases.

The biological diversity of Arctic ecosystems is generally low (Smith, 1990; Becker, 1994; Dayton et al., 1994), and there are large seasonal differences in the number of individuals present of a given species. Compared to similar temperate species, Arctic species usually are longer lived, have higher adult-survival rates, exhibit deferred age of first reproduction in females, and are characterized by a high-lipid content. Many of the species are migratory and are found only in large numbers in certain locales during the breeding season. Because species diversity is low, the ecosystems are potentially more vulnerable to pollution in general. Long-lived species, depending on their reproductive strategy, may be more vulnerable to radioactivity because of the potential for integration of dose in the reproductive organs with time, whereas high-lipid-content species may be more vulnerable to organic contaminants.

A factor critical to determining the distribution of different types of ecosystems and the variability of productivity in space in the Arctic is the mean minimum and maximum extent of the ice and snow cover. If the ice cover is near land or continuous with the land for most of the year, the kinds and numbers of food webs are limited as well as the number of individuals of a given species. Biota appear to respond to the ice cover either by adapting to the conditions physically or metabolically or by migrating from the area during adverse conditions. In general, primary productivity appears to increase with a decrease in latitude in the Arctic.

Various researchers (e.g., Smith, 1990; Codispoti et al., 1991; Becker, 1994; Dayton et al., 1994; and Savinova et al., 1995) have described characteristics of marine ecosystems in the Arctic. Important factors that limit the types of ecosystems are the light regimes, the low mean and extreme temperatures, the changes in permanent ice cover in some areas, and, in other areas, an ice pack fluctuating seasonally. In addition to phytoplankton, an important source of primary production is ice algae. Growth of ice algae is restricted seasonally and spatially and usually precedes that of phytoplankton (Smith, 1990). Because the ice cover, whether permanent or seasonal, thick or thin, with or without a snow layer, affects light penetration, the overall biological productivity in the Arctic is variable, and primary production is coupled to ice conditions. With the melting of the ice cover and increases in light intensity, blooms are initiated and continue as long as conditions are favorable. There is enhanced productivity at the ice edge, and the blooms in the marginal ice-zone area are a qualitatively significant feature (Smith, 1990). Considerable variability in the time of initiation of spring blooms as well as in overall productivity with area has been reported (Codispoti et al., 1991; Becker, 1994; Dayton et al., 1994; Savinova et al., 1995), and in specific situations where the ice conditions, nutrients, and light duration are appropriate, the productivity may be very high. The overall Arctic primary production rates are now being revised upwards because better data is available on "new" and total productivity and on the distribution of nutrients (Codispoti et al., 1991). Also affecting the characteristics of the ecosystem in a given area is the magnitude of the runoff from land.

A number of areas in the Arctic may be impacted by radioactivity. Those of special interest in our ecological risk assessment are the Kara Sea and Alaskan Shelf. The Kara Sea is included because it is a potential source of radioactivity and is included in the migratory pathway of some marine mammals and fishes consumed by humans. The Alaskan Shelf is important because the coastal areas are inhabited by populations that rely on the stability of the Alaskan fisheries. Of special concern is the part of the Alaskan Shelf that borders on the Beaufort and Chukchi seas because these areas are predicted to receive greater quantities of radionuclides from the Kara Sea than the coastal areas along the Bering Sea. The coastal zone in each of these areas is extensive and the amount of biological information available about each area is not uniform, in part because of the hostile climate and limited accessibility. For some areas there is only qualitative information and for others in-depth investigations were conducted on specific areas and populations (Grebmeier et al., 1989; Highsmith and Coyle, 1990; Grebmeier and Barry, 1991; Feder et al., 1994).

Arctic marine systems have been classified in different ways. In the overview by Savinova et al. (1995), they quote the work of Zenkevitch (1963) who divided the Arctic into subregions: (1) the abyssal Arctic subregion, (2) the shallow lower Arctic subregion, and (3) the shallow high Arctic subregion. The latter includes all the seas on the shelf of the FSU and North American sector, areas of our special interest. Three areas are contrasted by Grebmeier and Barry (1991) and include (1) the northern Bering and Chukchi seas, (2) the high Arctic Ocean with its marginal seas, and (3) the Arctic shelves influenced by warm Atlantic waters. Using these categories, parts of areas one and two include areas relevant to our risk assessment.

The types of ecosystems in the areas of interest include polar ice, benthic-pelagic, offshore pelagic, and coastal lagoons. In the Kara Sea, the polar-ice type ecosystem dominates with a benthic-pelagic type present seasonally; on the Alaskan Shelf along the Beaufort and Chukchi seas, all four types are present; in the Bering Sea, the benthic-pelagic and the offshore pelagic dominate and are relatively productive. It is important to note that even though different geographical areas may have similar types of ecosystems, the food webs and species present within these ecosystems may differ greatly. Appendix A presents information on the Arctic areas of primary interest, on the different types of ecosystems, and on ecologically and economically important marine populations potentially at risk. In addition, Table A-1 provides for the Beaufort and Chukchi seas' shelf areas the types of ecosystems present, some information about the trophic levels present, and a list of some of the species reported for each trophic level. Also, Figures A-1 to A-5 show food webs of important marine mammals.

5.2 Tier-I Assessment of Doses to Biota

5.2.1 Exposure Pathways of Aquatic Populations

It is important to consider the transport and fate of radionuclides in ecosystems and how this affects their availability to the biota and the ultimate radiation dose delivered. In this assessment, exposure pathways are taken into consideration, but it is assumed that the radionuclides are totally biologically available. Analysis of the transport and transformation of those radionuclides identified as potentially important in the Arctic and their partitioning among abiotic and biotic components of the ecosystem were considered previously in Sections 3 and 4. For most organisms, especially pelagic organisms, the most important exposure pathway is from radionuclides dissolved in the water column. Because organisms not only swim or are suspended in the water but also ingest or circulate water over respiratory tissues or through their gut, they may receive both internal and external doses and dose rates from radionuclides in the water.

Suspended and deposited particles can be another important source of radionuclides for some organisms. Particle-to-water distribution coefficients for radionuclides may be greater than 10^4, indicating that radionuclides may have a high affinity for particles and that these particles may be an important source of exposure for some organisms. Consequently, for particle feeders, whether pelagic or benthic, considerable radionuclide doses may be received from particles as well as food that is ingested. For benthic particle feeders, exposure from direct contact with the sediments becomes an important additional route of exposure.

Ingestion of food materials may contribute significantly to radionuclide burdens in biota. For those radionuclides that are transported readily across the gut and assimilated in other tissues, such as ^{137}Cs and ^{90}Sr, the contribution to the radionuclide burden from food may be important for dose considerations.

