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Radioiodine in the Savannah River Site Environment (U)

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Glossary

absorbed dose - The energy deposited per unit mass of matter when ionizing radiation interacts with matter. Conventionally expressed in units of rad. In International System (SI) units absorbed dose is expressed in units of gray (Gy), where 1 Gy equals 100 rad.

activation - See nuclear reaction.

activity - See radioactive decay.

alpha particle decay - See radioactive decay.

anion - A negatively charged ion that is produced when a molecule dissolves in water and produces two ions of opposite charge. For example, iodide is a negatively charged anion of iodine. See cation.

aquifer - A saturated, permeable geologic formation in which large quantities of water move as a result of natural hydraulic gradients.

atom - The smallest divisible unit of matter that still retains characteristics of an element. An atom consists of a center core (the nucleus) and a surrounding electron cloud. The nucleus consists of neutrons and protons. The number of protons in an atom determine which element the atom is.

atom percent - The number of atoms of a specific isotope relative to the total number of atoms of the element, expressed as a percentage.

beta particle decay - See radioactive decay.

beta-gamma activity - A term for the type of radioactivity that is detectable with a Geiger-Mueller detector or other detector for energetic beta particles and gamma rays. The detector does not distinguish individual radionuclides.

biological half-time - The time required for a biological system to eliminate one half of a given number of atoms of a specific isotope solely by the processes of elimination.

cation - A positively charged ion that is produced when a molecule dissolves in water and produces two ions of opposite charge. See anion.

CEDE - Committed Effective Dose Equivalent

cladding - Material that encapsulates nuclear fuel and targets. It serves as a container for the nuclear material and barrier between the moderator and the irradiated fuel and targets which contain fission and activation products.

collective dose - The effective or tissue-specific dose equivalents received by members of an exposed population. See also person-rem.

committed dose equivalent - The dose equivalent delivered to a tissue or organ over a 50-year period following intake.

committed effective dose equivalent - The committed dose equivalents to various tissues in the body, each multiplied by the appropriate weighting factor, and summed.

concentration - The quantity of a material per unit of volume or mass that the material is a constituent of. In this report, concentration typically refers to the quantity of an isotope of iodine per liter of liquid, per cubic meter of air, or per gram of a solid.

cooling period - The time interval that irradiated materials are stored before being reprocessed. During this time, short-lived radioiodine decays to inconsequential activity levels.

curie - A unit of radioactivity. Abbreviated as Ci. One curie is 3.7×10^{10} decays per second. Annual releases of radioiodine are expressed in units of curies. Environmental levels of radioiodine are expressed in units of pCi, where 1×10^{12} pCi equal 1 Ci. In International System (SI) units, radioactivity is expressed in units of bequerel, where 1 bequerel is 1 decay per second (equal to 27 pCi).

DCG - Derived Concentration Guide

decay - The process of a radioactive transformation

deionizer - See ion exchange.

deposition velocity - An empirical value that relates the concentration of a material in air to the flux density of a material at the ground or plant surfaces.

Derived Concentration Guide - The concentration of a radionuclide in air that would result in an effective dose equivalent of 100 mrem under conditions of continuous exposure to such air for one year. Also, the concentration of a radionuclide in water that would result in an effective dose equivalent of 100 mrem under conditions of continuous exposure to such water for one year.

detection limit - See definition #1 for sensitivity.

dose - See absorbed dose.

dose commitment - The radiation dose received over a specified time period that the radioactivity persists in the body.

dose equivalent - The product of the absorbed dose and a quality factor which is dependent on the relative biological effectiveness of the ionizing radiation. Typically expressed in units of rem or millirem (1 mrem = 0.001 rem). In International System (SI) units, dose equivalent is expressed in units of sievert, where 1 sievert is 100 rem.

dose factor - A conversion factor which, as applied to this report, indicates the 50-year committed dose equivalent or 50-year committed effective dose equivalent delivered per unit intake of radioactivity. Conventional units are mrem per pCi of intake.

EDE - Effective Dose Equivalent

effective dose equivalent - The summation of the dose equivalents received by specific tissues of the body when weighted with tissue-specific factors.

effective half-life - The apparent half-life that results when both biological elimination processes and radioactive decay occur simultaneously. See half-life.

equilibrium - See radioactive equilibrium

fission - See nuclear fission

fission product - Any nuclide, stable or radioactive, resulting from the process of nuclear fission. The definition includes the primary fission fragments formed at the instant of fission, as well as the daughter products from decay of the primary fission fragments.

gamma ray - A photon that is emitted from a nucleus when the nucleus undergoes a transition from a higher energy level to a lower energy level. In this report, this energy level transition typically is associated with radioactive decay of a nuclide. Because the transition is between specific energy levels, the gamma ray has a specific energy which is characteristic of the nuclide undergoing decay. Typical energies of gamma rays of interest range from tens of kiloelectron volts to thousands of kiloelectron volts.

gamma-emitting radionuclide - A nuclide that emits a gamma ray during the process of radioactive decay.

gamma spectrometry - A technique to identify gamma-emitting radionuclides by measuring the characteristic gamma rays emitted during their decay. When a gamma ray interacts with a detector for gamma spectrometry, it can deposit its full energy in the detector. This energy is converted into an electronic signal which is proportional to the gamma energy. Thus, gamma rays of different energies can be sorted into a spectrum. Interpretation of the spectrum results in identification of gamma-emitting radionuclides. Typical detectors include sodium iodide and germanium. These can be calibrated for detection efficiency as a function of gamma energy and sample geometry. The efficiency is used to quantify the activity of radionuclides.

GASPAR - A computer code developed for the Nuclear Regulatory Commission to evaluate doses from atmospheric routine releases of radionuclides.

grab sample - An instantaneous collection of an environmental sample.

half-life - The time interval required to halve the number of atoms of a specific isotope that are present in a given system. Half-life pertains to two processes in this report. Radioactive decay results in a radiological half-life. Elimination from biological systems results in a biological half-life. When both processes occur simultaneously, an effective half-life results. See radiological half-life, biological half-time, and effective half-life.

half-time - See half-life.

heavy water - Water in which deuterium has been substituted for the normal hydrogen atoms of the water molecule. Deuterium is a rare natural stable isotope of hydrogen. The nucleus of a deuterium atom contains one neutron whereas a normal hydrogen atom has none.

HEPA filter - High Efficiency Particulate Air filter. A type of filter that traps particles from air that is passed through the filter. It traps greater than 99.97% of particles with a diameter of 3×10^{-7} meters or larger.

internal dose factor - See dose factor.

ion - An atom, a group of atoms, or a molecule that has either lost or gained one or more electrons.

ion exchange - The process of removing anions and/or cations from a solution by passing the solution through ion exchange material, also known as a deionizer.

ionizing radiation - Radiation capable of removing electrons from atoms as the radiation passes through matter.

irradiation - The process of exposure of matter to radiation, which can be either ionizing radiation or neutrons.

isotope - A nuclide that belongs to a specific element. Each different isotope of an element has the same number of protons but a different number of neutrons. See nuclide.

isotopically enriched - When the abundance of an isotope in a sample of an element is greater than its abundance in nature.

keV - An abbreviation for kiloelectron volts.

kilo electron volt - A unit for the energy of the radiation emitted in radioactive decay.

LADTAP II - A computer code developed for the Nuclear Regulatory Commission to evaluate doses from liquid releases of radionuclides.

leach - The process by which water interacts with an insoluble material to remove those constituents that have some degree of water-solubility.

low energy photon spectrometry - A type of gamma spectometry that is used to measure photons having energies less than 100 keV. A special detector is required to detect such low energies. Typically, x-rays and low energy gamma rays are detected by this technique.

MAXIGASP - An SRS-modified version of the computer codes XOQDOQ and GASPAR. MAXIGASP calculates doses to offsite individuals from routine atmospheric releases of radionuclides. For doses to individuals, MAXIGASP does not use the population dose calculational routines of GASPAR.

metastable state - An energy state that is greater than the ground state of the nucleus of an atom and has a measurable half-life. See nuclide.

moderator - A material whose purpose is to reduce the kinetic energy (slow the speed) of the energetic neutrons emitted in the fission process. In SRS reactors, the moderator is heavy water.

millirem - A unit of dose equivalent which is one-thousandth of a rem.

molecular sieve - A hydrated aluminum and calcium or sodium silcate with an open molecular structure that allows trapping of certain vapors. See silver zeolite.

mrem - Abbreviation for millirem.

neoplasm - Any new or abnormal growth such as a tumor, whether malignant or benign.

neutron activation - In this report, a type of nuclear reaction in which a target nucleus interacts with a neutron to either capture the neutron (referred to as neutron capture) or emit two neutrons (referred to as (n,2n) reaction).

neutron capture - See neutron activation.

nuclear fission - The process of splitting a heavy nucleus generally into two lighter nuclei plus several neutrons. This process can occur naturally as a type of radioactive decay (spontaneous fission) or as neutron-induced fission. The two lighter nuclei, called fission products, are not unique in the fission process. Instead there is a distribution of fission products which include isotopes of iodine.

nuclear reaction - Nuclear reactions described in this report are limited to the interaction of a target nucleus with a neutron. Example reactions are neutron-induced fission, neutron capture, and (n,2n). See nuclear fission and neutron activation.

nucleus - The center part of an atom. It is composed of at least one proton and zero or more neutrons.

nuclide - An atom characterized by a specific number of protons and neutrons and by its energy state. A nuclide is either stable or unstable. An unstable nuclide is radioactive and is known as a radionuclide, for example ^{133}I. The nucleus of a given atom may also have the property that it can exist for some length of time in an energy state greater than the ground state. Such an excited energy state is known as a metastable state. The nuclide designation for an atom in the metastable state includes the label "m" for metastable, for example ^{133}I^{m}.

overhead - The vapor and condensate generated in the operation of an evaporator.

pCi - Abbreviation for picocurie. A unit of radioactivity. One pCi corresponds to 2.22 decays per minute. See curie; 1×10^{12} pCi equal 1 curie.

perched water - Water whose downward percolation is interrupted by low-permeability soil.

person-rem - Unit of collective dose to a population group. For example, a dose of 0.0002 rem to each of 10,000 individuals results in a collective dose of 2 person-rem.

photon - Electromagnetic radiation that for purposes of this report consists of either a gamma ray or an x-ray.

POPGASP - An SRS-modified version of the computer codes XOQDOQ and GASPAR. POPGASP calculates doses to offsite populations from routine atmospheric releases of radionuclides.

population dose - Another term for collective dose; see also person-rem.

rad - Unit of absorbed dose; see also absorbed dose.

radioactive decay - The process an unstable nucleus undergoes to achieve greater stability. Decay processes included in this report are alpha particle emission, beta particle emission, isomeric transition, and spontaneous fission. In alpha particle decay, an unstable nucleus emits a particle consisting of two neutrons and two protons. Gamma rays and x-rays may be associated with alpha decay. Heavy nuclides such as ^{238}U and ^{252}Cf undergo alpha particle decay. In beta particle decay, an unstable nucleus transforms a neutron into a proton and electron. The electron is ejected from the nucleus (a beta particle is an electron originating in the nucleus). Gamma rays and x-rays may be associated with beta decay. A metastable state may decay by emitting a beta particle or by isomeric transition to the ground state. Gamma and x-rays may be emitted in the isomeric transition process. For a description of the spontaneous fission decay mode, see nuclear fission.

The chain of nuclides involved in radioactive decay are referred to in terms of genetic relationships. The starting nuclide is called the parent. The nuclide the parent decays to is called the daughter, etc.

radioactive equilibrium - A steady-state condition in radioactive decay when the daughter nuclide is also radioactive and has a half-life shorter than that of the parent. Following chemical separation of parent and daughter nuclides, atoms of the daughter produced by subsequent decay of the parent will accumulate faster than they decay in the purified parent until equilibrium is established. For practical purposes, equilibrium is attained after a time interval equal to about 7 half-lives of the daughter. In this steady state condition, the daughter decays at a rate equal to its rate of formation; its rate of formation depends on the half-life of the parent. Therefore, in equilibrium, the apparent half life of the daughter is equal to the half-life of the parent.

The equilibrium relationship is demonstrated in Table 2-1. One of the products of irradiation in SRS reactors is ^{132}I with a half life of 2.3 hours. During the cooling period, it is in equilibrium with its ^{132}Te parent and decays with the ^{132}Te half-life of 3.3 days.

radioiodine - A collective term for radioactive isotopes of the element iodine.

radiological half-life - The time required for one half of a given number of atoms of a specific isotope to undergo radioactive decay.

radionuclide - A radioactive nuclide.

rem - Unit of dose equivalent. The typical unit is millirem, or one one-thousandth of a rem.

seepage basin - An excavated area that receives wastewater. It is designed to prevent overflow unless into another seepage basin. Thus water escapes from these basins by evaporation or migration through the soil. Soluble materials migrate with the water. The ion exchange properties of the soil retard the migration of various soluble materials to different degrees.

sensitivity - 1. The smallest quantity of a material that can be reliably measured in the analysis of a sample. 2. The degree to which a particular tissue is affected by ionizing radiation.

SHIELD - An SRL-developed computer code that calculates the inventory of fission products for given reactor operating conditions. The calculation of the inventories for over 800 fission products accounts for the conversion of fission products by neutron capture reactions as the fission products accumulate in the material being irradiated. Documentation of SHIELD is found in Finch, D.R., 1987, "The SHIELD System User's Manual", DPSTM-87-700-2, E.I. du Pont de Nemours and Company, Inc., Aiken, SC.

SHIELD was used to calculate radioiodine inventories for Table 2-1. Missing from Table 2-1 is ^{126}I, which is principally produced by the (n,2n) nuclear reaction on the stable isotope ^{127}I as this fission product accumulates (the SHIELD calculation does not include the (n,2n) nuclear reaction). However, the activity of ^{126}I is expected to be several orders of magnitude less than the activity of ^{128}I, which is a minor component.

Figures

short-cooled - Irradiated material that has not been allowed to cool for time sufficient to reduce the content of short-lived radioiodine to inconsequential levels.

silver zeolite - A zeolite is a hydrated aluminum and calcium or sodium silcate with an open molecular structure that allows trapping of certain vapors. Silver-substituted zeolite is particularly efficient for trapping of iodine. See molecular sieve.

spontaneous fission - See nuclear fission and radioactive decay.

stable isotope - A nonradioactive isotope.

supernate - Liquid above a precipitate or sediment.

surface mixing layer - A layer of the atmosphere characterized by a temperature inversion that inhibits further vertical mixing of releases in the atmosphere; mixing then progresses only horizontally. The height of the mixing layer is lower at night than during daylight. The average daily height is lower in winter than summer.

tritium - A radioactive isotope of the element hydrogen. Tritium is one of the principal products of the SRS.

unplanned release - The release of abnormal quantities of radioiodine to the environment. This results from abnormal operating conditions.

volatilization - The process of transforming a material to the vapor state from the solid state or from solution.

watershed - The area of the earth's surface that drains into a body of water.

weighting factor - A tissue-specific factor which equates the health risks from selective irradiation of a given tissue to the health risks from uniform, whole-body exposure.

XOQDOQ - A computer code developed for the Nuclear Regulatory Commission to evaluate the transport and dispersion of atmospheric effluents.

x-ray - A photon that is emitted from the electron cloud surrounding a nucleus when an electron undergoes a transition from a higher energy level to a lower energy level. In this report, this energy level transition typically is associated with radioactive decay of a nuclide. Because the transition is between specific energy levels, the x-ray has a specific energy which is characteristic of the element the daughter nuclide belongs to. In the decay of typical isotopes of iodine found at SRS, xenon x-rays of approximately 30 keV energy are emitted.

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Preface

Radioiodine, which is the collective term for all radioactive isotopes of the element iodine, is formed at the Savannah River Site (SRS) principally as a by-product of nuclear reactor operations. Part of the radioiodine is released to the environment during reactor and reprocessing operations at the site. The purpose of this report is to provide an introduction to radioiodine production and disposition, its status in the environment, and the radiation dose and health risks as a consequence of its release to the environment. A rigorous dose reconstruction study is to be completed by the Center for Disease Control during the 1990s.

The report covers the time period from the mid 1950s, when operations commenced, through 1989. The majority of the reported data for releases and environmental measurements are annual average values compiled from the published reports cited in the Bibliography. These are public reports that have been periodically issued since the site was established. Results of special environmental studies that did not appear in such reports, but were separately published, are cited within each chapter.

This report is one in a series to evaluate specific radionuclides. The environmental consequences of various scenarios for possible closure of waste sites are not addressed; such scenarios were published in SRS reports known as Environmental Information Documents. Accident scenarios for facilities are also not addressed in this report, but they were published in SRS reports known as Safety Analysis Reports and Safety Information Documents.

The reports of this series are intended to be "living documents" that will be updated in future years. The authors acknowledge the efforts of numerous colleagues at SRS who helped in providing data and information. Among those are Dave Hayes, Larry Heinrich, Doris Hoel, Terry Killeen, and Larry Spradley. Deserving special acknowledgment are Vickie Amos, Vonette Boylston, Felecia Brooks, Sara Cole, Gloria Overstreet, and Elaine Ray who extracted environmental radioiodine concentrations from published reports and entered the values into a database. The authors also extend their appreciation to Richard Shipley, Gary Evans, and Debra McCroskey for providing editorial support.

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Executive Summary

The offsite dose attributed to radioactivity released to the environment from the Savannah River Site (SRS) is small compared to the dose that individuals receive from all other sources. Radioiodine, a component of the released radioactivity, presents a negligible risk to the offsite population and environment in the vicinity of SRS.

The lifetime risk from radioiodine released to the atmosphere during 1955-1989 is 1.5 fatal cancer cases in the population residing within 80 km of SRS; the lifetime risk from all other causes of cancer is approximately 90,000 fatalities in that population. The lifetime risk due to aqueous releases of radioiodine is 0.007 fatal cancer cases in the population consuming treated water from the Savannah River; that population will experience a lifetime risk of approximately 11,000 fatalities from all other causes of cancer. A relationship between cancer deaths and SRS radioiodine is not discernible.

Released radioiodine principally consists of ^{131}I and ^{129}I, which have half-lives of 8 days and 16 million years, respectively. Annual atmospheric and aqueous releases of radioiodine significantly decreased beginning in the 1960s as stricter emission controls were implemented. No measurable quantities of short-lived ^{131}I released through 1989 remain in the environment as of 1990.

SRS released approximately 2,500 curies of ^{131}I and 6 curies of ^{129}I to the atmosphere from 1955 - 1989. The releases occurred during routine operations and one acute release incident. Nearly two-thirds of the total ^{131}I was released during routine operations in one year —1956. The acute atmospheric release incident occurred in 1961 when 153 curies of ^{131}I was released from the chemical separations area over several weeks. The incident was due to the inadvertent reprocessing of short-cooled irradiated materials.

Atmospheric transport mechanisms distributed much of the SRS releases into the local offsite environment where SRS ^{131}I was detected in most years from 1955-1961. Since then it has been detected only intermittently.

The annual maximum dose to an offsite adult due to SRS radioiodine releases to the atmosphere ranges between 17 and 0.04 mrem. The 17-mrem dose occurred in 1956, the year of greatest ^{131}I releases. Naturally-occurring radiation produces an annual dose of approximately 300 mrem in the average adult residing in the local area.

Aqueous releases of radioiodine were directly discharged to site streams or to earthen seepage basins. The aqueous releases were the result of routine operations and several failures of irradiated fuel elements.

Measured releases of ^{131}I to streams total approximately 300 curies from 1957 - 1978; direct discharges to streams ceased in 1978. The maximum annual release of 87 curies to site streams occurred in 1962. The streams are tributaries of the Savannah River, which is a water supply for two downriver water treatment plants. Consumers of treated Savannah River water received a radiation dose from ^{131}I that is minor compared to the dose from naturally-occurring radiation. The greatest annual maximum dose to the adult consumer is 0.3 mrem in 1957.

The seepage basins received approximately 2,600 curies of ^{131}I and 3 curies of ^{129}I during 1955 - 1989. An unknown fraction of radioiodine volatilized from the basins to the atmosphere. Other radioiodine migrated with the basin water into the ground. Near the seepage basins in the F chemical separations area, the migrating ^{129}I contaminates a shallow geological formation, the Santee Formation, which is a source of drinking water elsewhere at SRS. However, the drinking water is not contaminated with ^{129}I; the contaminated groundwater flows in the opposite direction from the B and Savannah River Laboratory areas, which are the only SRS areas that use Santee water. Furthermore, Upper Three Runs Creek cuts through the Santee Formation to physically isolate the source of the drinking water from the contaminated water. The absence of elevated tritium concentrations in the drinking water confirms that the drinking water is not contaminated with ^{129}I; contaminated groundwater contains approximately 10^5 greater concentrations of tritium than ^{129}I.

Migrating shallow groundwater principally resurfaces adjacent to Four Mile Creek after a transit time of at least 14 years. The maximum ^{129}I concentration observed in resurfaced groundwater is 410 pCi/L. The resurfaced groundwater becomes diluted as it drains into Four Mile Creek, which drains into the Savannah River. Concentrations of ^{129}I in the river water are typically less than 1 percent of the EPA drinking water standard of 1 pCi/L. Consumers of treated Savannah River water receive an insignificant dose from migrating ^{129}I.

This report serves as an introduction to the consequences of radioiodine releases. A rigorous reconstruction study is to be completed by the Center for Disease Control during the 1990s.

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The radioactive isotopes of the element iodine are collectively referred to as radioiodine. The two principal isotopes of interest in this report are ^{129}I and ^{131}I, which have radiological half-lives of 16 million years and 8 days, respectively.

Because all isotopes of iodine essentially have the same chemical and biological characteristics, radioiodine and stable iodine behave identically in the environment, the food chain, and the human body. Iodine, an essential element to nutrition, is principally found in the thyroid of the human body. Ingested radioiodine contributes to the radiation dose that humans receive from all sources.

Natural and manmade radioiodine are introduced in this chapter. They are discussed in the context of the global environment, regional environments in the vicinity of manmade sources, and specifically the environment in the vicinity of the Savannah River Site (SRS).

Radioiodine in the Global Environment

Mechanisms for production of natural radioiodine include the interaction of cosmic rays with the earth's atmosphere and fission of uranium and thorium in the earth's crust. Fission of uranium and thorium occurs principally by spontaneous fission (Edwards, 1962). Radioactive isotopes of iodine are among the products of the fission mechanism. Cosmic rays interact with xenon in the earth's atmosphere to produce radioiodine through nuclear reactions (Edwards, 1962). Essentially, the only radioactive isotope of iodine to survive the transit time between production in the atmosphere and deposition on the earth's surface is ^{129}I.

The global inventory of natural ^{129}I accumulated over the lifetime of the earth is estimated to be 10 curies in the terrestrial environment and 30 curies in oceans and the atmosphere (NCRP, 1983). Nuclear weapons testing, which began in 1945, has introduced approximately 10 curies of ^{129}I into the atmosphere (NCRP, 1983); radioiodine is produced by fission reactions when a nuclear weapon is detonated.

Atmospheric nuclear weapons testing is the greatest source of manmade radioiodine in the global environment. Weapons testing has released approximately 1.4 x 10^{10} curies of ^{131}I into the atmosphere (Eisenbud, 1987a). A significant part of the radioiodine from these sources has been deposited as global fallout on the earth's surface—predominantly in the northern hemisphere. The USA, UK, and USSR ceased atmospheric testing of nuclear weapons in 1962. Since then, as a result of radioactive decay, the ^{131}I in the global environment has generally decreased to immeasurably small concentrations . Occasional atmospheric tests conducted by China and France reintroduced ^{131}I to the environment for short periods of time; however, no atmospheric tests have been conducted by any nation since 1980.

Two nuclear reactor accidents—the satellite Cosmos 954 in 1978 and the Chernobyl nuclear reactor accident in 1986—released radioiodine to the global environment. The USSR satellite Cosmos 954 was powered by a nuclear reactor, which was estimated to have contained 4,900 curies of ^{131}I upon reentry into the earth's atmosphere (Eisenbud, 1987b). The satellite disintegrated upon reentry and distributed radioiodine throughout the upper atmosphere of the northern hemisphere.

The Chernobyl accident in the USSR is estimated to have released 7.3 x 10^6 curies of ^{131}I (USSR, 1987). Based on measurements of the ratio of ^{131}I to ^{129}I in environmental samples (Paul et al., 1987), it is estimated that 0.2 curies of ^{129}I were released to the environment. Accident conditions were severe enough to inject some of the radioiodine into high altitudes, which allowed radioiodine to be distributed throughout the northern hemisphere.

Radioiodine in Regional Environments

The impact of globally-distributed manmade radioiodine is generally greater near the source than at distant locations. Fallout from events such as the Chernobyl reactor accident and atmospheric testing of nuclear weapons was more significant in their respective regional environments than in the global environment.

A nuclear weapons test conducted underground may also release radioiodine to the atmosphere. Volatile forms of iodine may escape to the atmosphere through fissures, which form in the ground as a result of the explosion. Such an incident was detected in five European countries following an underground test by the USSR in 1987 (Bjurman et al., 1990).

In addition to Chernobyl, other reactor accidents have released radioiodine to regional environments. Fourteen land-based nuclear reactor accidents involving damage to the reactor core have been reported (Eisenbud, 1987c). The greatest quantities of ^{131}I were released in the four accidents shown in Table 1-1. Other than at Chernobyl, accident conditions were not severe enough to result in the distribution of large quantities of radioiodine beyond regional environments.

Under normal reactor operating conditions, radioiodine is essentially contained within the irradiated fuel and targets. Only small quantities of radioiodine are released to regional environments as a result of the normal operation of experimental, research, power, and production nuclear reactors.

Nuclear fuel reprocessing operations are the most significant sustained source of radioiodine releases since the 1940s. Reprocessing facilities are located in several countries and are used to chemically separate materials irradiated in experimental, research, power, and nuclear materials production reactors. As a result of the chemical separations process, part of the radioiodine content of irradiated materials can be released to the environment through atmospheric and aqueous effluents.

Table 1-1. Examples of radioiodine released in land-based nuclear reactor accidents

Year	Country	Reactor	Curies ^{131}I Released	Reference
1986	USSR	Chernobyl	7,300,000	USSR, 1987
1979	USA	Three Mile Island	15	Pasedag and Postma, 1986
1961	USA	SL-1	10	Eisenbud, 1987c
1957	UK	Windscale	20,000	Eisenbud, 1987c

Radioiodine in the SRS Environment

The US Department of Energy's Savannah River Site, which was previously known as Savannah River Plant, was established in 1951 as a government-owned contractor-operated facility to manufacture special nuclear materials—principally tritium and plutonium—for national defense purposes. The first production nuclear reactor began operations in late 1953. Reprocessing of irradiated materials in chemical separations facilities began in late 1954. SRS is located in the central Savannah River area of South Carolina (see Figure 1-1).

Numerous radioactive isotopes of iodine are formed as byproducts during the irradiation of fuel and targets. Most of the isotopes are short lived and decay to insignificant activity levels between the end of irradiation and the beginning of reprocessing. Because of half-life considerations, ^{131}I and ^{129}I are the only radioactive isotopes of iodine consistently released to the environment as a result of normal operations.

Atmospheric releases through 1989 totaled approximately 2,500 curies of ^{131}I and 6 curies of ^{129}I. Site streams received approximately 300 curies of ^{131}I; such discharges were terminated in 1978. All of the ^{131}I released through 1989 decayed to immeasurably low levels by the end of 1990; however, essentially all of the ^{129}I released throughout the history of SRS remains distributed in the environment.

Releases of ^{129}I represent a very small part of the total releases of all radionuclides from SRS. However, ^{129}I contributes a large part of the offsite population dose from SRS releases. In 1989, ^{129}I accounted for only 2 x 10^{-5} percent of the total radioactivity released from SRS to the atmosphere; however, it accounted for 13 percent of the offsite population dose. These facts illustrate the significance of the very long half-life of ^{129}I, its mobility in the environment, and the pathway to humans. (Corresponding values for ^{131}I in 1989 were 1 x 10^{-7} percent of releases and 2 x 10^{-3} percent of dose.)

The offsite population dose due to all SRS releases in 1989 was small compared to the overall dose that members of the general public received from all radiation sources. Figure 1-2 shows that SRS releases in 1989 accounted for approximately 0.006 percent of the dose from ionizing radiation received by members of the population in the vicinity of the site—those residing within 80 km of SRS and those consuming treated water from the Savannah River. Atmospheric ^{129}I releases comprised 13 percent of this 0.006 percent. Therefore, the importance of radioiodine is apparent only in the context of SRS dose contributions; on the scale of overall environmental doses, radioiodine is a minor consideration.

The chapters that follow describe the origin and disposition of radioiodine at SRS, measured releases and inventories, results of routine measurements and special studies in the environment, and an assessment of the dose and risk to the population due to SRS radioiodine releases.

