
2. SOURCE-TERM ANALYSES OF POTENTIAL RADIONUCLIDE RELEASES RELEVANT TO THE ARCTIC SEAS AND NORTHWEST PACIFIC OCEAN

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During the years 1960 to 1993, the FSU was responsible for the disposal of radioactive wastes into the Kara Sea; the Barents Sea; Sea of Japan; Sea of Okhotsk; the Pacific Ocean, east coast of Kamchatka; and the West Siberian Basin near the Ob and Yenisey Rivers. These releases came from several marine- and land-based sources. In 1993, the Russian report, *Facts and Problems Related to Radioactive Waste Disposal in Seas Adjacent to the Territory of the Russian Federation* (Yablokov et al., 1993), was released, presenting the findings presented of an October 1992 scientific study commissioned by the Office of the President of the Russian Federation. Related to the Arctic and Northwest Pacific Ocean areas, the *White Book*, as the report was later called, reported the following incidents:

- (1) Between 1965 and 1988, 16 marine reactors from seven FSU submarines and the icebreaker *Lenin*, each of which suffered some form of reactor accident, were dumped at five sites in the Kara Sea. Six of the 16 reactors contained their spent nuclear fuel (SNF);
- (2) Between 1960 and 1991, low-level *liquid* radioactive waste (LRW) was discharged at sites in the White, Barents, and Kara seas;
- (3) Between 1964 and 1991, low- and intermediate-level *solid* radioactive waste (SRW) was dumped at sites in the Barents and Kara seas;
- (4) Ten separate disposal sites were reported in the seas adjacent to the Russian Far East: six in the Sea of Japan, one in the Sea of Okhotsk, and three in the Pacific Ocean, east coast of Kamchatka.

The *White Book* rarely identified specific radionuclides and provided no estimates of the current levels of radioactivity or radionuclide releases to the environment. Supplementing the information presented in the *White Book*, Bradley and Jenquin (1995) have provided important information on the nuclear inventories for major nuclear-waste sites within the West Siberian Basin. The

primary sources of nuclear wastes in the Basin are the nuclear-weapons plants at Mayak, Tomsk 7, and Krasnoyarsk 26, and the nuclear weapons test site at Semipalatinsk. Each of the three weapons-production sites have nuclear reactors and related facilities for producing and processing plutonium (^{239}Pu) for nuclear weapons. Nuclear materials from these sites already have contaminated rivers that flow into the Kara Sea, and existing LRW stored at the various locations constitute a future threat to the river, and to the Arctic Ocean.

To assess the potential risks of these nuclear-waste sources, the RAIG must quantify the time-varying or instantaneous release of environmental radionuclides from a given source (e.g., a submerged reactor vessel or waste pond) to a transport medium (e.g., the Kara Sea or the Ob River). A source-term analysis has three principal components: (1) estimation of the total inventories (i.e., radioactivity) of radionuclides present in a given source; (2) selection of the most important nuclides for subsequent analysis; and (3) prediction of radionuclide releases from a given source to the environment. The RAIG begins the source-term analysis with an overview of waste sources, the identification of radionuclides in the waste sources, and the estimation of the amounts of radioactivity associated with the various nuclides. Because many nuclides are associated with the various sources, the RAIG conducts screening-level analyses to determine which nuclides are potentially most important from a radiological standpoint. Screening analyses incorporate information on parameters involving the estimated inventories, the half-lives for radioactive decay, mobility in the environment, and dose to provide a semi-quantitative ranking of nuclides. Once the key radionuclides are defined, the third phase of a source-term analysis begins: the estimation of continuous or discrete releases of the radionuclides to the environment. Various release scenarios are evaluated to bound the kinds of release mechanisms that could occur over time, for example, chronic releases that result from the corrosion and dissolution of reactor fuels or a catastrophic release caused by some external event.

This section summarizes the FSU disposal sites, radionuclide inventories, and potential release scenarios for sites in the Kara Sea; Sea of Japan; Sea of Okhotsk; the Pacific Ocean, east coast of Kamchatka; and the West Siberian Basin. The RAIG begins, however, with an overview of existing sources of radionuclides in Arctic waters to establish the basis for later comparisons with the FSU sources.

2.1 EXISTING SOURCES OF RADIONUCLIDES IN ARCTIC WATERS

Radionuclides in the Arctic Ocean include those from natural sources (e.g., primordial nuclides such as potassium (^{40}K) and uranium (^{238}U) together with its radioactive progeny) and nuclides derived from anthropogenic sources. From a radiological dose standpoint, the most important naturally occurring radionuclide in seawater is polonium (^{210}Po), an alpha-emitting radionuclide in the ^{238}U decay chain that has a half-life of 140 days (see Noshkin et al., 1994, Aarkrog et al., 1997). This radionuclide is ubiquitous in seawater and seafoods worldwide. The primary anthropogenic sources of radionuclides in Arctic waters include (1) nuclear fuel reprocessing facilities in Europe at Sellafield, United Kingdom, and La Hague, France; (2) global fallout from aboveground nuclear-weapons testing in the 1950s and 1960s; (3) liquid-waste discharges from Russian nuclear installations on the Ob and Yenisey rivers; and (4) the Chernobyl nuclear reactor accident. The principal nuclides of interest from such sources are fission products and actinides.

To put the various anthropogenic sources in perspective, Aarkrog (1994) estimated the contributions of these sources to the total inventories of cesium (^{137}Cs) and strontium (^{90}Sr) in Arctic waters (excluding radionuclides present in waste sites in the Kara Sea), as shown in Table 2-1.

Table 2-1. Estimated inventories of ^{137}Cs and ^{90}Sr in the Arctic Ocean from various sources (excluding ^{137}Cs in Kara Sea waste sites) (Aarkrog, 1994).

Source	Activity, PBq	
	^{90}Sr	^{137}Cs
Global fallout ^a	4.1	4.6
Sellafield discharges	1–2	10–15
Riverine discharges from the former Soviet Union	1–5	1–5
Chernobyl reactor accident	0	1–5
Total	6–11	17–30

^aIncluding runoff from land.

The dominant source of ^{137}Cs is the reprocessing facility at Sellafield, whereas for ^{90}Sr , worldwide fallout is the primary source. Kershaw and Baxter (1995) have provided a detailed review of radionuclide releases from Sellafield and their subsequent transfer to the Arctic Ocean. They report that a total of 41 PBq (i.e., 41×10^{15} Bq) of ^{137}Cs was discharged to the Irish Sea through 1992. Discharges from the site have fluctuated markedly over time, with peak releases for ^{137}Cs and other nuclides occurring in the 1970s. Changes in operations and control measures since that time have resulted in a significant decline in discharges (Gray et al., 1995). For comparison, the La Hague facility has released significantly less ^{137}Cs and ^{90}Sr than Sellafield (i.e., 2.3 and 1.2%, respectively, of the 41 and 6.2 PBq discharges of ^{137}Cs and ^{90}Sr from Sellafield; see Herrmann et al., 1995).

Many studies have dealt with the transport pathway of the soluble nuclides discharged from Sellafield to the Arctic, summarized by Kershaw and Baxter (1995), who indicate that the basic transport pathway of the released radionuclides follows this scheme: Irish Sea to North Sea via the North Channel, and North Sea to Norwegian Sea via the Norwegian Coastal Current where the plume divides into two parts, one leading to the Barents Sea and the other to the central portion of the Arctic Basin. Estimates of the transit time from Sellafield to the Arctic Ocean range from 4 to 6 years (Kershaw and Baxter, 1995). Based on measured concentrations of ^{137}Cs in ocean currents entering the Barents Sea, flow rates of the currents, and the transit time noted above, they estimate that 22% ($\pm 6\%$) of the ^{137}Cs released from Sellafield reaches the Barents Sea as it moves toward the central Arctic Ocean. This transfer factor produces an estimated input of 9 PBq through 1993 (i.e., 0.22×41 PBq). They note, however, that this may represent a “considerable underestimate” of the actual input because of limited data on the entire concentration profile of Sellafield-derived ^{137}Cs in ocean currents entering the Barents Sea.

Releases of $^{239,240}\text{Pu}$ from Sellafield have also reached Arctic waters, based on an analysis of the ratios of $^{238}\text{Pu}/^{239,240}\text{Pu}$, which can be used to distinguish between Pu derived from nuclear-fuel reprocessing facilities and from worldwide fallout (Holm et al., 1986). Specifically, the average $^{238}\text{Pu}/^{239,240}\text{Pu}$ ratio for the Sellafield facility is 0.31, while for global fallout the ratio is about 0.035 (Hallstadius et al., 1986). About 90% of the 590 TBq (i.e., 590×10^{12} Bq) of $^{239,240}\text{Pu}$ released since 1952 from Sellafield is estimated to reside in the sediments of the Irish Sea (Kershaw et al., 1995). Only about 4% of the Pu released is in soluble form, and Holm et al. (1986) estimated that 80% of that amount is removed by sedimentation prior to entering the Barents and Greenland Seas. Thus, nearly 20 TBq of $^{239,240}\text{Pu}$ has reached the Arctic Ocean from Sellafield, and according to Holm et al., the resulting concentrations of $^{239,240}\text{Pu}$ in seawater are 0.6–4 mBq/m³ (or 5–10% of the measured levels in the early 1980s). Baskaran et al. (1995) analyzed data on the concentrations of ^{238}Pu and $^{239,240}\text{Pu}$ in sediment samples from the Kara Sea and determined that $^{238}\text{Pu}/^{239,240}\text{Pu}$ ratio for the samples equals 0.034 ± 0.003 . This ratio is very close to the ratio of these Pu isotopes in worldwide fallout (i.e., 0.035), which suggests that fuel reprocessing facilities have contributed at most only small amounts of $^{239,240}\text{Pu}$ to the Kara Sea.

Other nuclides of potential interest include americium (^{241}Am), technetium (^{99}Tc), iodine (^{129}I), and antimony (^{125}Sb). Hallstadius et al. (1986) have reviewed data on the seawater concentrations of ^{241}Pu and ^{241}Am , which is produced from the decay of ^{241}Pu , concluding that radioactive decay of fallout ^{241}Pu , rather than reprocessing wastes, accounts for most of the ^{241}Am in Arctic waters. This is consistent with the fact that Am is more particle-reactive than Pu, and one would therefore expect that Am adsorption to river sediment (scavenging) would reduce greatly the amounts of Am reaching Arctic waters. Few data are available on ^{99}Tc ; Kershaw and Baxter (1995), however, suggested that this radionuclide has entered the Arctic Basin because of its long half-life and limited adsorption to suspended sediments. Measurements of ^{129}I within the Canadian Arctic Basin and the western Arctic in 1993/1994 have been reported by Kilius et al. (1995) and Beasley et al. (1995). Raisbeck et al. (1995) indicated that about 85% of the ^{129}I released to the North Sea is transferred to the Arctic Seas, and Yiou et al. (1995) estimated that about 90% of the ^{129}I discharged is from La Hague. La Hague has produced about 20 times more ^{125}Sb than Sellafield (Herrmann et al., 1995). Guegueniat et al. (1995) estimate that levels of ^{125}Sb in the Barents Sea that are attributable to La Hague range between 0.10 and 0.15 Bq/m³.

2.2 DESCRIPTION OF MARINE DISPOSAL SITES

2.2.1 Kara Sea Marine Nuclear Reactors

Of the discarded marine nuclear reactors, six of the 16 contained their SNF. In addition, approximately 60% of the SNF from one of the three icebreaker reactors was disposed of in a reinforced concrete and stainless steel (SS) shell container. The vast majority of the low- and intermediate-level SRW was disposed of in containers of unknown composition. The Kara Sea disposal sites for the 16 marine reactors and low- and intermediate-level SRW varied in depth from 12 to 380 m. In particular, the icebreaker reactors and part of their SNF, the single largest source of radioactivity, were reportedly disposed of in Tsviolka Fjord at an estimated depth of 50 m. Figure 2-1 shows a map of Novaya Zemlya with the approximate locations of the five disposal sites.

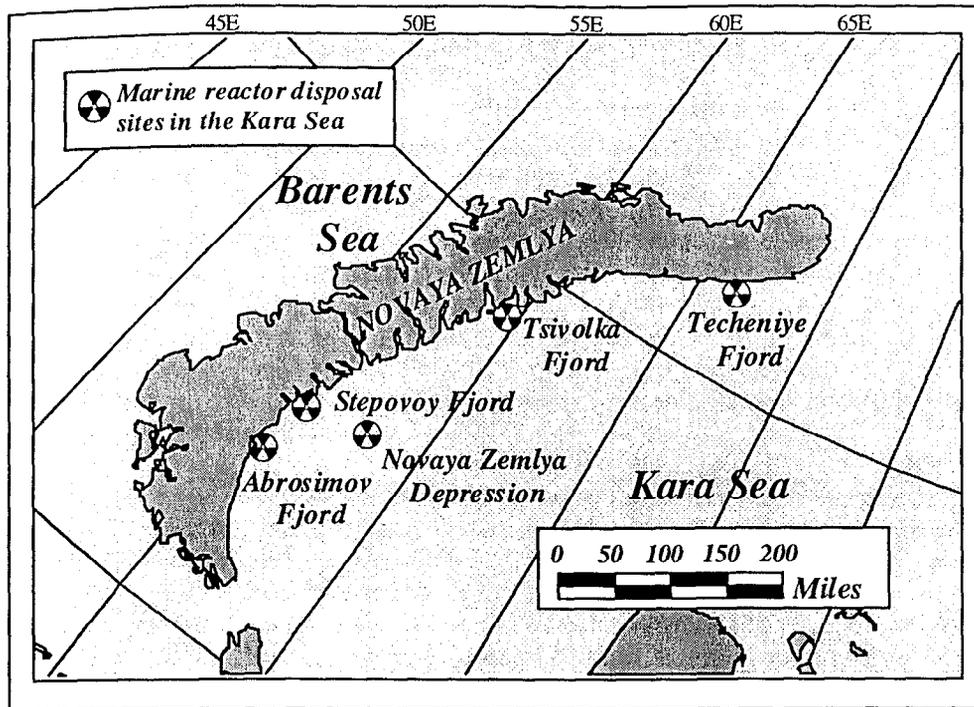


Figure 2-1. A map of Novaya Zemlya showing locations of the disposal sites.