5.2.2 Dosimetry Models

Considerable attention has been given to calculation of radiation doses and dose rates to biota. Dosimetry models were reviewed in a recent NCRP report (1991) and UNSCEAR report (1996). In the latter, a dosimetry model was defined as follows: "essentially as a mathematical construction that allows the energy deposition in a defined target to be estimated from a given radionuclide (source) distribution." Considerations included in the models are differences in dose absorbed by the biota because of their shape and size, because of the pathways by which they are exposed to the radioactivity, and because of the energies emitted by the radionuclides. In the dosimetry models, concentrations of radionuclides actually measured in the water, particles, and biota were used as well as those predicted from fate and transport models. In some calculations, steady-state conditions are assumed and bioconcentration factors (BCFs) in the biota and K_d values for the particles are used. Models used to estimate doses and dose rates include CRITR, EXTREM III, BIORAD, and Point Source Dose Distribution (PSDD) (UNSCEAR, 1996). The performance of different models was compared and agreement between CRITR and PSDD was reported to be excellent (NCRP, 1991).

The dosimetry models described above assume the same radiation quality factors (biological effectiveness) for nonhumans as for humans. However, there is still disagreement about the appropriate numbers to be applied to different taxonomic groups. In plants and nonhuman animals, the relative biological effectiveness of high-LET alpha particles has not been evaluated, and the use of the human radiation quality factor of 20 was questioned (UNSCEAR, 1996). In humans, the primary concern is with the induction of cancer, a stochastic effect, whereas in domestic and wild flora and fauna, it is the reduced ability to reproduce, a deterministic effect. In mammals, the most generally accepted number is 20, but values as high as 250 to 360 were reported (Jiang et al., 1994). Because in the PSDD approach a radiation quality factor of 20 was used and the proposed one is 5 (UNSCEAR, 1996), the dose-rate calculations given in subsequent tables for radionuclides emitting alpha particles may be high by a factor of 4. If a value in the hundreds is accepted for use, however, our dose-rate predictions may be low by a factor of about 10.

In the past, most of the doses determined for aquatic animals were calculated by one of the dosimetry models described above and primarily for the whole body, even though it is generally understood that the main ecological concern is the dose delivered to the reproductive organs. These types of calculations were made because little information was available on body distribution of radionuclides in aquatic animals. The tissue for which the most information is available is muscle, because this is consumed most frequently by humans. The errors introduced by using doses and dose rates calculated for the whole body instead of for the reproductive organs and early life stages are probably much greater for alpha particles than for beta and gamma emitters. Limited data are available about the transfer of alpha particles across the gastrointestinal tract of nonhumans. Information that would be useful would be the relative concentrations of radionuclides in the muscle tissue to that in the ovary and testis for those species with high radiosensitivity and identified as being critical for population stability.

The RAIG elected the PSDD dosimetry-model approach and used the database developed with that model in this risk assessment. In 1984, Woodhead described a generalized PSDD dosimetry model for bathy/pelagic and benthic fishes, for large bathy/pelagic and benthic crustaceans, for small bathy/pelagic and benthic crustaceans, and for benthic mollusks. (A dosimetry model for primary producers and for mammals was not included). For these seven groups of organisms, he determined the absorbed-dose fraction as a function of energy and considered the contribution to the dose rate from radionuclides dissolved in the water, in suspended and deposited particles, and in food. In a later publication, a generalized set of values per unit water concentration (mSv/h per Bq/m^3) was derived (IAEA, 1988). These values were calculated for 104 nuclides, were for the same seven groups of organisms, and included external exposure from water and particles and internal exposure from radionuclide incorporation. They used K_d and BCF values provided in the IAEA Publication (1985) and the same quality factors as those assumed for humans. The RAIG elected to use the PSDD-dosimetry model approach in this risk assessment and to use the unit-dose-rate conversion factors developed with that model for the radionuclides considered in this risk assessment (Table 5-1). However, because of the UNSCEAR report's recommendation (1996) that the term sievert be restricted to use for humans, the group will refer to the dose rates subsequently as mGy/h per Bq/m^3.

Table 5-1. Unit-dose-rate conversion factors for marine organisms. These factors were determined with the PSDD model and are given as dose rates to a model marine organism resulting from a unit concentration of a given radionuclide in seawater (IAEA, 1988).

Model Organism		Radionuclide ^{90}Sr	^{137}Cs	^{210}Po	$^{239,240}Pu$	^{241}Am
		Unit Dose Rate (mGy/h per Bq/m^3)				
Fish	Bathy/ pelagic	1.3×10^{-9}	1.8×10^{-8}	1.2×10^{-4}	2.4×10^{-6}	3.2×10^{-6}
	Benthic	1.3×10^{-9}	2.3×10^{-7}	1.2×10^{-4}	2.4×10^{-6}	9.6×10^{-6}
Large crustacean	Bathy/ pelagic	1.2×10^{-9}	4.7×10^{-9}	3.0×10^{-3}	1.8×10^{-5}	3.2×10^{-5}
	Benthic	2.4×10^{-8}	2.5×10^{-7}	3.0×10^{-3}	1.8×10^{-5}	3.9×10^{-5}
Small crustacean	Bathy/ pelagic	6.5×10^{-10}	3.8×10^{-9}	1.8×10^{-3}	5.9×10^{-5}	1.3×10^{-4}
	Benthic	1.1×10^{-7}	2.7×10^{-7}	1.8×10^{-3}	5.9×10^{-5}	1.3×10^{-4}
Mollusks	Benthic	5.3×10^{-8}	2.5×10^{-7}	6.1×10^{-4}	1.8×10^{-4}	1.3×10^{-3}

5.2.3 Standards for Protection of Aquatic Life

This assessment will evaluate first the doses and dose rates that were shown to produce significant detrimental effects on organisms and then compare these to the incremental dose rates contributed from the Arctic contamination source terms. Next, it will compare the incremental dose rates to that of an important naturally occurring radionuclide, ^{210}Po, and then to that of background radionuclides of the 1960s and 1990s.

Some efforts have been made to define limits to protect aquatic life (IAEA, 1988; NCRP, 1991). The following three ways for evaluating the potential significance of the impact of increased levels of radioactivity on a population were proposed (NRCP, 1991): "(a) the estimated dose rates may be compared with the variation in the natural radiation background, or indeed, the natural background itself; (b) comparisons may be made with the dose rates which have been shown to produce significant detrimental effects on populations of organisms in laboratory or field studies; (c) if, and when, limits are set on the incremental dose rates for the purposes of environmental protection, comparisons may be made against these criteria."

5.2.4 Doses and Dose Rates Potentially Producing Significant Detrimental Effects

It is well documented that radiation induces biological effects through the deposition of energy in the cells of the irradiated individuals (UNSCEAR, 1993 and 1994). These effects are produced from naturally occurring radionuclides as well as those released from anthropogenic activities. Within the cell, damage to the deoxyribonucleic acid (DNA) of the nucleus is of primary concern (Figure 5-1). If the damage from radiation is repaired, no adverse effects are apparent. When DNA repair is defective or the DNA-repair capacity of the cell is exceeded, the damage may be transmitted to the progeny of the cells. If the effects are produced in the somatic cells, they must become apparent, by definition, within the life of the irradiated organism and a consequence of concern in humans is the induction of cancer.