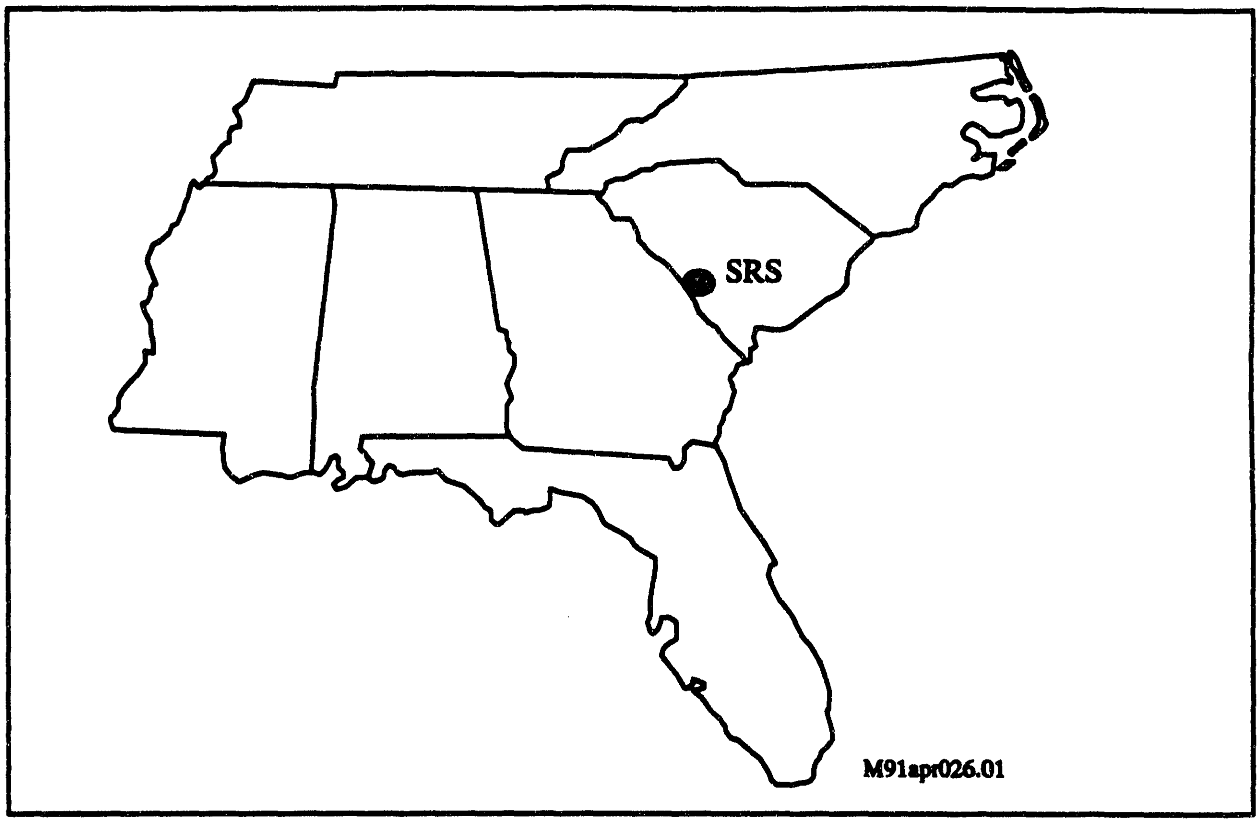

Figure 1–1. Location of SRS in the Southeastern United States

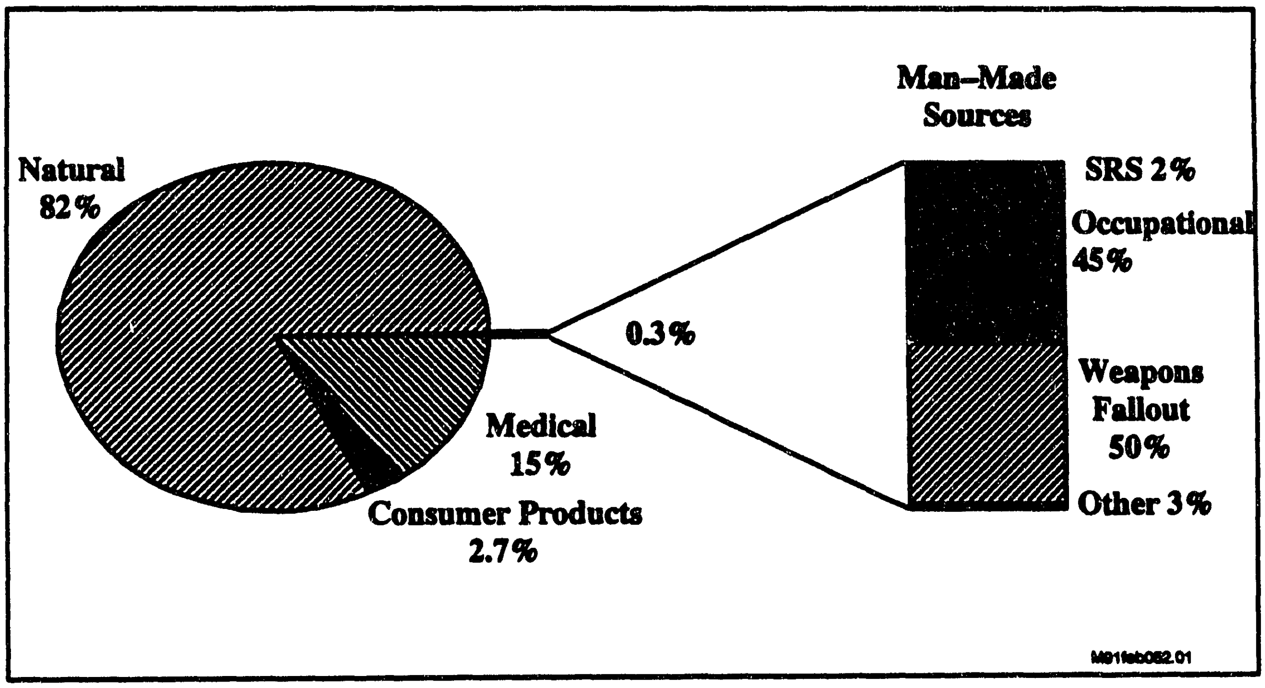

Figure 1–2. Relative Contributions from Various Sources to the Total Radiation Dose (225,200 person–rem) for the Population in the Vicinity of SRS in 1989

References for Chapter 1

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2

This chapter gives a general overview of the origin, uses, and disposition of radioiodine at SRS under normal operating conditions. The locations of SRS facilities that have or had the potential to release radioiodine are shown in Figure 2-1. The chemical separations and reactor facilities have the greatest potential to release radioiodine. They are located near the center of the 800-square-kilometer Savannah River Site. The history of measured releases from specific SRS facilities is presented in Chapter 3.

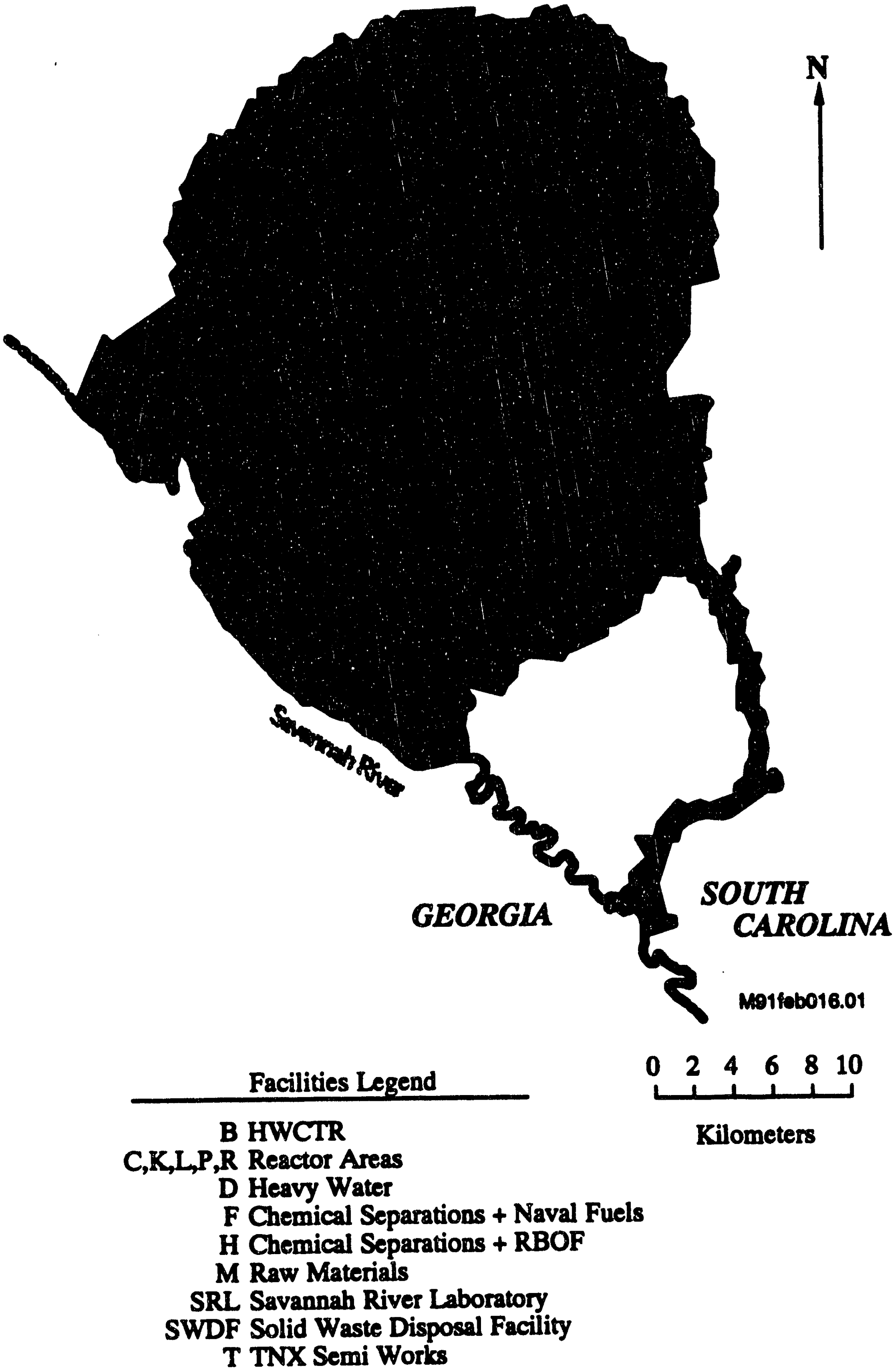

Figure 2–1. SRS Facilities with the Potential to Release Radioiodine

Origin of Radioiodine at SRS

Radioiodine at SRS predominantly originates in the fuel and targets that are irradiated in the nuclear materials production reactors. Other site operations and offsite sources contribute to the inventory of radioiodine at SRS.

Irradiation in Production Reactors

Most of the radioiodine at SRS is formed as a by-product in the nuclear fuel and targets during operation of the production reactors. The principal mechanism for production of radioiodine is neutron-induced fission. When a reactor is operating, neutron-induced fission reactions occur in the ^{235}U fuel of the reactor core. The reactions form a variety of fission products, which include isotopes of iodine.

In addition to fission products, each neutron-induced fission reaction in the fuel produces several neutrons, some of which induce additional fission reactions and maintain a chain reaction. Some of the remaining neutrons interact with target materials in the reactor. (Fuel and targets are encapsulated separately and referred to as elements.) One such interaction, neutron capture in a ^{238}U target, results in the production of a special nuclear material, ^{239}Pu.

In certain target materials, such as ^{238}U, neutron-induced fission is a competing reaction with neutron capture. Fission also occurs in some of the target's minor constituents, such as ^{235}U and ^{239}Pu in ^{238}U targets.

Table 2-1. Inventory of radioiodine produced in "representative" material irradiated in SRS production reactors.

		Curies Remaining at Various Times After End of Irradiation			
Isotope*	**Half-Life**	**0 sec.**	**24 hrs.**	**100 days**	**200 days**
^{128}I	25m	1.5×10^4	—	—	—
^{129}I	1.6×10^7y	2.3×10^{-1}	2.3×10^{-1}	2.5×10^{-1}	2.5×10^{-1}
$^{130}I^m$	9m	9.2×10^4	—	—	—
^{130}I	12h	1.4×10^5	3.5×10^4	—	—
^{131}I	8.0d	7.6×10^7	7.0×10^7	1.3×10^4	2.3×10^0
^{132}I	2.3h/3.3d**	1.1×10^8	8.9×10^7	—	—
$^{133}I^m$	9s	3.8×10^6	—	—	—
^{133}I	21h	1.8×10^8	8.2×10^7	—	—
$^{134}I^m$	4m	1.3×10^7	—	—	—
^{134}I	53m	2.0×10^8	1.3×10^0	—	—
^{135}I	6.6h	1.6×10^8	1.5×10^7	—	—
$^{136}I^m$	47s	5.0×10^7	—	—	—
^{136}I	83s	7.9×10^7	—	—	—
^{137}I	24s	7.9×10^7	—	—	—
^{138}I	7s	4.1×10^7	—	—	—
^{139}I	2s	1.8×10^7	—	—	—
^{140}I	<1s	4.1×10^6	—	—	—
^{141}I	<1s	8.6×10^5	—	—	—
^{142}I	<1s	1.1×10^5	—	—	—
Totals		1.0×10^9	2.6×10^8	1.3×10^4	2.5×10^0

* "m" designates a metastable energy state greater than the ground state of a particular isotope.

** Until chemically separated, ^{132}I (2.3-hour half-life) is in equilibrium and decays with the 3.3-day half-life of its ^{132}Te parent.

The total inventory of radioiodine in the core of an operating production reactor is approximately 10^9 curies. This inventory is achieved within several days after each startup of the reactor. It remains at that level until the reactor is shut down after a specified irradiation time. The radioiodine inventory comprises approximately 6 percent of the total fission-product inventory during reactor operation.

The inventory of the individual iodine isotopes is shown in Table 2-1 as a function of time after the end of irradiation. The data in Table 2-1 demonstrate that radioiodine activity levels quickly decrease after reactor shutdown. For example, only 26 percent of the inventory at the end of irradiation remains after 24 hours. The data also demonstrate that it is prudent to allow the irradiated material to cool for about 200 days before reprocessing through chemical separations.

The inventory of each iodine isotope was calculated with the SHIELD computer code for "representative" irradiated material. (See Glossary for a description of SHIELD.) This material represents the variety of operational parameters that were used from 1953 to 1988. These include composition of fuel and targets, irradiation time, and reactor power level.

Other Sources Due to SRS Operations

Small quantities of radioiodine have been produced at SRS by test reactors, spontaneous fission, and neutron activation analysis. The activity levels of radioiodine from these sources are insignificant when compared to the activity levels in irradiated nuclear fuel and targets; however, these sources are discussed in the following subsections to provide a complete overview of potential radioiodine releases.

SRS Test Reactors

Several small nuclear reactors were in use at two SRS locations from the 1950s through the 1970s. The Heavy Water Components Test Reactor (HWCTR), located in B Area, was used in the early 1960s to test prototype fuels for a proposed heavy water power reactor. The other test reactors were located in M Area and were used from the mid-1950s to the late 1970s. The Process Development Pile and the Lattice Test Reactor were used as zero-power mock-up facilities to test components for the production reactors. The Subcritical Experimental Pile was also used to test component designs. The Standard Pile provided neutrons for experiments such as neutron radiography and neutron activation.

Spontaneous Fission

Spontaneous fission is a radioactive decay mode that occurs in heavy elements such as uranium. It produces neutrons and fission-product radioiodine and is one of the mechanisms for the production of radioiodine in nature. The occurrence of spontaneous fission ranges from rare to frequent.

An example of a radionuclide that rarely decays by spontaneous fission is ^{238}U. The equilibrium concentration (see Glossary for explanation) of fission-product ^{131}I is approximately 0.0008 pCi/g of ^{238}U, which is essentially equal to its equilibrium concentration in natural uranium. (The ^{238}U content of natural uranium is 99.3 atom percent). The inventory of ^{131}I due to spontaneous fission of uranium in nature within the boundary of SRS is 10^4 pCi/cm of soil depth. This is based on the average abundance of 4 parts per million for uranium in the earth's crust. If a 100-cm soil depth were considered, the ^{131}I inventory within the site boundary would be 10^6 pCi, or 10^{-6} curies.

Uranium used in operations at SRS has been chemically purified. Because of its short half-life, ^{131}I attains the equilibrium concentration with the ^{238}U in SRS uranium within several weeks after chemical purification. In contrast, long-lived ^{129}I does not reach equilibrium; this would require tens of millions of years after chemical separation. Consequently, ^{129}I is essentially nonexistent in SRS uranium that has not been irradiated, but because of the age of the earth, ^{129}I does exist in uranium in nature.

An example of a radionuclide that frequently decays by spontaneous fission is ^{252}Cf. The equilibrium concentration of ^{131}I is approximately 2×10^{11} pCi/g of ^{252}Cf. This high concentration, almost one curie per gram, is due to the frequent occurrence of spontaneous fission and the 2.6-year half-life of ^{252}Cf.

Because ^{252}Cf frequently decays by spontaneous fission, it is a valuable neutron source. It has been produced in SRS reactors by the irradiation of ^{244}Cm targets. From the mid 1960s until 1987, the Savannah River Laboratory (SRL) fabricated industrial and medical ^{252}Cf sources for offsite use. Some of the sources have been retained at SRS for neutron calibrations in the ^{252}Cf fabrication facility and for use in two neutron activation facilities at SRS. One ^{252}Cf neutron activation facility was operated in M Area in the 1970s, and the other has been operational at SRL since the mid 1970s.

The spontaneous-fission-produced radioiodine present inside the encapsulated ^{252}Cf sources at the SRL ^{252}Cf Neutron Activation Analysis Facility (NAAF) as of 1979 was calculated for a safety analysis (Winn, 1979). The major iodine isotopes are shown in Table 2-2 together with the curies of each isotope that would be expected to volatilize in the event of a source failure. The potentially volatilized component represents 40% of the radioiodine inventory in the source.

Other isotopes of uranium and isotopes of thorium, plutonium, and curium also spontaneously fission. One or more of these materials have been used in M Area, the reactor areas, the chemical separations areas, the Naval Fuels Facility, TNX Semi-works, or SRL.

Neutron Activation Analysis

Neutron activation analysis is an analytical technique for measuring elemental compositions in materials. The ^{252}Cf neutron activation facilities, the Standard Pile, and the C and K production reactors have been used for that analysis. For example, from the late 1970s to mid 1980s, environmental samples were activated in C Reactor to determine uranium content. Traces of fission-product radioiodine were produced in the irradiated samples. From 1971 to 1984 iodine concentrated from environmental samples was activated in K Reactor to detect trace levels of ^{129}I; small quantities (typically less than 10^{-6} curies) of the activation products ^{130}I, ^{128}I, and ^{126}I were produced in the irradiated samples.

Table 2-2. Portion of radioiodine inventory in the ^{252}Cf NAAF as of 1979.

Isotope	Curies*
^{131}I	0.13
^{132}I	0.18
^{133}I	0.28
^{134}I	0.33
^{135}I	0.40
^{136}I	0.34
^{137}I	0.17
^{138}I	0.07
^{139}I	0.02
Total Radioiodine	1.92

* Activity represents only the portion of the inventory that is predicted to volatilize in the event of a source failure. This is 40 percent of the inventory.

Materials Originating Offsite

Certain fuels irradiated at offsite non-commercial facilities are shipped to SRS for reprocessing. While awaiting reprocessing, the fuel is stored in the Receiving Basin for Offsite Fuel (RBOF), which is located in H Area.

The isotopes ^{125}I, ^{129}I, and ^{131}I are purchased from commercial vendors typically in quantities less than 10^{-3} curies. The isotopes are used at SRS for experimental purposes, such as chemical yield determination, instrument calibration, and determination of retention characteristics and chemical properties of iodine on various absorbers. These isotopes are predominantly used in SRL. The amount of radioiodine present at SRS due to commercial purchases is insignificant when compared to the amount produced in fuel and targets at SRS production reactors or stored in RBOF.

SRS and it's vicinity have been subjected to global fallout of man-made radioiodine since the 1940s. Most global fallout has occurred because of atmospheric nuclear weapons tests.

Disposition of Radioiodine at SRS

The principal means for disposing of radioiodine are containment and time. Table 2-1 shows that very short-lived isotopes of iodine produced in SRS reactors decay to insignificant activity levels shortly after end of irradiation. By 200 days of cooling, ^{131}I and ^{129}I are the only isotopes of iodine remaining in measurable quantities. Their total activity is 2 parts per billion of the original radioiodine activity at the end of irradiation. To describe the disposition of the longer lived isotopes, the sources of radioiodine are categorized as follows:

Major Sources
- fuel and targets irradiated in production reactors
- certain fuels irradiated offsite

Minor Sources
- fuel and targets irradiated in SRS test reactors
- spontaneous fission
- neutron activation samples
- commercially purchased radioiodine
- global fallout

Radioiodine in Fuel and Targets Irradiated in Production Reactors

Under ideal operating conditions, radioiodine is contained within the cladding of fuel and target elements during irradiation and cooling. Cooling is the interval between the end of irradiation and the beginning of chemical separations. Irradiated materials are stored underwater in reactor basins for most of the cooling time, which was approximately 100 days during the initial years of operation. This was increased to approximately 200 days by the mid-1960s.

Under normal operating conditions, it is possible for traces of radioiodine to escape from irradiated fuel and target elements to the environment through small defects in the cladding. Air and water at reactors are monitored for such possible releases. Actual releases are described in greater detail in Chapter 3.

After the cooling period, fuel and targets are treated in the chemical separations areas to recover the desired products from the waste products such as radioiodine. In chemical separations part of the iodine volatilizes. The primary emission control is a silver-nitrate reactor to remove iodine from process air before the air is exhausted to the atmosphere through a 61-m stack. The silver-nitrate reactor consists of ceramic chips, also known as beryl saddles, coated with silver nitrate. Iodine passing through a bed of chips reacts with silver nitrate to form solid silver iodide.

Spent chips were washed and reused in the early years of operations. The wash solutions were processed and stored in underground tanks for radioactive waste. In later years spent chips were buried in the Solid Waste Disposal Facility (SWDF), which was previously known as the burial grounds. This practice was terminated in the early 1980s. Since that time chips have been stored at the SWDF.

Radioiodine contaminates aqueous wastes at the chemical separations facilities. The wastes are evaporated and then sent to underground storage tanks for radioactive waste. The condensate from evaporation was sent to the separations area seepage basins until November 1988 when use of the seepage basins was terminated. Since then condensate has been sent to the Effluent Treatment Facility where it is treated to remove radionuclides and chemicals before being discharged to Upper Three Runs Creek.

Most of the atmospheric and aqueous effluents in the chemical separations areas have been monitored for possible radioiodine releases. Monitoring techniques and measured releases are described in Chapter 3.

The disposition of long-lived ^{129}I produced during the first 20 years of chemical separations processing is estimated to be 40 percent stored in waste tanks, 30 percent buried in the SWDF, 20 percent released to the atmosphere, and 10 percent released to seepage basins (Cornman, 1974). The releases to the atmosphere and seepage basins and the burial in the SWDF have had a direct impact on the environment. Environmental radioiodine is discussed in detail in Chapter 4.

Radioiodine in Certain Fuels Irradiated Offsite

Irradiated fuels received from certain offsite facilities are stored underwater in RBOF until the fuel is ready for chemical separations. The fuel then enters the regular process stream in H Area. The disposition of radioiodine becomes the same as described for fuel irradiated in SRS production reactors.

Radioiodine in Fuel and Targets Irradiated in Test Reactors

Fuel and targets from the various test reactors were primarily sent to the RBOF for cooling before chemical separations. Some were sent to SRL for research or to reactor materials fabrication facilities. Reactor materials fabrication facilities only received fuel or targets that were not irradiated; this material was blended into the standard fabrication process for targets and fuel to be used in the production reactors.

Radioiodine from Spontaneous Fission

Radioiodine produced by spontaneous fission in heavy elements is retained by these elements as long as they remain encapsulated.

Neutron Activation Analysis Samples

Environmental samples are analyzed by high sensitivity neutron activation for the detection of trace levels of uranium and ^{129}I. Typically, the uranium analysis is non-destructive; therefore, any radioiodine produced from fission of uranium is not released. The radioiodine content is minute and the samples are disposed in the SWDF as solid radioactive waste.

The analysis for ^{129}I typically involves chemical purification of iodine at SRL after irradiation. In the purification steps, traces of the activation products ^{126}I, ^{128}I, and ^{130}I are released to the SRL exhaust system. None of these trace releases are significant enough to be detected in routine radioiodine monitoring of the stacks. Traces of radioiodine are retained in various liquids used in the purification steps and then stored in the SRL temporary waste tanks. Purified radioiodine in solid form is subjected to various measurements and allowed to decay to inconsequential levels before it is buried in the SWDF.

Commercially Purchased Radioiodine

The isotopes ^{125}I, ^{129}I, and ^{131}I are purchased from commercial vendors for use, typically, at SRL. The disposition of radioiodine is similar to that described for neutron activation analysis samples.

Global Fallout Radioiodine

During the era of extensive atmospheric nuclear weapons testing (1950s and early 1960s), global fallout was one of the principal sources of radioiodine in the local environment; SRS was the other principal source. Chapter 4 presents a discussion of environmental radioiodine.

References for Chapter 2

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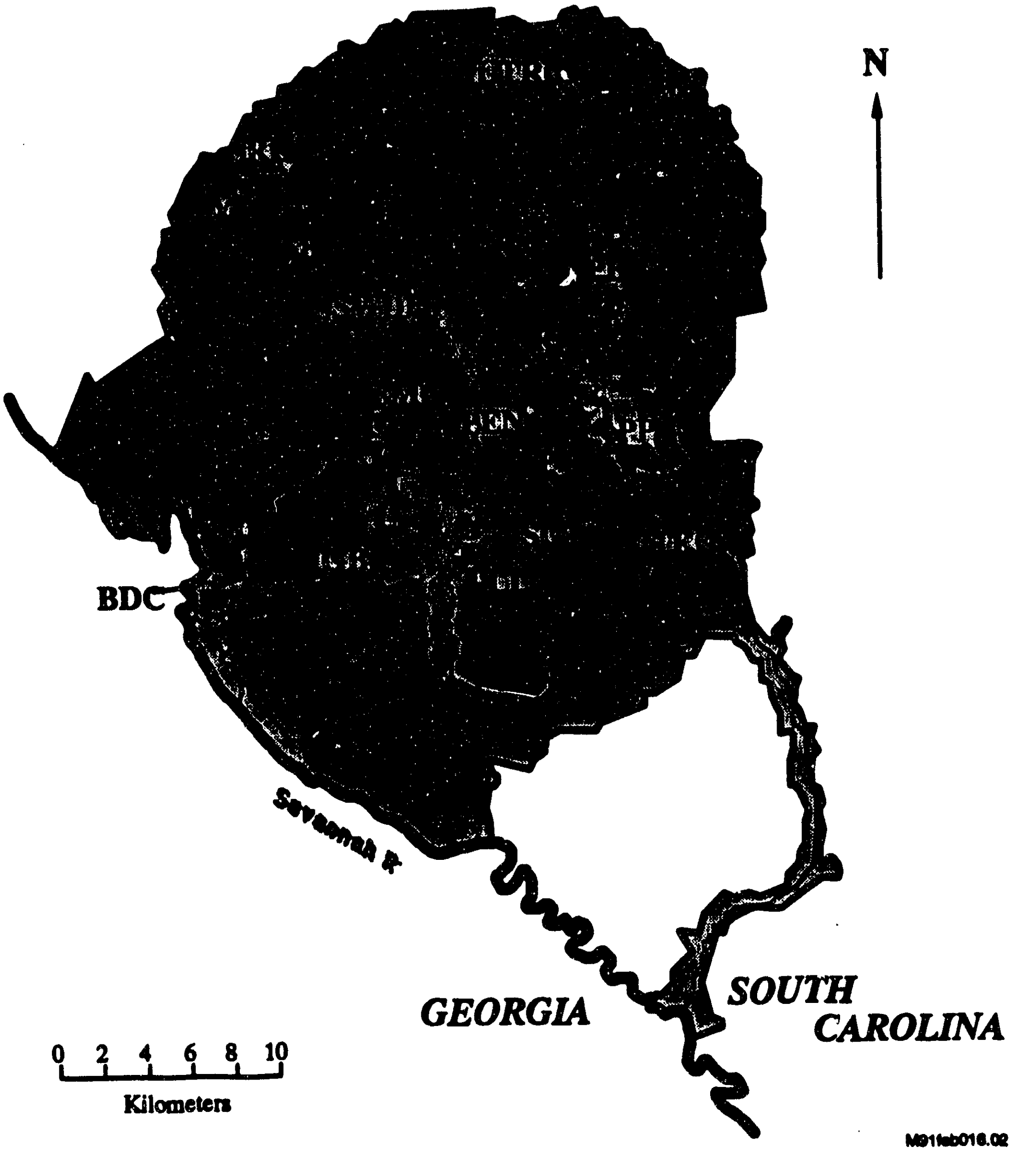
3

This chapter presents a discussion on radioiodine at SRS facilities in terms of possible release pathways, emission control features, and annual releases to the aqueous and atmospheric environments. The reported annual releases are taken from public reports cited in the Bibliography.

Routine operations at SRS facilities release radioiodine to the atmosphere, site streams, and seepage basins. The seepage basins allow short-lived radionuclides to decay to very low levels during the transit time (years) required for aqueous migration from the basins through the ground to site streams. The most significant releases of radioiodine occurred during the early years of site operations. The greatest releases originated in the F and H chemical separations facilities and the reactor facilities, all of which are located near the center of the 800-square-kilometer site. The proximity of these facilities to major streams within the boundary of SRS is shown in Figure 3-1, which also shows the location of other facilities that had the potential to release radioiodine.



Facilities Legend

B	HWCTR
C,K,L,P,R	Reactor Areas
D	Heavy Water
F	Chemical Separations + Naval Fuels
H	Chemical Separations + RBOF
M	Raw Materials
SRL	Savannah River Laboratory
SWDF	Solid Waste Disposal Facility
T	TNX Semi Works

Surface Waters Legend

BDC	Beaver Dam Creek
FMC	Four Mile Creek
IGB	Indian Grave Branch
LL	L Lake
LTRC	Lower Three Runs Creek
PB	Pond B
PC	Pond C
PEN	Pen Branch
PP	Par Pond
SC	Steel Creek
TB	Tims Branch
UTRC	Upper Three Runs Creek

Figure 3–1. Proximity of Site Streams to SRS Facilities that had the Potential to Release Radioiodine

Reactor Facilities

The five production reactors at SRS are identified by the letter designations C, K, L, P, and R. The reactors were designed to irradiate specific targets to produce special nuclear materials—principally tritium and plutonium—for national defense purposes. Certain radionuclides for other governmental purposes are also produced, such as ^{238}Pu, which is a power source for certain deep-space missions.

Facility Operations

After the reactors became operational in 1953-1955, they did not operate continuously. They were periodically shut down for maintenance, safety upgrades, or replacement of fuel and targets. In 1988 three reactors were shut down for extensive maintenance and safety upgrades; prior to 1988 one reactor had been placed on inactive status and another on standby.

The principal components of the reactor core are the fuel, targets, control rods, and moderator. When a nuclear reactor is operating, nuclear fission reactions occur in the ^{235}U fuel of the reactor core. Fission can also occur in certain target materials. Control rods are neutron-absorbing materials that are positioned in the core to control the rate of fission reactions in a reactor.

The moderator in SRS reactors is heavy water, which is circulated around the fuel and target elements in the reactor core. The moderator decreases the kinetic energy (slows the speed) of neutrons emitted in the fission process. Slow neutrons have a greater probability than fast neutrons for producing the desired nuclear reactions in fuel and target materials. The moderator also functions as the primary coolant to remove heat generated by nuclear reactions and radioactive decay in fuel and target elements. Heat is then transferred from the moderator by heat exchangers to the secondary coolant. SRS reactors use either river or pond water as the secondary coolant.

In reactors that use water from the Savannah River, the river water passes through the shell side of the heat exchanger and discharges to a site stream that flows to the Savannah River. Reactors that use pond water discharge the secondary coolant from the heat exchanger to a canal that flows to a series of cooling ponds. Cooled pond water is recirculated through the heat exchanger.

Under normal operating conditions, fuel and target elements are irradiated for a specified time, removed from the reactor core, and stored underwater in the disassembly basin of the reactor building. Storage in the disassembly basin for the cooling period, which by the mid 1960s was at least 200 days for most irradiated materials, allows short-lived radionuclides to decay to low levels before the reprocessing of the fuel and targets begins in the chemical separations areas.

The abnormal condition of a fuel or target failure, which consists of a major breech of the encapsulation material, is detected by sensors in the reactor. When a failure occurs, the reactor is shut down and the failed element is transferred into a container called a "harp". The harp is stored underwater in the reactor basin and vented to the reactor stack. Failures occurred more frequently in the early years of operation than in later years.

Pathways for Release

During normal reactor operations, traces of radioiodine escape from irradiated fuel or target elements if a defect develops in the cladding. The defect typically is a micro-scale crack. If a defect develops while the element is in the reactor core, radioiodine escapes from the irradiated material to the moderator by volatilization or by leaching of radioiodine from the exposed irradiated material.

In the more serious case of a failed element, some of the exposed uranium and plutonium can become dispersed in the moderator. After a failed element is removed, the dispersed uranium and plutonium continue to circulate with the moderator until they are removed from the moderator by an online ion exchange purification process. Neutrons generated during subsequent reactor operation cause fission in any residual circulating uranium and plutonium, thereby producing small quantities of radioiodine directly in the moderator.

Once radioiodine is in the moderator, it can circulate with the moderator, adsorb onto surfaces, migrate to the blanket gas above the moderator, or be removed by the online purification process.

Similar pathways occur in the disassembly basin during cooling of irradiated materials. Radioiodine escapes to basin water from cladding microdefects by volatilization or by leaching of radioiodine from the exposed irradiated material. Water also leaches adsorbed radioiodine from the exterior surfaces of fuel and targets. Once radioiodine is in the basin water, it circulates with the water, adsorbs onto surfaces, migrates to the air above the basin, or is removed by the basin deionizer.

Circulating moderator and basin water are the principal pathways by which aqueous radioiodine is released from reactors to the environment. If a leak exists in the reactor heat exchanger, moderator can mix with the secondary coolant. Any radioiodine released by this pathway is not quantified; however, the secondary coolant is monitored for beta-gamma activity. Leakage to the secondary coolant is a minor pathway compared to the disassembly basin pathway.

Initially, disassembly basin waters were purged directly to site streams to remove the heat generated by the stored irradiated fuel and targets. After installation of basin heat exchangers, deionizers, and filters in the 1960s, the volume of purge water decreased significantly as did the releases of radioactivity.