Table 2-2 shows disposal information for the marine reactors dumped in the Kara Sea. IAEA (1997) presents details of the nuclear reactors and their disposal.

Submarine Pressurized Water Reactors

Accident

Six of the seven nuclear submarines contained two pressurized water reactors (PWRs) each. Eleven of these PWRs were dumped into the Kara Sea between 1965 and 1988: eight with and three without their reactor compartments (RCs). All these nuclear submarines suffered some form of reactor accident; however, many specifics of the reactor design, maximum thermal power, compartment layout, detailed operating histories, and accident scenarios remain classified. A criticality accident aboard submarine factory number 421 is known to have caused over-pressurization of the right-board reactor pressure vessel (RPV). The fuel rods were reportedly undamaged; however, a decision was made to not reuse the RPV. As such, the SNF was not removed (Sivintsev, August 1994, March 1995, September 1995, and December 1995).

Disposal

With the exception of the right-board RPV from submarine factory number 421 and the two PWRs from submarine factory number 538, all PWRs were dumped in their separated RCs. The SNF was removed from the left-board RPV of submarine factory number 285 and both RPVs of submarine factory numbers 254, 260, and 538. The SNF remained in the right-board RPV of subma-

Table 2-2. Pertinent disposal information for the marine nuclear reactors dumped in the Kara Sea.

Disposal Site	Year of Disposal	Factory Number	Dumped Unit	Disposal Coordinates ¹	Disposal Depth ² (m)	Number of Reactors		Total Activity (PBq)	
						Without Spent Nuclear Fuel	With Spent Nuclear Fuel	At the Time of Disposal	1994
Abrosimov Fjord	1965	901	Reactor compartment	71° 56.03' N 55° 18.15' E	20 (10-15)	—	2	3.0	0.73
		285	Reactor compartment	71° 56.03' N 55° 18.08' E	20 (10-15)	1	1	12	0.66
		254	Reactor compartment	71° 55.22' N 55° 32.54' E	20	2	—	0.093	0.009
	1966	260	Reactor compartment	71° 56.03' N 55° 18.08' E	20	2	—	0.044	0.005
Tsivolka Fjord	1967	OK-150	Reactor compartment and special container with fuel	74° 26.10' N 58° 36.15' E	50	3	0.6 ³	20	2.2
Novaya Zemlya Depression	1972	421	Reactor	72° 40' N 58° 10' E	300	—	1	1	0.29
Stepovoy Fjord	1981	601	Submarine	72° 31.25' N 55° 30.25' E	50 (30)	—	2	1.7	0.84
Techeniye Fjord	1988	538	Reactors	73° 59' N 66° 18' E	35-40	2	—	0.006	0.005
Total						10	6.6	37	4.7

¹ Disposal site coordinates for all units except those from factory number OK-150 are from Yablokov et al., 1993. Disposal site coordinates for the OK-150 units are from reference Sivintsev, September 1995.

² The disposal depths were provided in May 1993 by the Russian Federation; those in parenthesis were obtained during joint Norwegian-Russian scientific cruises in 1993 and 1994.

³ Thermal shields, hardware, and approximately 60% of SNF discarded in special container.

rine factory number 285, the right-board RPV of submarine factory number 421, and both RPVs of submarine factory number 901 (Sivintsev, August 1994, March 1995, September 1995, and December 1995).

Before disposal, the primary circuit loops and equipment of all PWRs were washed, dried, and sealed. However, there is no indication that the seals were hermetic. Those RPVs containing SNF were filled with Furfurol (F), a hardening compound based on furfural, prior to disposal. Before filling each RPV with Furfurol (F), 28 of the 30 control or compensation rod (CCR) guide tubes were sealed and a 1.5-mm diameter hole was drilled through the upper wall of one. During filling, the RPV was heated, one unsealed CCR guide tube was used as the inlet, the other was used as the outlet, and air was withdrawn through the 1.5-mm diameter hole. Once the process was completed, the 1.5-mm diameter hole was to be capped. However, since the capping of this hole cannot be confirmed, for modeling purposes the hole is assumed to be open to allow ingress of water to the Furfurol (F).

The shallow waters of Abrosimov Fjord were used for four separate disposal operations. Separated RCs from submarine factory numbers 901, 285, and 254 were dumped in 1965 at estimated depths of 50 m (Sivintsev, September 1995), 20 m (Yablokov et al., 1993, and Sivintsev, September 1995), and 20 m (Yablokov et al., 1993), respectively. In 1966, the separated RC from submarine factory number 260 was also dumped at an estimated depth of 20 m (Yablokov et al., 1993). At the time of disposal, the RCs were allowed to flood, thereby exposing a significant portion of the external surface of each RPV and the cavities and internal constructions of those RPVs without SNF to seawater. As such, seawater is assumed to have been within the left-board RPV of submarine factory number 285, and both RPVs of submarine factory numbers 254 and 260 for 30, 30, and 29 years, respectively. The right-board RPV, with its SNF, was removed from the RC of submarine factory number 421, placed into a steel collar-like support structure within the hull of a barge, and covered with concrete. The concrete layer above the RPV lid was about 200-mm thick. The concrete between the outer surface of the RPV wall and the inner surface of the barge hull was no less than 800-mm thick. In 1972, the barge containing the right-board RPV of submarine factory number 421 was dumped in the Novaya Zemlya Depression at an estimated depth of 300 m (Yablokov et al., 1993; Sivintsev, September 1995).

Both RPVs of submarine factory number 538 and their associated steam generators (SGs) and primary circuit pumps, were removed from the RC of the submarine and placed into a steel collar-like support structure within the hull of a barge. The RPV lids and all penetrations into the lids were sealed by welding. No other protective barriers were provided. The barge containing both RPVs, their associated SGs, and their associated primary circuit pumps was sunk in the shallow waters of Techeniye Fjord in 1988 at an estimated depth of 35-40 m (Yablokov et al., 1993). The external surfaces, cavities, and internal constructions of each RPV are assumed to have been exposed to seawater since the time of disposal, a period of about 17 years.

Submarine Liquid Metal Reactors

Accident

The remaining nuclear submarine, designated as factory number 601, contained two liquid metal reactors (LMRs) of 70 MW maximum thermal power each and used Pb-Bi as the coolant or heat transfer medium. The steam-generating installation (SGI) began operation in December 1962 and operated successfully for the duration of the first core load. Both reactors were reloaded in Sep-

tember 1967 and operated at 10% of full power until May 24, 1968, when a portion of the left-board reactor core channels became blocked while the submarine was at sea. As a consequence, about 20% of the left-board reactor fuel was destroyed and deposited in the associated SG and volume compensator via the sealed primary circuit. The submarine subsequently returned to base on power from the right-board reactor, shut down, and was sealed on or about June 6, 1968 (Yefimov, 1994 and March 1995).

Disposal

The SNF remained in the two LMRs of submarine factory number 601. A number of actions were taken to secure the LMRs and prepare the RC for disposal. The primary means was the use of about 2 m³ of Furfurol (F) and 250 m³ of bitumen. In September 1981, over 13 years after the reactor accident, submarine factory number 601 was sunk in the shallow waters of Stepovoy Fjord at an estimated depth of 50 m (Yablokov et al., 1993). At the time of her sinking, the hatches of the RC were open. As such, seawater has been in the compartment above the bitumen filler for over 14 years (Sivintsev, December 1995; Yefimov, 1994, March 1995).

Icebreaker Pressurized Water Reactors

Accident

Launched in Leningrad in 1959, the icebreaker Lenin was the first nuclear merchant ship in the world. During 31 years in commission, the icebreaker had two SGIs. The first SGI contained three PWRs of 90 MW maximum thermal power each and operated from 1959 to February 1965, when during routine repair of the SGI, an operator error allowed the core of the center line (N2) PWR to be left without water for some period of time. As a consequence, a part of the reactor core was damaged because of residual heat. It is this first SGI that forms the basis for the icebreaker source term (Sivintsev, December 1993 and December 1995).

Disposal

Reactor Compartment. All SNF and the core barrel from the N2 reactor were removed from the three RPVs. Before disposal, the primary circuit loops and equipment were washed, dried, and sealed, and the ceiling of the RC was equipped with special pressure relief valves. The icebreaker, with the RC aboard, was towed from Murmansk to Tsivolka Fjord for the disposal operations. On September 19, 1967, the RC with three RPVs was dumped in the shallow water of Tsivolka Fjord at an estimated depth of 60 m directly from the icebreaker through the bottom of the hull. The disposal site was approximately 1 km from the site that was used for the damaged SNF and core barrel from the N2 RPV (Sivintsev, December 1993, March 1995, and September 1995).

Core Barrel and Spent Nuclear Fuel. As a consequence of the accident, only 94 of the 219 technical fuel channels (TFCs) from the N2 RPV could be disposed of in a normal manner. The remaining 125 TFCs and the core barrel from the N2 RPV, hereafter known as Configuration A, were placed within a reinforced concrete and SS shell container, hereafter known as Container B. The voids within the cavity of Container B were filled with Furfurol (F), and the lid was secured by welding. Once sealed, Container B was then moved to a temporary land storage facility constructed of concrete blocks. After about 18 months, Container B was removed from the temporary

storage facility and placed in a specially prepared caisson, hereafter known as Container C, aboard a 6.5-m diameter by 12.5-m long steel pontoon. The walls and lid of Container C were constructed of SS. The voids between Container B and the interior confines of Container C were filled with Furfurol (F), and the lid was secured by welding. Like the icebreaker, the pontoon was towed from Murmansk to Tsiivolka Fjord for disposal. During transit, a storm occurred in the region of the Kara Gate and the pontoon was temporarily lost because the towing cable ruptured. The pontoon was subsequently found, secured to the towing vessel *Lepse*, and towed to Tsiivolka Fjord. On September 18, 1967, the pontoon was dumped within 1 km of the site that would be used for the RC (Sivintsev, December 1993, March 1995, and September 1995).

2.2.2 Sea of Japan; Sea of Okhotsk; Pacific Ocean, East Coast of Kamchatka

The *White Book* (Yablokov et al., 1993) also reported a number of findings with respect to radioactive waste disposal in the seas adjacent to the Russian Far East. Overall, there were 10 separate disposal sites: six in the Sea of Japan, one in the Sea of Okhotsk, and three in the Pacific Ocean, east coast of Kamchatka. Again, like the information reported for the northern seas, the radionuclides were not identified and there was no estimate provided for the current levels of radioactivity or radionuclide release to the environment. Two accidental incidents have occurred that add to the level of radioactive contamination present in the seas adjacent to the Russian Far East: one in the Chazhma Bay, Sea of Japan and one in the Sea of Okhotsk.

Sea of Japan

Accident

On August 10, 1985, a radiation accident occurred aboard a nuclear submarine during refueling at the Pacific Fleet support facility in Chazhma Bay (Yablokov et al., 1993). An uncontrolled chain reaction is reported to have occurred in the left-board reactor. The resulting thermal explosion led to ejection of a fuel assembly from the reactor, a fire in the RC, and the loss of 10 lives. Damage to the submarine included a hole in the pressure hull in the aft section of the RC, which allowed radioactive water to enter and contaminate the seawater. Immediate atmospheric fallout of radionuclides following the explosion was reported to cover an area within a radius of 50 to 100 m around the submarine. Later, the fallout of aerosol particles were reported to cover seawater for a distance of up to 30 km from the accident site. The radionuclide release to the atmosphere was calculated at about 270,000 TBq, of which about 81,000 TBq were noble gases. Furthermore, of the total radionuclide release, the ^{137}Cs and ^{90}Sr fission product inventories were calculated to be 1.9 MBq (i.e., 1.9×10^6 Bq) and 1.5 GBq (i.e., 1.5×10^9 Bq), respectively (Soyfer, 1995).

Disposal

The Sea of Japan disposal sites (designated as Areas 1, 2, 5, 6, 9, and 10) varied in depth from 1.1 to 3.7 km and were used for the disposal of both LRW and SRW. At the time of the preparation of the *White Book*, the characteristics of the LRW were: (1) disposal dates between 1966 and 1992, (2) a total activity at the time of disposal of about 440 TBq of unspecified origin, and (3) a total volume of 82,892 m³. The characteristics of the SRW were: (1) disposal dates between 1968 and

1992, (2) a total activity at the time of disposal of about 170 TBq of unspecified origin, and (3) a total volume of 18,753 m³ within 5,332 containers, 35 ships, and 40 unenclosed objects.

Since the publication of the *White Book*, LRW was dumped again in Area 9 on October 17, 1993 (Danilyan and Vysotsky, 1995). The disposal reportedly covered a depth that varied from the surface to 2 m and an area 48 km long and 200–400 m wide. The total activity released to the sea was 0.014 TBq and consisted of ¹³⁷Cs (76%), ⁹⁰Sr (21%), ⁶⁰Co (1.5%), and ¹³⁴Cs (1.5%). Calculations of the dispersal of the radioactivity performed subsequent to the disposal operations indicated that background concentrations were reached within 15–25 hours after disposal. Table 2-3 presents a detailed summary of the pertinent disposal information contained in the *White Book* for the LRW and SRW dumped in the Sea of Japan, including the disposal-site coordinates.

Sea of Okhotsk

Accident

A 13,000-TBq ⁹⁰Sr radioisotope thermoelectric generator (RTG), a power source of the type used to power a remote lighthouse, was reported lost at sea during a helicopter transport operation near the Sakhalin Island coast in 1987 (Yablokov et al., 1993). No other specifics of the accident or general construction of the RTG are available.

Table 2-3. Pertinent disposal information for the liquid and solid radioactive waste dumped in the Sea of Japan, as presented in the White Book (Yablokov et al., 1993).