If the effects are produced in the germ cells, whose function is to transmit genetic information to new individuals, the effects may be detected in the descendants of the irradiated individual in the first or subsequent generations. Because preserving the health of aquatic environments requires insuring the maintenance of very diverse indigenous populations and the survival of individuals of endangered species, we need to understand the impacts of radiation on reproductive success. The types of databases the RAIG will use to assess current and potential exposures to radioactivity of indigenous populations of organisms are the effects of irradiation on mortality and on fertility and sterility, important components of reproductive success.

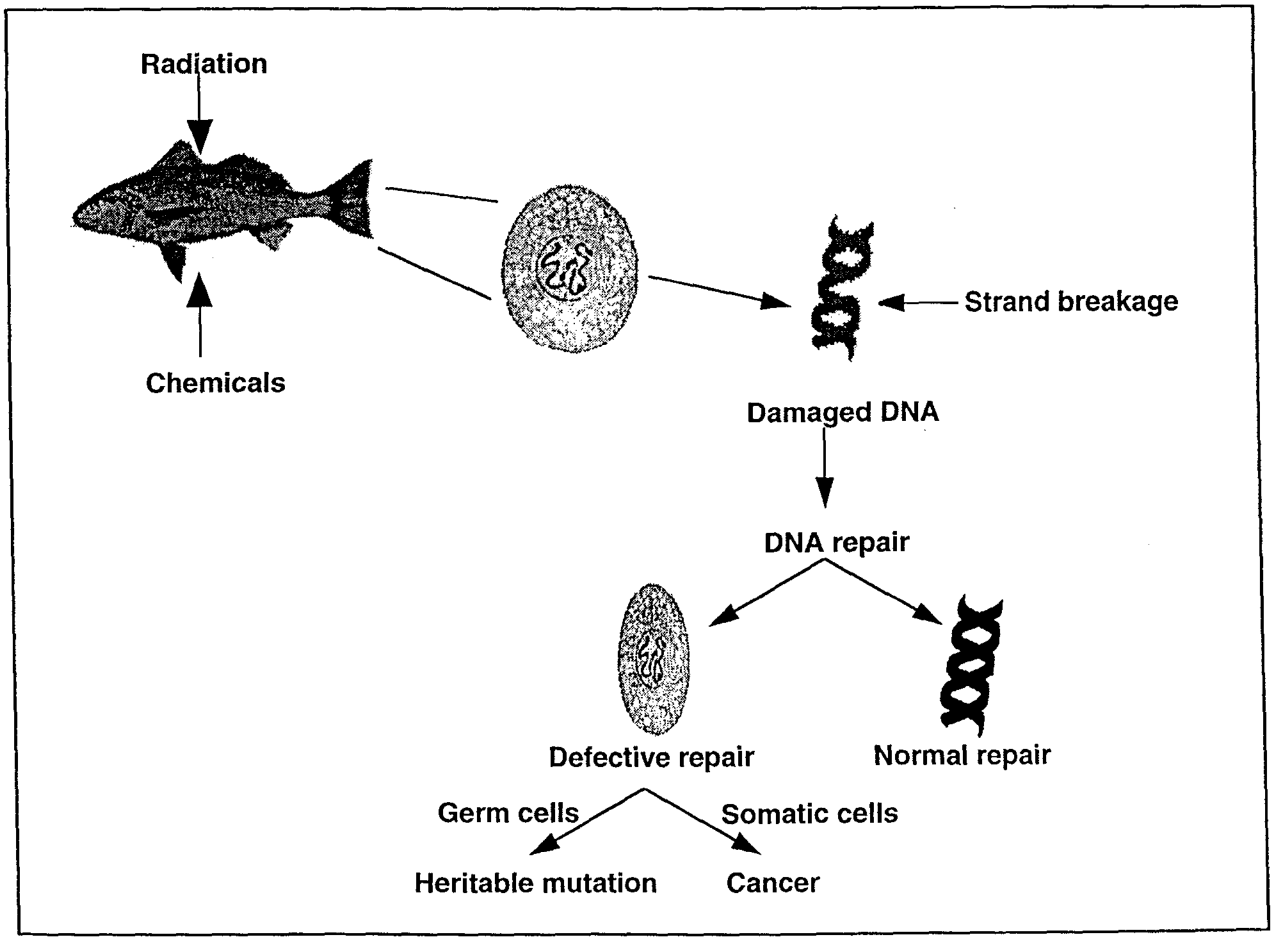

Figure 5-1. Impact of mutagens on genetic material of cells.

One of the largest databases on radiation effects is that on mortality induced after a single irradiation with relatively high doses. Lethal responses were examined in organisms from different phyla and from different types of ecosystems. In general, there appears to be a relationship between radioresistance to high doses of acute radiation and taxonomy of the organism, primitive forms being more radioresistant than complex vertebrates. Unfortunately, because experiments were carried out under different circumstances and for different time periods, the validity of comparing the results from different taxonomic groups is questionable. However, it is clear that mammals are the most sensitive group. Some representative data on mortality of both vertebrate and invertebrate organisms are presented in Appendix B. For our assessment purposes, the relevant information is that even though the range in the mortality response is large (<3 to >30,000 Gy) and the ranges in response for different groups of organisms overlap, it appears that doses less than 1 Gy and dose rates less than 0.1 Gy/h will not result in mortality in marine animal groups.

The databases more relevant than those of mortality for ecological risk assessment are those doses and dose rates potentially producing significant detrimental effects on reproductive success. Of primary concern are those irradiation exposures causing sterility and reduced fertility. Data from studies of the effects of acute and chronic exposure on development of gametes and zygotes indicate that, for some fishes and invertebrates, responses at the cellular and molecular levels show effect levels comparable to those observed in some mammals.

Reproductive success for a given population may be related to several characteristics of the species, including the inherent radiosensitivity of its reproductive tissues and early life stages, specific processes occurring during gametogenesis, and its reproductive strategy and life-style (Woodhead, 1984; Anderson and Harrison, 1986; ICRP, 1991; UNCSEAR, 1996). The RAIG considers inherent radiosensitivity factors to be those controlled by the genetic makeup of the organisms and that determine basic developmental processes and pathways and biological-repair processes (see Appendix C). First, it will review the effects of radiation on fertility and sterility, and then briefly describe how different processes of gametogenesis and strategies of reproduction and life style may affect the number of offspring that survive.

Problems encountered when assessing data on the effects of radioactivity on fertility and sterility are the heterogeneity in the kinds of tests performed. It must be reemphasized that factors other than total dose or dose rate affect the results. These include life stage, physiological factors, and exposure conditions. An example of effects of life stage on response in rainbow trout is shown in Table 5-2; early life-history stages appear to be more sensitive than latter stages. Numerous experiments were performed to characterize the responses of gametes and early life stages of fishes to low levels of radiation. Effects on gametes of fishes were reviewed in Egami and Ijiri (1979). In fishes, irradiation not only may retard development but also alter morphological and physiological characteristics of both early life stages and adults.