Periodic purges are necessary to eliminate tritium, the accumulation of which results in exposure to operating personnel. Any residual radioiodine that is not removed by the deionizer is released with the purge water, which has been discharged to earthen seepage basins or streams. Discharges to stream ceased in 1978.

A minor source of wastewater that contains radioiodine is the reactor process sewer system. This system principally receives cooling water from the shell side of the heat exchangers for reactor basins and miscellaneous process waters. Process sewer water is periodically sampled and analyzed for radionuclides.

Other wastewater is collected in process sumps at the reactor facilities. Occasionally, this wastewater contains moderator from leaks that developed during reactor operations or spills that occurred when line breaks were made during maintenance periods. The wastewater is collected and analyzed for both radionuclides and moderator. Possible disposition methods, depending on analysis results, have been to process through the Heavy Water Rework Facility or the waste evaporators in the separations areas, or to discharge to seepage basins or streams.

Circulating moderator and disassembly basin water are also the principal sources for atmospheric releases from reactors. Insoluble volatile forms of iodine in the moderator migrate, as vapor, into the blanket gas system above the moderator. Certain soluble forms of iodine volatilize in the moderator and migrate into the blanket gas system. Leaks and occasional purges of the blanket gas system allow radioiodine to be released to the process ventilation system. Volatilization from moderator leaks or spills also introduces radioiodine into the process ventilation system.

Air in the process ventilation system passes through a confinement system, which includes a carbon filter bed designed to remove radioiodine, and then exhausts to the atmospheric environment from a 61-m stack. During periods when the reactor is shut down for repairs, maintenance, or replacement of fuel and targets, air from the reactor building levels that are 6 and 12 meters below grade elevation bypasses the carbon bed enroute to the stack. This bypass occurs because a large air flow is required to remove tritium from process areas where work is performed. The capacity of the carbon bed is insufficient to handle an air flow that is 2 to 3 times greater than normal for an extended period of time. The capability exists to rapidly switch airflow to the carbon filter in an emergency.

In the disassembly basin water, insoluble volatile forms of iodine migrate, as a vapor, to the air above the basin. Certain soluble forms of iodine volatilize in the water and migrate to the air above the basin. The air above the disassembly basin is directly exhausted to the outside atmosphere, except for harp storage containers, which are vented to the process ventilation system.

Certain forms of iodine volatilize in the reactor seepage basins and migrate to the atmosphere at ground level—reactor seepage basins are open to the environment.

Emission Control

Various measures have been implemented to minimize releases of radioactivity. One of these is the extensive improvements in the quality of materials fabricated for irradiation. Such improvements minimize the formation of defects in the cladding of fuel and targets.

In the reactor areas, discharges of disassembly basin water to streams were reduced beginning in 1963 when the water was recirculated through heat exchangers for cooling and through deionizers for removal of radionuclides (except tritium). Prior to 1963 disassembly basin water was purged continuously to site streams. Although the deionizers were effective in removing particulates, permanent sand filters were installed in the late 1960s to maintain water clarity. The residues collected on the sand filters are transported to F Area to be processed and stored in waste tanks. Spent deionizer resin is transported to H Area for regeneration.

Because the heat exchangers, deionizers, and sand filters account for the removal of heat, most radionuclides, and particulates, only the accumulation of tritium prevents the indefinite recirculation of disassembly basin water. To reduce tritium exposure to workers in basin areas, basin water is periodically purged to site streams or to reactor seepage basins. No purges to site streams occurred after 1978.

Most of the radioiodine released in the early years of reactor operation originated in failed fuel elements. Reactor stack monitors and reactor core sensors detect failures or suspect failures of elements. In 1957 the practice of placing failed elements in harp storage containers was implemented.

Under normal operation conditions, moderator and some residual radionuclides adhere to fuel and target surfaces. When fuel and targets are removed from the reactor core, they are flushed with water to eliminate most of the adhered moderator and some residual radionuclides, such as radioiodine. This process was implemented in the late 1960s. After flushing, the elements are stored in the disassembly basin during the cooling time. The flush water is collected in drums for treatment in the D-Area Heavy Water Rework facility.

Atmospheric releases are minimized by the use of the confinement system, which is intended for routine operations and for a reactor incident. The system consists of a demister filter bank, a high efficiency particulate air (HEPA) filter bank, and the carbon filter bed. The latter consists of activated charcoal impregnated with 1 percent triethylenediamine (TEDA) and 1 percent potassium iodide to enhance the retention of several chemical forms of iodine. The carbon filter removes greater than 99.9 percent of the iodine from the process ventilation air.

Controls have been implemented to provide sufficient cooling time for short-lived radioiodine to decay to insignificant levels before reprocessing. The cooling time for most irradiated materials at SRS was approximately 100 days initially, but by the mid 1960s this increased to approximately 200 days. In response to an acute release incident in 1961, administrative controls (such as computer inventory of irradiated materials), and physical controls (such as an underwater ionization chamber to monitor shipments) were implemented to ensure sufficient cooling time has elapsed before reprocessing. In the 1961 incident, excessive ^{131}I was released due to the inadvertent reprocessing of very short-cooled material. After these controls were implemented, no other acute radioiodine release incident of this magnitude has occurred.

Release Monitoring

Although reactor stack air is continuously sampled for radioiodine, it has not always been quantified for the curies of radioiodine released. Continuous sampling is accomplished by passing a portion of stack air through a filter that traps iodine. The filter is changed weekly and analyzed for radioactivity. Until the early 1960s, a filter paper coated with silver nitrate was used; an activated charcoal filter, which is treated with TEDA to enhance collection of organic forms of iodine, replaced the filter paper.

The initial procedure to determine atmospheric radioiodine releases was to measure the air filters for gross beta-gamma activity. The filters were submitted for laboratory analysis of ^{131}I only if activity was detected above background. When the procedure was changed in the early 1970s to perform laboratory analysis specifically for ^{131}I by gamma spectrometry, the releases of ^{131}I from reactor stacks became quantified. Releases of ^{129}I from reactor stacks were not quantified until late 1988 when laboratory analysis for long-lived ^{129}I in the charcoal filters by low energy photon spectrometry began.

Similar procedures have been used to determine radioiodine releases to the air above the disassembly basins. The published annual atmospheric release value for ^{131}I from a reactor is the sum of stack and disassembly basin air releases. Analysis for ^{129}I is not performed.

Several techniques are used to monitor aqueous releases. Secondary cooling water from the reactors is monitored for beta-gamma activity, but not specifically for ^{131}I. Basin purge water was not routinely analyzed for ^{131}I until about 1960. Prior to that time, laboratory analysis was performed only if beta-gamma activity was detected. Specific analysis for ^{129}I is not performed because of the low probability of ^{129}I being present relative to ^{131}I (Table 2-1, page 10). Similarly, water from other release points, such as process sewers, is analyzed for ^{131}I but not for ^{129}I.

History of Measured Releases

Annual aqueous and atmospheric releases of ^{131}I from individual reactors are presented in the sections that follow. The aqueous releases are the measured discharges from the disassembly basins. Other releases (such as from heat exchangers, leaks, and sumps) are minor and are not included in the discussion of aqueous releases from individual reactors; they are included in a summary section for reactor releases.

The greatest measured aqueous releases of ^{131}I from reactors occurred during 1957 and the early 1960s. These releases were typically associated with failures of irradiated fuel elements. No measurements of short-lived isotopes, such as ^{132}I and ^{133}I, were reported at the time of these ^{131}I releases. The high-yield fission products ^{132}I and ^{133}I would still have been present in significant quantities if the releases occurred within several days after end of irradiation (Table 2-1, page 10).

No measurements for ^{129}I were conducted at the time of the greatest aqueous ^{131}I releases; however, the information in Table 2-1 leads to the conclusion that the corresponding ^{129}I releases were insignificant. For example, the activity of ^{129}I relative to ^{131}I is only one part in 300 million at 24 hours after shutdown of an SRS production reactor.

Routine atmospheric releases of ^{131}I from reactors were not quantified until approximately 1972. Releases of ^{131}I to the atmosphere are minor compared to the aqueous releases.

Atmospheric releases of ^{129}I were quantified after late 1988. No ^{129}I has been detected from any of the reactors. Although these ^{129}I measurements were conducted during the extended shutdown, the atmospheric release of ^{129}I during reactor operation is expected to be insignificant. This is because the activity of ^{129}I relative to ^{131}I is extremely small during reactor operation, and the atmospheric releases of ^{131}I have been small.

C Reactor

C Reactor was operational from 1955 until it was shut down for extensive maintenance in 1986; it was placed in standby mode in 1987. Disassembly basin purges were discharged into Four Mile Creek and three seepage basins designated 904-66G, -67G, and -68G. The seepage basins were used from 1959 to 1970 and from 1978 through 1986.

The total measured aqueous releases of ^{131}I from C Reactor during its operational lifetime are 21.8 curies to Four Mile Creek and 1.0 curie to the seepage basins. The greatest annual releases, approximately 4 curies to Four Mile Creek, occurred in 1961 and 1963 as shown in Figure 3-2.

The total measured atmospheric release of ^{131}I during the operational lifetime of C Reactor was 0.042 curies. Measured annual atmospheric releases of ^{131}I from C Reactor are shown in Figure 3-3.

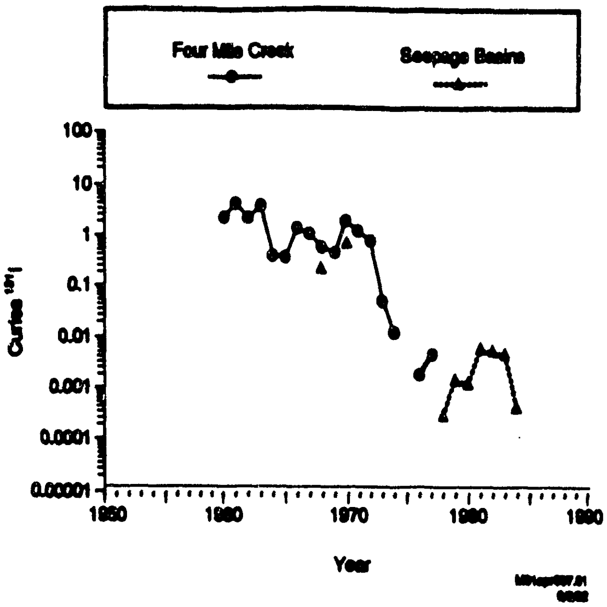


Figure 3-2. Measured Annual Aqueous Releases of ^{131}I from C Reactor

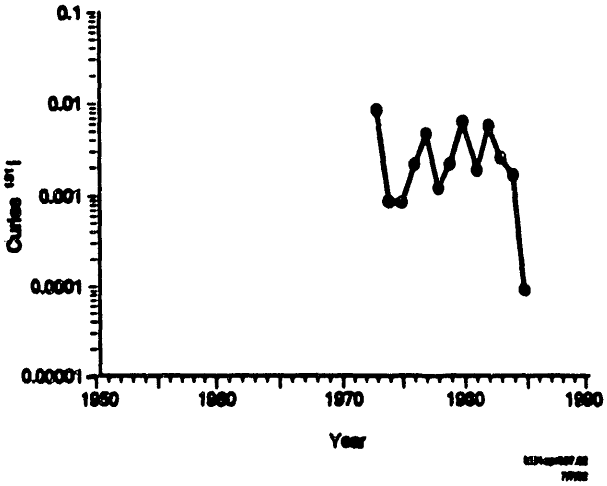


Figure 3-3. Measured Annual Atmospheric Releases of ^{131}I from C Reactor

K Reactor

K Reactor was operational from 1954 until the shutdown for extensive maintenance and safety upgrades, which began in 1988. Disassembly basin purges were discharged to Pen Branch and to two earthen basins–a seepage basin designated 904-65G and a retention basin designated 904-88G. Because seepage basin 904-65G had poor seepage characteristics, it received disassembly basin purges only in 1959, 1960, and 1965. Retention basin 904-88G received disassembly basin purges and occasional miscellaneous wastewater discharges from 1966 through 1989.

The total measured aqueous releases of ^{131}I from K Reactor through 1989 are 30.1 curies to Pen Branch and 21.6 curies to the earthen basins. Measured annual aqueous releases of ^{131}I are shown in Figure 3-4. The maximum annual release, 14 curies to the retention basin, occurred in 1969.

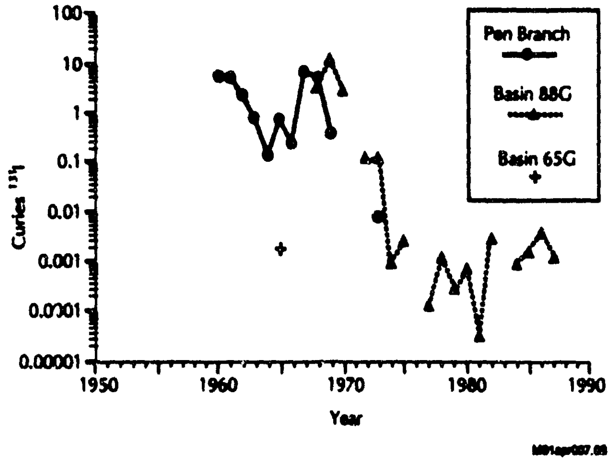


Figure 3-4. Measured Annual Aqueous Releases of ^{131}I from K Reactor

The total measured atmospheric release of ^{131}I from K Reactor through 1989 was 0.036 curies. Measured annual atmospheric releases are shown in Figure 3-5.

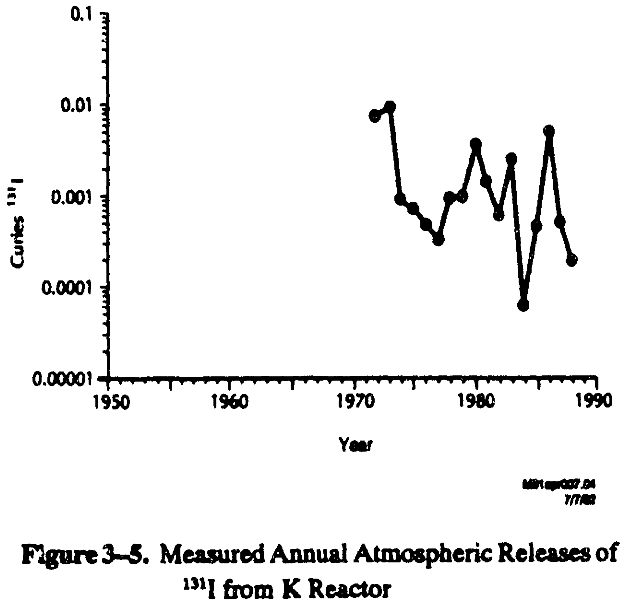


Figure 3-5. Measured Annual Atmospheric Releases of ^{131}I from K Reactor

L Reactor

L Reactor was operational from 1954 until 1968 when it was placed in standby. It was refurbished in the early 1980s, restarted in 1985, and operated until the 1988 shutdown. To provide thermal mitigation of secondary cooling water from the restarted reactor's heat exchanger, L Lake was constructed in the early 1980s by damming Steel Creek. Therefore, during L Reactor's second operational period, secondary cooling water from the reactor heat exchanger was discharged to L Lake which drains into Steel Creek.

Disassembly basin purges were discharged to Steel Creek and a seepage basin (904-64G). The seepage basin was used intermittently from 1958 to 1969 and 1985 through 1989. The L-Area Oil and Chemical Basin (904-83G) was used from 1961 to 1979 to receive wastewater containing various oils and chemicals from all of the reactor areas. It is adjacent to the 904-64G seepage basin.

The total measured aqueous releases of ^{131}I from L Reactor through 1989 are 34.7 curies to Steel Creek and 1.8 curies to the seepage basins. Measured annual aqueous releases of ^{131}I are shown in Figure 3-6. The greatest annual releases, approximately 8 curies to Steel Creek, occurred in 1961 and 1964. In its second operational period, aqueous ^{131}I was not discharged to Steel Creek.

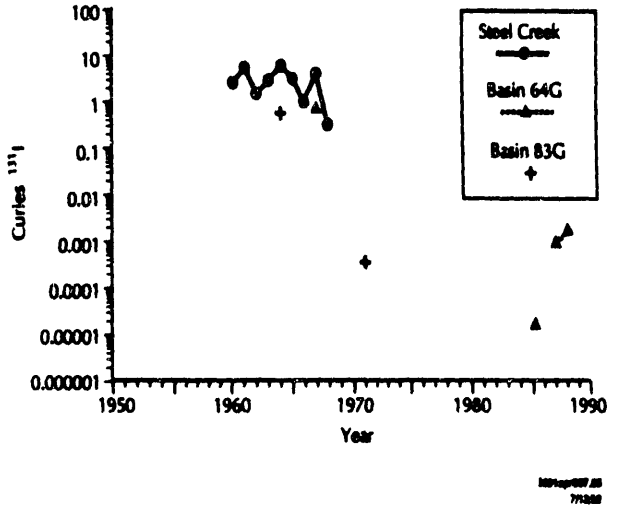


Figure 3-6. Measured Annual Aqueous Releases of ^{131}I from L Reactor

The total measured atmospheric release of ^{131}I from L Reactor through 1989 is 0.0007 curies. During L Reactor's first operational period, ^{131}I atmospheric releases were not quantified. Measured annual atmospheric releases of ^{131}I from L Reactor are shown in Figure 3-7.

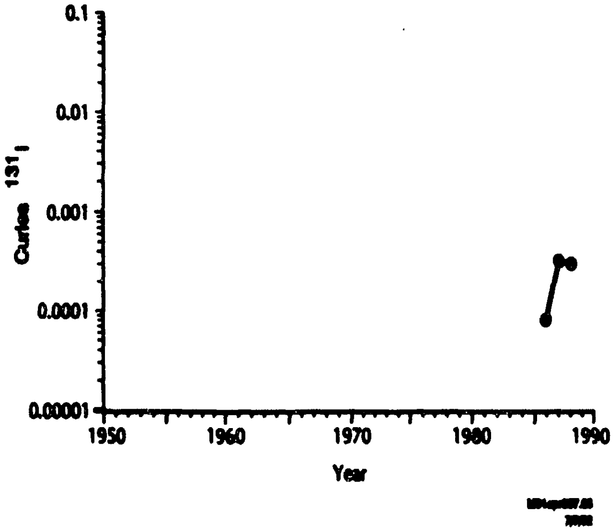


Figure 3–7. Measured Annual Atmospheric Releases of ^{131}I from L Reactor

P Reactor

P Reactor was operational from 1954 until the 1988 shutdown. Disassembly basin purges were intermittently discharged to Steel Creek and Par Pond (1954 to 1977) and to three seepage basins (1957 through 1989). The seepage basins have designations 904-61G, -62G, and -63G.

Reactor cooling water and miscellaneous effluents were discharged to Steel Creek until 1963; most of the cooling water was then diverted to Par Pond. When L Lake was constructed in the mid 1980s, all aqueous effluents from P Reactor were diverted to Par Pond.

The total measured aqueous releases of ^{131}I through 1989 are 159 curies to the Steel Creek/Par Pond system and 0.2 curies to the seepage basins. Measured annual aqueous releases of ^{131}I are shown in Figure 3-8. The greatest annual releases, approximately 70 curies to Steel Creek, occurred in 1957 and 1962 due to failed fuel elements. These were the greatest aqueous releases of ^{131}I from SRS to site streams.

Although aqueous ^{131}I was not routinely quantified prior to 1960, the value for 1957 probably represented special analyses in response to failures of newly designed fuel elements.

The total measured atmospheric release of ^{131}I from P Reactor is 0.055 curies through 1989. Measured annual atmospheric releases of ^{131}I from P Reactor are shown in Figure 3-9.

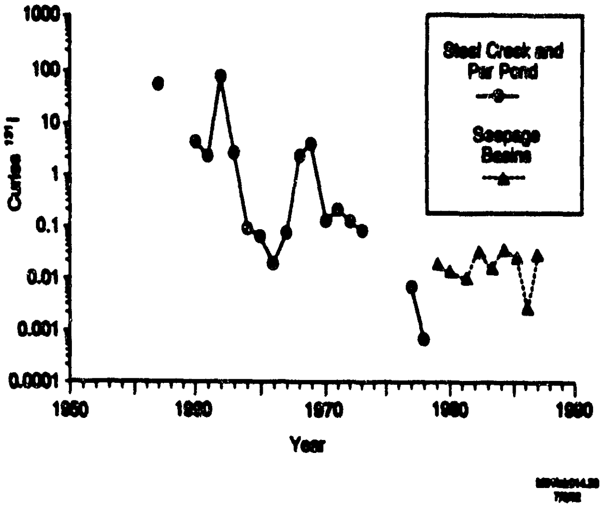


Figure 3–8. Measured Annual Aqueous Releases of ^{131}I from P Reactor

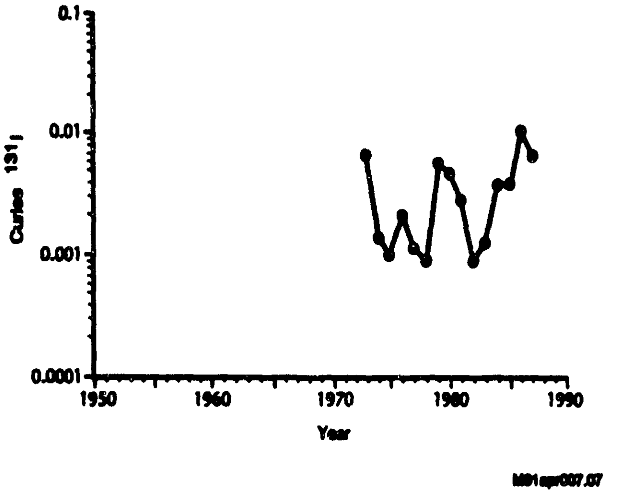


Figure 3–9. Measured Annual Atmospheric Releases of ^{131}I from P Reactor

R Reactor

R Reactor was the first operational production reactor at SRS. It operated from late 1953 until 1964 when it was placed on inactive status. Disassembly basin water was discharged into Lower Three Runs Creek from 1954 to 1958, into Par Pond from 1958 to 1964, and into a seepage basin system from 1957 to 1964. R Reactor had six seepage basins designated 904-103G, -104G, -57G, -58G, -59G, and -60G.

The original seepage basin, 904-103G, was used only from June to December 1957. Use of this basin was terminated due to surface outcropping and leakage into an abandoned sewer. The basin was backfilled in 1958. Basins 904-104G, -57G, -58G, and -59G were in use from 1957 to 1960. These four basins were deactivated, backfilled, and covered with asphalt in 1960. Basin 904-60G was in use from 1958 to 1964; it was backfilled in 1977.

Reactor secondary cooling water and miscellaneous effluents were discharged to Lower Three Runs Creek from 1954 to 1958. After 1958 these effluents were discharged to Par Pond, which was constructed by the damming of Lower Three Runs Creek. Beginning in 1961 discharges to a canal and pond system that eventually drained into Par Pond (Figure 3-1) replaced direct discharges into Par Pond.

The total measured aqueous release of ^{131}I from R Reactor during its operational lifetime is 56.2 curies (all of the ^{131}I was discharged to the Par Pond system). Measured annual aqueous releases of ^{131}I from R Reactor are shown in Figure 3-10. The maximum annual release of 35 curies occurred in 1963.

Atmospheric releases of ^{131}I were not quantified during the operational lifetime of R Reactor.

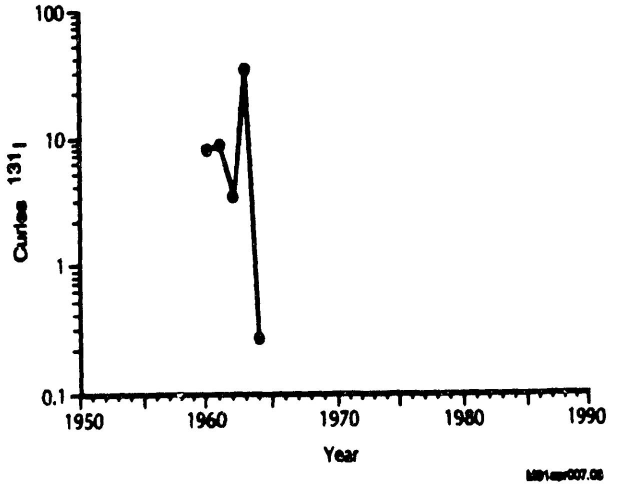


Figure 3–10. Measured Annual Aqueous Releases of ^{131}I from R Reactor into the Par Pond System

Summary of Reactor Releases

The total measured aqueous release of ^{131}I from reactor areas through 1989 is 327 curies. This includes disassembly basin releases of 302 curies to site streams and 24 curies to earthen basins. Minor aqueous releases (such as from heat exchangers, leaks, and sumps) contributed 1 curie to site streams. The total measured atmospheric release of ^{131}I is 0.13 curies.

The sums of the measured annual aqueous releases of ^{131}I from all reactors are shown in Figure 3-11. By the 1980s annual releases had decreased to approximately 0.1 percent of the annual releases in the early 1960s.

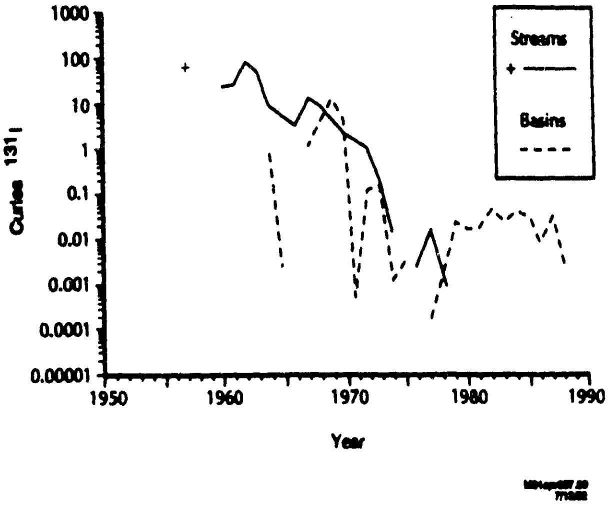


Figure 3–11. Combined Measured Annual Aqueous Releases from Reactors

Atmospheric releases of ^{131}I are shown in Figure 3-12. Although ^{131}I atmospheric releases were not quantified prior to 1972, those releases are estimated to have been small. This estimate is based on the lack of specific laboratory analyses for ^{131}I in stack air filters; such analyses would have been performed if excess beta-gamma activity were detected on the filters.

Atmospheric releases of ^{129}I were quantified beginning in 1988; ^{129}I releases are below detection limits. Prior releases when reactors were operational are expected to have been insignificant. This is because the activity of ^{129}I is orders of magnitude smaller that the activity of ^{131}I, and the atmospheric releases of ^{131}I have been small.

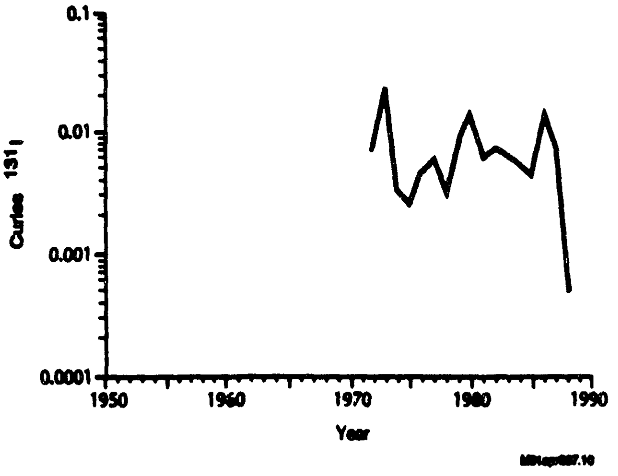


Figure 3–12. Combined Measured Annual Atmospheric Releases from Reactors

Separations and Liquid Waste Facilities

Two chemical separations facilities and their associated liquid waste storage facilities are located near the center of the site (Figure 3-1). The two separations areas are identified by letter designations F and H. In these facilities the products of interest in irradiated materials are chemically separated and purified from waste products; this procedure is called reprocessing. The greatest releases of radioiodine to the environment at SRS originated from these facilities.

Facility Operations

The two chemical separations facilities reprocess irradiated fuel and targets in buildings known as canyon buildings. Irradiated materials are dissolved and the products of interest are chemically separated and purified from waste fission and activation products.

Reprocessing began in late 1954 in F Area and mid 1955 in H Area. The Purex solvent extraction process is used to recover ^{239}Pu and ^{238}U from waste products in irradiated ^{238}U. During 1959-1960 the H-Area Purex process was modified to recover ^{235}U from the waste products in irradiated ^{235}U fuel using the HM (H Modified Purex) process. The principal chemical difference in the processes is that the HM process uses mercuric nitrate as a catalyst to enhance the dissolution of irradiated fuel.

Later modifications to both facilities allowed ^{237}Np to be recovered for use as target material. Irradiated ^{237}Np targets are dissolved and treated by the Frames ion exchange process in H Area to recover ^{238}Pu and ^{237}Np from the waste products. H Area also includes the RBOF storage facility.

Waste facilities in or adjacent to the separations facilities were designed for liquid waste handling. Depending on activity level, liquid wastes are stored in waste tanks or sent either to seepage basins or to the Effluent Treatment Facility (ETF) in H Area. In November 1988 the ETF became operational to treat the wastes that were previously sent to seepage basins. The seepage basins were then closed and, as of 1989, were being decommissioned.

Pathways for Release

Unlike the reactor facilities, which have released ^{131}I to the environment principally through aqueous effluents, the separations facilities have released ^{131}I almost equally through atmospheric and aqueous effluents. However, almost all aqueous ^{131}I releases from separations facilities were discharged to seepage basins and not directly to site streams.

During the chemical separations processes, radioiodine exists as a vapor, or is dissolved in aqueous and organic solvents. Emission controls minimize the atmospheric release of radioiodine volatilized during dissolution of irradiated materials. Volatilization of radioiodine also occurs in the separation and purification steps following dissolution. For example, the organic chemicals in these steps react with radioiodine to form volatile organic iodides. Releases then occur through the process vessel vent system, which exhausts to the atmosphere through a size-graded sand filter and a 61-m stack.

Volatile organic iodides that are expected to be present in atmospheric effluents include the alkyl iodides, such as methyl iodide or butyl iodide. Volatile inorganic forms may include I_2, ICl, IBr, HI, and HOI. Moreover, organic and inorganic forms of iodine may also be associated with particulate matter in the effluent (Brauer and Strebin, 1982).

Two studies examined the distribution of the organic and inorganic forms of iodine in air effluent from H Canyon. Eighty to 90 percent of the iodine was present in the organic form (SRL, 1973a; SRL, 1973b).

In a preliminary study of organic forms of iodine in atmospheric effluents from the SRS Purex process, butyl iodide and ethyl iodide were identified among numerous organic components that were detected; the other components were not identifiable (Smith and West, 1967).

Two later studies quantitatively measured the distribution of organic iodine compounds in air from H-Canyon. In one study essentially all of the organic iodine was present as methyl iodide (SRL, 1973a). The second study found that butyl iodide was 5 times more prevalent than methyl iodide (SRL, 1973b).

The presence of butyl iodide was attributed to radiation-induced reactions in the tributyl phosphate extractant of the Purex process. A butyl group formed in this manner can react with iodine to form butyl iodide. The different results obtained in these limited studies may indicate that the composition of iodine in stack air is dependent on which step of the separations and purification processes is being performed.

Aqueous waste streams contain residual radioiodine. High-level and low-level aqueous wastes are stored in tanks after their volume is reduced in an evaporator. The condensate from the evaporator contains radioiodine because some of the radioiodine distills. The condensate was sent to the seepage basins prior to 1988. The condensate is now sent to ETF. Another example of aqueous waste are occasional leaks in cooling water coils in process vessels.

The disposition of various process waters from the separations facilities depend on the results of analyses of the water for radioactivity. Water is concentrated in evaporators prior to storage, sent to seepage basins or to the ETF, temporarily retained in basins, or released to site streams.