Site Designation	Site Coordinates		Disposal Depth (km)	Time Period	Liquid Waste		Solid Waste					
	North Latitude	East Latitude			Total Volume (m ³)	Total Activity (TBq)	Form of Disposal			Total Volume (m ³)	Total Activity (TBq)	
							Containers	Ships	Unenclosed Objects			
Area 1	42° 0'	133° 10'	3.25 - 3.7	Not specified	16,250	0.056	—	—	—	—	—	
	42° 0'	134° 30'										
	41° 0'	133° 10'										
	41° 0'	134° 30'										
Area 2	41° 10'	131° 10'	2.9 - 3.3	Not specified	3,156	0.033	—	—	—	—	—	
	41° 10'	134° 30'										
	39° 30'	131° 10'										
	39° 30'	134° 30'										
Area 5	42° 26'	131° 37'	1.1 - 1.5	1966 - 1974	3,830	3.4	—	—	—	—	—	
	42° 26'	132° 20'		1986	259	0.006	—	—	—	—	—	
	42° 17'	131° 37'		1988	1,808	0.62	—	—	—	—	—	
	42° 17'	132° 20'		1990 - 1992	1,939	0.27	—	—	—	—	—	
Area 6	41° 55'	131° 47'	1.9 - 3.3	1968 - 1971	—	—	2,455	—	—	2,455	13	
	41° 55'	132° 13'		1973	—	—	241	2	—	861	6.8	
	41° 45'	131° 47'		1986 - 1987	5,072	18	—	—	—	—	—	
	41° 45'	132° 13'		1986 - 1988	-	-	219	3	2	1,093	12	
Area 9	41° 46'	133° 22'	3.25 - 3.7	1974 - 1978	14,827	2.2	990	10	-	2,809	34	
	41° 46'	134° 42'		1984 - 1992	18,143	400	698	10	26	7,017	79	
	41° 36'	133° 22'										
	41° 36'	134° 42'										
Area 10	41° 10'	131° 15'	2.9 - 3.3	1978 - 1983	—	—	729	10	12	4,518	23	
	41° 10'	131° 35'		1979 - 1984	17,608	20	—	—	—	—	—	
	40° 10'	131° 15'										
	40° 10'	131° 35'										
Total					82,892	440	5,332	35	40	18,753	170	

Disposal

The Sea of Okhotsk disposal site (designated as Area 3) depth was unspecified and was only used for the disposal of LRW. The *White Book* indicated that a total volume of 1,513 m³ of LRW was disposed, containing 0.004 TBq of unspecified radionuclides. Table 2-4 presents a detailed summary of the pertinent disposal information contained in the *White Book* for the LRW dumped in the Sea of Okhotsk, including the disposal site coordinates.

Table 2-4. Pertinent disposal information for the liquid radioactive waste dumped in the Sea of Okhotsk, as presented in the *White Book* (Yablokov et al., 1993).

Site Designation	Site Coordinates		Disposal Depth (km)	Time Period	Liquid Waste	
	North Latitude	East Latitude			Total Volume (m ³)	Total Activity (TBq)
Area 3	53° 0'	146° 40'	Not available	Not available	1,513	0.004
	53° 0'	148° 10'				
	51° 20'	146° 40'				
	51° 20'	148° 10'				

Pacific Ocean, East Coast of Kamchatka

The Pacific Ocean, east coast of Kamchatka, disposal sites (designated as Areas 4, 7, and 8) varied in depth from 1.4 to 2.57 km and were used for the disposal of both LRW and SRW. According to the *White Book*, 34,289 m³ of LRW were disposed of between 1966 and 1992, and contained a total activity at the time of disposal of about 13 TBq of unspecified origin. SRW were disposed between 1969 and 1992, with a total activity at the time of disposal of about 110 TBq of unspecified origin. The wastes included 1,502 containers, 2 ships, and 64 unenclosed objects.

Figure 2-2 shows the disposal sites for LRW in the Sea of Japan; Sea of Okhotsk; and the Pacific Ocean, east coast of Kamchatka. Figure 2-3 shows the disposal sites for SRW in the Sea of Japan, Sea of Okhotsk, and the Pacific Ocean, east coast of Kamchatka. Table 2-5 presents a detailed summary of the pertinent disposal information contained in the *White Book* for the LRW and SRW dumped in the Pacific Ocean, east coast of Kamchatka, including the disposal site coordinates.

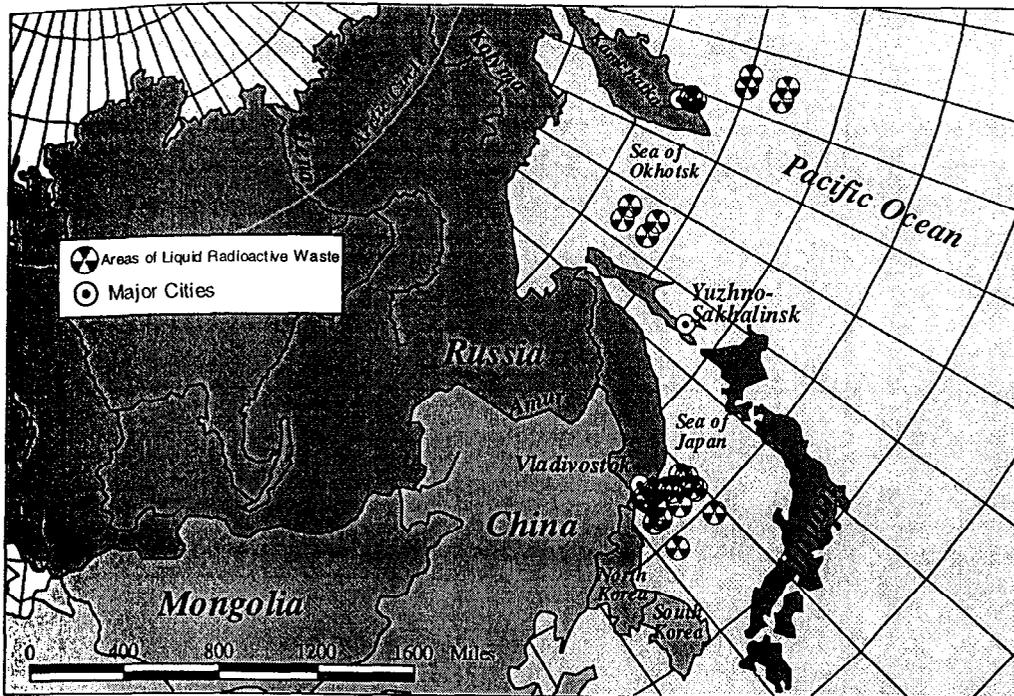


Figure 2-2. Liquid radioactive waste disposal sites in the Sea of Japan; Sea of Okhotsk; and the Pacific Ocean, east coast of Kamchatka.

Figure 2-3. Solid radioactive waste disposal sites in the Sea of Japan; Sea of Okhotsk; and the Pacific Ocean, east coast of Kamchatka.

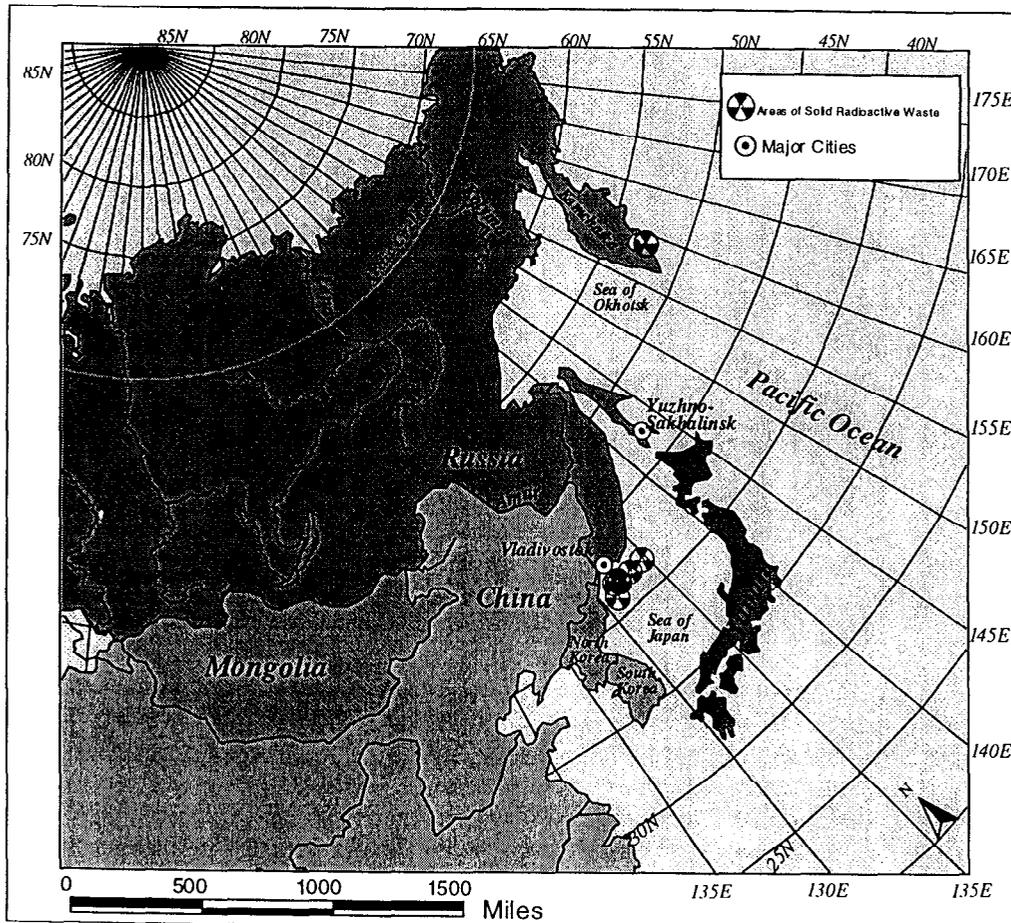


Table 2-5. Pertinent disposal information for the liquid and solid radioactive waste dumped in the Pacific Ocean, east coast of Kamchatka, as presented in the White Book (Yablokov et al., 1993).

Site Designation	Site Coordinates		Disposal Depth (km)	Time Period	Liquid Waste		Form of Disposal			Total Volume (m ³)	Total Activity (TBq)
	North Latitude	East Latitude			Total Volume (m ³)	Total Activity (TBq)	Containers	Ships	Unenclosed Objects		
	Area 4	50° 0'			161° 35'	Not specified	Not specified	4,803	0.007		
	50° 0'	162° 45'									
	48° 0'	161° 35'									
	48° 0'	162° 40'									
Area 7	52° 40'	159° 2'	1.4 - 1.5	1966 - 1975	10,456	0.27	—	—	—	—	—
	52° 40'	159° 12'		1977 - 1978	3,851	0.23	—	—	—	—	—
	52° 28'	159° 2'		1980 - 1992	19,982	12	—	—	—	—	—
	52° 28'	159° 12'									
Area 8	52° 34'	159° 6'	2.0 - 2.57	1969 - 1978	—	—	460	1	—	758	19
	52° 34'	159° 11'		1980 - 1983	—	—	297	—	—	297	13
	52° 28'	159° 2'		1986 - 1992	—	—	745	1	64	1,498	79
	52° 28'	159° 11'									
Total					34,289	13	1,502	2	64	2,553	110

2.3 SOURCES IN THE WEST SIBERIAN BASIN

The primary sources of nuclear wastes in the West Siberian Basin, as depicted in Figure 2-4, are the nuclear-weapons plants at Mayak, Tomsk 7, and Krasnoyarsk 26 and the nuclear-weapons test site at Semipalatinsk. Each of the three weapons-production sites have nuclear reactors and related facilities for producing and processing ^{239}Pu for nuclear weapons. Nuclear materials from these sites already have contaminated rivers that flow into the Kara Sea, and existing LRWs stored at the various locations constitute a future threat to the rivers (see Bradley and Jenquin, 1995).

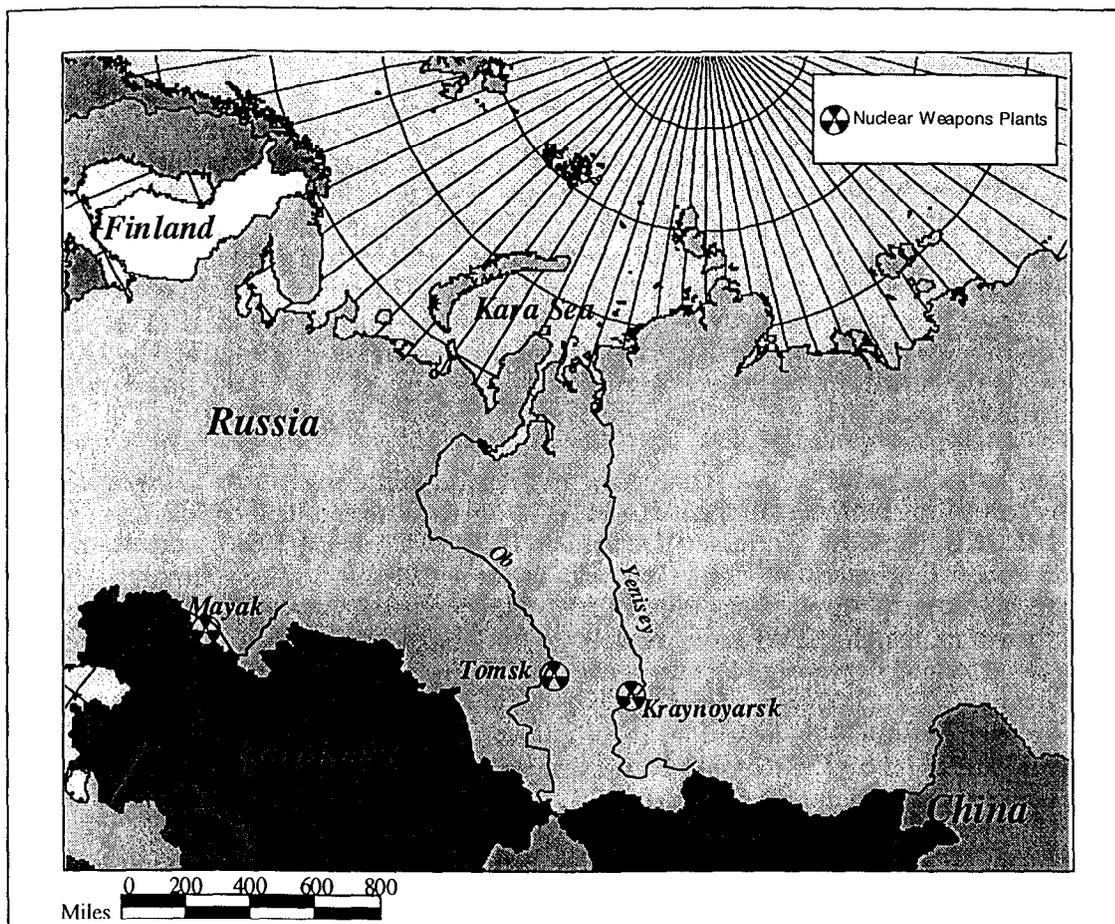


Figure 2-4. Locations of nuclear-weapons plants in the West Siberian Basin.