Table 5-2. Changes in the radiosensitivity of rainbow trout, *Salmo gairdnerii*, exposed to acute radiation during development (Welander, 1954; Welander et al., 1949).

Stage in Life Cycle	LD_{50} (Gy)
Gamete	0.5–1.0
1 cell	0.58
32 cell	3.1
Germ ring	4.5–4.6
Eyed	4.1–9.0
Adult	15

Acute irradiation doses that affect reproduction can be assessed from laboratory data on doses that have resulted in decreased fertility or sterility (Table 5-3). The results demonstrate that effects of acute irradiation on fertility in mammals, fishes, and invertebrates occur from doses ranging over at least two orders of magnitude (<0.1–20) and that doses between <0.1 and 0.5 Gy appear to define a critical range in which detrimental effects on fertility are first observed in a variety of radiosensitive organisms, but that doses less than 0.05 Gy are expected to have no observable effect. Also, induction of sterility occurs over a range from 1 to 1,000 Gy and doses less than 1 Gy are expected to have no observable effect.

Table 5-3. Comparison of sensitivity of reproductive tissues of invertebrates, fishes, and mammals exposed to acute radiation (dose in Gy). The doses for fertility are those at which significant changes were noted and for sterility were for when the effect was noted.

	Dose[a] (Gy)		
	Fertility	Sterility	References
Invertebrates			
Diaptomus clavipes (copepod, embryos)	10	—	Gehrs et al., 1975
Neanthes arenaceodentata (polychaete worm)	0.5	50	Harrison and Anderson, 1994a
Gammarus duebeni (amphipod, adult)	2.2	—	Hoppenheit, 1973
Artemia salina (brine shrimp, juveniles)	9	21	Holton et al., 1973
Crepidula fornicata (slipper limpet, larvae)	20	—	Greenberger et al., 1986
Physa acuta (freshwater snail, adults)	20	1,000	Ravera, 1966, 1967
Fishes			
Oryzias lalipes (medaka, adult males)	5	—	Hyodo-Taguchi, 1980
Oncorhynchus tschawytscha (chinook salmon, embryos)	2.5	—	Welander et al., 1948
Salmo gairdnerii (rainbow trout, 29-d embryos)	6	—	Konno, 1980
Mammals			
Mice (LD_{50}, primordial follicles)	0.1	1	UNSCEAR, 1982
Rat (LD_{50}, primordial follicles)	0.7	>8	UNSCEAR, 1982
Monkey (LD_{50}, primordial follicles)	10	≥20	UNSCEAR, 1982
Human male	0.1	2–6	UNSCEAR, 1982, 1993
Human female	<0.1 –0.5 [b]	3–10	UNSCEAR, 1982, 1988

[a] The radiation units provided in references were converted to Grays for comparative purposes and for some values are approximations.

[b] The data on sensitivity of oocytes indicate great differences with species as well as with developmental stages. Another complication is that oocytes of fetal primates show radiosensitivities similar to those of the mouse, indicating that *in utero* exposures may be a critical fertility consideration for the human female (Straume et al., 1988).

Data on chronic radiation show that the dose rates resulting in significant changes to fertility in invertebrates, fishes, and mammals had a large range of values (Table 5-4). Comparison of the dose rates affecting fertility shows that there is a range of lower values, 0.02–0.2 mGy/h, that result in detectable changes in fertility in both mammals and nonmammalian species, but that dose rates less than 0.2 mGy/h are expected to have no observable effect. The dose rates known to cause sterility in different species also have a large range, from 0.17 to 1,400 mGy/h, but sterility has not been reported for dose rates less than 0.1 mGy/h.

Table 5-4. Comparison of sensitivity of reproductive tissues of invertebrates, fishes, and mammals exposed chronically to radiation (dose rate in mGy/h). The dose rates for fertility are those at which significant changes were noted and for sterility were for when the effect was noted.

	Dose Rate[a] (mGy/h)		
	Fertility	Sterility	References
Invertebrates			
Neanthes arenaceodentata (worm, single generation)	0.19	20	Harrison and Anderson, 1994b
Ophyrotrocha diadema (worm, seven generations)	3.2	—	Knowles and Greenwood, 1994
Daphnia pulex (water flea, multiple generations)	550	1,400	Marshall, 1962
Fishes			
Ameca splendens (–, single generation)	<0.6	0.6	Woodhead et al., 1983
Poecilia reticulata (guppy, single generation)	1.7	13	Woodhead, 1977
Oryzias lalipes (medaka, adult males)	2.8	840	Hyodo-Taguchi, 1980
Oncorhynchus tschawytscha (chinook salmon, embryos)	4.2	—	Bonham and Donaldson, 1972
Gambusia affinis	13	—	Trabalka and Allen, 1977
Mammals			
Male human	0.05	0.23	UNSCEAR, 1982; 1992
Female human	0.023	—	" " "
Male dog	0.07	0.17	" " "

[a] The radiation units provided in references were converted to Grays for comparative purposes and for some values are approximations.

Another factor that differs greatly from species to species is the differences in doses and dose rates that cause sterility and decreased fertility. In the polychaete worm *Neanthes arenaceodentata,* the difference is two orders of magnitude, whereas for male dogs it is about a factor of two. The database on dose rates resulting in sterility in marine organisms is very limited. The RAIG assumes, however, that induction of sterility in radiosensitive species may occur at dose rates greater than 0.1 mGy/h. Because the database is small, any conclusions about the significance of the difference in dose rates affecting reproduction may not be valid. In a summary of data for mammals (UNSCEAR, 1993), the reader is cautioned that responses are dependent on the developmental stage of the gonadal tissue at the time and duration of the irradiation, and for any species the range in sensitivity may be large. Although such changes in sensitivity are not as well documented in other taxonomic groups, it may be an important factor.

The parameters that are more relevant to protection of ecosystems through limit setting are those values obtained from developmental responses rather than mortality. When comparisons (for the same group of species) are made of the relative radiosensitivity of adults, as measured by mortality, and of the relative radiosensitivity of early stages, as measured by developmental abnormalities or mortality, it is evident that radiosensitivity of early stages as measured by changes during development is not always in the same taxonomic relationship as that of the adult as measured by mortality. Because high radiosensitivity of gonadal tissues and early life stages affects reproductive success directly, radiosensitivity during development is of more concern. Also, the responses of special importance for limit setting are the low values obtained for fertility endpoints because these indicate greater radiosensitivity. It is of interest to note that for invertebrates the low values are in about the same range as those for some fishes and mammals. This indicates that at the cellular and molecular levels, radiosensitivity for these different organisms may not differ much if similar gametogenic stages are exposed.

Reproductive success also is affected significantly by the processes occurring during gametogenesis and their duration. Gametogenesis processes differ greatly from species to species. Important parameters include the ability to repopulate and repair damage to germ cells, the duration and synchrony of stages in gametogenesis, the overall time between production and release of gametes, and the overall time to sexual maturity. Also, within the same species, the responses of male and female gonads are commonly not the same. The testis generally is more radioresistant than the ovary. In some species, sterility requires dose rates and doses to the testis larger than those causing adult mortality. Recovery of gonads from radiation damage may reflect differences between sexes and among species in radioresistance of the stem-cell population and in cells' ability to repopulate.