The major sources of wastewater previously discharged to the seepage basins were the overheads from the nitric acid recovery unit and the process evaporators in both F and H Areas. In H Area, water from the Resin Regeneration Facility and RBOF was also discharged to the H-Area Seepage Basins. A flow sheet for liquid waste handling can be found in the reference Jacobsen et al., 1973. Because of the date of this reference, it does not show that ETF replaced the seepage basins.

Even after organic waste streams from the solvent extraction process receive a caustic wash, the solvent may contain residual traces of radioiodine due to the propensity of iodine to react with organic compounds. Spent organic solvent is sent to the SWDF for incineration. No studies of the radioiodine content of waste organic solvent have been performed.

Emission Control

Allowing sufficient time for ^{131}I to decay to low levels prior to reprocessing reduces ^{131}I releases. In the mid 1960s, the cooling time was lengthened from 100 to at least 200 days for most irradiated materials.

Administrative and physical controls, which were implemented in reactor areas after the 1961 acute release incident, minimize the possibility that very short-cooled fuel or targets might be inadvertently sent to the chemical separations areas.

The offgas from dissolution of irradiated materials is vented to the atmosphere through silver-nitrate-coated ceramic chips (beryl saddles), an acid recovery unit, and a 61-m stack. Volatilized iodine reacts with the silver nitrate to form highly insoluble silver iodide on the chips. This emission control system, known as a silver-nitrate reactor, has been in place since the startup of the chemical separations facilities. In the Frames process, offgases from the dissolution of ^{237}Np targets do not pass through the silver-nitrate reactor enroute to the stack. This is because fewer fissions occur in ^{237}Np than in uranium. The fewer fissions also allow ^{237}Np targets to be cooled for less time than uranium.

Some residual radioiodine that continues through the chemical separations process is retained in the precipitate of manganese dioxide that is used to clarify the solution of irradiated material in preparation for solvent extraction.

Some residual radioiodine that continues through the process is volatilized into the process vessel vent system.

Offgases from the process vessel vent system are exhausted through a sand filter and then through the 61-m stack. Although the sand filter is not expected to efficiently retain iodine, it is expected to retard the movement of some forms of iodine, which are successively being adsorbed and desorbed on the sand particles. The 61-m stack in each facility provides sufficient loft to the offgases to ensure that radioiodine is dissipated into very low concentrations over a large area. This minimizes exposure to people in and around the facility.

In the early years of operation, exhaust air was also passed through a caustic scrubber in F and H Areas to trap iodine. Erratic performance of the scrubber units contributed to the elevated ^{131}I releases observed in the initial years of operation.

The HM process has an inherent emission control feature. The mercuric nitrate catalyst used in the dissolution step reacts with iodine to produce a nonvolatile form of iodine.

Release Monitoring

Atmospheric releases of ^{131}I from separations facilities have been quantified since shortly after startup in 1954. Reprocessing of irradiated material began in F Canyon in November 1954. The first sampling period for ^{131}I in atmospheric effluents was December 14, 1954 to January 7, 1955. H Canyon began operations and ^{131}I release measurements in July 1955.

Continuous sampling of stack air for radioiodine is accomplished by passing a portion of the air through a filter that traps iodine. The air is extracted at the 59-m height and passes through approximately 90 meters of piping to the filter assembly at ground level. The filter is changed weekly and analyzed for radioiodine.

Limited studies of losses of iodine due to adsorption on the piping interior surfaces have been conducted. Adsorption is principally associated with elemental iodine. This appears to be a minor constituent of the chemical forms of iodine in stack air (SRL, 1973a; and SRL 1973b). Because adsorption is not well characterized for various operating conditions, no allowance for possible losses is made in quantification of the releases.

Three different techniques have been used to trap radioiodine. They represent the most suitable technologies that were available at the time. Initially, two different filters were used in parallel to trap radioiodine; one was a filter paper coated with silver nitrate, and the other was a caustic scrubber. In the early 1960s these were replaced by two TEDA-impregnated charcoal cartridges in series.

The change to charcoal filters was prompted by the release of 153 curies of ^{131}I from F Area over a 26-day period in 1961 (Jolly et al., 1968). This acute release incident was due to the inadvertent reprocessing of very short-cooled material. At the time of the release, the silver-nitrate filter did not provide an accurate measure of the severity of the release. The iodine collection efficiency of the silver-nitrate filter paper had dropped from 82 percent to 32 percent. This decrease was attributed to unusual concentrations of nitrogen oxides and other chemicals in the stack air (Marter, 1963). The release was quantified with the caustic scrubber system, which had a history of unreliable measurements because of erratic performance (Marter, 1963; Jolly et al., 1968).

The replacement charcoal filter system follows a filter paper that removes particulates. The first charcoal filter contains a radiation monitor that is set to alarm at an instantaneous release of approximately 0.3 curies. The daily change in cumulative activity is also monitored to detect unusually large daily releases that are less than the alarm point.

After a one-week exposure, the filters are analyzed for ^{131}I by gamma spectrometry and for ^{129}I by low energy photon spectrometry. Typically, the radioiodine is distributed greater than 97 percent in the charcoal and less than 3 percent in the particulate filter.

Since 1963 in F Area, dissolver offgases containing excess ammonia vapors have been diverted to an auxiliary 61-m stack after passing through the silver-nitrate reactor. Various methods have been used to quantify releases from the auxiliary stack; a molecular sieve filter proved to be the most effective. The filter is changed weekly and measured for ^{131}I; it is not routinely measured for ^{129}I.

Aqueous releases of ^{131}I have been quantified since 1955. Several techniques are used to monitor aqueous releases. Process cooling water in F Area is monitored in-line for beta-gamma activity. In H Area, batches of process cooling water are monitored prior to release. When liquid effluents were released to seepage basins, they were continuously sampled. Samples were collected weekly and analyzed by gamma spectrometry which detects ^{131}I but not ^{129}I. Effluent Treatment Facility discharges are sampled and analyzed by gamma spectrometry prior to release to Upper Three Runs Creek.

History of Measured Releases

The greatest aqueous and atmospheric releases are typically associated with the reprocessing of short-cooled fuel and targets, which occurred in the late 1950s and early 1960s. Short-cooled fuel and targets were reprocessed at that time because of the urgency to recover the special nuclear materials needed for nuclear weapons. In later years ^{131}I releases from separations areas were reduced as a result of a longer cooling period before reprocessing. Occasionally reprocessing of short-cooled materials continued into the 1970s.

At the time of the greatest ^{131}I releases, no measurements of short-lived isotopes such as ^{132}I and ^{133}I were reported. These isotopes would have accompanied any ^{131}I that was released if very short-cooled materials were reprocessed (Table 2-1, page 10).

Since the mid 1980s, atmospheric releases of ^{129}I have been quantified by measurement; prior releases were quantified by calculation.

F-Area Aqueous Releases

Aqueous wastes from F Canyon were initially sent to waste tanks and a seepage basin designated 904-49G, also known as Old F-Area Seepage Basin. The capacity of this basin proved to be inadequate. It was abandoned in 1955 after three additional seepage basins, (904-41G, -42G, and -43G) were constructed. These are known as F Seepage Basins 1, 2, and 3, and were used from 1955 until 1988 when waste waters were diverted to ETF; they are being dewatered and backfilled as of 1989.

The total measured aqueous release of ^{131}I to the F-Area seepage basin system through 1988 is 903 curies. Measured annual releases of ^{131}I are shown in Figure 3-13. The maximum annual release, 403 curies, occurred in 1956. This is attributed to the reprocessing of short-cooled materials. The increase during 1961 is associated with the acute release incident.

For ^{129}I, only estimates of releases to the seepage basins are available. The cumulative release is estimated at 2.0 curies for the F-Area seepage basins through 1985 (Killian et al., 1985a). This estimate is consistent with an inventory estimate of 0.7 curies through 1987, which is based on a few measurements of ^{129}I in basin water and basin sediments.

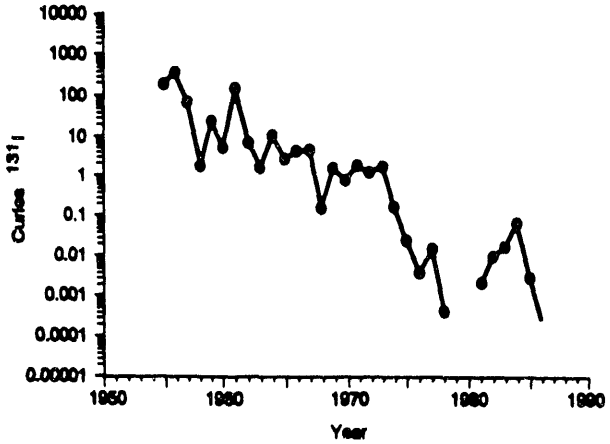


Figure 3–13. Measured Annual Aqueous Releases of ^{131}I to F–Area Seepage Basins

Various process waters that contain only low levels of radioactivity are discharged to Four Mile Creek. A limited study of the ^{129}I content of the effluent was conducted in 1979 and 1980. Concentrations did not exceed 0.06 pCi/L, which indicates the effluents are not a significant source of ^{129}I (Kantelo, 1987).

F-Area Atmospheric Releases

The total measured atmospheric release of ^{131}I from F Area through 1989 is 1,112 curies. Measured annual releases of ^{131}I are shown in Figure 3-14. The maximum annual release, 688 curies, occurred in 1956. During 1958 atmospheric releases of ^{131}I were below minimum detection limits because F Canyon was shut down for modifications from March 1957 until March 1959.

SRS's only acute atmospheric ^{131}I release incident due to the inadvertent reprocessing of very short-cooled material occurred in 1961 when 153 curies were released between May 29 and June 23 (Marter, 1963). About half of the ^{131}I was released in the first 5 days; the daily releases were 6, 24, 12, 18, and 14 curies. In comparison the average daily release during 1960 and 1961 (exclusive of this incident) was 0.02 curies. No measurements of ^{129}I were performed in response to this incident. Because the reprocessed material had only a very short cooling time, the quantity of ^{129}I released was insignificant relative to the quantity of ^{131}I released (Table 2-1, page 10).

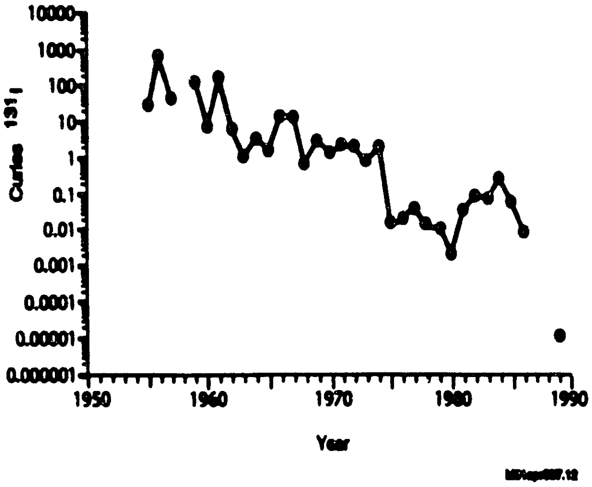


Figure 3–14. Measured Annual Atmospheric Releases of ^{131}I from F Area

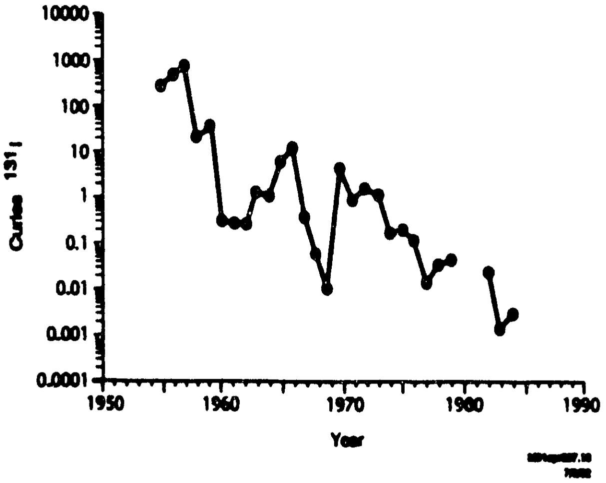


Figure 3–15. Measured Annual Aqueous Releases of ^{131}I to H–Area Seepage Basins

A malfunction of the silver-nitrate reactor was reported in the 1967 environmental report. A low collection efficiency apparently developed in the F-Area unit, which resulted in a short-term ^{131}I release at a maximum rate of 6 curies per day.

The history of annual atmospheric releases of ^{129}I from F Canyon will be discussed with H-Area atmospheric releases. Releases from the two areas have been historically published together as one combined annual release.

H-Area Aqueous Releases

Aqueous wastes from H Canyon were sent to waste tanks and four seepage basins (904-44G, -45G, -46G, and -56G). The basins, which are also known as H Seepage Basins 1, 2, 3, and 4, were used from 1955 until 1988 when waste waters were diverted to ETF. Basin 3 was not used after 1962. All basins are being dewatered and backfilled as of 1989.

The total measured aqueous release of ^{131}I to the H-Area seepage basin system through 1988 is 1,711 curies. Measured annual releases are shown in Figure 3-15. The maximum annual release, 798 curies, occurred in 1957. This is attributed to the reprocessing of short-cooled materials.

For ^{129}I only estimates of releases to the seepage basins are available. The cumulative release is estimated at 0.4 curies for the H-Area seepage basins through 1985 (Killian et al., 1985b). This estimate is consistent with an inventory estimate of 0.9 curies through 1987, which is based on a few measurements of ^{129}I in basin water and basin sediments.

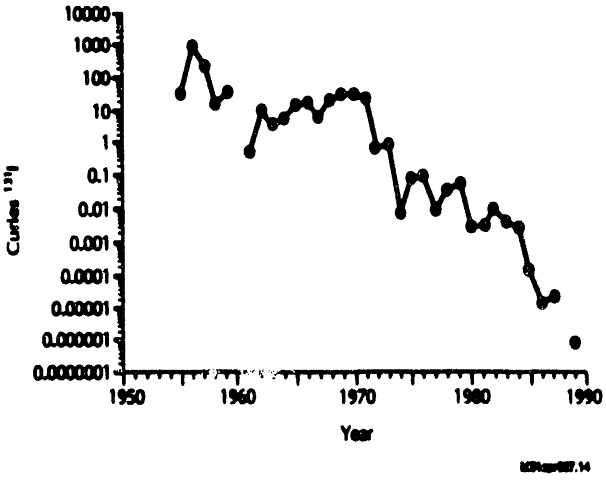


Figure 3–16. Measured Annual Atmospheric Releases of ^{131}I from H Area

H-Area Atmospheric Releases

The total measured atmospheric release of ^{131}I from H Canyon and RBOF to the atmosphere through 1989 is 1,399 curies. Essentially all of the ^{131}I originated in H Canyon. Measured annual releases are shown in Figure 3-16. The maximum annual release, 888 curies, occurred in 1956. During 1960 atmospheric releases of ^{131}I from H Area were below minimum detection limits because H Canyon was shut down from April 1959 until the end of 1960 for modifications to convert to the HM process.

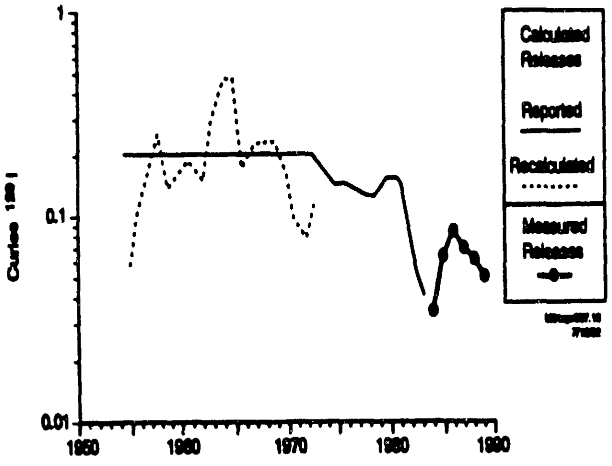


Figure 3–17. Combined Annual Atmospheric Releases of ^{129}I from Separations Areas

The annual atmospheric release of ^{129}I from the two separations areas is historically published as a combined release. The total combined ^{129}I atmospheric release through 1989 from F and H Areas is estimated to be 5.7 curies. The estimate is based on calculated and measured annual releases as shown in Figure 3-17.

Prior to 1984, annual ^{129}I releases were determined by calculation only. The calculation is based on the quantity, composition, and irradiation history of reprocessed materials; on the ^{129}I fission yield; and on the efficiency of iodine removal in the chemical separations processes. This efficiency is based on the experimentally measured distribution of ^{131}I in various steps of the separations processes.

For 1955 through 1973, two types of calculated annual releases, reported and recalculated, are plotted in Figure 3-17. The reported releases were determined by calculating atmospheric ^{129}I releases from the separations areas during 1955-1973 and prorating the total on an annual basis (Zeigler and Lawrimore, 1988). This method gives a constant value of 0.21 curies of ^{129}I released per year. Releases were recalculated to estimate the year-to-year variations in the combined releases for a study of ^{129}I deposition (Boone et al., 1985).

Measured releases have been reported in the site release documents since 1984. From 1981, when measurements were begun, through 1983, calculated releases agreed satisfactorily with the measured releases. For example, in 1981 the measured release of 0.139 curies is 13 percent different from the calculated release of 0.160 curies.

The maximum recalculated ^{129}I releases occurred in 1964 and 1965. These annual releases of 0.5 curies are 2.5 times greater than the average release for 1955 through 1973. The maximum releases reflect the quantity, composition, and irradiation history of reprocessed materials at that time. The decrease from approximately 0.15 curies in 1981 to 0.06 curies in 1982 reflects a change in irradiation conditions. The change required targets to be irradiated to only half of their previous exposure; correspondingly, the quantity of ^{129}I produced is approximately half of the previous quantities.

Effluent Treatment Facility

Aqueous discharges are analyzed for ^{131}I by gamma spectrometry. No ^{131}I has been detected. No analyses for ^{129}I have been performed as of 1989.

Stack air was tested for compliance with National Emission Standards for Hazardous Air Pollutants during startup testing in December 1988 and January 1989, when ETF was operating at about 30 percent of capacity. Stack air was continuously sampled for radioiodine using a particulate filter followed by two TEDA-impregnated charcoal filters in series. The filters were changed weekly and analyzed for ^{131}I by gamma spectrometry and for ^{129}I by low energy photon spectrometry. Although no ^{131}I was detected, low levels of ^{129}I were detected (Taylor, 1989). By scaling the ^{129}I release to an annual basis and to 100 percent of operating capacity, an annual release of approximately 0.0002 curies of ^{129}I is estimated. This is less than 1 percent of the combined atmospheric release of ^{129}I from F and H Canyons for 1989.

Waste Tank Farms

High level radioactive liquid waste storage tanks, located in both F and H Areas, contain ^{129}I. Approximately 40 percent of the SRS-produced ^{129}I as of 1974 is predicted to have been disposed of in the waste tanks (Cornman, 1974).

The waste in the tanks consists of sludge and supernates; iodine is expected to be in the supernate rather than the sludge. Limited analyses for the ^{129}I content of the sludge and supernate in certain waste tanks have been performed. Samples of decontaminated supernate from F-Area Tank 24 were analyzed in 1983. No ^{129}I was detectable at a detection limit of 450 pCi/L (Ryan, 1983).

Studies of supernate in H-Area waste tanks show ^{129}I concentrations on the order of 10^5 pCi/L for Tanks 11H, 12H, 15H, and 32H (Fowler and Cook, 1984). No ^{129}I was detectable in sludge samples from Tanks 9H, 10H, 13H and 15H at a detection limit of 5 pCi/g (Fowler and Cook, 1984). Based on these studies, Fowler and Cook estimated that the inventory of ^{129}I in SRS waste tanks, as of the early 1980s, was 9.1 curies.

Radioactivity leaked into the ground from certain waste tanks. In 1960, H-Area Tank 16 leaked about 100 liters of waste into the surrounding soil (Poe, 1974). In 1961 F-Area Tank 8 leaked about 6000 liters of waste onto the soil (Odum, 1976). The reported inventory of radionuclides for these tanks did not include ^{129}I; contaminated soil was not analyzed for ^{129}I.

Waste tanks have the potential to release radioiodine to the atmosphere. Because the vapor space in the waste tanks is flushed with air, volatile forms of radioiodine are released to the atmosphere. Vented air from the tanks is not monitored for radioiodine.

Summary of Separations and Liquid Waste Facilities

Total aqueous releases of ^{131}I to seepage basins from separations facilities are nearly equal to the total atmospheric releases. The total measured ^{131}I releases through 1989 are 2,614 and 2,511 curies to seepage basins and the atmosphere, respectively. For ^{129}I the corresponding estimated releases are 3 and 6 curies. (Nine curies of ^{129}I are estimated to be stored in waste tanks.) The distribution of ^{129}I between basins and the atmosphere differs from the distribution of ^{131}I. This is likely due to the considerable uncertainty in the estimate for the ^{129}I inventory in the basins.

The combined annual releases of radioiodine from separations areas to the atmosphere and the seepage basins are shown in Figure 3-18. This includes the annual atmospheric releases of ^{129}I using the recalculated values for 1955 through 1973. The long-term trend is that ^{131}I releases decreased significantly with time following the maximum releases in the early years of operation. Annual releases in the 1980s were approximately 0.01 percent of the releases in the late 1950s. In general since 1975, ^{131}I releases to the atmosphere have been less than ^{129}I releases.

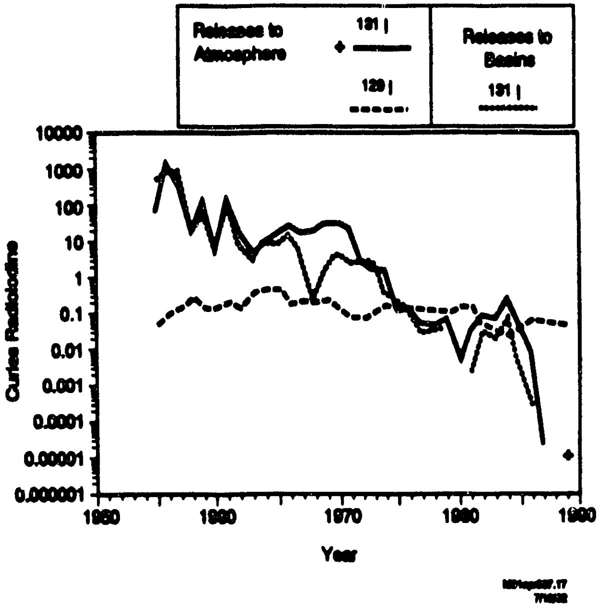


Figure 3–18. Combined Annual Releases of Radioiodine from Separations Areas

Solid Waste Disposal Facility

The SWDF is a site for burial or above-ground storage of radioactive solid wastes and certain liquid wastes. The first facility, designated 643-G, was used from 1952 to 1972. The new facility, designated 643-7G, is adjacent to the original and has been used since 1969.

The predominant radioiodine-containing material buried at SWDF is the spent ceramic chips coated with silver nitrate that were used for emission control during the dissolution of fuel and targets in F and H Areas. The buried inventory of ^{129}I, as of 1982, is estimated to be 14 curies (Hawkins, 1983). The practice of burying spent chips was terminated in the early 1980s. They are now stored at SWDF without burial.

Other solid waste buried at the SWDF is potentially contaminated with ^{129}I (for example, spent reactor deionizers). No analyses of these materials for ^{129}I content have been performed.

Liquid waste that potentially contains radioiodine has been stored at the SWDF. Degraded solvent from the separations areas was burned in open pans at the SWDF in the early years of operation. Since 1975 it has been stored in steel tanks at the SWDF pending combustion in an incinerator for radioactive waste. As of 1987, 655 of the 755 cubic meters of degraded solvent stored in the tanks since 1975 had been incinerated (Jagge et al., 1987). Given the propensity of iodine to react with organic compounds, the degraded solvent may contain ^{129}I, even though the solvent has been scrubbed with caustic. No ^{129}I analyses of the spent solvent have been performed.

Savannah River Laboratory

SRL provides research and development support for the production facilities of SRS. Laboratory facilities handle radioactivity ranging from high levels, which are manipulated in shielded cells, to environmental levels.

Certain liquid radioactive wastes are accumulated in holding tanks pending shipment to F Area for treatment and disposition. From 1954 to 1982, a seepage basin system was used for low activity wastes. The first two seepage basins (both designated 904-53G) were placed into service in 1954. The last two basins (904-54G, and -55G) were added in 1958 and 1960. No radioiodine is documented as having been released to the SRL seepage basins.

Experimental studies using high levels of radioactivity have been performed in shielded cells. Air vented from the shielded cells passes through a carbon bed to remove iodine, then through HEPA filters before exhausting through a 50-m stack. In the early years of operation a caustic scrubber was also used to purify air from the shielded cells. Before the air is discharged to the stack, it passes through a sand filter, which was installed in the early 1970s.

Air exhausted from other parts of the laboratory passes through HEPA filters and exhausts through 30-m stacks. If air radioactivity exceeds specific limits, the air is diverted to the sand filter before being exhausted through the stack. All stack air is monitored for radioiodine using TEDA-impregnated charcoal filters. The offgas exhaust from the SRL liquid storage facility is also monitored for ^{131}I using TEDA-impregnated charcoal filters.

The total measured atmospheric releases of ^{131}I from SRL through 1989 is 6.4 curies. Measured annual atmospheric releases of ^{131}I are shown in Figure 3-19. The maximum annual release, 2.9 curies, occurred in 1958 when experimental studies of short-cooled fuel elements were performed.

Figure 3-19 also shows that 0.010 curies of ^{133}I were released to the atmosphere from SRL in 1973. This was the only documented release of short-lived ^{133}I at SRS (Zeigler and Lawrimore, 1988; Cummins et al., 1991). The origin of the ^{133}I is undocumented. If it originated from a short-cooled target or fuel element, ^{132}I might also have been present in the release. The presence of ^{132}I depends on when the iodine was chemically separated from tellurium (Table 2-1, page 10).

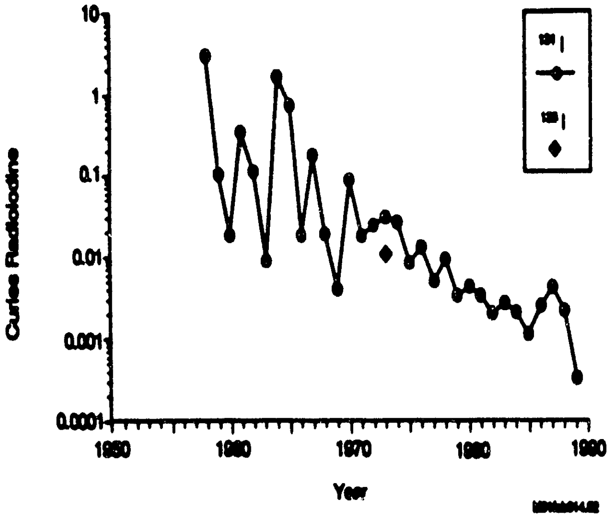


Figure 3–19. Measured Annual Atmospheric Releases of Radioiodine from SRL

Other Facilities

All documented radioiodine releases from SRS facilities are listed in the previous sections. A number of other facilities handle materials containing radioiodine. The facilities that have or had the potential to release radioiodine are the test reactors, the Heavy Water Rework Facility, and any facility handling spontaneously fissioning elements.

No releases of radioiodine are documented for any of the test reactors described in Chapter 2. Had there been releases from these facilities, the releases would be small compared to the releases from production reactors. Operation of the test reactors was discontinued by the 1970s.

The Heavy Water Rework Facility, located in D Area, consists of distillation towers and associated equipment for removal of ordinary water that accumulates in the heavy water moderator used in the reactors. Leaks in the system are vented through a 21-meter stack. The aqueous waste stream is sent to Beaver Dam Creek after verification that activity levels do not exceed specific limits. There is a potential for traces of radioiodine to be present in the moderator, as is described in the section on reactor facilities. No specific monitoring for radioiodine has been performed. The predominant radioactivity associated with this facility is tritium.

Facilities, other than those already described in this chapter, that have handled spontaneously-fissioning elements are the reactor material facilities in M Area, (where uranium fuel and targets are fabricated), the TNX Semi Works (where various processes are tested on a pilot scale with uranium), and the Naval Fuels facility (where uranium fuel for the US Navy was fabricated from the late 1980s until shutdown in 1989). The presence of low levels of radioiodine due to spontaneous fission has no impact on operations at those facilities because the radioiodine is contained within the uranium metal; radioiodine in this case is only of academic interest.

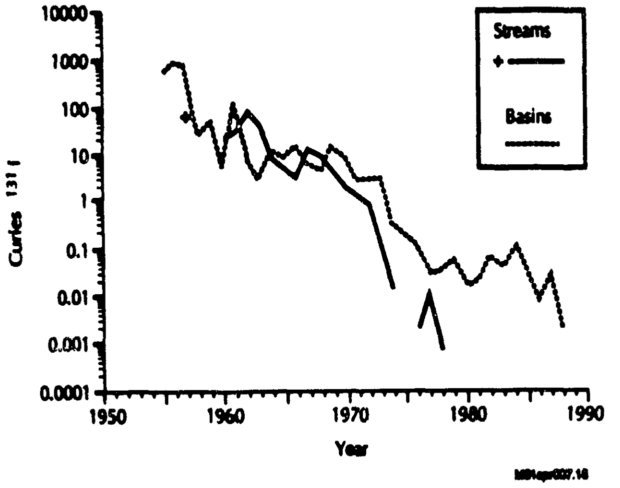


Figure 3–20. Combined Annual Aqueous Releases of ^{131}I from SRS

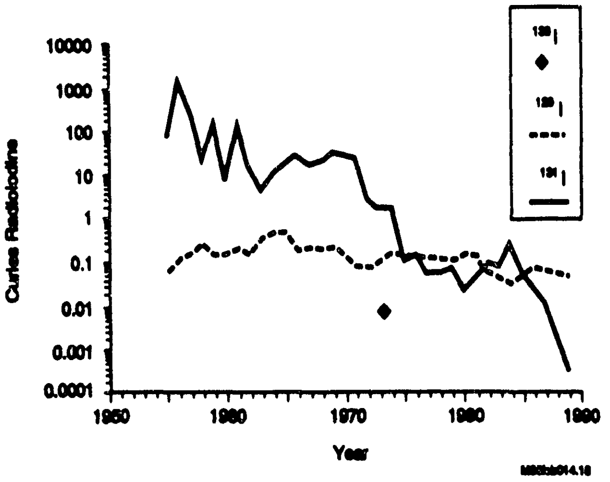


Figure 3–21. Combined Annual Atmospheric Releases of Radioiodine from SRS

Summary of Measured Releases from SRS

Measured annual releases of ^{131}I from SRS to the aqueous environment are summarized in Figure 3-20. Annual releases of radioiodine from SRS to the atmospheric environment are summarized in Figure 3-21. In general, since 1975, ^{129}I has been the predominant component of annual radioiodine releases to the atmosphere from the site. The annual aqueous and atmospheric releases of ^{131}I and ^{129}I are presented in tabular form in Chapter 5.

By the 1980s the annual ^{131}I releases had been reduced to approximately 0.01 percent of the releases of the late 1950s. However, reductions in releases began by the early 1960s. Consequently, releases from 1963-1989 account for only approximately 8 percent of the total atmospheric ^{131}I. Ninety-two percent was released during the 8-year period from1955-1962.

The total releases of ^{131}I to the aqueous and atmospheric environments through 1989 are summarized in Table 3-1. Releases from the separations areas account for 89 percent of the total aqueous releases and greater than 99 percent of the total atmospheric releases. The separations areas discharged ^{131}I almost equally to seepage basins and the atmosphere. Reactor areas discharged ^{131}I almost exclusively to the aqueous environment. Direct discharges from reactor areas to site streams account for 10 percent of the total aqueous release from the site; direct discharges to streams ceased in 1978.