The LRWs produced from these sites have been managed in various ways. During the years 1949 to 1951 at the Mayak site, for example, LRWs were discharged directly to the Techa River (a tributary of the Ob), contaminating marshes, river water, and related sediments for hundreds of kilometers downstream (Trapeznikov et al., 1993). Later, the LRWs were diverted to Lake Karachai (denoted Reservoir 9) and a series of other surface reservoirs. High-level radioactive wastes are now stored in single-wall tanks. At Tomsk 7 and Krasnoyarsk 26, LRWs also are stored in surface reservoirs; however, the principal method of LRW disposal at those two sites has been the injection of wastes into subsurface geologic media via special wells. Discharges of once-through cool-

ing waters from the nuclear reactors at Tomsk 7 (to the Tom River, a tributary of the Ob) and Krasnoyarsk 26 (to the Yenisey River) also have resulted in the contamination of river water and sediments with fission products, actinides, and activation products. The dominant radionuclides from those releases, though, are short-lived activation products such as ^{51}Cr , ^{32}P , and ^{59}Fe (Vakulovsky et al., 1995). At the nuclear test site at Semipalatinsk, the principal source of potential contamination to the Irtysh River (a tributary of the Ob) is runoff from lands contaminated with radionuclides from nuclear testing.

2.4 RADIONUCLIDE INVENTORIES

2.4.1 Kara Sea

The *White Book* reported estimates of total radioactivity at the time of disposal; these were 85,000 TBq of fission products in the SNF; 3,700 TBq of activation products in the reactor components; 890 TBq of unspecified origin in the low-level LRW, over 50% of which was discharged in the Barents Sea; and 590 TBq of unspecified origin in the low- and intermediate-level SRW, over 95% of which was discarded in the Kara Sea (Yablokov et al., 1993). With rare exception, the report identified no radionuclides and provided no estimate for the current levels of radioactivity. The methods and models used to quantify the radionuclide inventories are described by IAEA (1996), which presents the results of the Source Term Working Group (STWG) of the International Arctic Seas Assessment Project (IASAP).

Two independent estimates were prepared for the radionuclide inventories. One estimate was prepared by consulting members of the IAEA STWG from the Russian Research Center "Kurchatov Institute" (RRCKI), Moscow (Sivintsev, December 1993 and August 1994), and the Institute of Physics and Power Engineering (IPPE), Obninsk (Yefimov, 1994). Another estimate was prepared by the State Institute of Applied Ecology (SIAE), Moscow (Rubtsov and Ruzhansky, 1995). In preparing the estimates, the STWG core models for the PWRs and LMRs were represented by the icebreaker OK-150 and submarine factory number 601 core models, respectively, while the SIAE estimates used a VVER-1,000 core model to represent both the PWR and LMR cores. Values of the fuel burn-up used in the STWG inventory calculations came from RRCKI and IPPE records; those for SIAE were supplied by the Russian Navy. Computer programs used in both inventory estimates are well established and benchmarked.

Results from the STWG estimate, when compared with those from SIAE, showed the following:

- (1) Fission products are in good agreement for the icebreaker and no worse than a factor of 0.5 for the nuclear submarines; and
- (2) Actinides agree within a factor of 0.5 for the icebreaker and are no worse than a factor of 0.1 for the nuclear submarines.

Upon consideration of the above, the STWG concluded that even though SIAE results tend to be higher, and therefore more conservative, they do not represent the best estimate for the IASAP effort. First and foremost, the core models used in the STWG estimate for the icebreaker and submarine factory number 601 represent the actual configurations; the SIAE models do not. Second, even though there are differences between the core configurations of the nuclear submarine

and icebreaker PWRs, the OK-150 model is more representative of the true core configurations than that of the VVER-1,000. This is further substantiated by the fact that comparisons of the STWG actinide results to those in other Russian reports (Sivintsev, September 1995) indicated differences of no more than $\pm 20\%$. Table 2-6 shows estimated 1994 radionuclide inventories of fission products, activation products, and actinides in the marine reactors dumped in the Kara Sea. Table 2-7 shows estimated 1994 activity of long-lived radionuclides at each disposal site from the marine reactors dumped in the Kara Sea. With respect to an upper estimate of the limit on the total radioactivity dumped in the Kara Sea, the STWG estimate of 37,000 TBq is approximately a factor 2.4 less than the 88,000 TBq estimate of the *White Book*, and approximately a factor of 6 less than the 210,000 TBq estimate obtained from an independent calculation of the reactor fuel burn-up based on the operating characteristics and power requirements of the vessels (Mount et al., 1994).

One other potential source of radioactivity in the SNF associated with the dumped reactors is a nuclear criticality event. Criticality occurs when, on average, more than one fission event is produced per fission neutron generated. The key parameters influencing a criticality event are the geometry of the fissionable material and its composition. The STWG examined alternative mechanisms that could lead to criticality in SNF (e.g., corrosion of control rod material, leading to enhanced neutron fluency in residual fuel) and concluded that in some special circumstances the nuclear fuel could achieve a critical state. Nevertheless, the amount of additional fission products created in such an event would be insignificant compared to the estimated inventories presented in Table 2-7.

Table 2-6. Estimated 1994 radionuclide inventories of fission products, activation products, and actinides in the marine reactors dumped in the Kara Sea (from IAEA, 1997; based on October 1995).

Factory Number	Activity in 1994							
	Fission Products		Activation Products		Actinides		Total	
	Bq	Percent (%)						
901	7.2×10^{14}	15	6.0×10^{12}	0.13	3.4×10^{12}	0.073	7.3×10^{14}	15
285	6.3×10^{14}	14	1.3×10^{13}	0.27	8.1×10^{12}	0.17	6.5×10^{14}	14
254	-	-	9.5×10^{12}	0.20	-	-	9.5×10^{12}	0.20
260	-	-	5.1×10^{12}	0.11	-	-	5.1×10^{12}	0.11
OK-150 ¹	1.8×10^{15}	39	2.3×10^{14}	5.0	8.3×10^{13}	8	2.2×10^{15}	46
421	2.9×10^{14}	6.1	2.9×10^{12}	0.062	2.8×10^{12}	0.061	2.9×10^{14}	6.2
601	5.3×10^{14}	11	3.0×10^{14}	6.5	3.6×10^{11}	0.008	8.4×10^{14}	18
538	-	-	4.5×10^{12}	0.096	-	-	4.5×10^{12}	0.096
Total	4.0×10^{15}	86	5.7×10^{14}	12	9.7×10^{13}	2.1	4.7×10^{15}	100.00

¹ The fission product, actinide, and 27% of activation product activities were discarded in a reinforced concrete and stainless steel container.

Table 2-7. Estimated 1994 activity of long-lived radionuclides at each disposal site from the marine reactors dumped in the Kara Sea (from IAEA, 1997; based on October 1995).

Radionuclide	Abrosimov Fjord	Tsvolka Fjord	Novaya Zemlya Depression	Stepovoy Fjord	Techeniye Fjord	Kara Sea Total
<i>Fission products</i>						
³ H	3.1×10^{12}	8.3×10^{11}	1.6×10^{11}	4.8×10^{13}	—	5.2×10^{13}
⁸⁵ Kr	1.1×10^{13}	1.7×10^{13}	3.3×10^{12}	7.2×10^{12}	—	3.9×10^{13}
⁹⁰ Sr	3.2×10^{14}	4.4×10^{14}	7.4×10^{13}	1.2×10^{14}	—	9.5×10^{14}
⁹⁰ Y	3.2×10^{14}	4.4×10^{14}	7.4×10^{13}	1.2×10^{14}	—	9.5×10^{14}
⁹⁹ Tc	9.6×10^{10}	1.3×10^{11}	1.9×10^{10}	3.0×10^{10}	—	2.8×10^{11}
¹²⁵ Sb	2.8×10^{10}	5.8×10^{10}	2.2×10^{10}	5.9×10^{10}	—	1.7×10^{11}
¹²⁹ I	9.4×10^7	2.1×10^8	1.9×10^7	6.5×10^7	—	3.9×10^8
¹³⁷ Cs	3.5×10^{14}	4.8×10^{14}	6.8×10^{13}	1.3×10^{14}	—	1.0×10^{15}
¹³⁷ Bam	3.4×10^{14}	4.6×10^{14}	6.5×10^{13}	1.2×10^{14}	—	9.9×10^{14}
¹⁴⁷ Pm	5.7×10^{11}	1.4×10^{12}	4.7×10^{11}	9.2×10^9	—	2.4×10^{12}
¹⁵¹ Sm	7.8×10^{12}	1.1×10^{13}	1.7×10^{12}	3.7×10^{12}	—	2.4×10^{13}
¹⁵⁵ Eu	—	—	—	9.6×10^{10}	—	9.6×10^{10}
Subtotal	1.4×10^{15}	1.8×10^{15}	2.9×10^{14}	5.3×10^{14}	—	4.0×10^{15}
<i>Activation products</i>						
¹⁴ C	1.5×10^{11}	1.6×10^6	1.2×10^{10}	—	2.4×10^9	1.6×10^{11}
⁶⁰ Co	6.2×10^{12}	4.2×10^{13}	9.8×10^{11}	9.3×10^{13}	1.0×10^{12}	1.4×10^{14}
⁵⁹ Ni	4.9×10^{12}	1.8×10^{12}	1.0×10^{11}	1.6×10^{12}	1.9×10^{11}	8.6×10^{12}
⁶³ Ni	2.2×10^{13}	1.8×10^{14}	1.8×10^{12}	1.4×10^{14}	3.3×10^{12}	3.5×10^{14}
¹⁵² Eu	—	—	—	6.0×10^{13}	—	6.0×10^{13}
¹⁵⁴ Eu	—	—	—	1.1×10^{13}	—	1.1×10^{13}
²⁰⁵ Pb	—	—	—	1.9×10^8	—	1.9×10^8
²⁰⁷ Bi	—	—	—	1.7×10^{10}	—	1.7×10^{10}
²⁰⁸ Bi	—	—	—	6.2×10^9	—	6.2×10^9
²¹⁰ Bim	—	—	—	3.4×10^9	—	3.4×10^9
Subtotal	3.4×10^{13}	2.3×10^{14}	2.9×10^{12}	3.0×10^{14}	4.5×10^{12}	5.7×10^{14}
<i>Actinides</i>						
²³⁸ Pu	5.2×10^{11}	1.0×10^{12}	8.7×10^{10}	8.6×10^9	—	1.6×10^{12}
²³⁹ Pu	7.5×10^{11}	5.0×10^{12}	1.1×10^{11}	3.4×10^{11}	—	6.2×10^{12}
²⁴⁰ Pu	3.4×10^{11}	2.3×10^{12}	3.9×10^{10}	6.6×10^9	—	2.7×10^{12}
²⁴¹ Pu	8.8×10^{12}	6.7×10^{13}	2.4×10^{12}	2.8×10^9	—	7.8×10^{13}
²⁴¹ Am	1.1×10^{12}	7.1×10^{12}	1.6×10^{11}	1.1×10^9	—	8.3×10^{12}
Subtotal	1.2×10^{13}	8.3×10^{13}	2.8×10^{12}	3.6×10^{11}	—	9.7×10^{13}
Total	1.4×10^{15}	2.2×10^{15}	2.9×10^{14}	8.4×10^{14}	4.5×10^{12}	4.7×10^{15}

Table 2-8. Estimated 1994 activity of long-lived activation products in the reactor-related unenclosed solid radioactive waste dumped in the Sea of Japan and the Pacific Ocean, east coast of Kamchatka.

Site Designation ¹	Year of Disposal ¹	Dumped Unit ¹	Disposal Coordinates ¹	Disposal Depth (km) ²	Radionuclide			
					¹⁴ C	⁶⁰ Co	⁵⁹ Ni	⁶³ Ni
					Activity in 1994 (GBq) ³			
Area 8	1986	Primary loop circulating pump (50 pieces)	52° 31' N 159° 8' E	2.0 - 2.57	0.00021	500	0.018	3.3
	1988	Steam generator (10 pieces)	52° 30' N 159° 9' E	2.0 - 2.57	0.00031	990	0.028	5.0
	1989	Submarine core plate	52° 30' N 159° 9' E	2.0 - 2.57	0.90	1,000	29	330
		Primary loop circulating pump (50 pieces)	52° 30' N 159° 9' E	2.0 - 2.57	0.0000045	16	0.00040	0.073
Area 9	1991	Steam generator (5 pieces)	41° 40' N 134° 0' E	3.25 - 3.7	0.000035	160	0.0030	0.56
Area 10	1978	Two submarine reactors	41° 10' N 131° 15' E	2.9 - 3.3	0.59	180	19	200
	1983	Reactor lid (8 pieces)	41° 40' N 131° 26' E	2.9 - 3.3	1.2	690	39	420
Total					2.7	3,500	87	960

¹ Information is that presented in the White Book (Yablokov et al., 1993).

² The exact depth of disposal was not specified. The depth presented is that associated with the area designation in the White Book.

³ Inventory is based on the total activity at the time of disposal as presented in the White Book.