Reproductive success for a given species may be related also to its reproductive strategy (Woodhead, 1984; Anderson and Harrison, 1986). For example, in a species producing many offspring, the survival of early life stages may be very low, and the loss of abnormal embryos induced from radiation exposure may be masked completely by those lost from other ecological factors, such as food limitation and predation. Species that produce fewer offspring may have strategies for protecting early life stages, such as brooding of the early stages, guarding of nests, and viviparous development (having early life stages develop within the body of the female).

Other important factors of reproductive strategy, in addition to the total number produced, rate of division, and sensitivity of the gametes, include the time between the formation of the primary germ cells and the release of mature gametes. This becomes important in long-lived species, such as whales with a 100-year life span, that are exposed to chronic irradiation, because integration of dose may occur. For some marine mammals and nonmammals, if no repair of radiation damage occurs, the dose to reproductive tissues may be integrated over decades. Unfortunately, in many marine mammals, fishes, and invertebrates the processes involved in radiosensitivity and in gametogenic and reproductive strategies are not known.

Reproductive success of a species in natural ecosystems is affected also by the changes in the population gene pool from multigeneration exposures to radiation. In most radioactivity-contaminated ecosystems, the exposure to the biota is chronic and at low levels, resulting in multigeneration exposures. However, the data available on the effects of this type of irradiation on reproductive success are limited. The database from laboratory studies contains information from only two multigeneration studies (Table 5-4). The duration of most studies was less than a complete life cycle, and the stages in the life cycle irradiated were not always comparable. The database from field studies includes results from multigeneration investigations, but the results from many of these studies were confounded by the presence in the ecosystem of contaminants other than radioactivity (NCRP, 1991). The effect of multigeneration exposure becomes important because the dose-response curves for specific species having the same LD_{50} may differ greatly (Figure 5-2). One can expect that selection of radioresistant individuals will occur upon continuous exposure, and species having a broader range in sensitivity may have a greater survival potential.

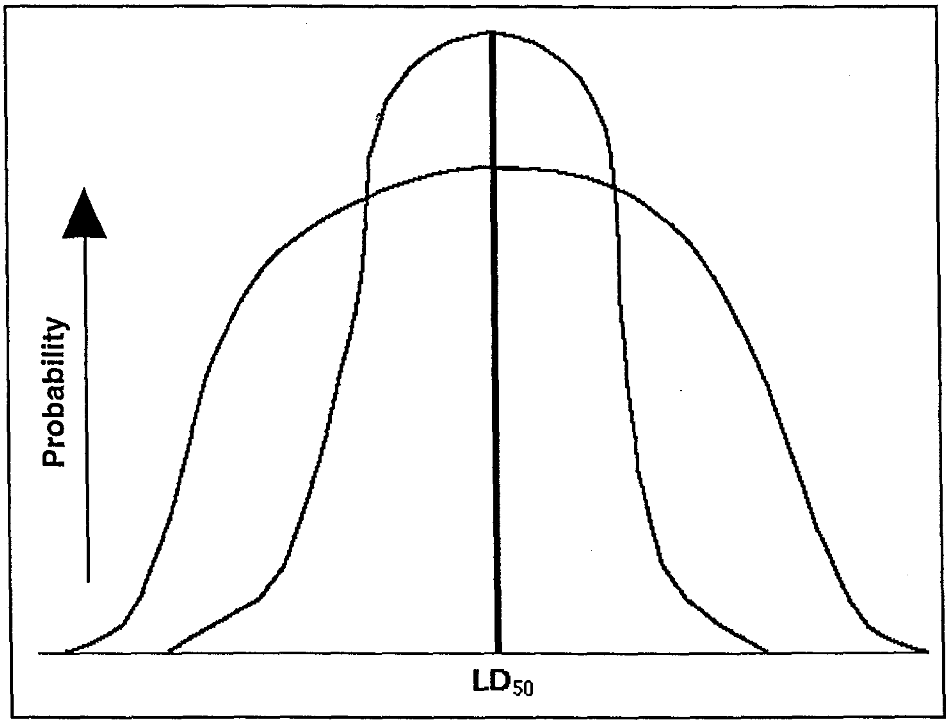

Figure 5-2. For the same LD_{50} value, the shape of species dose-response curve that occurs from radiation may differ significantly.

5.2.5 Dose Rates from Potential FSU Radionuclide Contamination of the Arctic Seas

Dissolved radionuclide concentrations were predicted using the RAIG model for four different scenarios: (1) Kara Sea acute release (instantaneous release and distribution of all known Kara Sea sources), (2) Kara Sea chronic release (predicted time-varying release from Kara Sea source), (3) riverine acute release (instantaneous release and distribution of all known inland sources potentially releasing into the Ob and Yenisey rivers), and (4) riverine chronic release (predicted gradual release and distribution of all known inland sources potentially releasing into the Ob and Yenisey rivers). The predicted dissolved radionuclide concentrations resulting from these release scenarios were highest in the Beaufort Sea (see Section 3). These concentrations and the unit-dose-rate conversion factors (see Table 5-1) were used to calculate the dose rates for bathy/pelagic and benthic fishes, large bathy/pelagic and benthic crustaceans, small bathy/pelagic and benthic crustaceans, and benthic mollusks (Table 5-5).

Table 5-5. Dose rates for fish and invertebrates resulting from background and instantaneous releases of radionuclide inventories in the Kara Sea and Ob and Yenisey rivers. The total doses from anthropogenic radionuclides of concern and from naturally occurring ^{210}Po are provided.