The purpose of the seepage basins is to allow short-lived radionuclides to decay to low levels during the time required (years) for the aqueous migration to site streams. An unknown portion of the radioiodine in the seepage basins volatilized to the atmosphere.

The best estimate of the status of ^{129}I releases and inventories resulting from reprocessing through the mid-1980s is 3 curies to seepage basins, 6 curies to the atmosphere, 9 curies stored in waste tanks, and 14 curies buried in the SWDF. Table 3-2 compares this distribution with the estimated 1974 distribution (Cornman, 1974). Both estimation methods show that approximately 70 percent of ^{129}I is stored and 30 percent is released to the aqueous and atmospheric environment.

Table 3-1. Summary of radioiodine releases (curies) to the environment through 1989

SRS Area	Aqueous		Atmospheric		
	^{131}I	^{129}I	^{131}I	^{129}I	^{133}I
Separations					
to basins	2614	3	—	—	—
to atmosphere	—	—	2511	5.7	—
Reactors					
to basins	24	—	—	—	—
to streams	303	—	—	—	—
to atmosphere	—	—	0.13	—	—
SRL					
to atmosphere	—	—	6.4	—	0.01
Totals	2941	3	2518	5.7	0.01

Table 3-2. Comparison of the disposition of ^{129}I in 1974 and the mid-1980s

Disposition	Estimated Percent Distribution as of	
	mid-1980s	1974
SWDF	44	30
Waste Tanks	28	40
Atmosphere	19	20
Seepage Basins	9	10

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4

Releases from SRS and global fallout from atmospheric nuclear weapons tests have been the principal sources of radioiodine in the environment surrounding SRS. Beginning in 1954 SRS routinely analyzed environmental samples for ^{131}***I. Although routine analyses of*** ^{129}***I did not begin until the mid 1980s, short-term studies of its distribution in the environment have been conducted since the early 1970s.***

Results of routine analyses are taken from periodic SRS environmental reports (see Bibliography). Typically, annual average values are used in this report. Results for short-term studies are taken from other reports published at SRS or in the scientific literature. The routine analyses and short-term studies were designed to determine the fate of radioiodine as it moved through the environment by atmospheric and aqueous transport mechanisms.

Atmospheric Transport

As a plume containing radioiodine moves through the atmosphere, the concentration of radioiodine in the air generally decreases with distance from the source. This decrease is the result of dilution and deposition.

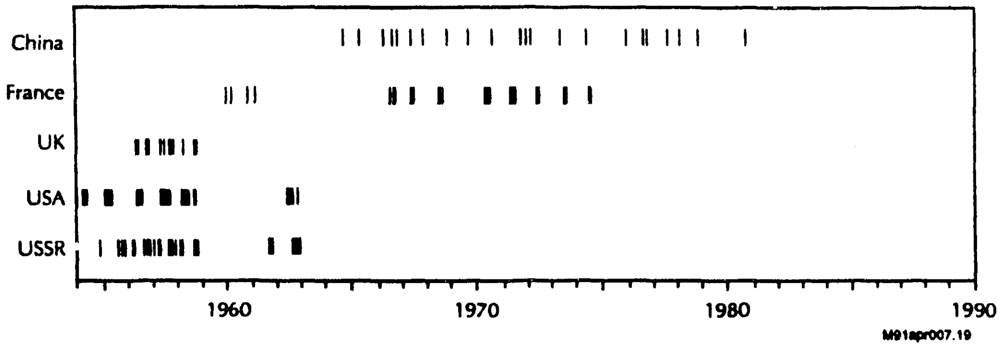

Figure 4–1. The Occurrence of Nuclear Weapons Tests in the Atmosphere from 1954 through 1989

Dilution occurs as the plume spreads throughout an ever increasing volume of air. The degree of dilution depends on such conditions as wind speed, turbulence, air buoyancy, and the height of the surface mixing layer. Under ideal conditions the decrease in concentration will be inversely proportional to distance from the release point. Over the course of a year, the inverse relationship applies to annual average concentrations along the circumference at given radial distances from the release point.

The inverse relationship is used in this report to distinguish SRS from global fallout as the principal source of radioiodine in the offsite local environment. This is determined by the slope of the straight line drawn through the annual average ^{131}I data plotted on logarithmic scales as a function of distance from the center of the separations areas at SRS. Distances are measured from this point, rather than the geographic center of the site, because atmospheric releases originate almost exclusively in the separations areas. If global fallout is the sole source of the radioiodine, the slope is 0. Local concentrations do not change with distance from SRS because the fallout originates thousands of kilometers from SRS. If SRS is the sole source, the slope is -1 under ideal dilution conditions. On logarithmic scales a slope of -1 means that concentration is halved for every doubling of distance.

When deposition is considered, the slope will be steeper (more negative) than the slope under ideal dilution conditions. Deposition consists of physical and chemical processes that remove radioiodine from the air and deposit it on the earth's surface. These include gravity settling of particulate forms of iodine, entrainment with rain, and physical and chemical interactions of iodine with vegetation, soil, and other features of the earth's surface.

The environment near SRS routinely contained detectable levels of ^{131}I from 1955 through 1962. Nuclear weapons testing in the atmosphere was intense during that time, and SRS releases of ^{131}I to the atmosphere were the greatest in its history. Ninety-two percent of the total atmospheric ^{131}I released from SRS and 72 percent of the total ^{131}I produced in atmospheric nuclear weapons tests occurred from 1955 through 1962. (Seventy-two percent of the total fission energy yield produced in the history of worldwide atmospheric tests occurred during 1955 through 1962 [UNSCEAR, 1982].)

Figure 4–1 shows the frequency of worldwide nuclear weapons tests in the atmosphere from 1954 through 1989. Earlier atmospheric tests are not included because the ^{131}I produced in those tests decayed to insignificant levels before measurements began at SRS. This figure is based on tabulations of nuclear weapons tests (Bennett, 1978; Carter and Moghissi, 1977; Perkins and Thomas, 1980).

As a result of the Limited Test Ban Treaty, the US, UK, and USSR ceased atmospheric testing at the end of 1962. Residual ^{131}I was not detectable in the environment after 1963. However, ^{131}I was intermittently reintroduced to the environment due to atmospheric testing of nuclear weapons by China and France—two nations that did not participate in the treaty. No nation has tested nuclear weapons in the atmosphere since 1980.

Fallout from all but three of the Chinese atmospheric tests was detected in the vicinity of SRS. Fallout from the French tests after 1962 was generally not detected near SRS because these tests were conducted in the South Pacific and there is little mixing of fallout from the southern hemisphere to the northern hemisphere.

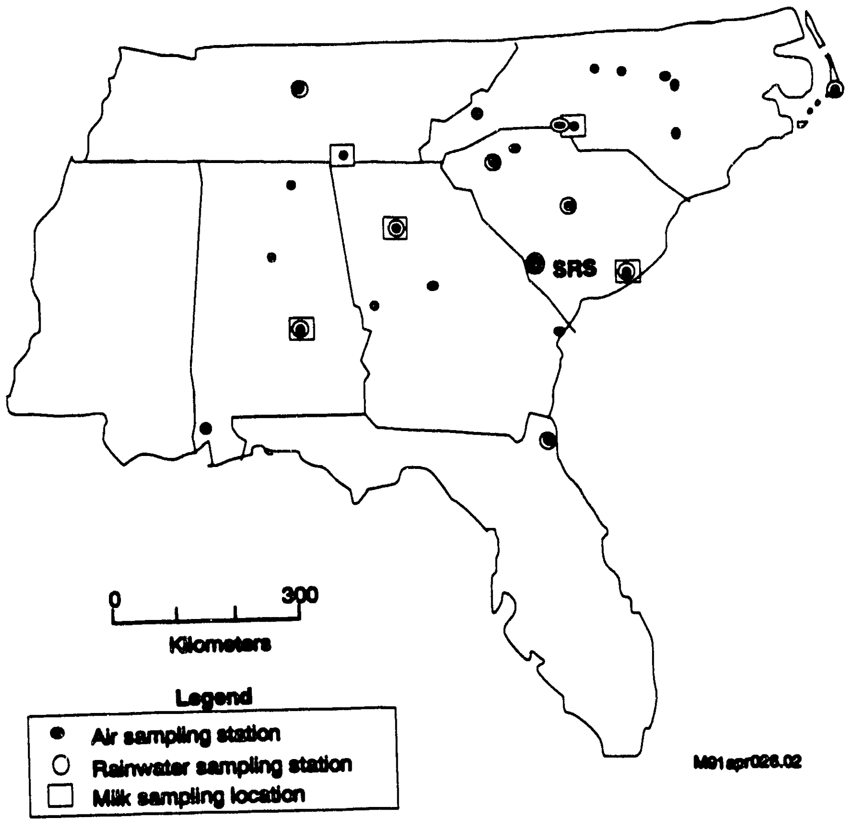

Figure 4-2. Location of PHS Network Sampling Stations in the Southeast

To assess the environmental impact of global fallout in the United States, the US Public Health Service (PHS) established several nationwide networks to collect and analyze environmental samples for radioactivity. Data from the PHS networks are compared in this chapter with the SRS ^{131}I measurements. Figure 4-2 shows the locations of the network sampling stations that are used for the comparison. Only stations located in the southeast are used and most are located in major population centers.

Air and rainwater stations belonged to the National Air Sampling Network and the Environmental Radiation Ambient Monitoring System (the latter was known as the Radiation Surveillance Network from inception until 1967 and then as the Radiation Alert Network until 1974). Milk locations belonged to the Raw Milk Network and the Pasteurized Milk Network. In 1971 the PHS networks that were still operating became the responsibility of the EPA.

Because of incomplete mixing of weapons test debris in the atmosphere, fallout levels often widely varied between nearby locations. Such variations in network data have been minimized by averaging the data from the large number of locations in Figure 4-2. This provides a comprehensive southeastern value for comparison with SRS measurements of ^{131}I in air, rain, vegetation, and milk in the local environment.

Air

Beginning in 1955 SRS sampled the air for volatile and particulate forms of ^{131}I at locations onsite, near the site perimeter, and near the 40-km radius from the geographic center of the site. (In SRS terminology, 40-km radius means the circumference at a radius of 40 km.) Four air sampling stations near the 160-km radius were added in 1962 to provide distant background locations for measurement of global fallout. The background stations happen to be located in cities that are part of the nationwide air sampling networks.

In most years ^{131}I has been detected at locations close to the separations areas. Near the perimeter and 40-km radius, ^{131}I was detected from 1955 through 1963. Near the 160-km radius, it was detected only in the first two years of operation—1962 and 1963.

No ^{131}I has been detected on an annual average basis since 1963 at any offsite location. The samples from the 40-km radius typically were not measured from 1964-1980 unless ^{131}I was detected at the site perimeter. No ^{131}I results were published in SRS environmental reports from 1974 to 1979 and 1982 to 1986.

Figure 4-3 illustrates the slope analysis technique to determine when SRS was the principal source of radioiodine in the local offsite atmospheric environment. Annual average ^{131}I concentrations at individual sampling stations for two representative years are plotted against distance. The plotted distance is measured from the center of the two chemical separations areas rather than the geographic center of the site because greater than 99 percent of the atmospheric releases of radioiodine originates from the chemical separations areas.

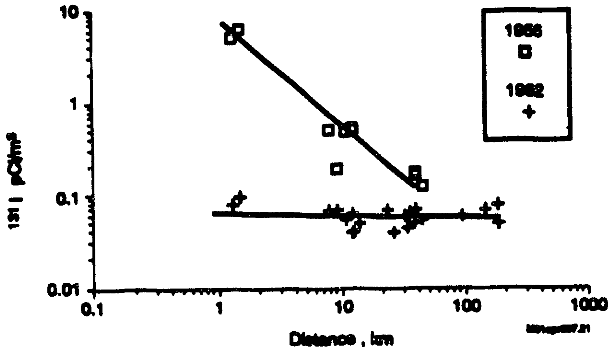

Figure 4-3. Variation in Annual Average Concentrations of ^{131}I in Air with Distance from the Center of the Chemical Separations Areas at SRS for Two Representative Years

The results for 1962 illustrate a year in which SRS releases had no influence on atmospheric ^{131}I at offsite locations. The slope of the regression line through the data is –0.06. Although concentrations decreased between locations near the separations facilities and site perimeter, they did not decrease between locations near the site perimeter and the 160-km radius. Therefore, before the SRS ^{131}I reached the site boundary, it was overwhelmed by global fallout ^{131}I.

The results for 1956 show that SRS was the principal source of atmospheric ^{131}I in the vicinity of SRS. The slope of the regression line through the data is approximately -1.1. The lack of a background component demonstrates that SRS ^{131}I overwhelmed global fallout ^{131}I within the 40–km radius.

Slope analysis is only a semi–quantitative technique. Although slopes near –1 and 0 can be attributed almost exclusively to SRS and global fallout, respectively, the relative contributions of both sources to intermediate slopes cannot be quantified because of the variable conditions in the environment. However, an intermediate slope of –0.2 or steeper (more negative) is adopted as the criterion for SRS being the principal source of ^{131}I in the local offsite environment. This is based on Figure 4–4, which illustrates hypothetical concentrations due to global fallout and SRS releases.

In Figure 4–4 the SRS component has the ideal slope of -1.0 and comprises 50 percent of the total concentration at the 11–km radius from the center of the separations facilities; the site perimeter ranges from approximately 11 to 20 km measured from the center of the chemical separations facilities. The slope of the total concentration curve between the 11–km radius and the 40–km radius is –0.37. The slope would be steeper if the SRS contribution were greater than 50 percent at 11km.

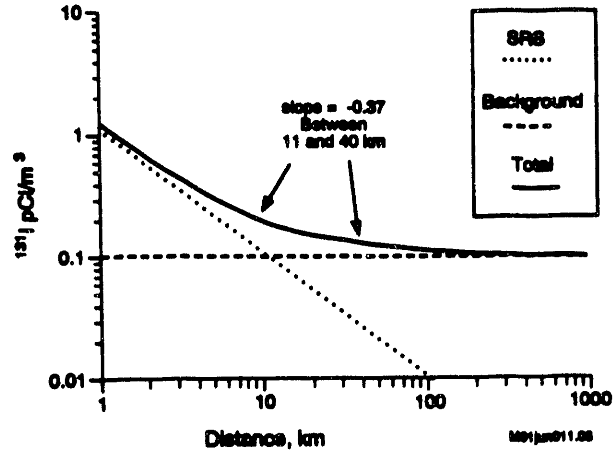

Figure 4–4. Example of SRS and Global Fallout Components of Hypothetical ^{131} I Concentrations beyond 1 km from the Separations Facilities

The criterion for SRS being the principal source is based on the slope when SRS contributes 50 percent at the nearest offsite location (11 km). This slope of –0.37 is conservatively reduced by a factor of two in order to account for less than ideal conditions and for the scatter in the ^{131}I measurement results (see Figure 4–3). Thus, for this report a slope of –0.2 or steeper distinguishes SRS from global fallout as the principal source of ^{131}I in the local environment.

Table 4-1 shows the slopes of the regression line for the years that ^{131}I was detected. SRS releases were the principal source in 1955-1959 and 1961—slopes ranged from -0.3 to -1.1. Similar negative slopes (-0.5 to -1.2) were observed for atmospheric tritium concentrations in the vicinity of SRS (Murphy et al., 1990).

Table 4-1. Slopes of annual average ^{131}I concentrations in air as a function of distance

Year	Slope
1955	-0.46
1956	-1.05
1957	-0.52
1958	-0.27
1959	-0.62
1960	-0.11
1961	-0.32
1962	-0.06
1963	0.00

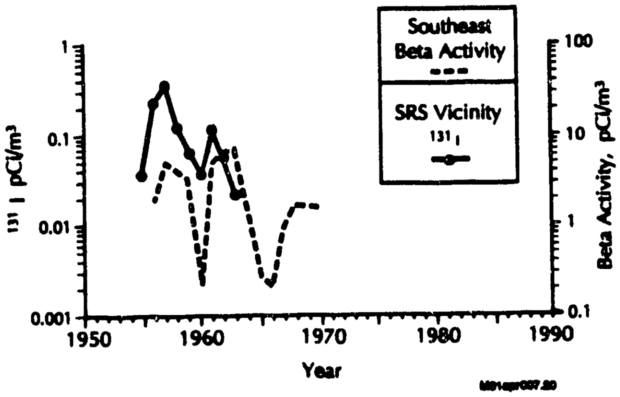

Figure 4-5. Annual Average Concentrations of ^{131}I in Air at the 20-km Radius from 1955 through 1989 Compared to Annual Average Concentrations of Particulate Beta Radioactivity in Southeastern Air from 1956 through 1970

The regression line is used to determine the annual average concentration of ^{131}I in air at the 20-km radius for comparison with the PHS network results. The 20-km radius from the center of the chemical separations areas represents all offsite concentrations that are close to the site perimeter. Annual average concentrations of ^{131}I are shown in Figure 4-5 for the years that ^{131}I was detected. The ^{131}I concentrations are compared to annual average concentrations of particulate beta radioactivity in southeastern air. Because the PHS networks did not measure ^{131}I, only the patterns in the PHS and SRS data can be compared.

In the absence of ^{131}I measurements in the PHS networks, the concentrations of beta activity in the particulate fraction of southeastern air serve as an indicator for the presence of elevated global fallout ^{131}I. This is because the arrival of fresh fallout from an atmospheric test causes beta activity in air to increase; fresh fallout contains short-lived beta-emitting radionuclides including ^{131}I.

Although PHS network measurements continued beyond 1970, only the data through 1970 are used for this comparison. The data were taken from the references: PHS, 1960a; PHS, 1962a; PHS, 1963a; PHS, 1964a; PHS, 1965a; PHS, 1966a; PHS, 1967a; PHS, 1970a; PHS, 1971; and EPA, 1971a.

The ^{131}I concentrations prior to 1960 were greater than expected by the beta activity concentration pattern. Qualitatively, the excess is attributed to SRS releases, which for those years was the principal source of ^{131}I according to the slope analysis. The slope analysis also shows that global fallout was the principal source of ^{131}I in 1960, 1962, and 1963. The similarity in the patterns of ^{131}I and beta concentrations after 1960 is consistent with that analysis.

In 1960 the ^{131}I concentration did not undergo the order-of-magnitude decrease observed for beta activity. This is partially due to the numerous monthly average concentrations that were below detection limits during July to December; these are not included in the annual average—only the positive measurements are averaged. This results in an average annual concentration that is conservatively greater than the actual annual concentration.

In 1963 the more rapid decrease in ^{131}I concentration than in beta activity may have been due to the short half-life of ^{131}I. Longer-lived beta activity persisted in the atmosphere well into 1963 following the intense testing in 1962 (Figure 4-1). Consequently, the annual beta activity concentration in 1963 is the greatest in the history of the southeastern PHS network stations despite the cessation of atmospheric tests in December 1962.

The lack of detectable ^{131}I after 1963 is consistent with the general absence of fresh fallout as evidenced by the decrease in beta activity concentrations. Although beta activity concentrations increased from 1967-1970, they did not approach the level that was attained in 1962-1963. Consequently, annual average ^{131}I concentrations in 1967-1970 remained below detection limits in the absence of SRS releases similar to those of the late 1950s.

Another source of radioiodine releases to the atmosphere is volatilization from the seepage basins. A limited study of ^{131}I volatilization from the seepage basins in the separations areas was conducted in the early years of site operations when aqueous releases were the greatest. The average atmospheric ^{131}I concentration approximately 8 meters downwind from F Seepage Basin 1 was 2,000 pCi/m^3 in 1956; it was 5,000 pCi/m^3 approximately eight meters downwind from H Seepage Basin 1. Dilution and deposition during atmospheric transport would have reduced the concentrations to approximately 0.01 pCi/m^3 at the 20–km radius. Figure 4–6 shows that the annual average concentration at 20 km in 1956 was greater than 0.1 pCi/m^3. Stack releases of ^{131}I from the separations areas overwhelmed the ^{131}I volatilized from the seepage basins.

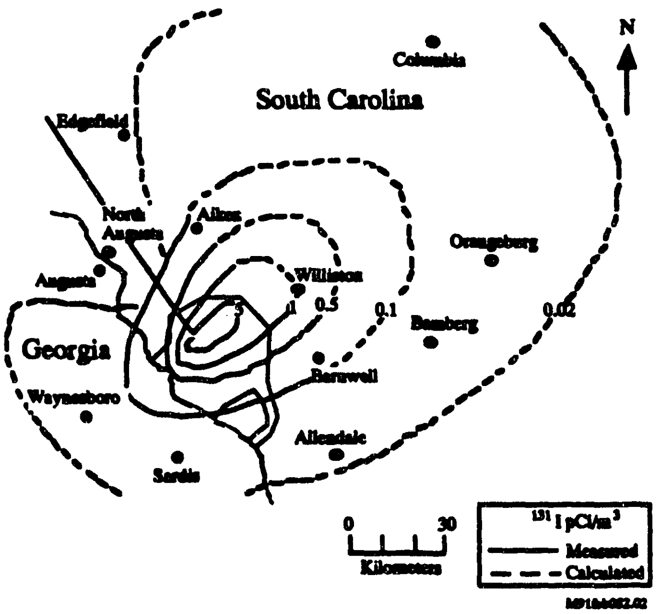

Figure 4–6. Measured and Calculated Concentrations of ^{131}I in Air (pCi/m^3) Near SRS for May 30 through June 7 During the 1961 Release Incident at SRS

The only acute ^{131}I release incident due to the inadvertent reprocessing of very short-cooled materials occurred during May and June 1961. The magnitude of this release clearly influenced the offsite ^{131}I concentrations as shown by the contours of measured and calculated concentrations in Figure 4–6 (Marter, 1963). Because calculated concentrations agreed favorably with measured concentrations, the calculated results were assumed to represent the actual concentrations at distances beyond the locations sampled. The maximum offsite concentrations of ^{131}I were in the range of 1 to 5 pCi/m^3, which did not exceed the 1961 SRS guideline of 100 pCi/m^3 for the population at large (Marter, 1963).

Similar elevated concentrations of ^{131}I in local air were observed following a Chinese atmospheric nuclear weapons test on December 28, 1966. The maximum concentration in individual local offsite air samples was 2.6 pCi/m^3 when the fallout arrived in early 1967.

Elevated ^{131}I concentrations in local air were also observed following the Chernobyl accident in 1986. The maximum ^{131}I concentration detected was 1.1 pCi/m^3. Volatile forms comprised an average of 60 percent of the total particulate and volatile ^{131}I measured. The maximum particulate concentration of 0.4 pCi/m^3 is consistent with the maximum observed in the southeast in the EPA network. The maximum of 0.6 pCi/m^3 occurred in Jacksonville, FL (EPA, 1986a). (In response to the Chernobyl accident, EPA analyzed particulate air samples and precipitation samples by gamma spectrometry in addition to the beta activity measurement.)

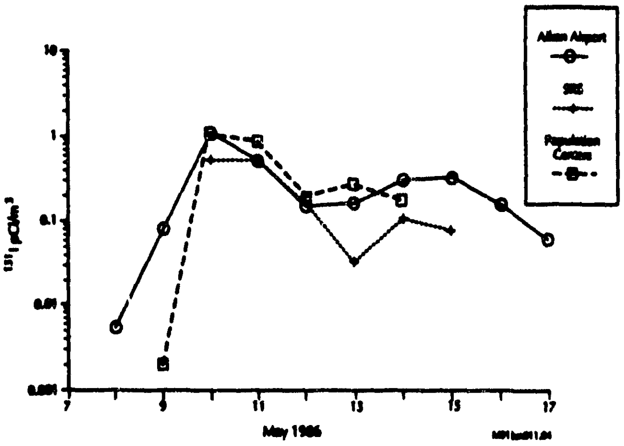

Figure 4-7. Measured Air Concentrations of ^{131}I Near SRS Following the 1986 Chernobyl Reactor Accident

Measured local concentrations at SRS, the Aiken Airport, and several nearby population centers are shown in Figure 4–7. The TRAC mobile laboratory (Appendix B) monitored air at selected population centers within the 160-km radius of SRS (Sigg, 1986; Sigg, 1989). The TRAC was at the following locations on the following dates in 1986:

- May 9—Madison, GA
- May 10—between Barnwell, SC, and Interstate-95
- May 11—Savannah, GA
- May 12—Augusta, GA
- May 13—Columbia, SC
- May 14—Greenville, SC

At SRS and Aiken Airport, air was continuously sampled with fixed air samplers. The Administration area (next to SRL in Figure 3-1, page 17) was sampled at SRS. The Aiken Airport is approximately 40 km north of the geographic center of SRS.

After peaking on May 10, total ^{131}I concentrations slowly decreased with occasional increases such as the one on May 13-14. No ^{131}I was detectable at the Aiken Airport after June 25 (Kantelo and Winn, 1986). Longer–lived radionuclides in the Chernobyl debris were detectable at Aiken Airport until the end of August 1986 (Winn and Kantelo, 1989).

Measurement uncertainties (one standard deviation) in the total ^{131}I concentrations were typically less than 10 percent. Results for May 11 and 12 at the three sampling locations are in good agreement. The spread in the concentrations for the other days reflects several factors. During rapidly changing concentrations such as on May 9, the concentration determined from the short sampling interval on board TRAC is expected to differ from the average concentration for a 24-hour sampling interval at the Aiken Airport. Also, concentrations at geographically separate locations are expected to differ. Such geographic differences, which are due in part to inhomogeneous mixing of the debris in the atmosphere, were observed in global fallout during the 1950s and 1960s (Bruner, 1963).

Measurements of ^{129}I in air are sparse. Atmospheric concentrations of ^{129}I near SRS were only determined in 1975. High sensitivity neutron activation analysis was used to determine the low environmental levels of ^{129}I. Concentrations of approximately 6 x 10^{-5} pCi/m^3 were detected from the site perimeter to the 160-km radius (Hochel, 1976). (Background ^{129}I air concentrations in North America are on the order of 10^{-6} pCi/m^3 [Brauer, 1974].) The absence of a decrease with distance from SRS may have been due to a bias in performing ^{129}I analyses at very low concentrations (Hochel, 1976).

Deposition in Rain

Entrainment in rainwater is one of the mechanisms for removal of radioiodine from air. Rainwater was one of the first types of environmental samples specifically analyzed by SRS for ^{131}I. These analyses began in mid 1954. Rainwater is collected at locations onsite, near the site perimeter, and near the 40-km radius from the geographic center of the site. Four rainwater stations near the 160-km radius were added in 1978. Two of the distant stations—Columbia and Greenville, SC—are located in cities that are part of the nationwide rain sampling networks.

Measurement results from each sampling station have been published in two different formats in the SRS environmental reports. The initial results were reported as annual average concentrations (activity per unit volume). The results after 1961 have been reported as annual average deposition (activity per unit surface area); these are determined by multiplying a measured concentration by the corresponding volume of water collected and dividing by the area of the collecting surface.

For this report all the rainwater results are expressed in units of deposition (pCi/m^2). The initial data that were reported as annual average concentrations in the environmental reports have been converted to deposition using the average SRS rainfall for each corresponding year.

Locations close to the separations areas have shown detectable ^{131}I in most years. Near the perimeter and 40-km radius, ^{131}I was detected from 1955 through 1963. After 1963 it was detected at the offsite locations only occasionally.

Table 4-2 shows the results of slope analysis of the rainwater data. SRS was the principal source of ^{131}I in local rainwater in 1955-1957, 1959, 1961, 1976, and 1977. Slopes ranged from -0.3 to -0.9. This is consistent with the slope of -0.8 observed for tritium in rain near SRS (Murphy et al., 1990).

Table 4-2. Slopes of annual average ^{131}I concentrations in rainwater as a function of distance

Year	Slope
1955	-0.88
1956	-0.65
1957	-0.26
1958	0.01
1959	-0.35
1960	-0.09
1961	-0.60
1962	-0.02
1963	0.01
1966	0.14
1967	-0.17
1970	-0.05
1976	-0.57
1977	-0.31

Table 4–2 does not include 1972, 1978, and 1986 when deposition was below 1,000 pCi/m^2. Values below 1,000

pCi/m^2 are near the detection limits and the slope analysis is not accurate with such statistically poor data.

Annual average deposition values for ^{131}I in rainwater at the 20-km radius from the separations areas are shown in Figure 4-8. These are compared to the annual average deposition values for beta radioactivity in southeastern rainwater and the annual average concentrations of particulate beta radioactivity in southeastern air.

Although PHS rain measurements have continued each year, the only network data used beyond 1970 corresponds to the years that ^{131}I was detected in the local environment. Beta activity deposition values are taken from the references: PHS, 1960b; PHS, 1961; PHS, 1962b; PHS, 1963b; PHS, 1964b; PHS, 1965b; PHS, 1966b; PHS, 1967b; PHS, 1968a; PHS, 1969a; PHS, 1970a; PHS, 1970b; EPA, 1971a; EPA, 1972a; EPA, 1973a; EPA, 1976; EPA, 1977a; EPA, 1978a; and EPA, 1986b.

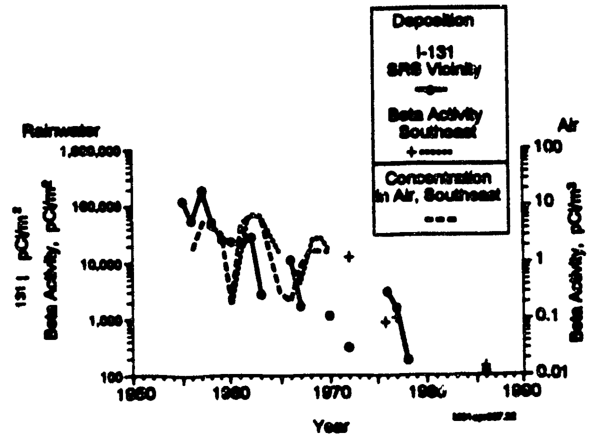

Figure 4-8. Annual Average Deposition of ^{131}I in Rain at the 20-km Radius from 1955 through 1989 Compared to Annual Average Deposition of Beta Radioactivity in Southeastern Rain (1960 through 1970 and Intermittently to 1986) and Annual Average Concentrations of Particulate Beta Radioactivity in Southeastern Air (1956 through 1970)

In the absence of ^{131}I measurements in the PHS rain network, the deposition of beta activity in southeastern rain serves as an indicator for the presence of rain that contains fallout ^{131}I. Because the patterns of beta activities in rain and air agree, the pattern of air concentrations is considered to represent the pattern of rain deposition for the years prior to the beginning of network rain analyses in 1960.

Figure 4-8 shows only marginal agreement in the pattern of local ^{131}I deposition and southeastern beta activity deposition from 1956 through 1970. One reason for the disagreement is that rainfall is often a local condition; comparison to the average rain in the southeastern region is not always valid.

The increases in local and southeastern deposition after 1970 are principally due to fallout from Chinese nuclear weapons tests.

Fallout ^{131}I from the Chernobyl accident was detected in 1986. The local maximum ^{131}I concentration was 100 pCi/L (not converted to deposition). This occurred on May 14 and was not coincident with the maximum ^{131}I air concentrations of May 10 and 11; rain did not fall on those days. The local maximum concentration is consistent with the maximum concentration in the EPA network in the southeast. This is 110 pCi/L in Nashville, TN (EPA, 1986a).

The 1954 measurements of ^{131}I in rainwater are not included in the previous discussion of annual deposition because the analyses began in mid 1954. Concentrations of ^{131}I in rainwater during July-December 1954 ranged from 20-50 pCi/L. The concentrations show no consistent change with distance. Therefore, the ^{131}I is attributed to global fallout.

Terrestrial Deposition

Radioiodine is deposited from the air onto the terrestrial environment by wet and dry deposition. Short-lived ^{131}I has not been detected in gamma spectrometric analysis of periodic soil samples. Preliminary studies of ^{129}I concentrations in surface soil near SRS were conducted in 1971 and 1975. Concentrations in the 1971 study were 6 x 10^{-4}, 5 x 10^{-4}, and 2 x 10^{-5} pCi/g at Jackson, SC (13 km from the center of the separations areas), Augusta, GA (28 km), and Madison, GA (100 km), respectively (Brauer, 1974). (The North American background of ^{129}I in soil is on the order of 10^{-4} pCi/g [Brauer, 1974].) The slope of the concentration change with distance is approximately -2.