2.4.2 Sea of Japan and Pacific Ocean, East Coast of Kamchatka

Reactor-related parts, such as primary loop circulating pumps, SGs, a reactor core plate, RPVs, and a reactor lid, reportedly were dumped as unenclosed objects in both the Pacific Ocean, east coast of Kamchatka (Area 8) and the Sea of Japan (Areas 9 and 10) (Yablokov et al., 1993). In the case of the reactor core plate, RPVs, and reactor lid, the radionuclides contained in each result from neutron activation of the SS or low-alloy steel from which they are constructed. For the primary loop circulating pumps and SGs, the radionuclides contained results from primary system corrosion. Whether activation or corrosion product, the long-lived radionuclides of consequence are the same: ^{14}C , ^{60}Co , ^{59}Ni , and ^{63}Ni . The difference is in the relative quantity of each.

Using information reported for the activation product inventories in the submarine PWRs without SNF that were dumped in the Kara Sea (Sivintsev, August 1994), the RAIG can estimate the activation product inventories in the reactor core plate, RPVs, and reactor lid. Conservatively, assuming that each reactor-related object was dumped a year after reactor shutdown, the RAIG can estimate the inventory from the product of the total reported activity at time of disposal and the average fraction of ^{14}C , ^{60}Co , ^{59}Ni , and ^{63}Ni contained in the submarine PWRs without SNF at 1 year after shutdown. Simple radioactive decay then provides the inventory in 1994.

In a similar manner, the RAIG also can estimate the corrosion-product inventories in the primary loop circulating pumps and SGs. With one exception, the procedure is exactly the same as that for the activation products. The difference is the source of the fraction of ^{14}C , ^{60}Co , ^{59}Ni , and ^{63}Ni contained in the corrosion products: a British calculation for a generic nuclear submarine one year after shutdown (House of Commons Defense Committee, 1990).

Table 2-8 presents the estimated 1994 activity of long-lived activation products in the reactor-related unenclosed SRW dumped in the Sea of Japan and the Pacific Ocean, east coast of Kamchatka. Overall, the reactor-related unenclosed solid objects in 1994 contained about 4.6 TBq of radioactivity, with ^{60}Co and ^{63}Ni constituting 77% and 21% of the total, respectively. Furthermore, the reactor core plate in Area 8 and the reactor lid in Area 10 were the two greatest sources of this radioactivity. Figure 2-5 shows the disposal sites for the reactor-related unenclosed solid objects in the Sea of Japan and the Pacific Ocean, east coast of Kamchatka.

2.4.3 Sea of Okhotsk

Between 1987 and 1990, scientists conducted extensive annual surveys of the area in which the ^{90}Sr RTG was lost (Danilyan and Vysotsky, 1995), but they located no RTG and observed no detectable activity. Assuming the reported strength of the ^{90}Sr RTG source (13,000 TBq) to be that at the time of its loss near the Sakhalin Island coast in 1987, the RAIG assigned the 1994 activity a concentration of 11,000 TBq.

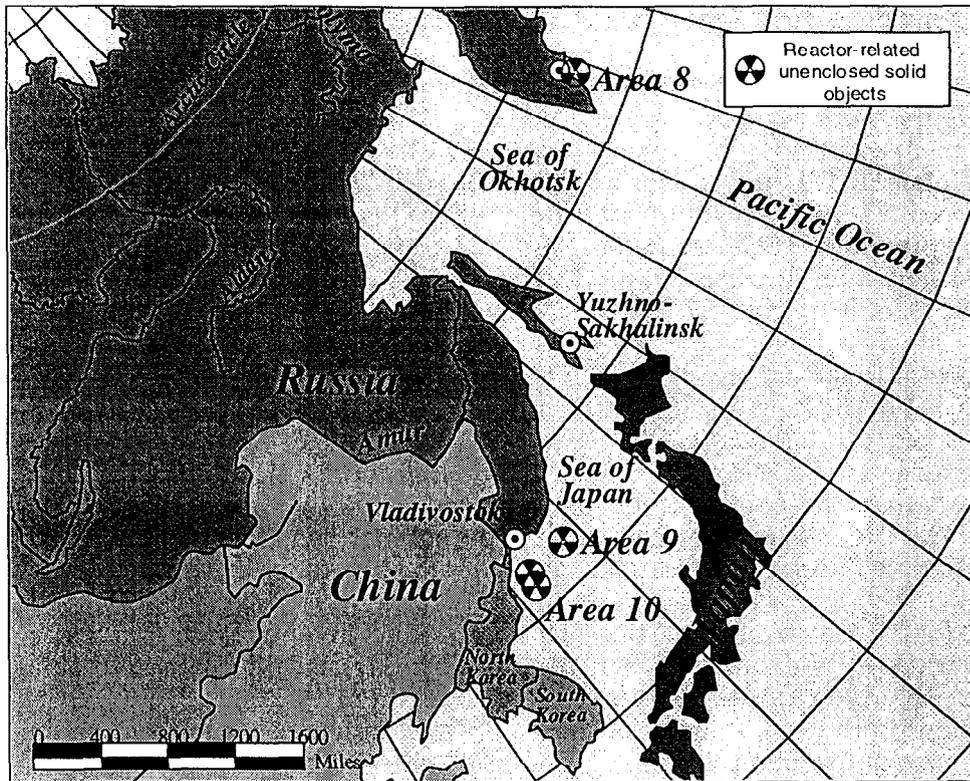


Figure 2-5. Disposal sites for reactor-related unenclosed solid objects in the Sea of Japan and the Pacific Ocean, east coast of Kamchatka.

2.4.4 West Siberian Basin

Bradley and Jenquin (1995) have prepared a review of the nuclear inventories for the major sites within the West Siberian Basin. Table 2-9 summarizes estimates of the total activities for the sites and includes a qualitative assessment of the likelihood of release from a given source to an adjacent river. Deep-well injection of LRW accounts for 85% of the total radionuclide inventory, and given the geologic isolation of these injected wastes, they are not considered to be available for riverine transport. The remaining waste sources are basically associated with existing surface reservoirs and marshes that contain elevated levels of dissolved radionuclides. The inventories associated with sites categorized in Table 2-9 as having some potential for migration are as follows:

- 4,800,000 TBq (for Tomsk-Tom River);
- 4,900,000 TBq (for Mayak-Techa River);
- 5,800 TBq (Krasnoyarsk-Yenisey River); and
- 3,300 TBq (Semipalatinsk-Irtysh River).

Table 2-9. Summary of the total radionuclide inventories for nuclear wastes at Tomsk 7, Krasnoyarsk 26, and Mayak (from Bradley and Jenquin, 1995).

Site/Source	Current Activity, TBq (% of total)	River Affected	Status
Tomsk-7/Injection	3.7×10^7 (58.0%)	Tom-Ob	Contaminant migration reported to be minimal, because of depth of discharges, contamination of Tom-Ob unlikely in near future
Krasnoyarsk-26/Injection	1.7×10^7 (26.7%)	Yenisey	(same as above)
Tomsk-7/Reservoirs	4.8×10^6 (7.5%)	Tom-Ob	Unknown, some migration possible
Mayak/Reservoir 9	4.4×10^6 (6.9%)	Techa-Ob	Already migrating into river system
Mayak/Solid Waste	4.4×10^6 (0.7%)	Techa-Ob	Likely marginal contributor
Mayak/Reservoir 17	74,000 (0.1%)	Techa-Ob	Already migrating into river system
Mayak/Techa Reservoirs	7,400 (<0.01%)	Techa-Ob	Already migrating into river system
Mayak/Production Reactors	4,900 (<0.01%)	Techa-Ob	Already migrating into river system
Krasnoyarsk-26/Production Reactors	3,900 (<0.01%)	Yenisey	Inventory in Yenisey River and has migrated at least 1,500 km
Semipalatinsk	3,300 (<0.01%)	Irtysch-Ob	Derived from weapons testing, likely migrating into river systems
Krasnoyarsk-26/Reservoirs	1,900 (<0.01%)	Yenisey	Unknown, some migration to Yenisey possible
Tomsk-7/Production Reactors	1,400 (<0.01%)	Tom-Ob	Inventory in Tom River and likely migrating
Mayak/Tank Explosion	1,600 (<0.01%)	Techa-Ob	Small contributor to Techa River contamination
Ob-Yenisey Rivers	1,200 (<0.01%)	Kara Sea	Amount of Sr^{90} (1,100 Tbq) and Cs^{137} that has migrated to the Kara Sea from the West Siberian Basin
Mayak/Wind Erosion	22 (<0.01%)	Techa-Ob	Likely marginal contributor
Nuclear-Power-Reactor Operations	3 (<0.01%)	Ob	Discharges to marshes at Beloyarsk reactor site, unlikely contributor

Although estimates of the total radionuclide inventories available for riverine transport are useful in determining the relative magnitudes of the different sources, the actual radionuclide composition of the wastes must be characterized to assess the potential risks associated with those sources. In this regard, little is known about the radionuclide composition of the LRWs stored at Tomsk or Krasnoyarsk, although they are probably similar to the wastes at the Mayak facility. Table 2-10 presents the inventories of ^{90}Sr and ^{137}Cs that are associated with the LRWs stored in the lakes and reservoirs at Mayak (see Figure 2-6). These inventories do not reflect the radioactivity in the reservoir sediments. The largest inventories are in Lake Karachai (also referred to as Reservoir 9), which poses no direct threat for an inadvertent discharge to the Techa River, but the transport of radionuclides in groundwater from the aquifer beneath the lake is a source of long-term discharge to the Techa River. Lake Kyzyltash (Reservoir 2) drains to Reservoir 3, but there is apparently no threat of a catastrophic release from the lake, as there is a natural shoreline serving as a barrier. The wastewater reservoirs that constitute a threat to the Techa River are primarily Reservoirs 3, 4, 10, 11, and 17. The dominant radionuclide in those particular reservoirs is ^{90}Sr , with an estimated inventory of 9,200 TBq, which is about a factor of 100 higher than the inventory of ^{137}Cs . Tritiated water represents about 2,500 TBq of the total inventory (from Bradley and Jenquin, 1995). Measured concentrations of ^{90}Sr and ^{137}Cs in the Techa River below Reservoir 11 show the same relationship between those two nuclides in the radioactive liquid wastes, that is, ^{90}Sr levels are about a factor of 100 greater than those for ^{137}Cs (see Trapeznikov et al., 1993).

Table 2-10. Inventories of ^{90}Sr and ^{137}Cs dissolved in wastewaters stored in reservoirs associated with Mayak facility (from Bradley and Jenquin, 1995).

Reservoir	Volume m^3	^{90}Sr		^{137}Cs	
		Concentration Bq/m^3	Inventory Bq	Concentration Bq/m^3	Inventory Bq
2 ^a	8.4×10^7	4.1×10^5	3.4×10^{13}	1.7×10^5	1.4×10^{13}
3	7.8×10^5	5.9×10^7	4.6×10^{13}	7.4×10^6	5.8×10^{12}
4	4.2×10^6	6.3×10^6	2.6×10^{13}	2.7×10^6	1.1×10^{13}
6	1.8×10^7	1.4×10^4	2.4×10^{11}	7.4×10^2	1.3×10^{10}
9 ^b	4.0×10^5	6.3×10^{10}	2.5×10^{16}	4.4×10^{11}	1.8×10^{17}
10	7.7×10^7	1.3×10^7	9.9×10^{14}	3.2×10^5	2.4×10^{13}
11	2.2×10^8	1.9×10^6	4.1×10^{14}	7.4×10^2	1.6×10^{11}
17	3.0×10^5	2.6×10^{10}	3.9×10^{15}	1.5×10^8	2.2×10^{13}

^aLake Kyzyltash

^bLake Karachai

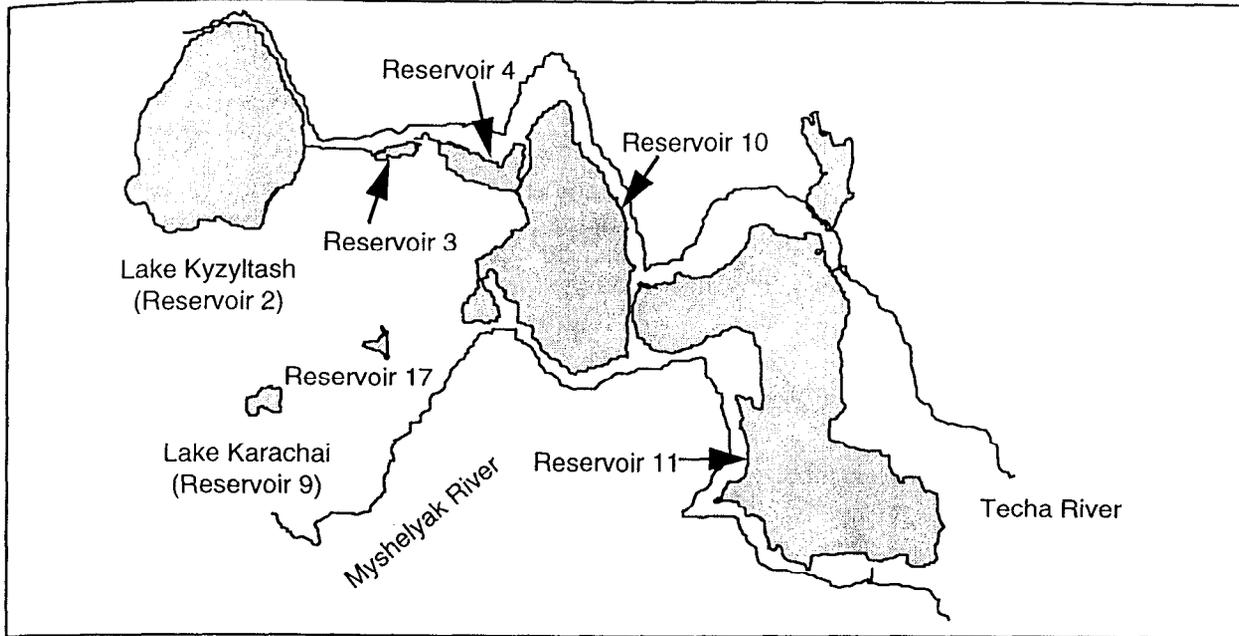


Figure 2-6. The reservoirs constructed to prevent Techa River contamination from liquid radioactive discharges from the Mayak operation.