		Dose Rate (mGy/h)					
		Background Radionuclides		Kara Sea Sources		Riverine Sources	
Organism	Nuclide	1960s	1990s	Acute	Chronic	Acute	Chronic
Bathy/pelagic fish	^{90}Sr	1.2×10^{-8}	1.3×10^{-9}	3.9×10^{-11}	7.8×10^{-13}	6.0×10^{-11}	2.7×10^{-11}
	^{137}Cs	2.5×10^{-7}	3.6×10^{-8}	5.8×10^{-10}	1.3×10^{-11}	1.4×10^{-11}	0.0
	^{239,240}Pu	1.2×10^{-7}	1.2×10^{-8}	3.8×10^{-10}	1.5×10^{-12}	0.0	0.0
	^{241}Am	2.9×10^{-8}	3.2×10^{-9}	7.0×10^{-10}	5.8×10^{-13}	0.0	0.0
Total anthropogenic		4.1×10^{-7}	5.2×10^{-8}	1.7×10^{-9}	1.6×10^{-11}	7.4×10^{-11}	2.7×10^{-11}
	^{210}Po	1.2×10^{-7}	1.3×10^{-7}	1.3×10^{-7}	1.3×10^{-7}	1.3×10^{-7}	1.3×10^{-7}
Benthic fish	^{90}Sr	1.2×10^{-8}	1.3×10^{-9}	3.9×10^{-11}	7.8×10^{-13}	6.0×10^{-11}	2.7×10^{-11}
	^{137}Cs	3.2×10^{-6}	4.7×10^{-7}	7.4×10^{-9}	1.6×10^{-10}	1.8×10^{-10}	0.0
	^{239,240}Pu	1.2×10^{-7}	1.2×10^{-8}	3.8×10^{-10}	1.5×10^{-12}	0.0	0.0
	^{241}Am	8.6×10^{-8}	9.6×10^{-9}	2.1×10^{-9}	1.7×10^{-6}	0.0	0.0
Total anthropogenic		3.4×10^{-6}	4.8×10^{-7}	9.9×10^{-9}	1.7×10^{-10}	2.4×10^{-10}	2.7×10^{-11}
	^{210}Po	1.3×10^{-7}	1.3×10^{-7}	1.3×10^{-7}	1.3×10^{-7}	1.3×10^{-7}	1.3×10^{-7}
Large bathy/pelagic crustacean	^{90}Sr	1.1×10^{-8}	1.2×10^{-9}	3.6×10^{-11}	7.2×10^{-13}	5.5×10^{-11}	2.5×10^{-11}
	^{137}Cs	6.6×10^{-8}	9.4×10^{-9}	1.5×10^{-10}	3.3×10^{-12}	3.7×10^{-12}	0.0
	^{239,240}Pu	9.0×10^{-7}	9.0×10^{-8}	2.9×10^{-9}	1.2×10^{-11}	0.0	0.0
	^{241}Am	2.9×10^{-7}	3.2×10^{-8}	7.0×10^{-9}	5.8×10^{-12}	0.0	0.0
Total anthropogenic		1.3×10^{-6}	1.3×10^{-7}	1.0×10^{-8}	2.1×10^{-11}	5.9×10^{-11}	2.5×10^{-11}
	^{210}Po	3.3×10^{-6}	3.3×10^{-6}	3.3×10^{-6}	3.3×10^{-6}	3.3×10^{-6}	3.3×10^{-6}
Large benthic crustacean	^{90}Sr	2.1×10^{-7}	2.4×10^{-8}	7.2×10^{-10}	1.4×10^{-11}	1.1×10^{-9}	5.0×10^{-10}
	^{137}Cs	3.5×10^{-6}	5.0×10^{-7}	8.0×10^{-9}	1.8×10^{-10}	2.0×10^{-10}	0.0
	^{239,240}Pu	9.0×10^{-7}	9.0×10^{-8}	2.9×10^{-9}	1.2×10^{-11}	0.0	0.0
	^{241}Am	3.5×10^{-7}	3.9×10^{-8}	8.6×10^{-9}	7.0×10^{-12}	0.0	0.0
Total anthropogenic		5.0×10^{-6}	6.5×10^{-7}	2.0×10^{-8}	2.1×10^{-10}	1.3×10^{-9}	5.0×10^{-10}
	^{210}Po	3.3×10^{-6}	3.3×10^{-6}	3.3×10^{-6}	3.3×10^{-6}	3.3×10^{-6}	3.3×10^{-6}

Table 5-5, continued.

		Dose Rate (mGy/h)					
		Background Radionuclides		Kara Sea Sources		Riverine Sources	
Organism	Nuclide	1960s	1990s	Acute	Chronic	Acute	Chronic
Small bathy/pelagic crustacean	^{90}Sr	5.9×10^{-9}	6.5×10^{-10}	2.0×10^{-11}	3.9×10^{-13}	3.0×10^{-11}	1.4×10^{-11}
	^{137}Cs	5.3×10^{-8}	7.6×10^{-9}	1.2×10^{-10}	2.7×10^{-12}	3.0×10^{-12}	0.0
	$^{239,240}Pu$	3.0×10^{-6}	3.0×10^{-7}	9.5×10^{-9}	3.8×10^{-11}	0.0	0.0
	^{241}Am	1.1×10^{-6}	1.3×10^{-7}	2.8×10^{-8}	2.3×10^{-11}	0.0	0.0
Total anthropogenic		4.2×10^{-6}	4.3×10^{-7}	3.8×10^{-8}	6.4×10^{-11}	3.3×10^{-11}	1.4×10^{-11}
	^{210}Po	2.0×10^{-6}	2.0×10^{-6}	2.0×10^{-6}	2.0×10^{-6}	2.0×10^{-6}	2.0×10^{-6}
Small benthic crustacean	^{90}Sr	9.9×10^{-7}	1.1×10^{-7}	3.3×10^{-9}	6.6×10^{-11}	5.1×10^{-9}	2.3×10^{-9}
	^{137}Cs	3.8×10^{-6}	5.4×10^{-7}	8.6×10^{-9}	1.9×10^{-10}	2.1×10^{-10}	0.0
	$^{239,240}Pu$	3.0×10^{-6}	3.0×10^{-7}	9.5×10^{-9}	3.8×10^{-11}	0.0	0.0
	^{241}Am	1.2×10^{-6}	1.3×10^{-7}	2.9×10^{-8}	2.3×10^{-11}	1.7×10^{-8}	0.0
Total anthropogenic		8.9×10^{-6}	1.1×10^{-6}	5.0×10^{-8}	3.2×10^{-10}	2.2×10^{-8}	2.3×10^{-9}
	^{210}Po	2.0×10^{-6}	2.0×10^{-6}	2.0×10^{-6}	2.0×10^{-6}	2.0×10^{-6}	2.0×10^{-6}
Mollusks	^{90}Sr	1.1×10^{-5}	5.3×10^{-8}	1.6×10^{-9}	3.2×10^{-12}	2.4×10^{-9}	1.1×10^{-9}
	^{137}Cs	3.5×10^{-6}	5.0×10^{-7}	8.0×10^{-9}	1.8×10^{-10}	2.0×10^{-10}	0.0
	$^{239,240}Pu$	9.0×10^{-6}	9.0×10^{-7}	2.9×10^{-8}	1.2×10^{-10}	0.0	0.0
	^{241}Am	1.2×10^{-5}	1.3×10^{-6}	2.9×10^{-7}	2.3×10^{-10}	0.0	0.0
Total anthropogenic		2.5×10^{-5}	2.5×10^{-6}	3.2×10^{-7}	5.6×10^{-10}	2.6×10^{-9}	1.1×10^{-9}
	^{210}Po	6.7×10^{-7}	6.7×10^{-7}	6.7×10^{-7}	6.7×10^{-7}	6.7×10^{-7}	6.7×10^{-7}

The detrimental effects considered in this risk assessment are the doses and dose rates causing mortality, sterility, and decreased fertility. The database provided above shows that the doses and dose rates affecting these parameters differ widely among the animals examined. In Table 5-6, the ranges of doses and dose rates eliciting these responses are presented as well as the no-observable-effects levels (NOELs) that are adopted in this assessment. For purposes of ecological risk assessment, the following assumptions about the potential effects on radiation on marine biota in the Arctic are made: (1) the doses and dose rates causing no observable effects on fertility in the most radiosensitive species will not result in decreased reproductive success in species of invertebrates, fishes, and mammals and (2) doses and dose rates lower than those causing mortality and sterility in the most radiosensitive species will not result in loss of endangered species or reduced biodiversity in Arctic ecosystems.