In the 1975 study, the slope is approximately -1.3 for ^{129}I concentrations of 0.021, 0.005, and 0.0010 pCi/g at the site perimeter and the 40- and 160-km radii, respectively (Hochel, 1976). These concentrations are one to two orders of magnitude greater than the 1971 results at corresponding distances. SRS did not release sufficient ^{129}I between 1971 and 1975 to account for such large discrepancies in surface concentrations. They are attributed to differences in the number and depth of surface samples at each location, the length of time the soil had been undisturbed, forested compared to unforested sampling locations, and the soil retention characteristics for iodine.

The preliminary studies led to a comprehensive study of the cumulative ^{129}I deposition in the terrestrial environment surrounding SRS. During 1978-1979 vegetation and litter from the surface and soil from 60-cm deep cores were collected at distances of approximately 20, 40, 80, and 170 kilometers from the center of the chemical separations facilities along radial lines in five compass directions (Anderson, 1978; Kantelo et al., 1982).

Most sampling locations were in forested areas. All locations had been undisturbed for at least 25 years. Multiple cores and vegetation/litter samples were collected at each location. A composite sample of the vegetation/litter was prepared for each location. Each core was sectioned into 2.5-cm segments at depths of 0, 15, 30, and 60 cm and a composite sample for each specified depth was prepared for each location. The composite samples were analyzed for ^{129}I using neutron activation analysis.

Concentrations of ^{129}I in the 2.5-cm surface layer are intermediate between the values in the two preliminary studies at corresponding distances. However, the purpose of the comprehensive study was to determine the cumulative deposition rather than surface concentrations alone.

Cumulative deposition was determined by integrating the vertical soil deposition profile and adding the deposition in the vegetation/litter layer, which typically comprised less than 10 percent of the total. This small degree of iodine retention in the vegetation/litter might have been caused by biological degradation, which did not allow the fallen litter layer to retain iodine for many years. The cumulative deposition of ^{129}I (pCi/m^2) at each of the four sampling locations along each radial sampling line is shown in Figure 4-9.

The slope of the regression line through the data for all compass directions is -1.5, which demonstrates that SRS was the source of the ^{129}I. A slope of –1.5 means that deposition decreases by a factor of 2.8 for every doubling of distance. The slope is slightly steeper than the -1.2 slope observed for tritium in soil in the vicinity of SRS (Murphy et al., 1990).

Two data points were not used to fit the regression line. These are at distances of 19 and 80 km in the south-southeast (SSE) direction and represent the only sampling locations in open fields. Figure 4-9 shows that deposition in the forested locations was three to four times greater than the deposition in open fields. Although this conclusion is based on very limited data, it is consistent with similar observations of ^{129}I deposition near a former nuclear fuel reprocessing plant in West Valley, New York (Brauer and Strebin, 1982). Similar behavior is observed for zinc, lead, and cadmium near a smelting plant (Little and Martin, 1972).

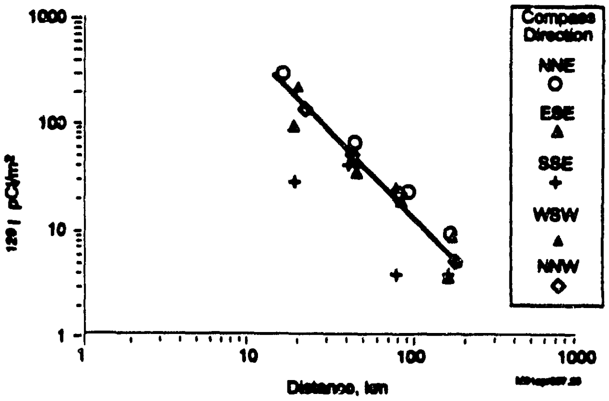

Figure 4-9. Variation in the Cumulative Terrestrial Deposition of ^{129}I with Distance from the Center of the Chemical Separations Facilities , as of 1978–1979

Deposition in forested areas is greater than in unforested areas because the forest acts as a filter for iodine. The turbulence created as air passes through leaves, pine needles, branches, and vegetation provides greater opportunity for removal of iodine by physical and chemical mechanisms in forests than in open areas. Iodine-containing leaves, pine needles, bark, and vegetation eventually become part of the litter layer from which iodine is leached into the soil.

Contours of the terrestrial deposition of ^{129}I in the vicinity of SRS were obtained by combining the results for forested locations shown in Figure 4-9 with SRS wind rose data for 1976-1978. The contours shown in Figure 4-10 converge on SRS and clearly demonstrate the concept that strongly negative slopes in all compass directions indicate atmospheric transport from a release point. At a distance of 200 km, the extrapolated deposition is still an order of magnitude greater than the expected background, which is on the order of 0.1 pCi/m^2 in North America (calculated from Rickard et al., 1974, assuming a 7-cm half-depth).

By the time of this terrestrial distribution study, SRS had released a cumulative total of 4.7 curies of ^{129}I to the atmosphere. Based on the results in Figure 4-10, approximately 45 percent (2.1 curies) was present in the terrestrial area ranging in radial distance from 7 to 200 km (Kantelo et al., 1982). The 2.1 curie inventory is an overestimate because it is based entirely on the forested data; open areas

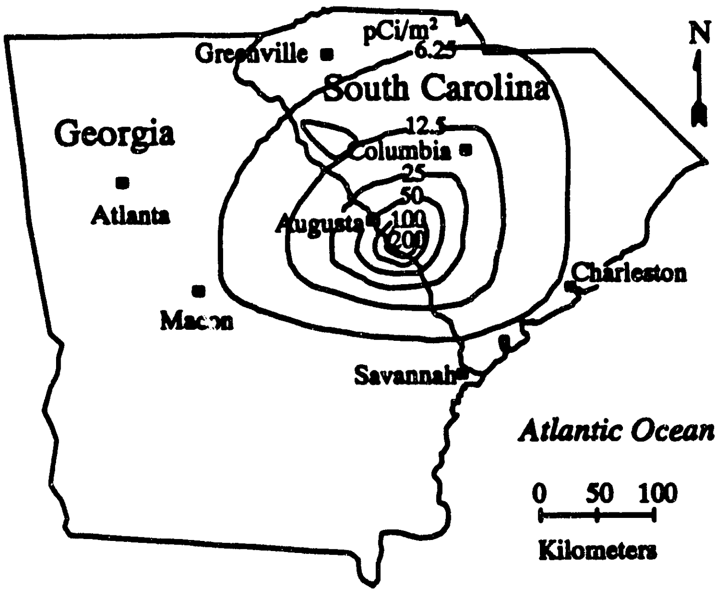

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Figure 4-10. Contours of the Cumulative Terrestrial Deposition of ^{129}I (pCi/m^2) Surrounding SRS, as of 1978–1979

Table 4-3. Half-depths for downward transport of deposited ^{129}I through southeastern soil

Direction	SSE				ESE				NNE				NNW				WSW			
Location Number*	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
Half-depth (cm)	7.0	6.9	9.9	9.9	7.4	8.8	5.6	6.2	8.2	9.8	3.8	6.7	7.9	5.3	3.9	4.5	6.7	5.0	4.8	8.9

*Sampling locations are numbered along each compass direction by increasing distance from center of reprocessing areas. Typical distances are 20, 43, 84, and 166 km.

were not taken into account. Thus, the actual inventory is less than calculated.

Despite the limitations of the inventory method, the 2.1-curie inventory compares favorably with a predicted inventory of 2.4 to 3.8 curies based on deposition from the 4.7-curie cumulative release. Calculations predict that 50 to 80 percent (depending on deposition velocity) of the ^{129}I should be deposited in the 7- to 200-km range (Murphy, 1991a; Murphy, 1991b). The calculations do not consider iodine removal processes such as volatilization or rainwater runoff. These result in an actual inventory smaller than predicted. This comparison demonstrates that the calculational method for ^{129}I atmospheric releases (Chapter 3) likely did not underestimate the actual releases.

The vertical soil profile at each location shows an exponential decrease in deposition with depth. Half-depths (analogous to half-life in radioactive decay) range from 4 to 10 cm with a mean of 7 cm, as shown in Table 4-3. The variations in half-depth do not correlate with any measured soil characteristics, which consisted of pH, cation exchange capacity, and the content of clay, sand, silt, and organic matter.

The small values of half-depth demonstrate that airborne ^{129}I deposited in the terrestrial environment is efficiently retained near the soil surface. The half-depth values were included in a database to develop a mathematical model for the movement of deposited ^{129}I in southeastern soils. Modelling results predict a 30-year mean effective residence time in the top 30 cm (Boone et al., 1985).

Deposited ^{129}I has also been observed to be efficiently retained near the surface in the vicinity of other nuclear fuel reprocessing facilities that experience different climatic and soil conditions. The facilities studied were Nuclear Fuels Services in New York (Rickard et al., 1974; Brauer and Strebin, 1982), Hanford in Washington (Rickard et al., 1974; Brauer and Strebin, 1982), the Karlsruhe Reprocessing Plant in Germany (Schuttelkopf and Pimpl, 1982; Robens et. al., 1989), the Sellafield Works in England (Stewart and Wilkins, 1985), and the Power Reactor and Nuclear Fuel Development Corporation in Japan (Muramatsu and Ohmomo, 1986).

Vegetation

Radioiodine principally accumulates in vegetation by wet and dry deposition. The accumulation is generally proportional to the exposed surface area of vegetation. Large or dense foliage is expected to accumulate the greatest concentrations of airborne radioiodine. Accumulation depends on the growing season. Uptake through the root system is less significant for ^{131}I than ^{129}I due to half–life considerations.

SRS began periodic sampling of vegetation for ^{131}I at various locations between the center of the site and the perimeter, near the perimeter, and near the 40-km radius in 1955. Sampling near the 160-km radius was added in 1974. Although the type of vegetation collected is not always specified in the environmental reports, Bermuda grass is collected when available. This is the principal forage crop for dairy cattle in the SRS vicinity.

In most years ^{131}I was detected at locations close to the separations areas. Near the perimeter and 40-km radius, ^{131}I was detected in the eight years from 1955 through 1962. After 1962 it has been sporadically detected at offsite locations.

The slope analysis technique cannot be rigorously applied to the results because annual average concentrations of ^{131}I in vegetation are reported for general radial distances from the geographic center of the site rather than for each specific sampling location. However, the general radial distances provide an approximation for the slope analysis.

Table 4-4 shows the results of such slope analysis. SRS was the principal source of ^{131}I in local vegetation during 1956, 1959, 1961, and 1967. Estimated slopes range from -0.2 to -0.7. A slope of -1.0 was observed for tritium in vegetation near SRS (Murphy et al., 1990). Slopes for 1959, 1960, 1962, and 1967 were determined only from the January-June results; July-December results in those years were below detection limits. Slopes for 1958 and 1970 could not be determined by this technique. In 1958 no 40-km samples were collected. In 1970 ^{131}I was detected only near the 40-km radius.

Table 4-4. Approximate slopes of annual average ^{131}I concentrations in vegetation as a function of distance

Year	Slope
1955	0.1
1956	-0.7
1957	0.0
1959	-0.2
1960	-0.1
1961	-0.3
1962	0.0
1967	-0.2

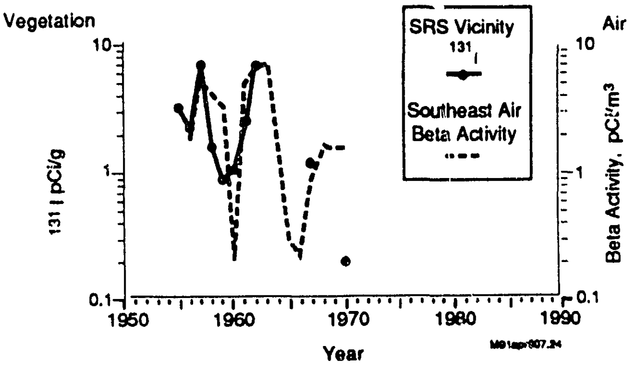


Figure 4-11. Annual Average Concentrations of ^{131}I in Vegetation at the 20-km Radius from 1955 through 1989 Compared to Concentrations of Particulate Beta Radioactivity in Southeastern Air from 1956 through 1970

The annual average concentrations of ^{131}I in vegetation at the 20–km radius from the center of the separation areas are shown in Figure 4–11. The year–to–year variations in the data are compared to particulate beta radioactivity in southeastern air; radioactivity in vegetation was not measured by the nationwide networks.

The pattern of local ^{131}I vegetation concentrations closely follows the pattern of beta activity concentrations in southeastern air. The 1960 plotted local concentration is conservatively high in that it represents only the January–June semi–annual average; the July–December value was below the detection limit.

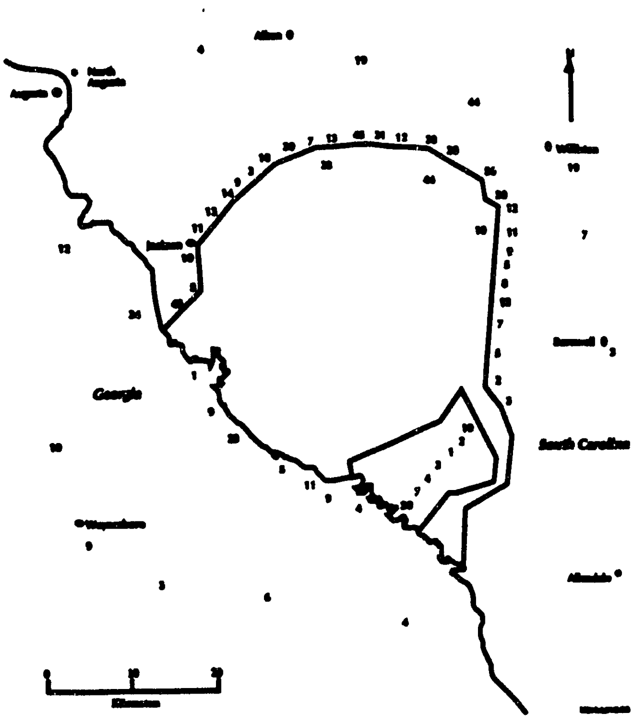


Figure 4-12. Measured Concentrations of ^{131}I in Bermuda Grass (pCi/g) Near SRS on June 5 Following the 1961 Release Incident

Following the 1961 acute release of ^{131}I from F-Area separations facilities in late May and early June, Bermuda grass was collected onsite and offsite. Results for the samples collected on June 5, are shown in Figure 4-12 (Marter, 1963).

The pattern of ^{131}I concentrations in vegetation is consistent with the pattern of air concentrations for the incident. The activity of ^{131}I on Bermuda grass decreased with an apparent half-life of 5 days compared to its radiological half-life of 8 days. The difference is attributed to dilution by new vegetative growth (Marter, 1963). The maximum offsite concentration was 56 pCi/g near the northeast perimeter of the site. For comparison the maximum offsite local ^{131}I concentration due to intermittent global fallout after 1963 is 9 pCi/g; this occurred in 1976 and 1977.

The maximum concentration of ^{131}I within the SRS boundary at the height of the 1961 release incident was 1,110 pCi/g. Even greater concentrations were observed later in June near the F-Area seepage basins after they received aqueous wastes from the incident. The maximum concentration of ^{131}I in vegetation was 485,000 pCi/g due to volatilization of ^{131}I from the adjacent seepage basins. The mechanism of volatilization from the seepage basins in the separations areas had been demonstrated in 1956; fifty meters from H Seepage Basin 1 the annual average concentration of ^{131}I in vegetation was 4,600 pCi/g and decreased with distance from the basins.

Measurements of ^{129}I in vegetation are sparse. Preliminary studies of ^{129}I in vegetation near SRS were conducted in 1971 and 1975. Concentrations in the 1971 study were 2×10^{-4}, 3×10^{-5}, and 4×10^{-6} pCi/g at Jackson, SC, Augusta, GA, and Madison, GA, respectively (Brauer, 1974). (The North American background of ^{129}I in vegetation is on the order of 10^{-6} pCi/g [Brauer, 1974].) The slope of the concentration change with distance is approximately -2.3, which is consistent with the slope in the corresponding study of surface soil.

In the 1975 study, ^{129}I concentrations of 9×10^{-3}, 4×10^{-3}, and 5×10^{-3} pCi/g were observed at the site perimeter and the 40- and 160-km radii, respectively (Hochel, 1976). The slope between the perimeter and the 40-km radius is -0.9. This is consistent with the slope in the corresponding study of surface soil. The inconsistent concentration at the 160-km radius reflects the difficulty of analyzing for low levels of ^{129}I.

Concentrations in the 1975 study are one to two orders of magnitude greater than the 1971 results at corresponding distances. As in the preliminary soil studies, the disagreement between concentrations in 1971 and 1975 is attributed to sampling differences. The type of vegetation, its location in forested or unforested areas, and the time of year affect the ^{129}I concentrations.

Food

Radioiodine principally accumulates in food by wet and dry deposition. SRS began periodic analysis of local food, which consisted of food crops and animal products, in 1957. Milk and eggs are the only foods in which ^{131}I has been detected.

Milk was recognized during the 1950s as a sensitive indicator for ^{131}I exposure through the grass-to-cow-to-human pathway. Consequently, SRS began local milk analyses in 1957. Also in 1957, the PHS began the Raw Milk Network, which operated through 1966. The PHS began the Pasteurized Milk Network in 1960.

SRS principally collected local milk samples at individual farms, dairies, and major milk distributors between the 20- and 40-km radius. The number of locations sampled in any given year is insufficient to perform a statistically valid slope analysis. However, given the influence of SRS radioiodine releases on other environmental media, local milk concentrations of ^{131}I are probably influenced by SRS releases in the years that SRS is the principal source of ^{131}I in the local environment.

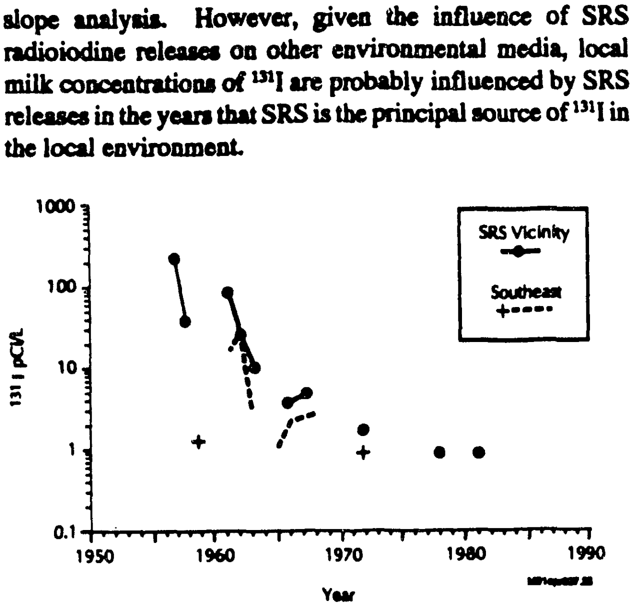


Figure 4-13. Comparison of Annual Average Concentrations of ^{131}I in Milk Near SRS (1957–1989) and in the Southeast (1958–1970 and Intermittently to 1981)

The annual average concentrations of ^{131}I in all types of local milk are shown in Figure 4-13. These are compared to the annual average concentrations of ^{131}I in southeastern milk. The concentrations are taken from the references Campbell and Murthy, 1963; Robinson, 1968; PHS, 1962c; PHS, 1963c; PHS, 1964c; PHS, 1965c; PHS, 1966c; PHS, 1967c, PHS, 1968b; PHS, 1969b; PHS, 1970c; EPA, 1971b; EPA, 1972b; EPA, 1973b; EPA, 1978b; EPA, 1979a; and EPA, 1981.

The Raw Milk Network began in mid 1957 with five locations, none of which were in the southeast. The first annual result from a southeastern location in the network was in 1959 from Atlanta, Georgia, 230 km northwest of SRS. The plotted southeastern milk concentrations during 1961-1966, when the Raw Milk Network and Pasteurized Milk Network concurrently operated, are the average of the two networks. Annual average Pasteurized Milk Network concentrations are plotted from 1967 through 1970. The only network data used beyond 1970 corresponds years that ^{131}I was detected in local milk.

In general, local and southeastern ^{131}I concentrations are in good agreement considering differences in sampling frequency. The local concentrations are expected to be greater than the southeastern concentrations because the SRS milk program often sampled more extensively after arrival of fresh global fallout than the networks sampled.

The sporadic appearance of ^{131}I in local milk after 1963 generally occurred when global fallout intermittently reappeared. The local concentrations of 0.9 pCi/L in 1978 and 1981 were close to the detection limit, which generally was 1 pCi/L. In 1978 elevated ^{131}I was detected at only one of the five PHS southeastern stations; the annual average concentration in Montgomery, AL, was 1.4 pCi/L.

The reason for the appearance of ^{131}I in local milk in 1981 is unknown. Residual ^{131}I from the last atmospheric nuclear weapons test in October 1980 would not have survived into 1981. Also, SRS releases of ^{131}I in 1981 were not excessively large; they were consistent with releases since 1975. Results for the southeastern stations in the Pasteurized Milk Network did not exceed detection limits in 1981.

Local concentrations prior to 1959 are compared to concentrations in network cities more distant than Atlanta. The closest cities for comparison are Cincinnati, OH, 670 km north northwest of SRS, and St. Louis, MO, 960 km northwest of SRS. Because the cities are so distant, their results serve only as a guideline for the environmental levels existing in the eastern US at that time.

In Cincinnati and St. Louis, the average ^{131}I concentrations for August–December 1957 were 190 and 410 pCi/L, respectively. The local average ^{131}I concentration of 500 pCi/L for July–December 1957 is greater than, but still consistent with these values. The average concentrations in Cincinnati and St. Louis, for January–June 1958 were 5 and 17 pCi/L respectively; the local value was 14 pCi/L.

The first result from Atlanta was in July 1958. The average ^{131}I concentration for July-December 1958 in Atlanta was 26 pCi/L. The corresponding local average concentration was 60 pCi/L. As in 1957, the local and regional concentrations were consistent, but the local value was greater. In general this relationship of local and southeastern ^{131}I concentrations continued for the annual results in subsequent years.

Following the 1961 acute release of ^{131}I from F-Area separations facilities in late May and early June, the greatest concentration of ^{131}I in milk was 5,451 pCi/L at a farm several kilometers north of the site perimeter. Results from the June 5-7 collections are shown in Figure 4-14 (Marter, 1963). The pattern of ^{131}I concentrations in milk is consistent with the air pattern.

The only other perishable foods of dietary significance that were available following the release incident were eggs and peaches. Fresh eggs, collected from the farm having the greatest concentration of ^{131}I in milk, contained an average of 33 pCi of ^{131}I per egg. No ^{131}I was detected in peaches

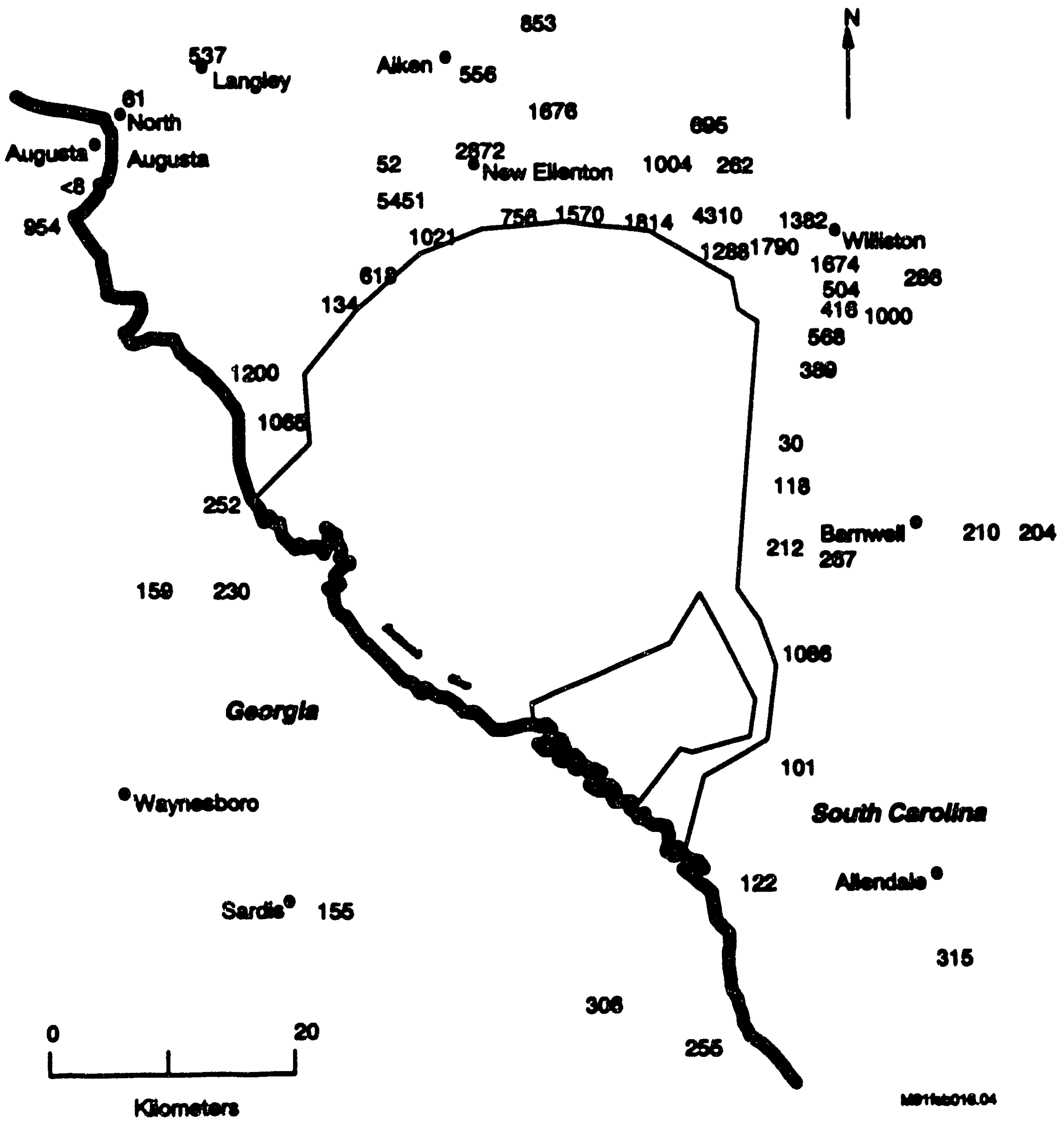


Figure 4-14. Measured Concentrations of ^{131}I in Milk (pCi/L) Near SRS for June 5–7 During the 1961 Release Incident at SRS

collected from an area in the path of the release plume (Marter, 1963).

Fallout ^{131}I from the Chernobyl accident was detected in one local milk sample. The ^{131}I concentration was 11 pCi/L on May 29, 1986, which was almost three weeks after the arrival of the fallout. Similar concentrations and delay times after arrival were observed in the Pastuerized Milk Network. Concentrations of 14–26 pCi/L were observed in Atlanta, Charlotte, and Charleston 3-4 weeks after arrival of the Chernobyl fallout (EPA, 1986a).

In the early 1960s, a significant difference was observed in ^{131}I concentrations in farm and dairy milk near SRS. Concentrations of ^{131}I in farm milk were approximately a factor of 3 greater than concentrations in dairy milk. This is attributed to the difference in feeding habits. Farm cows drink surface water and feed almost exclusively on pasture grass, both of which are exposed to fresh fallout. In contrast, the diet of dairy cows includes harvested fodder and nonfodder supplements in which the ^{131}I content is not replenished by new fallout. Therefore, dairy cows ingest less ^{131}I than farm cows ingest. This conclusion is supported by fallout studies of ^{90}Sr in farm and dairy milk conducted in Minnesota hundreds of kilometers from nuclear production facilities (PHS, 1962).

During the periods of intermittent global fallout after 1963, the maximum ^{131}I concentrations in local dairy and distributor milk were 240 and 330 pCi/L, respectively. These occurred in early 1967 after the Chinese test on December 28, 1966. The maximum in the southeastern Pasteurized Milk Network in early 1967 was 114 pCi/L (PHS, 1967d). In milk farm cows, the maximum local concentration of 778 pCi/L was observed after the Chinese test of September 26, 1976; this concentration is not consistent with the maximum concentration of 21 pCi/L in the southeastern Pasteurized Milk Network following this test (EPA, 1977b).

A difference between farm and dairy milk was also observed in the only study of ^{129}I concentrations in milk near SRS. In general, milk from individual cows contained greater concentrations of ^{129}I than milk from dairies (Hochel, 1976). The average ^{129}I concentration in dairy milk from within the 40-km radius is 0.06 pCi/L. The maximum concentration observed is 9 pCi/L from an individual cow near the site perimeter. One reference states that the North American background concentration of ^{129}I in milk is 0.006 pCi/L (Brauer, 1974).

Network measurements indicate a smaller North American background value. During 1976-1978 ^{129}I was measured in the Pasteurized Milk Network at two locations in the southeast—Montgomery, AL, and Charleston, SC. All but one of the ^{129}I concentrations are less than detection limits of approximately 0.001 pCi/L (EPA, 1977c; and EPA, 1979b). The concentration in the May-June 1976 sample from Charleston was 0.0056 pCi/L.

Summary of Atmospheric Transport

SRS releases and global fallout have been the principal sources of ^{131}I in various local environmental media. The years when SRS was judged to have been the principal source of ^{131}I in the environment near SRS are shown in Table 4-5. The criterion for this judgment is a slope more negative (steeper) than -0.2 in the logarithmic plot of ^{131}I data as a function of distance from SRS. Milk is not included in Table 4-5 because of an insufficient number of sampling locations to perform slope analysis.

Table 4-5. Years when SRS was the principal source of ^{131}I in various local offsite environmental media

Year	Vegetation	Rain	Air
1955		X	X
1956	X	X	X
1957		X	X
1958			X
1959	X	X	X
1961	X	X	X
1967	X		
1976		X	
1977		X	

Table 4-5 shows that SRS releases had a significant impact on all three environmental media only in 1956, 1959, and 1961. SRS releases had an impact on only one or two of the media in 1955, 1957, 1958, 1967, 1976, and 1977. Such behavior is expected because atmospheric releases of ^{131}I from SRS are not continuous; releases depend on specific operations underway in the separations facilities. Therefore, the detection of SRS ^{131}I in vegetation depends on the timing of the periodic sample collections relative to recent SRS releases. Similarly, the detection of SRS ^{131}I in rain depends on the coincidence of rainfall with SRS releases.

The slope analysis shows that air, soil, and vegetation have slopes steeper than -1.0 in some years. A slope steeper than -1.0 indicates the effect of the iodine removal processes. These include gravity settling of particulate forms of radioiodine and physical and chemical interactions with vegetation, soil, and other features of the earth's surface. Entrainment in rain apparently is not a major removal process on an annual basis; the slope for rain deposition does not exceed -0.9.

Maximum concentrations rather than annual average concentrations are used to assess the short–term impact of the acute release of ^{131}I from SRS in 1961. The maximum offsite concentrations in air, vegetation, and milk due to this incident range from a factor of 2 to 7 greater than the corresponding concentrations due to global fallout that occurred after 1963.

Low concentrations of ^{129}I have been detected in air, vegetation, soil, and milk near SRS. The cumulative deposition pattern of ^{129}I in the local terrestrial environment as of 1979 shows that SRS is the source of the ^{129}I. The inventory in the local terrestrial environment is consistent with the calculated releases; this indicates that calculated releases probably were not underestimated.

The terrestrial results also show that forested areas are more efficient than open areas in removing ^{129}I from the air. Deposited ^{129}I is shown to be efficiently retained near the surface. These two observations are consistent with observations near other nuclear facilities.