2.5 RADIONUCLIDE SCREENING

The nuclear wastes dumped into the Arctic and Pacific Oceans by the FSU, and those currently entering or postulated to enter through the Russian river systems, contain a variety of radionuclides. Some of these radionuclides, though, are far more important than others in terms of their potential radiological impact. Therefore, to provide a focus for our assessment of the risks posed by the various nuclear wastes, the RAIG performed a screening analysis to identify those radionuclides that could pose the greatest potential risks, and conversely, to identify those that pose the least risk. The screening analysis was divided into two parts, the first addressing the Kara Sea as a source area (for disposed wastes and riverine influges from Russian rivers) and the second part, the wastes dumped in the northwest Pacific Ocean.

2.5.1 Kara Sea and West Siberian Sources

The relative importance of each radionuclide present in the nuclear wastes dumped in the Kara Sea and at nuclear facilities in the Siberian Basin is a function of several factors. The first is the overall quantity, or inventory, of each radionuclide. The more of any radionuclide there is in the Arctic Ocean, the greater its potential importance as a radiological hazard. The second factor is the radiological half-life of each radionuclide. If a radionuclide decays quickly, it will be less important than one that remains radioactive for a longer time. The third factor is the solubility of the radionuclides in seawater; relatively insoluble radionuclides such as plutonium and americium adsorb to suspended particulate matter and are scavenged from the water column as the particles settle to the sea bottom. Particle scavenging of such nuclides, therefore, reduces their mobility in the marine environment and, subsequently, potential human exposures. A fourth

factor is the bioavailability of the radionuclides, that is, the effectiveness by which a given radionuclide is absorbed or taken up into marine organisms that are part of food chains. Finally, the human metabolism and retention of radionuclides, and their dosimetry, are also important factors.

The RAIG has based the radionuclide screening methodology on the estimation of potential human population doses resulting from the long-term ingestion of fish, marine mammals, and mollusks contaminated as result of potential radionuclide releases to the Arctic Ocean. For the i th radionuclide, the collective (population) dose at time t is

$$D_i(t) = \text{Pop} \left[\sum_{j=1}^n I_j \times \text{BCF}_{ij} \right] \times \text{DCF}_i \times C_i(t) \quad (2-1)$$

where

- $D_i(t)$ = annual population dose for the i th radionuclide in year t , person-Sv/yr;
- I_j = annual average intake of the j th seafood item, kg/yr;
- Pop = number of individuals consuming contaminated seafood, persons;
- BCF_{ij} = bioconcentration factor of the i th radionuclide for the j th seafood item, m^3/kg ;
- DCF_i = dose-conversion factor for ingestion of the i th radionuclide, Sv/Bq; and
- $C_i(t)$ = time-varying concentration of the i th radionuclide, Bq/ m^3 .

For screening analysis, the RAIG can estimate the time-varying concentration of a given radionuclide by assuming that its estimated inventory is instantaneously released to the Arctic Ocean, with no credit taken for containment properties of the waste forms. This is clearly a conservative assumption; however, it does allow for a dose-based ranking that uses the estimated radionuclide inventories. The time-varying concentration of a radionuclide in the Arctic Ocean at time t after an instantaneous release is calculated as

$$C_i(t) = \frac{S_i}{V} \exp [-(\lambda_d^i + \lambda_w + \lambda_s^i)t], \quad (2-2)$$

where

$$\lambda_d^i = \frac{0.693}{T_{1/2}^i}, \quad (2-3)$$

$$\lambda_w = \frac{W}{V}, \quad (2-4)$$

$$\lambda_s^i = \frac{K_d^i F_s}{h(1 + K_d^i P)}, \quad (2-5)$$

and

- $C_i(t)$ = concentration of the i th radionuclide at time t , Bq/m³;
- S_i = inventory of the i th radionuclide at time 0, Bq;
- V = volume of Arctic Ocean, m³;
- λ_d^i = radioactive decay rate of the i th radionuclide, 1/yr;
- λ_w = loss rate constant for water exchange, 1/yr;
- λ_s^i = loss rate constant for the i th radionuclide from sedimentation, 1/yr;
- $T_{1/2}$ = half-life for radioactive decay, y;
- W = exchange rate of water through the Arctic Ocean, m³/yr;
- K_d^i = sediment/water distribution coefficient of the i th radionuclide, m³/kg;
- F_s = sedimentation rate, kg/m²-y;
- P = mass loading of particulate matter in water, kg/m³; and
- h = average depth of the Arctic Ocean, m.

Integrating Eq. 2-1 from the time of release to $t = \infty$ obtains an estimate of the population dose resulting from long-term dietary exposures (composed of two dietary items: fish/marine mammals and shellfish) to the i th radionuclide (UNSCEAR, 1977), or

$$\int_0^{\infty} D_i(t) dt = \text{Pop} \left[\sum_{j=1}^2 I_j BCF_{ij} \right] DCF \frac{S_i}{V} (\lambda_d^i + \lambda_w + \lambda_s^i)^{-1} \quad (2-6)$$

A ranking formula can now be developed that expresses the relative importance of each nuclide as percent of the total population dose for all nuclides, or

$$\text{Rank}_i(\%) = \frac{100 \left[\sum_{j=1}^2 I_j BCF_{ij} \right] S_i DCF_i (\lambda_d^i + \lambda_w + \lambda_s^i)^{-1}}{\sum_{i=1}^n \left[\sum_{j=1}^2 I_j BCF_{ij} \right] S_i DCF_i (\lambda_d^i + \lambda_w + \lambda_s^i)^{-1}} \quad (2-7)$$

Note that Eq. 2-7 has been simplified; both the population and water-volume terms were canceled.

The loss-rate constant for water exchange is λ_w (Eq. 2-4) equal to the exchange rate of water from the ocean compartment divided by the compartmental volume. The volume of the Arctic Ocean is about $1.9 \times 10^{16} \text{ m}^3$ and its average depth is 1,000 m. Estimates of the combined water exchanges from the North Atlantic and Pacific Oceans range from 2.9 to $15 \times 10^6 \text{ m}^3/\text{s}$ (or $3.2 \times 10^{13} \text{ m}^3/\text{yr}$). For example, Ostlund and Hut (1984) used data on isotopic tracers to estimate an oceanic exchange rate of $9.0 \times 10^{13} \text{ m}^3/\text{yr}$, whereas Nielsen (1995) used an ocean circulation model to estimate a water exchange rate of nearly $4.8 \times 10^{14} \text{ m}^3/\text{yr}$. Aagaard and Greisman (1975) estimated an exchange of $9.4 \times 10^6 \text{ m}^3/\text{s}$, based largely on current data (see Section 3 for additional discussion of water exchange rates among the Arctic Seas.). Assuming that the 2.9×10^6 and $15 \times 10^6 \text{ m}^3/\text{s}$ values represent lower and upper bounds of the exchange rates, then the geometric mean value is $6.6 \times 10^6 \text{ m}^3/\text{s}$. Therefore, λ_w is 0.011/yr.

The decline in the concentration of a radionuclide in water from the deposition of sediments containing adsorbed radionuclides is controlled by both the sedimentary characteristics of the water column (mass loading particles and their average settling rate) and the sediment-water distribution of a radionuclide (i.e., as the partition coefficient increases, the loss via sedimentation increases). The loss-rate constant for a highly sorptive nuclide such as ^{239}Pu resulting from particle settling, λ_s (Eq. 2-5), is $9.1 \times 10^{-4}/\text{yr}$ or a factor of 12 lower than the rate constant for volumetric exchange of water through the Arctic Ocean (using a K_d of $100 \text{ m}^3/\text{kg}$, an average depth of 1,000 m for the Arctic Ocean, and a sediment loading and deposition rate of $0.001 \text{ kg}/\text{m}^3$ and $0.01 \text{ kg}/\text{m}^2\text{-yr}$, respectively [Nielsen, 1995]). Table 2-11 summarizes the K_d values used to represent the nuclides (from IAEA Technical Report 247, [IAEA, 1985]) and Table 2-12 presents the rate constants for radioactive decay, sedimentation, and water exchange for each of the radionuclides. The remaining terms that need to be estimated for the ranking formula (Eq. 2-7) are the dose-conversion factors (DCFs) for each radionuclide, the bioconcentration factors (BCFs) for fish/marine mammals and mollusks, and the related consumption rates. Table 2-11 contains the DCFs and BCFs used in the screening analysis. For the purposes of the screening-level analysis, the RAIG also has used the default BCF values from IAEA Technical Report 247 (IAEA, 1985) (refer to Section 4 for an analysis of BCFs related marine species from Arctic waters), while the DCFs are from ICRP (1994) and Eckerman (1988). To represent the elevated ingestion of seafoods in subsistence diets of Arctic populations, the RAIG selected an ingestion rate of $200 \text{ kg}/\text{yr}$, based on the range of intakes summarized in Section 6. An ingestion rate of $1 \text{ kg}/\text{yr}$ was used to represent the ingestion of shellfish (UNSCEAR, 1977). This reduced intake is consistent with the limited use of shellfish in subsistence diets.

The initial screening analysis of the Kara Sea source region considered only the inventories of the various radionuclides for the nuclear wastes disposed in the Kara Sea listed in Table 2-7. Results of that analysis are shown in Table 2-12. The principal radionuclides of potential concern are ^{239}Pu , ^{241}Am , and ^{137}Cs . These nuclides represent 86% of the potential population dose, but account for only 39% of the total radionuclide inventory (excluding the short-lived decay products of ^{137}Cs and ^{90}Sr). A second set of nuclides consisting of ^{60}Co , ^{63}Ni , ^{90}Sr , and ^{238}Pu represent another 13% of the potential population dose. However, when the large inventories of ^{90}Sr in the reservoirs at Mayak are considered (5,000 TBq in Reservoirs 10, 11, and 17 that could discharge to the Ob River system), this nuclide becomes more important than the activation products ^{60}Co and ^{63}Ni . Consequently, ^{239}Pu , ^{241}Am , ^{90}Sr , and ^{137}Cs amount to 90% of the screening dose recalculated using the higher inventories for ^{90}Sr . The RAIG focused on these four nuclides because they collectively represent the greatest contributors to potential doses. Moreover, the activation products ^{60}Co and ^{63}Ni are incorporated within metal matrices in reactor components and are gradually

released as corrosion takes place. The actinides and fission products are associated with the nuclear fuel assemblies that are presumably more susceptible to dispersal in seawater. To examine the sensitivity of the screening model to the rate constants for sedimentation, the RAIG set the K_d values of all the radionuclides to zero. This adjustment, however, did not alter the relative ranks of the screening set, although the long-lived actinides ^{239}Pu and ^{241}Am accounted for more of the total population dose. An increase in the water exchange rate also did not affect the ranking of the various nuclides.

2.5.2 Northwest Pacific Ocean Sources

The screening analysis of radionuclides associated with the radioactive wastes dumped in the Northwest Pacific Ocean is restricted to the activation products associated with reactor components (disposal sites in the Sea of Japan and the East Coast of Kamchatka) and a radioisotope thermoelectric generator fueled by ^{90}Sr . There is no information on the true radionuclide inventories associated with non-reactor-related solid wastes, but this is not surprising because radiological measurements of such wastes prior to disposal are usually very limited or nonexistent. As a means of determining whether the releases of the activation products from reactor components constitute a viable threat to Alaska, the RAIG develops simple, screening-level estimates of the resulting concentrations of the radionuclides dissolved in a volume of water that is equivalent to that of the Bering Sea (i.e., $3.6 \times 10^{15} \text{ m}^3$). This clearly is a conservative approach because it neglects the dispersion that would occur as the radionuclides are transported from the wastes sources to the Bering Sea. Nevertheless, if the predicted concentrations are insignificant under these conditions, there is no need to conduct more detailed assessments of the sources.

The primary nuclides associated with the reactor-related unenclosed solid objects are the activation products consisting of ^{14}C , ^{60}Co , ^{59}Ni , and ^{63}Ni . Releases of these radionuclides will occur over time through the corrosion of the construction materials. The primary activation product listed in Table 2-8 for Areas 8, 9, and 10 is ^{60}Co , which also has the highest dose per unit of radioactivity ingested of the four activation products. Therefore, for the purposes of the screening analysis, the RAIG will use this nuclide to assess the radiological hazard of the reactor-related wastes. The estimated inventory of ^{60}Co as well as predictions of the corrosion-driven release rates will serve as the basis of the screening-level estimates of ^{60}Co in seawater. The screening-level analysis for the RTG will be based on its inventory of ^{90}Sr . The annual dose attributable to consumption of fish/marine mammals from the hypothetical contamination of a volume of water equivalent to the Bering Sea is given as

$$D_i = I_i \text{ BCF}_i \text{ DCF}_i \text{ C}_i, \quad (2-8)$$

where

- D_i = annual dose resulting from the ingestion of the i th radionuclide, Sv/yr;
- I_i = annual average intake of fish/marine mammals, kg/yr;
- BCF_i = bioconcentration factor for the i th radionuclide for the fish/marine mammals consumed, m^3/kg ;
- C_i = concentration of the i th radionuclide in seawater, Bq/m^3 ; and
- DCF_i = dose conversion factor for ingestion of the i th radionuclide, Sv/Bq.

Table 2-11. Input parameters for use in the dose-based screening analysis of radionuclides. The sediment-water distribution coefficients (K_d) and bioconcentration factors (BCF) values are from IAEA (1985), while the dose conversion factors (DCF) are from ICRP (1994) and Eckerman et al. (1988).