Table 5-6. The doses and dose rates resulting in mortality, sterility, and decreased fertility in groups of mammals, fishes, and invertebrates and the recommended no-observable-effects levels (NOELs).

	Dose (Gy)		Dose Rate (mGy/h)	
	Range	NOEL	Range	NOEL
Mortality	<3–>30,000	<1	0.1–0.48	<0.1
Sterility	1–1,000	<1	0.17–1,400	<0.1
Fertility	<0.1–20	0.05	0.023–550	0.02

The predicted highest dose rates (for the specific scenarios addressed) were for mollusks living in the Beaufort Sea. It is expected that the highest dose rates would be obtained for mollusks because many live in or on the bottom sediments and may consume organic-rich, radionuclide-contaminated particles. The highest incremental dose rate from FSU-related sources was for mollusks following a hypothetical instantaneous release into the Kara Sea of contained sources. This dose rate was 3.2×10^{-7}mGy/h, which is about five orders of magnitude lower than our dose-rate NOEL for fertility of 2.0×10^{-2} mGy/h. Therefore, this dose rate should not affect adversely the reproductive success of benthic living organisms.

The predicted lowest dose rate from the total incremental FSU radionuclides was 1.70×10^{-9} mGy/h, which was calculated for bathy/pelagic fishes in the Beaufort Sea after a predicted instantaneous release into the Kara Sea of known sources. Currently, no data are available on fishes from the Beaufort Sea. However, data on irradiation of salmon eggs and larvae indicate that salmon are relatively radiosensitive (Tables 5-2, 5-3, and 5-4). Because the dose rate predicted for bathy/pelagic fishes was about seven (7) orders of magnitude lower than our fertility dose-rate NOEL of 2.0×10^{-2} mGy/h, the RAIG proposes that the radiation dose rates predicted for the Beaufort Sea are too low to affect reproductive success of any species of fishes indigenous to or migrating into the Alaskan seas.

No data on irradiation effects on fertility, sterility, and mortality in marine mammals were identified, and no dosimetry models for them are available. However, let us assume that the dosimetry models for humans can be used for these marine mammals and that the food they consume is similar to that of some Alaska Natives, i.e., a mixture of marine fishes and mammals. The RAIG calculated the doses for different Alaskan coastal communities (Table 6-5) for the Kara Sea instantaneous-release scenario and determined dose rates from 1.2×10^{-4} to 1.2×10^{-1} µSv/yr to the Native populations. Let us assume that the marine mammals might receive the highest dose, to 1.2×10^{-1} µSv/yr, or to 1.4×10^{-8} mGy/h, and that they are no more sensitive than the most sensitive mammal tested to date. Then, this dose of to 1.4×10^{-8} mGy/h assumed for marine mammals is more than 5 orders of magnitude lower than our fertility dose-rate NOEL of 2.0×10^{-2} mGy/h and should result in no marine-mammal population changes.

Marine birds and their eggs comprise a significant fraction of the diets of some Alaskan indigenous populations. As with marine mammals, no data were identified on radiation effects on seabirds. The limited acute-irradiation data on wild and domestic birds was reviewed (UNSCEAR, 1996), and mortality LD_{50}s were reported to be in the same range as small mammals (5-12 Gy). Chronic irradiation in the range of 8.4 to 42 mGy/h to field populations of birds caused embryo mortality, and chronic irradiation at dose rates greater than 10 mGy/h until hatching essentially sterilized both sexes of chickens. Because our dose-rate NOEL for fertility of 2.0×10^{-2} mGy/h is considerably lower than those known to impact bird reproduction, the RAIG concludes that bird populations should suffer no adverse effects.

The response to irradiation of all organisms of most interest to us is effects on fertility. This will be determined not only by dose rate but also on the duration that the species is sexually active. It can be expected that if biological repair is minimal, there is the potential for integration of dose from the time of production of the primary gametocytes to the end of sexual activity. Species of some mollusks, crustaceans, and fishes live as long as 10 years, and most marine mammals have a normal life span of tens of years. Among the species of interest, the one with one of the longest life span is the bowhead whale, which can live as long as 100 years. If worst-case scenarios are assumed, i.e., the dose rate to a female bowhead whale is the maximum for marine species (3.2 10^{-7} mGy/h) for mollusks assuming acute release from Kara Sea sources), there is no biological repair of radiation damage, and the female is sexually active for 50 years, the total dose to the whale is about 1.4×10^{-1} mGy ($3.2 \times 10^{-7} \times 8.76 \times 10^{3}$ h/yr $\times$ 50 yr). This total dose more than two orders of magnitude lower than that of the 0.05 Gy (50 mGy) acute dose affecting fertility in the most sensitive mammal examined.

Migratory patterns may be another factor that affects the exposure of marine species to radioactivity. Again the bowhead whale provides the worst-case exposure. This whale has an extensive migratory range, including most of the Arctic seas. Because it is a planktivorous whale, it has a very short food chain, feeding primarily on phytoplankton and zooplankton. During its migration in the summer to feed in plankton-rich areas in or near the Kara Sea, it may be exposed to waters with higher radionuclide concentrations than those predicted in the Beaufort Sea from acute releases from Kara Sea sources. Thus, until data are available on radionuclide concentrations in the reproductive organs of bowhead whales, considerable uncertainty exists on the actual doses that may be received and the effects that may occur.

5.2.6 Dose Rates from Radionuclides Occurring Naturally

Comparing the quantities received from natural sources to those received from anthropogenic sources provides a perspective from which the harmful effects of increased radiation exposure can be considered. Woodhead (1984), who calculated the dose rates to marine organisms from natural background radiation, global fallout, and waste radionuclides, provided such a perspective. The dose rates in the marine environment caused by radionuclide inputs arising from human activities range from less than the natural background exposure for typical nuclear power stations in routine operations up to a few tenths of mGy/h for the rather exceptional case of when Windscale was discharging large amounts of waste into the northeast Irish Sea.

The naturally occurring nuclide considered to be the most significant contributor to the dose is ^{210}Po (see Section 4). Comparison of the dose rates to mollusks, which have the highest predicted dose rate, from the total FSU incremental anthropogenic radionuclides to those of ^{210}Po shows that the ^{210}Po dose rate is more than twice that from the FSU sources, 6.7×10^{-7} and 3.2×10^{-7} mGy/h, respectively.

Some information is available on the absorbed dose rates from ^{210}Po in specific tissues. For certain pelagic organisms, the dose rate is high, 150 mGy/h in the hepatopancreas and 4 mGy/h in the testis of a small mid-water shrimp, and 30 mGy/h in the intestine and 1 mGy/h in the gonads of a sardine (UNSCEAR, 1996).