Groundwater Transport

The movement of radionuclides in groundwater has been described in two fundamental equations (Codell and Duguid, 1983). One equation describes the movement of the carrier fluid (water). The other equation describes the mass transport of dissolved constituents (such as radionuclides). If the movement of the carrier in a given region is known, the distribution coefficient (K_d) can be used to predict radionuclide transport.

The K_d for a given radionuclide is defined as the ratio of the activity per unit mass of soil divided by the activity per unit volume of water in the soil. This relationship indicates that the smaller the K_d, the more mobile the radionuclide is. Measured K_d values for a given radionuclide typically show a wide variability depending on factors such as soil particle size, soil characteristics, chemical form of the radionuclide, and water chemistry (NCRP, 1984).

With a typical K_d of 3.0, ^{129}I has a mobility intermediate between tritium (K_d of 0.001) which is very mobile and ^{137}Cs (K_d of 100) which is not very mobile. Therefore, some migration of ^{129}I in groundwater at SRS is expected.

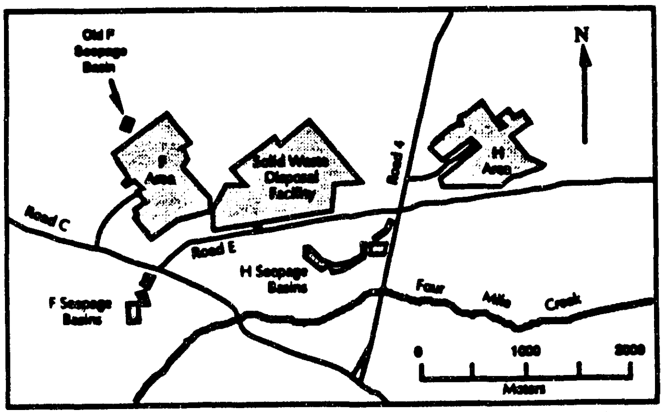


Figure 4-15. Locations of Waste Sites Near the Center of SRS

Sources of ^{129}I in groundwater are principally the effluents released to earthen seepage basins at the separations areas and the solids buried in the SWDF. These waste sites, which are shown in Figure 4-15, arose from past waste disposal practices that were consistent with general industry methods. Use of the seepage basins in the separations areas was terminated in 1988.

The groundwater near these waste sites becomes contaminated with radioiodine when water percolates through the soil. In the SWDF, rainwater that percolates through the soil leaches radioiodine from certain buried wastes such as the spent ceramic chips coated with silver nitrate. In the seepage basins, dissolved radioiodine migrates with the percolating basin water. Groundwater moves downward and horizontally and resurfaces adjacent to site streams, principally Four Mile Creek. Because migration times from these waste sites to the outcrops are measured in years, ^{129}I is the only radioactive isotope of iodine present in resurfaced water in measurable quantities.

A thorough description of the geological and hydrological characteristics near the waste sites is presented in the reference Murphy et al., 1990.

F-Area Seepage Basin System

The F-Area Seepage Basins that were used from 1955 to 1988 are located south of F Area as shown in Figure 4-15. The Old F Area Seepage Basin, which was abandoned in 1955, is located north of F Area. The basins are approximately 9 km from the nearest site boundary. The elevation of the basins is approximately 85 meters above sea level. The nearest major site streams to the F-Area Seepage Basins used since 1955 are Four Mile Creek (600 meters to the southeast) and Upper Three Runs (1,600 meters to the northwest).

A limited study of the ^{129}I content of F Seepage Basin 3 was conducted in 1977–1978. Concentrations of ^{129}I in monthly grab samples of basin water ranged from 50 to 300 pCi/L with an average of 160 pCi/L (Anderson, 1978; Kantelo, 1987). Month–to–month variations are attributed to rainfall and operational conditions in the separations process. The

^{129}I in a basin may remain in solution with the water in the basin, volatilize to the atmosphere, or be transported with the basin water as it migrates through the ground.

Because extensive data on the migration of radioiodine do not exist, tritium migration data can be used to illustrate possible radioiodine movement. The travel time for tritium to migrate from the seepage basins to the outcrops adjacent to Four Mile Creek ranges from approximately 15 to 30 years (Murphy et al., 1990). Because the K_d for iodine is greater than the K_d for tritium, iodine moves at a slower rate than tritium. Groundwater resurfaces along the seepline that is shown in Figure 4-16.

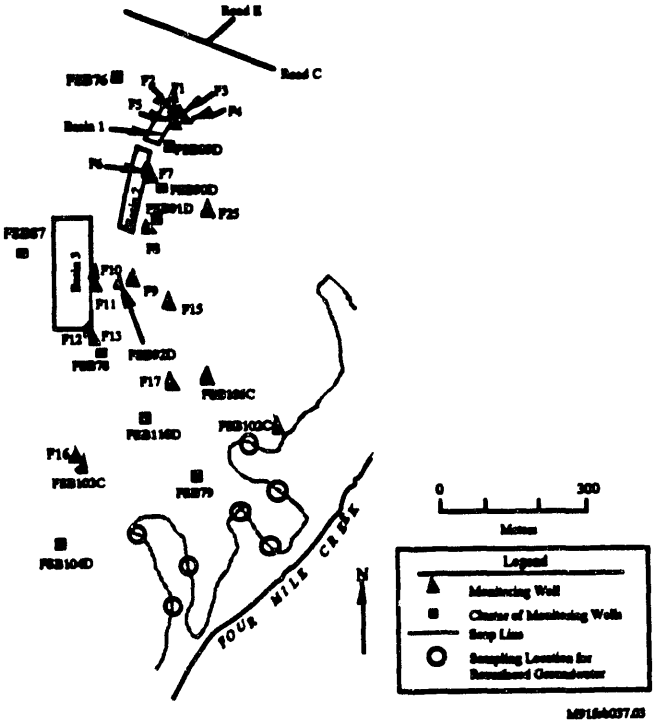


Figure 4-16. Locations of Monitoring Wells Selected for Analysis of Radioiodine in Groundwater Near F-Area Seepage Basins

Although ^{131}I has a short half life, its migration was detected in monitoring wells for perched water and the water table during the early years of operation. The locations of these wells, designated F1 through F13, are shown in Figure 4-16. The greatest concentrations were in the shallow wells closest to the seepage basins. The greatest annual average concentration of ^{131}I was approximately 5x10^4 pCi/L in well F6 in 1956, the year of maximum aqueous releases to the F–Area Seepage Basins.

Migration of ^{129}I in groundwater at the F seepage basins was measured in 1977 when water table samples were analyzed by neutron activation analysis. In monitoring wells F15, F16, F17, and F25, shown in Figure 4-16, concentrations of ^{129}I ranged from 0.1 to 300 pCi/L (Anderson, 1978; Kantelo, 1987).

Routine analysis for ^{129}I using low energy photon spectroscopy began in 1989. Although it is less sensitive (detection limit of approximately 1 pCi/L) than neutron activation analysis, it is more amenable to extensive routine analyses. Groundwater from 24 monitoring wells was analyzed for ^{129}I several times during 1989. The locations of these wells, which have the letter designations "FSB", are shown in Figure 4-16. The individual wells of a well cluster are not shown. A well cluster consists of several monitoring wells, with each well typically collecting water in different geological formations. The progressively deeper waters that were analyzed in 1989 are the water table, the McBean Member of the Santee Formation, the Upper Congaree Formation, and the Lower Congaree Formation.

At the water table depth, ^{129}I was detected in 8 of 9 wells. The average concentration is 80 pCi/L and the maximum is 290 pCi/L. In McBean ^{129}I was detected in 5 of 7 wells at an average concentration of 60 pCi/L and a maximum of 140 pCi/L. No ^{129}I was detected in the eight wells of the Congaree formation, which is a minor source of drinking water at SRS.

The shallower Santee Formation is also a minor source of drinking water at SRS. The only SRS wells that draw drinking water from the Santee Formation are located in the B and SRL areas, which are at least 5 km from the seepage basins (Figure 3-1, page 17). Although the concentrations observed in the McBean Member of the Santee Formation near the F seepage basins exceed the EPA standard of 1 pCi/L for ^{129}I in drinking water (EPA, 1977d), the part of the Santee Formation tapped for drinking water is not contaminated with ^{129}I. This conclusion is based on the lack of elevated tritium concentrations in the drinking water; ^{129}I concentrations in McBean are associated with tritium concentrations that are a factor of approximately 10^5 greater. The source of the drinking water is not likely to become contaminated because the contaminated groundwater flows in the opposite direction toward Four Mile Creek. Furthermore, Upper Three Runs Creek cuts through the Santee Formation to physically isolate the two portions of the formation. This is shown in Figure 4–17, which is based on the reference Aadland and Bledsoe, 1990.

Figure 4-17. Hydrogeologic Profile Along a Straight Line Connecting the F-Area Seepage Basins and B Area (SRL is Offset from the Line; also, See Figure 3-1)

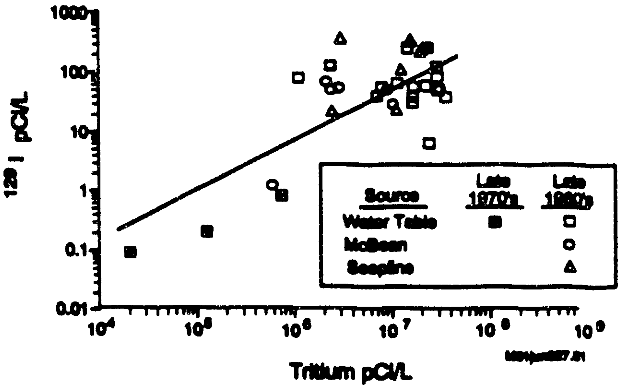


Figure 4-18. Correlation of ^{129}I and Tritium Concentrations in the Groundwater Near the F-Area Seepage Basins

Groundwater resurfaces at outcrops along a seepline adjacent to Four Mile Creek at an elevation approximately 20 meters lower than the elevation of the basins. The location of the seepline is shown in Figure 4-16.

The first measurement of ^{129}I in outcrop water was in 1977. The ^{129}I concentration in a grab sample was 224 pCi/L (Anderson, 1978). In 1988 and 1989, a comprehensive survey for numerous radionuclides, including ^{129}I, was conducted along the seepline; Figure 4-16 shows most of the locations sampled close to the basins. Concentrations of ^{129}I in grab samples of resurfaced groundwater ranged from 24 to 340 pCi/L in 1988, and from less than 23 (detection limit) to 410 pCi/L in 1989 (Haselow et al., 1990).

The outcrop results for 1977, 1988, and 1989 are not directly comparable because sampling locations are not the same in each year. The results only show that the elevated concentrations of ^{129}I in outcrop water are in the hundreds of pCi/L. These concentrations are consistent with the concentrations observed in the groundwater monitoring wells.

Most of the groundwater samples analyzed for ^{129}I were also analyzed for tritium. Figure 4-18 shows the concentrations of ^{129}I plotted against the corresponding tritium concentrations. Groundwater ^{129}I concentrations are associated with tritium concentrations that are greater by a factor of 10^5 to 10^6. The correlation of ^{129}I and tritium concentrations is shown by the regression line through the data. The slope of the line is 0.81, which means for every doubling of tritium concentration, the ^{129}I concentration increases a factor of 1.8.

Sediments in the path of the migrating basin water also show evidence of ^{129}I. Sediment cores taken inside the F-Area seepage basins in 1984 show a maximum concentration of 120 pCi/g (Corbo et al., 1985). The vertical profile of ^{129}I concentrations can not be determined because only the top and bottom of the cores were analyzed for ^{129}I; this provides only two data points for the vertical profile.

Sediments cores were taken along the seepline in 1988 and 1989. Concentrations of ^{129}I in sediments from various depth intervals range from 1 to 5 pCi/g in 1988 and are less than detection limits of approximately 5 pCi/g in 1989 (Haselow et al., 1990). (Because detection limits are not fixed values, it is possible to measure 1 pCi/g in one set of analyses and have a detection limit of 5 pCi/g in another set.) Coring locations are not the same in each year.

Sediment cores were taken from inside the abandoned Old F-Area Seepage Basin in 1986 to study migration of radionuclides. Sections from various depth intervals in four basin-sediment cores were individually analyzed for ^{129}I. The maximum concentration is 3 pCi/g (Shedrow, 1986). Vertical profiles of ^{129}I concentrations show no consistent pattern. In two cores ^{129}I is still detected at the bottom of the 400-cm deep cores. In the other two cores, the ^{129}I content is below the detection limit of approximately 0.3 pCi/g beginning at depths of 100 and 300 cm.

H-Area Seepage Basin System

The H-Area Seepage Basins are located southwest of H Area (see Figure 4-15). The basins are about 11 km from the nearest site boundary and have an elevation of approximately 75 meters above sea level. The nearest major site streams are Four Mile Creek, which is 300 meters to the southeast, and Crouch Branch (a tributary of Upper Three Runs Creek), which is 1,100 meters to the north.

A limited study of the ^{129}I content of H–Seepage Basin 4 was conducted in 1977–1978. Concentrations of ^{129}I in monthly grab samples of basin water ranged from 15 to 80 pCi/L, with an average of 46 pCi/L (Anderson, 1978; Kantelo, 1987). As described for the F seepage basin system, month–to–month variations are attributed to rainfall and operational conditions in the separations process. Also, tritium migration data can be used to illustrate possible ^{129}I migration. The travel time for tritium to migrate from the seepage basins to Four Mile Creek ranges from 14 to more than 50 years (Murphy et. al., 1990). Groundwater resurfaces along the seepline shown in Figure 4-19.

Migration of ^{131}I was detected in monitoring wells for the water table during the early years of operation. The locations of these water table wells, designated H1 through H8, are shown in Figure 4–19. The greatest annual average concentration of ^{131}I was approximately 3 x 10^6 pCi/L in well H5 in 1956.

Migration of ^{129}I in groundwater at the H Seepage Basins was measured in 1977 when water table samples were analyzed. In monitoring wells H13, H14, H15, H18, and H19, shown in Figure 4-19, concentrations of ^{129}I ranged from 0.1 to 16 pCi/L (Anderson, 1978; Kantelo 1987). An ^{129}I concentration of 63 pCi/L was measured in 1980 for the water table monitoring well BG10 adjacent to basin 4 (Kantelo, 1987). This is the maximum ^{129}I concentration detected in the groundwater near the H–Area Seepage Basins.

Groundwater from 19 monitoring wells was analyzed for ^{129}I several times during 1989. The locations of these wells, which have the letter designations "HSB", are shown in Figure 4-19. The progressively deeper waters that were analyzed in 1989 were the water table, the Upper Congaree Formation, and the Lower Congaree Formation. At the water table depth, ^{129}I was detected in 13 of 16 wells. The average concentration is 14 pCi/L and the maximum is 50 pCi/L. No ^{129}I was detected in the three wells for the Congaree Formation.

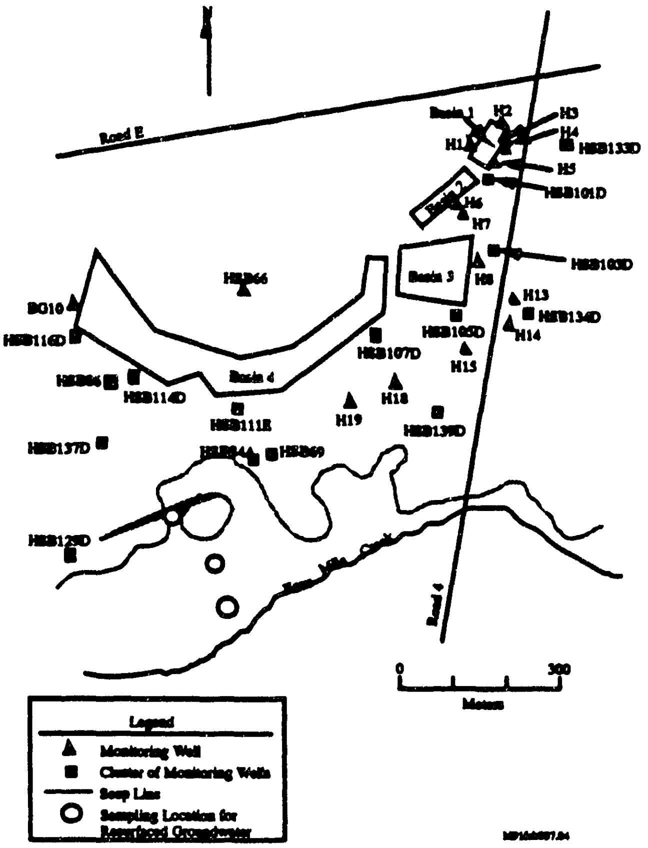


Figure 4-19. Locations of Monitoring Wells Selected for Analysis of Radioiodine in Groundwater Near H-Area Seepage Basins

Groundwater resurfaces along a seepline adjacent to Four Mile Creek at an elevation approximately 15 meters lower than the elevation of the basins. The location of the seepline is shown in Figure 4-19. Concentrations of ^{129}I in grab samples of resurfaced groundwater at two locations along the seepline were 5 and 50 pCi/L in 1988, and were below detection limits of approximately 50 pCi/L at five locations in 1989 (Haselow et al., 1990). Figure 4-19 shows the locations sampled close to the basins.

The outcrop results for 1988 and 1989 are not directly comparable because sampling locations are not the same in each year. The results only show that the elevated concentrations of ^{129}I in outcrop water are in the tens of pCi/L. These concentrations are consistent with the observed concentrations in the groundwater monitoring wells and are an order of magnitude lower than the results for the F-Area seepage basin system.

Figure 4-20 shows concentrations of ^{129}I plotted against corresponding tritium concentrations. Groundwater ^{129}I concentrations are associated with tritium concentrations that are greater by a factor of 10^5 to 10^6. The correlation of ^{129}I and tritium concentrations is shown by the regression line through the data. The slope of the line is 0.73, which is not as steep as the slope of 0.81 for the groundwater data near the F-Area seepage basins. The difference is meaningless because insufficient data exists at the lower concentrations to determine precise slopes for the data in Figures 4-18 and 4-20.

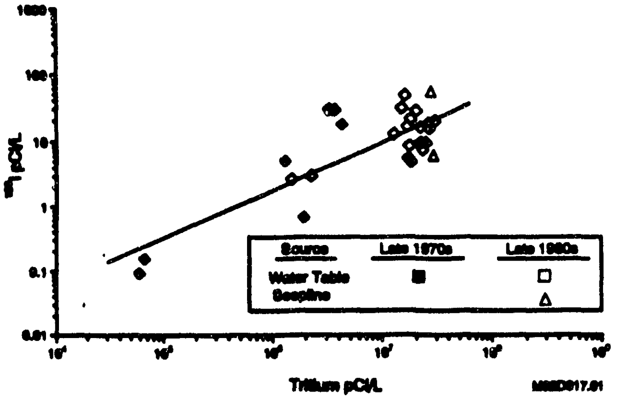


Figure 4-20. Correlation of ^{129}I and Tritium Concentrations in the Groundwater Near the H-Area Seepage Basins

An incorrect tritium concentration of 1.6 x 10^5 pCi/ml on May 23, 1989 is reported for monitoring well HSB116D in the 1989 annual environmental report. The correct value is 1.5 x 10^4 pCi/ml (EMS, 1989a and EMS, 1989b). The correct value, converted to units of pCi/L, appears in Figure 4-20.

Sediments in the path of the migrating basin water also show evidence of ^{129}I. Sediment cores taken from H-Area Seepage Basins in 1984 show a maximum concentration of 190 pCi/g (Corbo et al., 1985).

Sediment cores have also been taken along the seeplines. Concentrations of ^{129}I in sediments from various depth intervals range from 1 to 5 pCi/g in 1988 and are less than detection limits of approximately 5 pCi/g in 1989 (Haselow et al., 1990). Coring locations are not the same each year.

Solid Waste Disposal Facility

The SWDF, which is located between F and H Areas (see Figure 4–15), is approximately 10 km from the nearest site boundary and has an elevation of approximately 88 meters above sea level. The nearest major site streams are Four Mile Creek, which is 900 meters to the south, and Crouch Branch, which is 700 meters to the north. The horizontal flow of groundwater beneath the SWDF is divided between southerly flow toward Four Mile Creek and northerly flow toward Upper Three Runs Creek.

Tritium migration data can be used to illustrate possible ^{129}I migration. The travel time for tritium migration from the SWDF to Four Mile Creek ranges from 11 to 50 years. The travel times to Upper Three Runs Creek are considerably longer; essentially all the tritium (12-year half-life) is expected to decay before outcropping (Murphy et al., 1990).

The principal source of ^{129}I buried in the SWDF is spent ceramic chips that were used for radioiodine removal during dissolution of irradiated materials. The presumed chemical form of ^{129}I on the chips is silver iodide, which is only very slightly soluble in water.

An experimental study of the leachability of ^{129}I from buried chips demonstrates that a small fraction, 1.8 x 10^{-7} (0.18 parts per million), is leachable from the waste (Hawkins, 1983). This fraction was relatively constant on an annual basis during the four-year study, although the leachable fraction for individual samples during a given year fluctuated with the quantity of rainfall. This small leachable fraction is consistent with the very slight solubility of silver iodide. The leached iodine is expected to exist in the iodide chemical form in groundwater.

An experimental determination of the distribution coefficient for ^{129}I in soil from the SWDF shows that K_d ranges from 3.6 to 10 depending on the concentration of stable iodine. Smaller K_d values are associated with the greater stable iodine concentrations. With a K_d of 3.6, ^{129}I is predicted to move at 5 percent of the groundwater velocity (Hoeffner, 1985).

Until 1989 only limited analyses were performed to determine the extent of ^{129}I migration from the SWDF. The monitoring well PDQ5 was of particular interest in the earlier studies. This well is 180 meters from the south boundary of the SWDF (see Figure 4-21). It has a 2-meter screen at the 14-meter depth, which corresponds to the middle of the of migrating tritium plume. Concentrations of ^{129}I in well PDQ5 increased from 0.25 pCi/L in April 1982 to 1.5 pCi/L in December 1982 to 12 pCi/L in August 1983 (Oblath, 1986). These limited results demonstrate the arrival of the leading edge of migrating ^{129}I. The concentration of tritium is approximately 10^6 greater than the ^{129}I concentration in the August 1983 sample. The ^{129}I concentration is less than the detection limit of 7 pli/L in 1987 (McIntyre and Wilhite, 1988).

Measurable concentrations of ^{129}I were detected in groundwater from monitoring wells BG56, BG57, BGC3C, I13, and BGC2C along the south boundary of the SWDF in 1979, 1980, and 1983. Concentrations are no greater than 0.03 pCi/L with the exception of well BGC3C, which has a concentration of 0.9 pCi/L (Kantelo, 1987). The tritium concentration exceeds the ^{129}I concentration by factors of 10^7 to 10^8 in the August 1983 samples. The purpose of wells BGC3C and BGC2C is to determine the depth and breadth of the migrating tritium plume. The other wells are water table wells.

Groundwater from 19 monitoring wells was analyzed several times during 1989. The locations of these wells, which have the letter designation "BGO" and "HSB", are shown in Figure 4-21. The progressively deeper waters analyzed in 1989 were the water table (9 wells), the McBean Member of the Santee Formation (7 wells), and the Congaree Formation (3 wells). No ^{129}I was detected in the water samples from these 19 wells at a detection limit of approximately 1 pCi/L. The 1989 results are consistent with the earlier studies; monitoring well PDQ5 was not sampled in 1989.

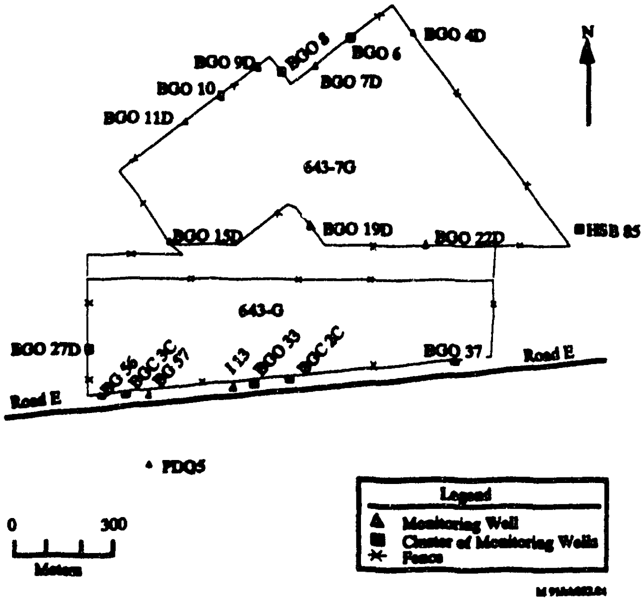


Figure 4-21. Locations of Monitoring Wells Selected for Analysis of Radioiodine in Groundwater Near SWDF

Surface Water Transport

The proximity of major site streams to SRS facilities involved with radioiodine is shown in Figure 3-1 (see page 17). The major streams are tributaries of the Savannah River, which flows to the Atlantic Ocean. One small unnamed stream in the northeastern sector of the site drains to the east into the Salkehatchie River. No effluent from SRS facilities drains into this unnamed stream, which is not shown in Figure 3-1.

Surface Waters at SRS

The principal sources of the natural flow in site streams are springs and rainfall. Cooling water from SRS reactors and other SRS processes augments the natural flow in certain streams. The majority of radioiodine transported by site surface waters originates in direct aqueous releases from SRS facilities or migration from separations area seepage basins and the SWDF. A minor source is rainwater runoff, which transports part of the radioiodine deposited from SRS atmospheric releases and global fallout.

Upper Three Runs Creek is the only site stream that originates offsite. It is 40 km long, has a watershed of about 540 km^2, and an average natural flow rate of approximately 7 m^3/sec. One of its principal tributaries is Tim's Branch, which receives effluents from M Area and SRL. These effluents add approximately 0.3 m^3/sec to the flow rate of Upper Three Runs Creek. The Effluent Treatment Facility in H-Area began discharging treated water to Upper Three Runs Creek in 1988.

Beaver Dam Creek is the smallest of the SRS streams that drain into the Savannah River. Its length is about 5 km and its watershed is about 26 km^2. Beaver Dam Creek receives approximately 3 m^3/sec from D Area effluents. In the Savannah River swamp, overflow from Four Mile Creek mixed with Beaver Dam Creek when C Reactor discharged secondary cooling water.

Four Mile Creek is approximately 24 km long and has a watershed of about 57 km^2. Its natural flow is 0.14 m^3/sec. It receives effluent from the F and H chemical separations areas and migrating water from the SWDF and F- and H-Area Seepage Basins. Effluent from the chemical separations areas adds approximately 0.3 m^3/sec. Until the mid 1980s the lower reaches of Four Mile Creek received secondary cooling water from C Reactor. Cooling water added a flow of approximately 11 m^3/sec when the reactor was operating. Miscellaneous C-Area discharges added approximately 0.14 m^3/sec.

Pen Branch flows approximately 24 km to the Savannah River swamp where its flow path becomes relatively undefined. It generally flows through the swamp parallel to the river for about 8 km before entering Steel Creek near its confluence with the Savannah River. Pen Branch has a watershed of about 55 km^2. Its natural flow rate of 0.2 m^3/sec increases to about 11 m^3/sec when K Reactor discharges secondary cooling water.

Steel Creek is approximately 18 km long and has a watershed of about 90 km^2. Flow rates vary depending on reactor effluents. For example the flow was 22 m^3/sec when both P and L Reactors discharged aqueous effluents to Steel Creek in the early 1960s, 0.7 m^3/sec for most of the 1970s, and 11 m^3/sec in 1985 after the restart of L Reactor. A large impoundment, L Lake, was formed by damming Steel Creek south of L Reactor in the mid 1980s.

Lower Three Runs Creek has the second greatest drainage area (150 km^2) of the SRS streams. Both P and R Reactors discharged effluents to the Lower Three Runs Creek system. In the upper reaches of the creek a large impoundment, Par Pond, was constructed in 1958 to provide cooling water for recirculation through P and R Reactors. Several smaller ponds were constructed between the reactors and Par Pond to provide cooling for reactor effluents reentering Par Pond. The average flow rate from Par Pond into Lower Three Runs Creek is 1 m^3/sec. From the Par Pond dam, Lower Three Runs Creek flows approximately 38 kilometers to the Savannah River. For most of this distance the creek is outside the main part of the site; however, the creek is bordered by a narrow strip of SRS-owned land.

Site steams, other than Upper Three Runs Creek and Lower Three Runs Creek, pass through an extensive swamp along the South Carolina side of the Savannah River. The swamp typically floods at least one day during every month of the year (Marter, 1974). The most frequent flooding occurs in January through April and the annual average flooding is 23 percent of the year.

When flooding occurs, the flow in the Savannah River swamp generally runs parallel to the river and does not enter the main channel until high ground is encountered approximately 10 km downriver from Steel Creek. This condition allows mixing of water from site streams with the water in the flood plain.

Figure 4-22 shows the annual average concentrations of ^{131}I measured in the SRS surface waters. No ^{131}I has been detected in Upper Three Runs Creek, Beaver Dam Creek, or the swamp. The concentration in Four Mile Creek during 1962 was below the detection limit. No annual average ^{131}I results were reported before 1960 or during 1965-1977; concentrations have been below detection limits after 1977.

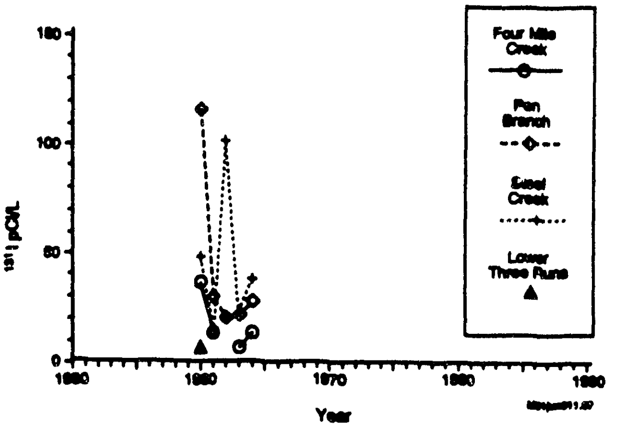


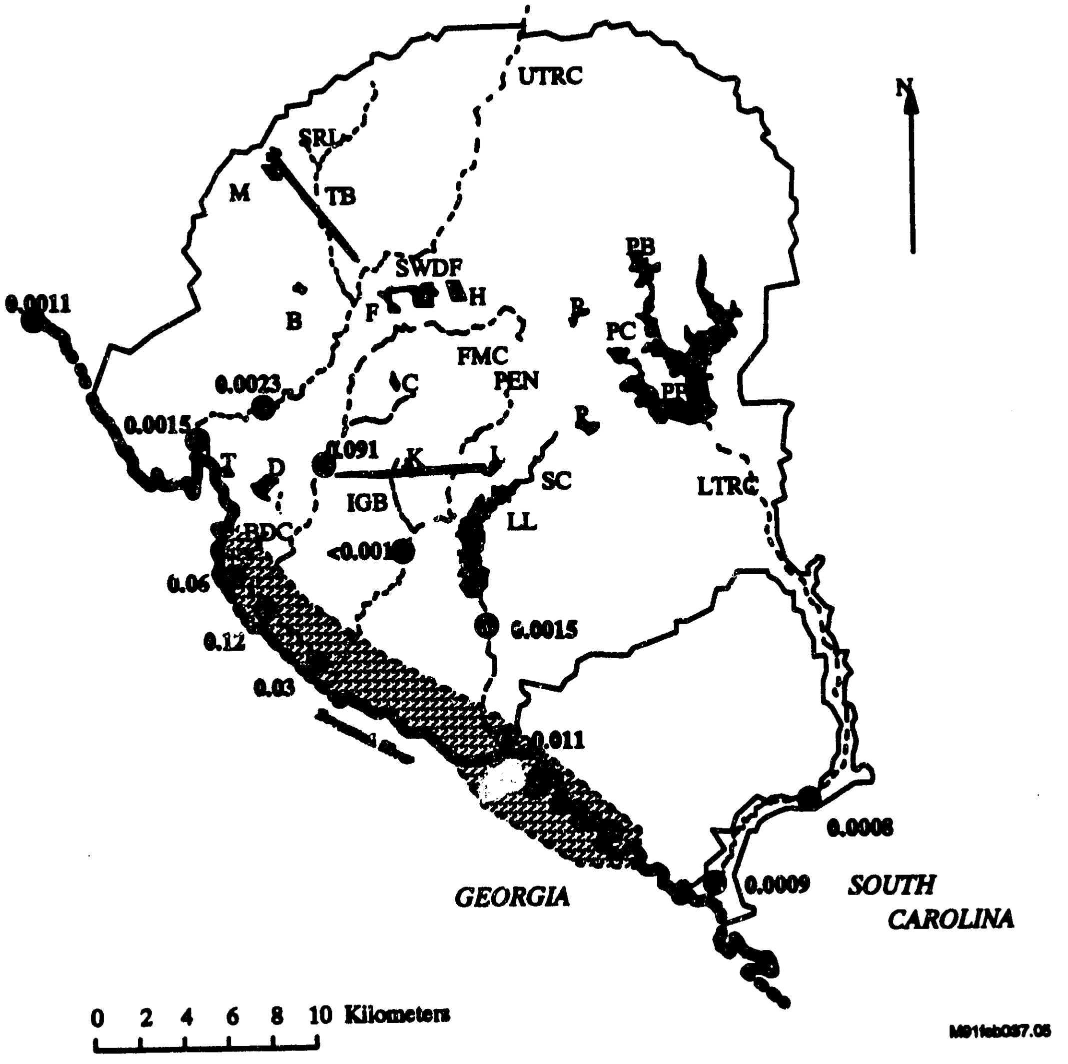
Figure 4-22. Annual Average Concentrations of ^{131}I in SRS Streams from 1960 through 1989

The greatest annual average ^{131}I concentrations are associated with fuel element failures in the reactors. These concentrations were 116 pCi/L in Pen Branch during 1960 and 102 pCi/L in Steel Creek during 1962. The greatest ^{131}I concentration in an individual sample of stream water was 22,000 pli/L, which was observed in Steel Creek following a fuel element failure in P Reactor in 1957.