Radionuclide	Half-Life y	K_d m^3/kg	DCF Sv/Bq	Bioconcentration Factors	
				Fish m^3/kg	Mollusk
^{241}Am	432	2,000	2.1×10^{-7a}	0.05	20
^{239}Pu	24,100	100	2.5×10^{-7a}	0.04	3
^{238}Pu	88	100	2.3×10^{-7a}	0.04	3
^{137}Cs	30	3	1.4×10^{-8a}	0.1	0.03
^{60}Co	5	200	3.4×10^{-9a}	1	5
^{63}Ni	96	100	1.5×10^{-10a}	1	2
^{90}Sr	29	1	2.8×10^{-8a}	0.002	0.001
^{152}Eu	13	500	1.8×10^{-9b}	0.3	7
^{154}Eu	9	500	2.6×10^{-9b}	0.3	7
^{14}C	5,730	2	5.8×10^{-10a}	20	20
^{59}Ni	75,000	100	6.3×10^{-11a}	1	2
^{151}Sm	90	2,000	1.1×10^{-10b}	0.5	5
^{99}Tc	213,000	0	6.4×10^{-10a}	0.03	1
^{147}Pm	3	2,000	2.8×10^{-10b}	0.5	5
^{155}Eu	5	500	4.1×10^{-10b}	0.3	7
^{125}Sb	3	1	7.6×10^{-10b}	0.4	0.2
^{205}Pb	14,000,000	200	4.4×10^{-10b}	0.2	1
^{207}Bi	38	1	1.5×10^{-9b}	0.4 ^c	0.2 ^c
^{208}Bi	368,000	1	3.4×10^{-9d}	0.4 ^c	0.2 ^c
^{210m}Bi	3,000,000	1	2.6×10^{-8b}	0.4 ^c	0.2 ^c
^{129}I	15,700,000	0	1.1×10^{-7a}	0.01	0.01
3H	12	0	1.8×10^{-11a}	0.001	0.001

a ICRP (1994)

b Eckerman et al. (1988).

c Bismuth (Bi) should have properties similar to those of antimony (Sb) based on their position in the Periodic Chart. The RAIG has therefore used the K_d and BCF values of antimony to represent those of bismuth.

d A dose-conversion factor was not available for ^{208}Bi , so the RAIG used the DCF of ^{60}Co because its gamma-ray energy is comparable to that of ^{208}Bi .

Table 2-12. Results of the dose-based ranking of the radionuclides present in the Kara Sea dump sites.

Radionuclide	Inventory Bq	Rate Constants			Ranking (%)
		Radioactive Decay	Water Exchange	Particle Scavenging	
			1/yr		
¹³⁷ Cs	1.0×10^{15}	2.3×10^{-2}	1.1×10^{-2}	3.0×10^{-5}	68
²³⁹ Pu	6.2×10^{12}	2.9×10^{-5}	1.1×10^{-2}	9.1×10^{-4}	11
²⁴¹ Am	8.3×10^{12}	1.6×10^{-3}	1.1×10^{-2}	6.7×10^{-3}	7.5
⁶⁰ Co	1.4×10^{14}	1.3×10^{-1}	1.1×10^{-2}	1.7×10^{-3}	4.7
⁶³ Ni	3.5×10^{14}	7.2×10^{-3}	1.1×10^{-2}	9.1×10^{-4}	4.2
⁹⁰ Sr	9.5×10^{14}	2.4×10^{-2}	1.1×10^{-2}	1.0×10^{-5}	2.5
²³⁸ Pu	1.6×10^{12}	7.9×10^{-3}	1.1×10^{-2}	9.1×10^{-4}	1.5
¹⁵² Eu	6.0×10^{13}	5.2×10^{-2}	1.1×10^{-2}	3.3×10^{-3}	0.58
¹⁵⁴ Eu	1.1×10^{13}	7.9×10^{-2}	1.1×10^{-2}	3.3×10^{-3}	0.11
¹⁴ C	1.6×10^{11}	1.2×10^{-4}	1.1×10^{-2}	2.0×10^{-5}	0.28
⁵⁹ Ni	8.6×10^{12}	9.2×10^{-6}	1.1×10^{-2}	9.1×10^{-4}	0.07
¹⁵¹ Sm	2.4×10^{13}	7.7×10^{-3}	1.1×10^{-2}	6.7×10^{-3}	0.03
^{210m} Bi	3.4×10^9	2.3×10^{-7}	1.1×10^{-2}	1.0×10^{-5}	0.01
²⁰⁸ Bi	6.2×10^9	1.9×10^{-6}	1.1×10^{-2}	1.0×10^{-5}	<0.01
⁹⁹ Tc	2.8×10^{11}	3.3×10^{-6}	1.1×10^{-2}	1.0×10^{-6}	<0.01
¹⁴⁷ Pm	2.4×10^{12}	2.6×10^{-1}	1.1×10^{-2}	6.7×10^{-3}	<0.01
²⁰⁷ Bi	1.7×10^{10}	1.8×10^{-2}	1.1×10^{-2}	1.0×10^{-5}	<0.01
¹²⁵ Sb	1.7×10^{11}	2.5×10^{-1}	1.1×10^{-2}	1.0×10^{-5}	<0.01
¹⁵⁵ Eu	9.6×10^{10}	1.4×10^{-1}	1.1×10^{-2}	3.3×10^{-3}	<0.01
¹²⁹ I	3.9×10^8	4.4×10^{-8}	1.1×10^{-2}	2.0×10^{-7}	<0.01
³ H	5.2×10^{13}	5.6×10^{-2}	1.1×10^{-2}	1.0×10^{-8}	<0.01
²⁰⁵ Pb	1.9×10^8	5.0×10^{-8}	1.1×10^{-2}	1.7×10^{-3}	<0.01

The screening calculations will be based on an elevated ingestion rate of 200 kg/yr of fish and marine mammals. For ⁶⁰Co and ⁹⁰Sr the RAIG will use BCFs of 1 and 0.002 m³/kg, and dose-conversion factors of 3.4×10^{-9} and 2.8×10^{-8} Sv/Bq, respectively (Table 2-11).

Sea of Japan and East Coast of Kamchatka

The inventories of ⁶⁰Co in the reactor-related components dumped in Areas 8, 9, and 10 are 2,500, 160, and 870 GBq, respectively. If these amounts of radioactivity were mixed into 3.6×10^{15} m³ of water, the resulting concentrations would be 6.9×10^{-4} , 4.4×10^{-5} , and 2.4×10^{-4} Bq/m³. The associated annual dose estimates are below 5×10^{-7} mSv/yr, which is more than a million times lower than the annual background dose of approximately 2 mSv that most of the world's population receives annually from various sources (UNSCEAR, 1988). The presence of these wastes is unlikely to represent a radiological hazard even to the local seas in which they are located because releases will take place slowly over time as corrosion products are emitted to the ocean/sediment system at the waste sites. Mount et al. (1996) estimated that the total release of activation products from the submarine-reactor core plate in Area 8 starts below 0.01 GBq/yr and over 1,000

years, falls to 100 Bq/yr. The predicted release rates for the two reactors and the reactor lid in Area 10 were added together, taking into account their different dates of dumping. The total release rate begins at about 1 GBq/yr, dropping off to a level less than 0.1 GBq/yr, extending past the year 4000.

Sea of Okhotsk: ⁹⁰Sr Radioisotope Thermoelectric Generator

From a screening standpoint, the ⁹⁰Sr-powered RTG represents a significant nuclear source, as the dilution of 11,000 TBq of ⁹⁰Sr in 3.6×10^{15} m³ of water would result in a concentration of 3 Bq/m³, which is above background levels in the Pacific Ocean. The associated dose rate, however, is far below background levels at an estimated 3.4×10^{-5} mSv/yr. While instantaneous release of the ⁹⁰Sr source at the time of loss is the most conservative as to environmental impact, considering the reported hermetically sealed solid construction of the RTG (Danilyan and Vysotsky, 1995) and the robust construction expected, it is not unreasonable to assume that the steel construction will survive long enough for radioactive decay to make the source strength of much less consequence to the environment when it finally begins to be released. Furthermore, once release does occur, the portion of the ⁹⁰Sr released that would end up in the Bering Sea would depend on the actual location of the RTG in Sea of Okhotsk and movement of currents from that location to the Pacific Ocean and then the Bering Sea. Given that the dose estimate for instantaneous release is very low and that transport from the Sea of Okhotsk to the Bering Sea would further reduce the levels of ⁹⁰Sr in seawater, the RAIG concludes that the RTG constitutes no significant threat to the Arctic.

2.6 RELEASE SCENARIOS

Our review of the available information on FSU nuclear wastes dumped in the Arctic and Pacific Oceans and stored at inland locations adjacent to the Ob and Yenisey Rivers in the West Siberian Basin indicates that the risk assessment should focus on radionuclide releases from naval reactors disposed in the Kara Sea and from the waste storage ponds and reservoirs at the inland sites. As a means of addressing the uncertainties inherent in deriving estimates of the releases from such sources, the RAIG presents both acute and chronic release scenarios. Acute release scenarios are meant to represent catastrophic or upper-bound releases of radionuclides to the Kara Sea or rivers, whereas the chronic release scenarios represent the slow discharge of nuclides to seawater via the corrosion/dissolution of fuels or leakage from wastewater reservoirs to surface waters of the Ob/Yenisey river systems. Thus, the acute scenarios are assumed to be less likely than the chronic cases, but have the potential for substantially higher releases of radionuclides.

2.6.1 Kara Sea

The acute or upper-bound release case for the Kara Sea sources was defined as the instantaneous discharge of the inventories of ⁹⁰Sr, ¹³⁷Cs, ²³⁹Pu, and ²⁴¹Am (i.e., 0.95, 1.0, 0.006, and 0.008 PBq) to seawater. This release scenario is unlikely to occur because it is difficult to postulate the mechanism(s) by which the nuclides of concern could be abruptly transferred to seawater. Nevertheless, the hypothesized release does constitute an upper-bound case and the available inventories will decline with time because of radioactive decay. After 100 years, for example, the inventories of ⁹⁰Sr and ¹³⁷Cs will be about a factor of 10 lower than the present inventories. The decay

rates for ^{239}Pu and ^{241}Am are much slower, and consequently the inventories of those nuclides will remain elevated for centuries.

The more realistic release mechanism for the fission products and actinides in spent reactor fuels is the corrosion of fuel elements by seawater. Corrosion-based releases of the nuclides have been estimated by the IASAP STWG (IAEA, 1997). The time-varying release rates are calculated as the product of the activity of a given radionuclide present at time t in SNF and the ratio of the volume of SNF corroded at time t to the initial volume of the SNF. The effective corrosion rate is estimated by adjusting a base corrosion rate to account for the degree of SNF containment and the use of Furfurol (F) as a barrier material. Corroded material is assumed to be soluble and available for subsequent transport. This is clearly a conservative case, as corroded material may largely remain as insoluble debris at the various waste locations (IAEA, 1997).

The STWG prepared alternative scenarios to describe future radionuclide releases from the reactors dumped in the Kara Sea. Most relevant for the risk assessment is the "best estimate" case, in which radionuclides are gradually released to seawater as the SNF corrodes. Other scenarios included abrupt, catastrophic releases associated with external events or glacial scouring of the fjords after 1,000 years. These types of scenarios are essentially the same as the acute release described above, except there is a prolonged period that allows the inventories of some radionuclides to decline substantially. The time-varying releases of ^{90}Sr , ^{137}Cs , ^{239}Pu , and ^{241}Am are shown in Figure 2-7. The initial peak represents the early entry of seawater to unprotected RCs, while a secondary peak occurs in the year 2300 when containers holding damaged SNF are breached and fission products and actinides are then released to seawater.

2.6.2 West Siberian Basin

Releases of radioactive effluents to the Ob and Yenisey River systems from the nuclear facilities at Mayak (Techa/Ob Rivers), Tomsk (Tom/Ob Rivers), and Krasnoyarsk (Yenisey River) could occur as accidental discharges of LRWs from storage reservoirs or as chronic releases in the form of waste water seeping through earthen retaining dams or contaminated groundwater that discharges to the river. To characterize the potential release from these source terms, the RAIG reviews pertinent information on the radionuclide inventories associated with waste-storage ponds and develops preliminary estimates of effluent discharges for both accidental and chronic release modes.

Accidental or Acute Release Scenarios

Mayak: Releases from Waste-Water Reservoirs

RWs at the Mayak site (Figure 2-6) are stored in a combination of lakes and man-made reservoirs that are linked in series by canals or separated by earthfill dams over a linear distance of 30 km. Four dams are associated with Reservoirs 3, 4, 10, and 11. In the past, discharge from the final reservoir in the series (Reservoir 11) was directly into the Techa River. The height of the earthfill dam on this reservoir has been raised by 2 m over time to prevent discharge of contaminated water into the Techa River (Bradley and Jenquin, 1995) and canals were built in 1963 (left-bank canal) and 1972 (right-bank canal) to allow the Techa River flow to bypass the series of reservoirs named above. The dam on Reservoir 11 has leaked in the past, and continues to leak now, resulting in the continued contamination of the Techa River (Bradley and Jenquin, 1995).