5.2.7 Dose Rates from Fallout Radionuclides in the 1960s and 1990s

The dose rates for the same seven groups of organisms in the Beaufort Sea from background radionuclides existing in the 1960s and 1990s were calculated (Table 5-5). In the 1960s, peak concentrations resulted from nuclear-weapons testing, and the dose rate to mollusks from background radiation was 2.47×10^{-5} mGy/h. This dose rate is about two orders of magnitudes higher than the 3.2×10^{-7} mGy/h predicted from the hypothetical instantaneous release into the Kara Sea of all contained sources and about three orders of magnitude lower than our dose-rate NOEL for fertility of 2.0×10^{-2} mGy/h (Table 5-6).

5.3 Uncertainties

5.3.1 Temperature Effects

One area of concern is that very little data are available on the effects of the low temperatures in the Arctic on accumulation by and loss of radionuclides from ecologically and economically important Arctic species. From the information available on other biological systems, it is likely that the accumulation and loss of radionuclides will occur at slower rates, but the concentration factors (ratio of the concentration in the organisms to that dissolved in the water at steady-state conditions) will be the same. It also is likely that the amount of radiation damage accumulated will be the same, but its manifestation and repair will occur at lower rates.

5.3.2 Interaction of Radionuclides with Other Contaminants

In most marine environments, both organic and inorganic contaminants are present. Yet, little information is available on how these may interact to potentially cause damage. The vast majority of laboratory experiments deal with the effects of contaminants singly. Because our concern is effects of contaminants on fertility and the ultimate effect on the reproductive success of populations, of special interest are the interactions of radiation with those organic contaminants reported to be endocrine disrupters. These are organic contaminants that mimic, block, or disrupt the action of natural reproductive hormones, such as estrogen and testosterone, and are reported to have caused significant reproductive effects in wildlife populations (Hileman, 1996).

5.4 Tier-II and Tier-III Risk Assessments

The radionuclide concentrations predicted from the four hypothetical release scenarios into the Kara Sea from FSU radioactive sources were sufficiently low that the dose rates calculated for the biota in the Beaufort Sea were not likely to affect their reproductive success; they were orders of magnitude lower than the dose rate causing a fertility effect in the most radiosensitive mammalian species. In situations where radionuclide concentrations potentially may impact populations because of reduced reproductive success, however, more in-depth assessments are required. For species potentially at risk, the following types of information may be needed:

- Radiosensitivity and duration of the different stages of gametogenesis;
- Reproductive strategy and life style;
- Species domain and niche competitors;
- Species resilience and extinction potential; and
- Effects of interactions with other contaminants.

It is not expected that under current regulations there would be releases that would result in dose rates sufficiently high to require either a Tier-II or Tier-III risk assessment. The exceptions might be if information becomes available that interactions among contaminants result in greatly increased adverse effects or if there is a large-scale nuclear accident.

5.5 Summary

The RAIG assessed the potential risks to Alaskan marine aquatic populations from exposure to radioactive wastes released into the Arctic seas by the former Soviet Union. This risk assessment considered worst-case scenarios, which include very conservative assumptions (the Tier-I approach). The dosimetry model selected for use was the Point Source Dose Distribution (PSDD) model. This model takes into consideration differences in dose absorbed by biota because of their shape and size, because of the pathways by which they are exposed, and because of the energies emitted by the radionuclides and their bioeffectiveness.

The RAIG evaluated first the doses and dose rates potentially producing significant detrimental effects in radiosensitive organisms. The databases used were the doses and dose rates resulting in mortality and in sterility and decreased fertility, important components of reproductive success. Then, for a group of marine organisms, the dose rates were compared to the dose rates predicted in Alaskan marine biota from (1) the worst-case-scenario release of FSU nuclear wastes in Arctic seas, (2) the important naturally occurring radionuclide, ^{210}Po, and (3) the radionuclide levels present in the 1960s and 1990s. These comparisons provided a measure of the potential risk to marine populations in Alaskan coastal waters from the release of FSU-nuclear wastes.

Examination of the data on mortality from acute and chronic irradiation shows that it is expected that doses less that 1 Gy and dose rates less than 0.1 Gy/h will not result in mortality of radiosensitive organisms. The RAIG considers the dose of 1 Gy and the dose rate of 0.1 G/h to be mortality no-observable-effects levels (NOELs), i.e., the dose and dose rate below which no mortality is observed.

Results from the studies of the effects of acute and chronic irradiation on sterility and fertility indicate that for some fishes and invertebrates, responses at the cellular and molecular levels show effect levels comparable to those observed in some mammals. In radiosensitive species, acute doses greater than 1 Gy and dose rates greater than 0.1 mGy/h may induce sterility. The RAIG considers the dose of 1 Gy and the dose rate of 0.1 G/h to be sterility NOELs. Also, in radiosensitive vertebrates and nonvertebrates, doses between <0.1 and 0.5 Gy and dose rates between 0.02 and 0.2 mGy/h define a critical range in which detrimental effects on fertility are first observed. Thus, the RAIG considers the dose of 0.05 Gy and the dose rate of 0.02 mGy/h to be fertility NOELs.

The RAIG predicted the highest dose rates from a hypothetical instantaneous release into the Kara Sea of known FSU sources for mollusks living in the Beaufort Sea (3.2×10^{-7} mGy/h). But this dose rate is five orders of magnitude lower than our fertility dose rate NOEL (2.0×10^{-2} mGy/h) and only about half of that (6.7×10^{-7} mGy/h) from naturally occurring ^{210}Po present in Alaskan marine systems. Because the dose rates predicted using the worst-case scenario were so low, no Tier-II and Tier-III assessments were made.

The conclusions from this risk assessment are the following.

- The doses and dose rates that cause mortality in marine species are not expected to occur in the Arctic except in the case of future releases associated with large-scale accidents, such as that at Chernobyl.

- The doses and dose rates causing sterility are not expected in the Arctic except in the case of future releases associated with large-scale accidents. Thus, no loss of either human or nonhuman species is expected.

- The doses and dose rates causing no detectable effects on fertility in the most radiosensitive species will not result in deleterious fertility effects in: (1) individual or progeny of marine mammals, such as whales, dolphins, or seals; or (2) the reproductive success of aquatic birds, fishes, and invertebrates. However, the very limited database on doses and dose rates resulting in sterility requires extreme conservation in any Tier-II and Tier-III risk assessment until species-specific information is available.

- An important uncertainty is in the exposures that may not cause sterility but impair fertility to the degree that reproductive success is impacted to the point when the species is not able to compete successfully in the ecosystem.

- The increased resilience of populations to continuous exposure to radioactivity expressed (1) through changes in reproductive strategies, (2) by selection of resistant organisms, and (3) from the up-regulation of genes involved in repair and detoxification is not defined, but such processes would make our results even more conservative.

There is no indication that there will be any decrease or loss in indigenous populations or damage to ecosystems through decreases in biodiversity. However, because our NOELs include no consideration of interactions with other contaminants in the ecosystem or physiological differences resulting from low environmental temperatures, marine biota may be at a greater risk than predicted from just radionuclide contamination.