An extreme-event release would occur if there were a common initiating event such as an earthquake or flood that caused the failure of the earthen dams associated with Reservoirs 3, 4, 10, and 11. Reservoir 17 also has a retaining dam; however, it is not in the main chain of reservoirs, as shown in Figure 2-7. Another mechanism of dam failure is the wetting-induced collapse of compact soil. Collapse (once wetting starts to occur) would be essentially instantaneous (e.g., 4 hours or less) (Lawton et al., 1992). This failure mechanism would presumably be site-specific; however, the wetting-induced collapse of the earthen dam at Reservoir 10 could lead to the failure of Reservoir 11, the next reservoir in the chain, which would then discharge to the Techa River. Depending on the extent of the dam failure(s) (i.e., a small section of a dam versus a major col-

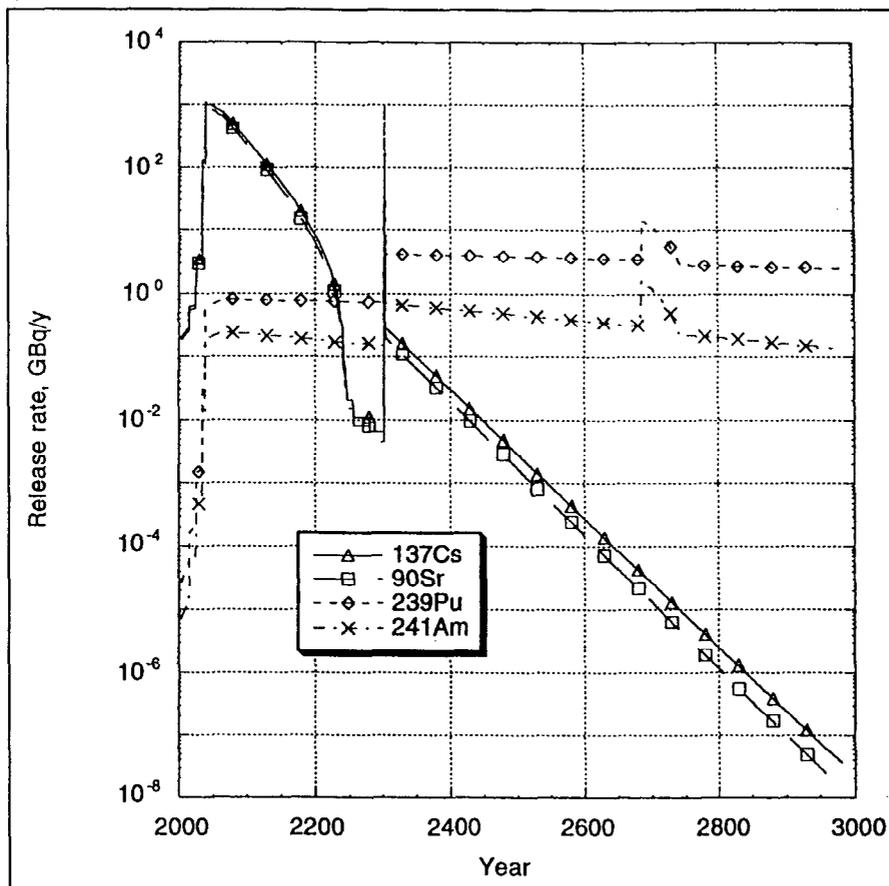


Figure 2-7. Time-varying release rates of radionuclides from naval reactors dumped in the Kara Sea (from IAEA, 1997).

lapse), it is conceivable that remedial actions could be taken to limit the total amount of LRWs discharged as well as the rate of discharge.

Without detailed analyses of the probabilities of different earthquake intensities of this area of Russia, the likelihood of wetting-induced collapse, the magnitude and frequency of floods, and the structural response of the dams to such events, it is not possible to determine which failure modes are most plausible for the Mayak reservoirs. Nevertheless, it appears that Reservoirs 10 and 11 are the most vulnerable based on their collocation and the possibility that the failure of one reservoir could lead to the failure of the other. The radioactive liquid wastes in these reservoirs that could be released to the Techa River contain 1,400 TBq of ⁹⁰Sr and 24 TBq of ¹³⁷Cs (see

Table 2-10). Based on previous problems and remedial actions at Mayak, discussed in Bradley and Jenquin (1995), the RAIG expects that efforts to contain a major release would be made at Mayak by the addition or reinforcement of earthfill dams, or similar efforts.

To account for the fact the dam-reservoir system is under active institutional control and that remedial actions would undoubtedly be instituted to deal with a breached dam, the RAIG assumes that the release of radioactive liquids from Reservoirs 10 and 11 would occur continuously over a 1-year period. This type of release would produce a discharge that is comparable in magnitude to the releases occurring during the period 1949 to 1951, when an estimated 1,000 TBq of ^{90}Sr and ^{137}Cs in LRWs were discharged directly to the Techa River from the Mayak facility (each nuclide contributed about half of the total activity; Trapeznikov et al., 1994). However, the release scenario described here would be dominated by ^{90}Sr and would occur over a shorter period of time. The constant discharge rates for ^{90}Sr and ^{137}Cs would be 4 and 0.07 TBq/d, respectively, for one year (based on 1,400 TBq and 24 TBq of those radionuclides in the LRWs stored in the two reservoirs). Because of the preliminary nature of this scenario, it should be viewed primarily as a screening-level case that constitutes a plausible release event for use in the risk assessment.

Tomsk and Krasnoyarsk: Releases from Waste-Water Reservoirs

At Tomsk, the potential for an accidental release is not as clear. There are apparently eight reservoirs at the site: three natural ponds, three man-made ponds, and two sludge pits (personal communication, D. Bradley, Pacific Northwest National Laboratory, August 19, 1996). Collectively, these reservoirs contain one million TBq (10^{18} Bq) of LRW. They are located <5 km from the Tom River and may be connected to it by a small creek or river. Bradley and Jenquin (1995) also mentioned a "special sewer" for the disposal of plutonium machining waste. The low-lying, flat nature of the local terrain near the Tomsk reservoirs along with the prospect for backwater phenomena (the spring freshet from the Irtysh River entering the Ob River at Belogorye prevents the release of the spring freshet on the upper Ob and Tomsk Rivers) increases the possibility that flooding could cause discharges of LRWs from the reservoirs. Unfortunately, data on the actual radionuclide composition of the wastes are not available and the characteristics of the ponds/reservoirs, dams, and canals are unknown, and therefore it is not possible to develop an accidental release scenario for this facility. As a default, the RAIG will use the Mayak release described above to represent an accidental discharge of LRW from either facility into the Ob River system and subsequently the Kara Sea.

The Krasnoyarsk facility also has surface reservoirs for the storage of LRWs, but according to Bradley and Jenquin (1995), the total inventory of radionuclides is much lower than for Mayak and Tomsk (i.e., less than 2,000 TBq) as well as the total reservoir volume (under 300,000 m³). The principal reason for this is that the subsurface injection of LRWs has been the primary method of disposal—not surface storage in reservoirs. Historically, the primary radioactive effluents from Krasnoyarsk that have contaminated the Yenisey River have been discharges from the once-through cooling systems of its production reactors.

Although the limited information available on Krasnoyarsk prevents the development of a plausible release scenario of LRWs to the Yenisey River, the reference Mayak event is certainly larger than any release that could occur at Krasnoyarsk.

Remobilization of Asanov Marsh ⁹⁰Sr Inventory

Trapeznikov et al. (1994) reported that about 10,000 TBq of ⁹⁰Sr was released as a result of Mayak operations between 1949 and 1951, about 80% of this material is estimated to remain in the Mayak Reservoir system; 15% was discharged to the Ob River system, and the remaining 5% is in the sediments and flood plain of the Techa River. Decay correcting this amount to 1994 leaves about 200 TBq of the original ⁹⁰Sr inventory. The bulk of this inventory is thought to remain in the Asanov Marsh. Remobilization of this inventory requires a scenario in which the marsh soil and flora are dewatered and subsequent flood events wash organic matter and loose soil into the Techa River. Because this release scenario would not result in the discharge of as much ⁹⁰Sr as the hypothesized release from the Mayak Reservoirs 10 and 11, the RAIG will use the Mayak release to represent an acute discharge of radioactive effluents to the Kara Sea from inland sources. However, the RAIG cannot, at this point, determine which release scenario is more likely to occur.

Chronic-Release Scenarios

The Ob and Yenisey rivers have discharged ⁹⁰Sr and other radionuclides to the Kara Sea since the early 1950s. The primary sources of the radionuclide discharges have been the runoff of nuclides deposited onto watersheds in the form of nuclear fallout and the discharge of LRWs to rivers. Unless there are major new releases, the discharges should gradually decline as the inventories of the various radionuclides undergo radioactive decay. To establish the basis for a chronic-release scenario of ⁹⁰Sr into the Kara Sea from the two rivers, the RAIG reviews data on the magnitude of historic discharges and assesses the potential magnitude of chronic releases produced by discharges of contaminated groundwater and leakage from earthen dams. The RAIG then presents the release rates used to represent these scenarios in the risk assessment.

Historical ⁹⁰Sr Fluxes from the Ob and Yenisey Rivers

The observed flux of ⁹⁰Sr at Salekhard (near the point where the Ob River discharges into the Kara Sea estuary) largely is due to washoff from the watershed of atmospheric fallout. The historical ⁹⁰Sr flux from the Ob River since 1970 has been about 10 to 40 TBq/yr, based on an average Ob River flow rate of 12,680 m³/s (data from Bobkin and Bobrovitskaya, 1995) and ⁹⁰Sr concentrations in river water of 25-100 Bq/m³ (Chumichev, 1995). In contrast, Vakulovsky et al. (1995) estimate that the flux of ⁹⁰Sr from the Yenisey River averaged about 1.5 TBq/yr during the years 1985 to 1991.

Near-Surface Groundwater Plumes from Reservoir Leachate

Seepage of leachate from the surface reservoirs containing LRWs has contaminated groundwater beneath the storage reservoirs and lakes at Mayak. Subsequent discharge of the contaminated groundwater to the Techa River represents a potential source of river contamination. Similar groundwater contamination undoubtedly occurs at Tomsk and Krasnoyarsk, but less is known about its extent. The magnitude of the groundwater transport mechanism depends on a number of factors, including the hydrologic gradient beneath the sites, the direction of groundwater flow, the extent of radionuclide adsorption to aquifer materials, etc. In the most simple analysis, a contaminant plume beneath a reservoir is estimated to be advected with the steady-state flow rate of the groundwater. The flux of ⁹⁰Sr in groundwater can therefore be calculated as the product of the uniform concentration of ⁹⁰Sr in groundwater (assumed to be equal to its concentration

in a reservoir) and the volumetric discharge of groundwater. The groundwater flow rate is equal to the velocity of groundwater and the cross section of the plume (based on a plume thickness of 2 m and the width of a reservoir).

As an example calculation, Bradley and Jenquin (1995) reported that the plume below Lake Karachai has migrated 2.5 km in 40 years, and this implies a plume velocity of approximately 60 m/yr. For Lake Karachai (Reservoir 9 with a ^{90}Sr concentration of 0.063 TBq/m^3 , an inventory of 25,000 TBq, and an area of 0.25 km^2) the flux is estimated to be 4,400 TBq/yr, and therefore the plume will discharge for nearly six years (i.e., $25,000/4,400$). The assumption that this groundwater velocity is representative of all contaminant plumes is unlikely to be valid across the various reservoir/aquifer systems; however, it is commensurate with the preliminary nature of this source-term analysis. Using the methodology described above, the RAIG has estimated the non-decay-corrected flux rate of ^{90}Sr from each of the Mayak Reservoirs to groundwater and then to surface water. Table 2-10 presents the estimated fluxes for near-surface reservoir leachate entering the groundwater and then discharging to the Techa River. The release periods are calculated by dividing the inventory by the flux rate. The estimated fluxes vary considerably in their magnitude and duration. Because the fluxes estimated for Lake Karachai and Reservoir 17 are relatively short in duration, but large in magnitude, they can be represented by the surface-water release described earlier consisting of a 1,400 TBq discharge of ^{90}Sr from Reservoirs 10 and 11 that lasts continuously for one year. Given the various simplifying assumptions used to estimate the groundwater fluxes of ^{90}Sr from Lake Karachai and Reservoir 17, the RAIG believes that those predicted fluxes are unlikely to exceed the surrogate surface-water release scenario—especially if remedial actions were instituted to limit the groundwater discharges.

The longer-term discharges to surface water can be simulated as the product of a base discharge rate and a decaying source term, or

$$F(t) = F_b e^{-\lambda_d t}, \quad (2-9)$$

where $F(t)$ is the flux to the Kara Sea (in Bq/yr) at time t (in years), F_b is the base flux of ^{90}Sr at the beginning of the assessment period, and λ_d is the rate constant for the radioactive decay of ^{90}Sr (i.e., $0.024/\text{yr}$). This formulation can be used to represent the long-term discharge of contaminated ground water as well as seepage from the earthen dams at the various reservoirs. Based on the magnitude of the historic discharges of ^{90}Sr and the estimated discharges from groundwater shown in Table 2-13, the RAIG has selected a base flux of 40 TBq/yr to reflect the chronic discharges of ^{90}Sr to the Ob and Yenisey rivers in the form of groundwater discharges, reservoir leakage, and watershed runoff. This flux is on the upper end of the historic discharges from the Ob River since 1970 and encompasses the release rates presented in Table 2-13 for chronic groundwater discharges (i.e., up to 8 TBq/yr) plus a significant amount of dam seepage for both Mayak and Tomsk.

Table 2-13. Estimates of the maximum fluxes of ^{90}Sr to the surface water from the discharge of contaminated groundwater.

Reservoir	Computed Flux Rate (TBq/yr)	Release Duration (y)
2	0.25	140
3	6.7	7
4	1.0	26
6	0.004	65
9	4,400	6
10	7.9	130
11	1.8	230
17	15	5

2.7 SUMMARY

- Operation of the Sellafield nuclear reprocessing facility in Great Britain has resulted in the discharge of approximately 41,000 TBq of ^{137}Cs to the Irish Sea through 1992. Of this amount, an estimated 9,000 TBq has entered the Arctic Ocean. In contrast, only about 20 TBq of the 590 TBq of the $^{239,240}\text{Pu}$ discharged from Sellafield is estimated to have reached the Arctic Ocean.
- Source-term analyses of the marine reactors dumped in the Kara Sea indicate that about 1,000 TBq of ^{137}Cs and 6 TBq of $^{239,240}\text{Pu}$ remain in the SNF. Unlike the Sellafield-derived nuclides that are already present in the water column, the reactor-based nuclides constitute a potential threat to the Arctic Ocean because future releases will depend on the rate of corrosion/dissolution processes. In any event, the historic releases from Sellafield are a more significant source of ^{137}Cs and $^{239,240}\text{Pu}$.
- Screening analyses of the radionuclides present in the Kara Sea, the Northwest Pacific Ocean, and LRWs in ponds and reservoirs in the West Siberian Basin indicate that the principal radionuclides of potential concern are ^{90}Sr , ^{137}Cs , ^{239}Pu , and ^{241}Am . In addition, the most credible sources in terms of future impacts to Alaskan waters are the Kara Sea wastes and the LRWs stored at inland locations in Russia.
- Radionuclides released from the Kara Sea and West Siberian Basin were characterized by using two different types of scenarios, one to represent acute or accidental releases and the other to represent chronic releases. For the inland sources, the acute scenario consists of the discharge of 1,400 TBq of ^{90}Sr and 24 TBq of ^{137}Cs from Reservoirs 10 and 11 at Mayak to the Ob River system. The chronic scenario consists of a base discharge of 40 TBq/yr of ^{90}Sr from reservoir leakage that declines with time as a function of the radioactive decay of ^{90}Sr .