Surveys to determine ^{129}I concentrations in grab samples of site streams and swamp waters were conducted in August 1977 and June 1978 (Anderson, 1978; Anderson, 1980; Kantelo, 1987). Concentrations for a given stream in the two surveys generally agree within a factor of two. This difference is not considered significant because concentrations depend on flow conditions, which were not documented at the time of sampling. The 1978 survey results are shown in Figure 4-23.

Local background concentrations of ^{129}I were approximately 0.001 pCi/L, as shown by the values in Lower Three Runs Creek and the Savannah River upriver from SRS. This local background is approximately 100 times greater than the North American river water background, which is on the order of 10^{-5} pCi/L (Brauer, 1974). The local background in streams is elevated because of the deposition of airborne SRS ^{129}I onto the terrestrial environment. Rainfall runoff leaches part of the deposited ^{129}I and transports it to the streams.

Concentrations in Upper Three Runs Creek—the stream with the largest watershed—are approximately twice the local background. This is because the proximity of Upper



Facilities Legend

B	HWCTR
C,K,L,P,R	Reactor Areas
D	Heavy Water
F	Chemical Separations + Naval Fuels
H	Chemical Separations + RBOF
M	Raw Materials
SRL	Savannah River Laboratory
SWDF	Solid Waste Disposal Facility
T	TNX Semi Works

Surface Waters Legend

BDC	Beaver Dam Creek
FMC	Four Mile Creek
IGB	Indian Grave Branch
LL	L Lake*
LTRC	Lower Three Runs Creek
PB	Pond B
PC	Pond C
PEN	Pen Branch
PP	Par Pond
SC	Steel Creek
TB	Tims Branch
UTRC	Upper Three Runs Creek
▨	Swamp

*L Lake did not exist at the time of these measurements

Figure 4-23. Concentrations of ^{129}I (pCi/L) in SRS Streams and the Savannah River Swamp in 1978

Three Runs Creek to the separations areas results in greater deposition in its watershed than in the other watersheds. Calculations show that the elevated ^{129}I in Upper Three Runs Creek does not originate from spontaneous fission of SRS uranium dispersed into the sediments of Tim's Branch. Although tons of unirradiated uranium have been dispersed, the quantity of uranium cannot produce enough ^{129}I by spontaneous fission to account for the elevated concentrations.

The greatest ^{129}I concentrations are in Four Mile Creek and in the swamp impacted by Four Mile Creek. These concentrations are approximately 100 times greater than the local background. The elevated concentrations are principally due to migration of ^{129}I from the separations area seepage basins.

Four Mile Creek has been studied more than any other stream. Grab samples downstream of the outcrops from F-Area seepage basins but upstream of the confluence with C-Reactor effluent, showed concentrations of 33 and 2 pCi/L in 1973 and 1977, respectively (Kantelo, 1987). In 1988 the concentrations were below the detection limit of 1 pCi/L (Haselow et al., 1990).

Sampling of Four Mile Creek in 1977, 1979, and 1980 between the outcrops from F- and H-Area seepage basins showed ^{129}I concentrations of approximately 1 pCi/L (Kantelo, 1987). In 1988 the concentrations were below the detection limit of 1 pCi/L (Haselow, et al., 1990).

Although a simultaneous sampling of Four Mile Creek upstream and downstream of the outcrops from F-Area seepage basins has not been not performed, the limited results obtained in the 1970s suggest that the majority of ^{129}I in Four Mile Creek originates in the F-Area seepage basins. That conclusion is supported by the 1977, 1988, and 1989 groundwater results along the seeplines if it is assumed that the input rate of ^{129}I to Four Mile Creek from the outcrops near the H-Area seepage basins does not exceed the input rate from the outcrops near the F-Area seepage basins.

Sediments from site streams and the Savannah River swamp have periodically been analyzed. Gamma spectrometric analysis has not revealed any detectable ^{131}I. A comprehensive survey of sediments for numerous radionuclides, including ^{129}I, was conducted in the mid 1980s (Lower, 1987). Sediment cores were taken to a depth of 100 cm. A composite sample of the entire core for each sampling location was analyzed. Results of ^{129}I analyses are shown in Figure 4-24.

Concentrations of ^{129}I generally are at the detection limit of 0.1 pCi/g, or within a factor of two of it. The greatest sediment concentrations of ^{129}I are in Four Mile Creek and the swamp impacted by Four Mile Creek. These concentrations are approximately 5 times greater than the detection limit. The elevated concentrations principally are caused by ^{129}I that migrates from the separations area seepage basins and adsorbs on the sediments.

Despite the history of multi–curie releases of short-lived ^{131}I to Steel Creek, greatly elevated concentrations of ^{129}I were not detected. This is because the high activity ^{131}I releases were associated with failures of freshly irradiated fuel elements; the ^{129}I content of the releases was extremely small (Table 2-1; page 10).

A significant portion of any ^{129}I absorbed on sediments in Steel Creek was probably leached by the hot secondary coolant effluent from L and P reactors. This condition also applies to Pen Branch and K Reactor, although an elevated concentration of 0.36 pCi/g was detected in Pen Branch.

The results for Four Mile Creek may also indicate the effects of hot water leaching. The factor of three difference in ^{129}I concentrations upstream and downstream of the confluence with effluent from C-Reactor may be related to leaching of iodine from sediments by hot water.

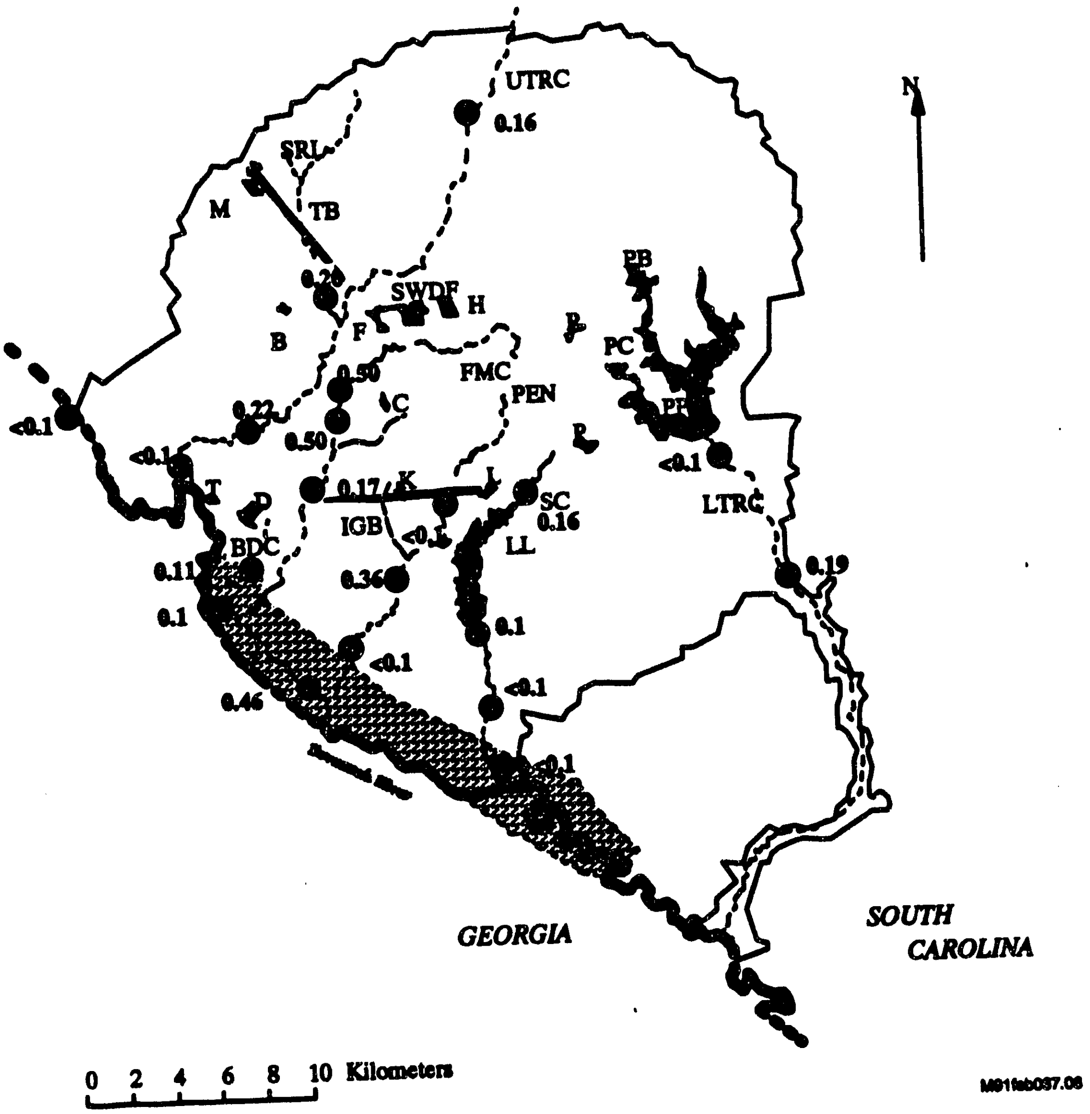
The lack of extensive sampling and analyses for ^{129}I limits conclusions to the generalities stated in this section on SRS surface waters.

Savannah River

The Savannah River forms the southwestern SRS site boundary for about 33 km. It has an average flow rate that increases from 285 m^3/sec upriver from SRS near Augusta, GA, to 340 m^3/sec downriver near Clyo, GA. The upriver and downriver locations are approximately 300 and 100 river-kilometers, respectively, from the Atlantic Ocean. SRS is located between 261 and 228 river-kilometers from the Atlantic Ocean.

SRS has three pump stations on the river. Two supply cooling water for the production reactors while the third supplies cooling and process water for D Area. Except for evaporative losses, all river water is returned to the Savannah River.

The annual average concentrations of ^{131}I in Savannah River water at Highway 301 Bridge, approximately 37 km downriver from Steel Creek, are shown in Figure 4-25. Concentrations prior to 1960 were not measured. The plotted concentration is the net concentration, which is the difference between measured concentrations at Highway



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PC	Pond C
PEN	Pen Branch
PP	Par Pond
SC	Steel Creek
TB	Tims Branch
UTRC	Upper Three Runs Creek
▨	Swamp

Figure 4-24. Concentrations of ^{129}I (pCi/g) in Sediments from SRS Streams and the Savannah River Swamp in the Mid 1980s

301 Bridge and Shell Bluff. The latter is a background location approximately 8 km upriver from Upper Three Runs Creek. The net concentration of 0.2 pCi/L in 1966 is based on downriver and upriver annual average concentrations of 0.3 and 0.1 pCi/L, respectively. In all other years, the upriver concentrations have been either greater than the downriver values or less than detection limits. In the latter case, no subtraction is performed; the downriver concentration is considered to be the net concentration.

The source of the ^{131}I in the Savannah River was direct discharges of disassembly basin water from reactors to site streams. The maximum annual average concentration of 10 pCi/L occurred in 1960; another peak occurred in 1962 at 8.5 pCi/L. Concentrations of ^{131}I in 1960-1963 are greater than the 1977 EPA drinking water standard of 3 pCi/L (EPA, 1977d).

The greatest concentration of ^{131}I in an individual sample of river water is associated with the 1957 fuel element failure in P Reactor. Just below the confluence of Steel Creek with the Savannah River the maximum concentration was 3,800 pCi/L. Further downriver at Highway 301 Bridge, the maximum was 800 pCi/L. The decrease in concentration reflects dilution of Steel Creek water by the Savannah River.

The net concentrations are compared in Figure 4-25 to calculated downriver concentrations. The latter are determined by dividing annual total curies of ^{131}I released to SRS streams that feed the Savannah River by the water volume transported in the Savannah River past SRS for each individual year. Decay during the transit time to Highway 301 is not considered. Releases of ^{131}I from R-Reactor are also not considered; these were discharged to Par Pond. Except for 1960, measured and calculated results compare favorably.

Savannah River water has been intermittently analyzed for ^{129}I between 1970 and 1981. Grab samples of water were collected downriver from SRS at Highway 301 Bridge and upriver from SRS at Shell Bluff (Anderson, 1978; Anderson, 1980; Kantelo, 1987). Concentrations of ^{129}I at both locations are shown in Figure 4-26. SRS is the principal source of ^{129}I in the river. Figure 4-26 shows that the downriver concentration typically exceeds the upriver concentration by an order of magnitude. Downriver ^{129}I concentrations are typically less than 1 percent of the EPA drinking water standard of 1pCi/L (EPA, 1977d).

Detection limits for individual samples are in the range of 0.0001 to 0.001 pCi/L. The value of the detection limit for a given sample is governed by a number of factors, such as volume sampled and measurement interferences. The North American river water background is on the order of 10^{-5} pCi/L (Brauer, 1974).

The occasional detection of ^{129}I in upriver water reflects the surface water transport of ^{129}I from the watershed. Rainwater runoff in the watershed transports ^{129}I that is deposited from SRS atmospheric releases and global fallout. Anomalous results were obtained in the summer of 1978 when elevated concentrations of ^{129}I were detected in the upriver samples. As a result, concentrations upriver are essentially equal to corresponding concentrations downriver. The origin of this anomaly is unknown.

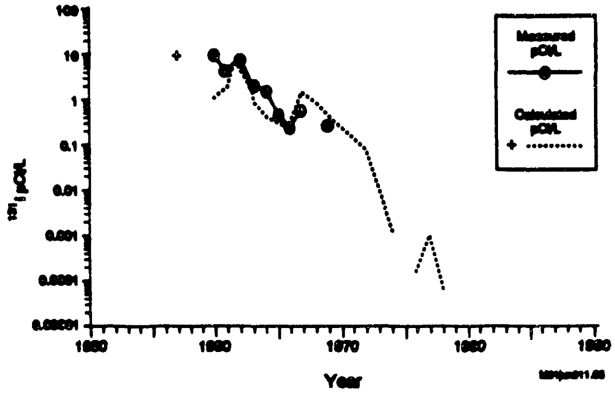

Figure 4-25. Measured and Calculated River Concentrations of ^{131}I due to SRS operations from 1957 through 1989 (Savannah River at Highway 301 Bridge Downriver from SRS)

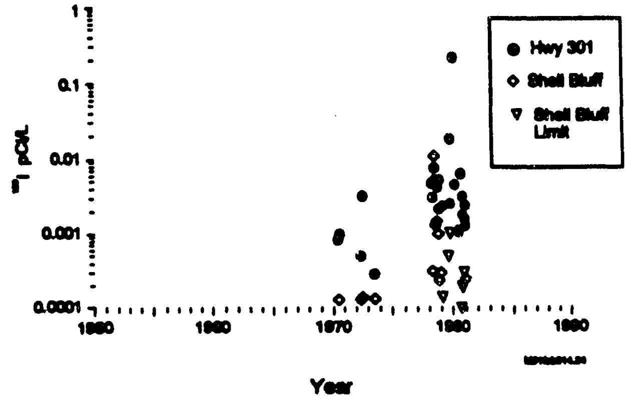

Figure 4-26. Comparison of ^{129}I Concentrations in the Savannah River Upriver and Downriver from SRS Measured Intermittently from 1970–1981

The transport rate for SRS ^{129}I in the Savannah River was initially estimated to be between 0.016 and 0.027 curies per year (Anderson, 1978; Anderson, 1980). When later measurements were also considered, the transport rate was determined to be 0.022 curies per year. This value continues to be used in annual environmental reports and release reports as the estimate for ^{129}I migration from the separations area seepage basins since 1980. For this purpose the 0.022 curies is equally divided between the F– and H–Area seepage basin systems. Migration from the seepage basins prior to 1980 is not considered. Figure 4–26 shows that ^{129}I was detected in the Savannah River in 1970, although at lower concentrations than in the late 1970s. Migration from the separations area seepage basins is the most likely source of the ^{129}I in 1970.

Downriver Water Treatment Plants

The Beaufort-Jasper Water Treatment Plant supplies water to Beaufort and Jasper counties in South Carolina. Operation of the plant, which has a capacity of 38 million liters per day, began in 1965. Savannah River water is withdrawn near river-kilometer 65 and transported in a 29-km canal to the plant.

The Port Wentworth Water Treatment Plant supplies water to parts of Chatham County Georgia. The plant, which has a capacity of 170 million liters per day, was operating before SRS existed. Savannah River water is withdrawn from Abercorn Creek at a point about 3 km from its confluence with the Savannah River and transported through an 11-km pipeline to the plant. The mouth of Abercorn Creek is near river-kilometer 46.

Although ^{129}I at the water treatment plants has not been analyzed, concentrations are expected to be about 20% smaller than those at Highway 301. These are well below the EPA drinking water standard of 1 pCi/L for ^{129}I.

Savannah River Estuary and Atlantic Coastal Waters

No studies of radioiodine in the estuary or coastal waters have been performed.

Summary of Aqueous Transport

Migration of ^{129}I occurs from the separations area seepage basins and the SWDF; the extent of ^{129}I migration from SWDF is not well defined. Groundwater ^{129}I migrating from the separations area seepage basins began resurfacing adjacent to Four Mile Creek probably as early as 1970. This is consistent with expected migration rates.

Groundwater concentrations of ^{129}I, which near the F-Area seepage basin system reach hundreds of pCi/L, are associated with tritium concentrations that are a factor of 10^5 to 10^6 greater. This relationship is used as an indicator for the presence or absence of ^{129}I in other water. For example, groundwater from the Santee Formation is a minor source of drinking water for the B and SRL areas at SRS. The lack of elevated tritium concentrations in this drinking water indicates that the portion of the formation tapped for drinking water is not contaminated with the ^{129}I that contaminates the McBean Member of the Santee Formation near the F-Area seepage basins. The source of the drinking water is not likely to become contaminated. The contaminated groundwater flows in the opposite direction, and Upper Three Runs Creek cuts through the Santee Formation to physically isolate F Area from the source of the drinking water.

Groundwater in the deeper Congaree Formation is also a minor source of drinking water at SRS. No ^{129}I has been detected in groundwater from the Congaree Formation near the separations area seepage basins. The still deeper groundwater in the Black Creek-Middendorf formations (Tuscaloosa aquifer) is the major source of drinking water at SRS.

The shallower groundwater principally resurfaces along seeplines adjacent to Four Mile Creek. Consequently, Four Mile Creek water contains the greatest ^{129}I concentrations of site streams.

Drainage of the streams into the Savannah River is the principal source of ^{129}I transported by the river. The large volume of river flow dilutes the elevated stream concentrations to levels typically less than 1 percent of the EPA drinking water standard.

Direct discharges of disassembly basin water from reactors to site streams resulted in elevated concentrations of ^{131}I in the Savannah River that during 1960-1963 exceed the 1977 EPA drinking water standard. No ^{131}I has been detected after 1968.

Concentrations of radioiodine at the downriver water treatment plants are expected to be similar to the concentrations measured in the Savannah River approximately 37 km downriver from SRS.

Concentrations in Animals

Atmospheric and aqueous transport of radioiodine results in the intake of radioiodine by wildlife. This principally occurs by animals consuming vegetation and water that contains radioiodine from SRS releases and global fallout. Ingested iodine is principally found in the thyroid gland of animals. Thyroids were one of the first types of environmental samples specifically analyzed for radioiodine by SRS. Routine measurements of ^{131}I began in 1954 and routine measurements of ^{129}I began in 1984. Because animal thyroids are generally not eaten by humans, the dose to humans from this pathway is minimal.

Thyroids from a variety of animals, both small and large, have been analyzed. Table 4-6 contains a summary of measurement results for those years when radioiodine was detected. Small animals include rabbit, fox, raccoon, bobcat, opossum, squirrel, rat, owl, and quail. Large animals consist of deer and cattle. Except for cattle, all of these animals are collected onsite; cattle are collected from local offsite farms or slaughter houses.

The maximum concentrations occurred in 1956 and 1957 when releases of ^{131}I from the separations areas to both the atmosphere and seepage basins were at a maximum. A rat collected near the H-Area seepage basin in 1956 contained the greatest observed ^{131}I thyroid concentration—7.1×10^5 pCi/g. Other animals collected farther from the seepage basins in 1956 and 1957 had ^{131}I thyroid concentrations of about 2,000 pCi/g. For comparison, global fallout in Tennessee during 1957 resulted in ^{131}I concentrations of 26,000 and 67,000 pCi/g in cattle and sheep thyroids, respectively, (Van Middlesworth, 1988).

In years other than 1956 and 1957, ^{131}I thyroid concentrations in animals never exceeded 1,000 pCi/g. In many of those years, global fallout was a significant contributor of ^{131}I in thyroids. In 1954, 1955, 1958, 1961, and 1962 thyroids were collected coincident with the arrival of fallout from atmospheric nuclear weapons tests by the USSR or the USA. In 1966, 1967, and 1968 cattle thyroids were collected specifically for research purposes soon after the arrival of fallout from Chinese tests.

The 1970 results represent two separate samplings of deer thyroids. The first set was collected in February to study the effects of atmospheric releases of ^{131}I from H Area. About 10 curies of ^{131}I were released during the first eight weeks of 1970, due to the reprocessing of short-cooled neptunium targets. Eight thyroids were analyzed and found to have an average ^{131}I concentration of 21 pCi/g. The second 1970 sampling occurred soon after the arrival of fallout from a Chinese test in October. The average ^{131}I concentration in thirty nine thyroids was 32 pCi/g. The elevated results for 1976, 1977, and 1980 are also attributed to fallout from Chinese tests.

The PHS operated a nationwide network, known as the Bovine Thyroid Network, from late 1964 to early 1969. The purpose was to analyze for ^{131}I in bovine thyroids from regions of the country near nuclear facilities. Network results can be compared to the SRS results for cattle thyroids in 1966–1968.

Only the network results for thyroids collected at least 100 kilometers from SRS in the southeastern states are considered for the comparison. These provide the southeastern background and are taken from references PHS, 1967e; PHS, 1968c; and PHS, 1969c. Network maximum ^{131}I concentrations are consistent with the local maximum values in 1966 and 1968. The network maximum concentration in 1966 was 702 pCi/g compared to the local maximum of 785 pCi/g. The network thyroid was collected in Gwinett County, GA, approximately 230 km northwest of SRS. In 1968 the network and local maximum concentrations were 45 and 41 pCi/g, respectively. The network thyroid was from Forsyth County, GA, approximately 260 km northwest of SRS. In 1967 the network maximum was 1,856 pCi/g, which exceeds the local maximum of 290 pCi/g by a factor of six. The network thyroid was collected in Tennessee in early January shortly after the arrival of fallout from the Chinese test of December 28, 1966.

A limited number of flesh samples from deer and hogs were analyzed for ^{131}I since 1965. Only in 1987 was ^{131}I detected; its concentration was 3 pCi/g.

Thyroid concentrations of long-lived ^{129}I are generally not affected by passing global fallout. This is because the activity of ^{129}I produced in atmospheric nuclear weapons tests is exceedingly small relative to the activity of ^{131}I (see Chapter 1).

Some of the thyroids from the Bovine Thyroid Network were included in a comprehensive nationwide survey for ^{129}I during 1964 to 1969 (Brauer et al., 1974). The average ^{129}I concentration in bovine thyroids from South Carolina and Georgia is 0.2 pCi/g. Similar results are observed in eastern Washington and northeastern Oregon near the Hanford nuclear fuel reprocessing facility. Concentrations are at least an order of magnitude lower in states that are more distant from SRS and Hanford.

In 1974, SRL conducted a preliminary unpublished study of ^{129}I concentrations in deer thyroids from the SRS site. The average thyroid concentration was 16 pCi/g. Results of deer thyroid measurements conducted in later years generally proved to be consistent with this value. Routine measurements of deer thyroids from SRS began in 1984. Until 1989, the annual average concentrations never exceeded the 1974 value of 16 pCi/g. In 1989, the average ^{129}I concentration rose to 63 pCi/g. This is due to one deer thyroid with a concentration of 2,145 pCi/g. The average concentration for 1989 without this thyroid is 11 pCi/g, which is consistent with previous years.

The deer with the 2,145 pCi/g concentration had likely ingested the ^{129}I by consuming water and vegetation from the seepline for the separations area seepage basins. The value of 2,145 pCi/g is similar to the maximum concentration of 3,700 pCi/g reported at the West Valley nuclear fuel reprocessing plant in New York (Matusek et al., 1974).

Table 4-6. Radioiodine in animal thyroids

Year	Animal		Detected Concentration pCi/g*			
			^{131}I		^{129}I	
	Number Analyzed	Type	Average	Maximum	Average	Maximum
1954	3	small	103	142		
1955	27	small	33	96		
1956	60	small	**	7.1 x 10^5		
1957	**	**	**	2.8 x 10^5		
1958	32	**	**	680		
1961	3	deer	135	190		
	34	small	120	655		
	3	cow	**	250		
1962	7	deer + small	50	**		
1966	**	cattle	**	785		
1967	**	cattle	**	290		
1968	**	cattle	**	41		
1970	47	deer	30	93		
1974	12	deer			16	91
1976	65	deer	150	670		
1977	70	deer	**	410		
1980	48	deer	6	9		
1984	58	deer			15	210
1985	22	deer			4	31
1986	38	deer			2	5
1987	24	deer			3	16
1988	24	deer			3	20
1989	41	deer			63	2,145

*Concentration units are pCi per gram wet weight.
**Unspecified in published reports.

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5

This chapter describes the health impacts associated with exposure to radioiodine. The methodology used to produce dose estimates is presented. The results of the dose assessment for radioiodine releases from 1955 to1989 are discussed.

Radiation doses are sometimes expressed in terms of rad (radiation absorbed dose). A dose of one rad is equal to the absorption of 100 ergs of energy per gram of absorbing material. Since one rad of absorbed dose from different types of radiation can have different biological effects in humans, the unit of rem was developed. Rem doses, or more precisely, dose equivalents, are obtained by multiplying rad doses by a quality factor. The quality factor places doses on a common scale of biological effect (carcinogenesis) for all types of ionizing radiation. The quality factor for x-rays, gamma rays, and beta particles is 1 (one). Thus, in the case of internal doses from x-rays, gamma rays, and beta particles, one rad is equal to one rem.

The rem is too large to conveniently represent doses from environmental radiation. The millirem (mrem), which is one one-thousandth of a rem, is used for this purpose.

Relationship of Dose to Risk and Health Effects

Exposure of humans to radiation increases the risk of cancer, genetic changes, and fetal abnormalities. Risk factors adopted by the EPA are used to assess the impact of radioiodine releases from SRS.

Ionizing Radiation

Ionizing radiation is radiation that strips the electrons from the matter through which it passes. The interaction of ionizing radiation with biological systems can induce a series of chemical reactions that can cause permanent changes in the genetic material of cells. These changes (mutations) may cause abnormal functioning within the cell or may lead to cell death.

The nature of radiation-induced cellular changes is dependent upon the magnitude of the dose and the rate at which it is received. For the low doses and dose rates encountered in the environment from SRS releases, the most significant type of effect is cancer induction.

This is believed to be a stochastic effect; in other words, an increase in dose increases the probability of the effect, but the severity of the effect is independent of the dose. An illustration of this approach to the dose-response relationship is provided in Figure 5-1.

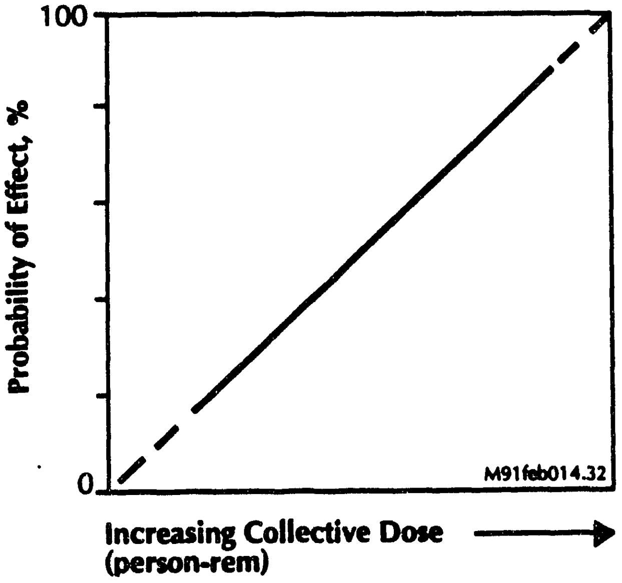

Figure 5–1

As seen in the figure, a characteristic of stochastic risks is the absence of a threshold. In other words, it is conceivable that any dose of radiation, no matter how small, might give rise to a cancer. On the other hand, there is no way to be certain that a given dose, no matter how large, has caused or ultimately will cause a cancer in an individual.

Cancer Risk Estimates

The most comprehensive estimates of cancer induction by exposure to ionizing radiation come from studies of the atomic bomb survivors at Hiroshima and Nagasaki. Less definitive studies include those of medical patients exposed to therapeutic and diagnostic radiation. Studies of laboratory animals have increased the understanding of dose-effect relationships. The International Commission on Radiological Protection (ICRP) has evaluated all of these studies and concluded that the best estimate of lifetime risk of fatal cancer for members of the general population is approximately 500 cases per 1,000,000 person-rem (ICRP, 1991). This is equivalent to one case per 2000 person-rem.

Radioiodine Exposure and Dose to Humans

Stable iodine is an essential trace constituent in the human body. The adult body contains an average of 0.011 grams; in a typical adult, approximately 0.010 of the 0.011 grams are in the thyroid gland. The thyroid, an endocrine gland in the throat below the larynx, secretes iodine-bearing hormones, which are essential for maintaining normal metabolism in all body cells.

Radioactive isotopes of iodine have the same biochemical behavior in the human body as stable iodine, i.e., a large fraction of intake is found in the thyroid gland. Radioiodine released to the environment from the nuclear industry and from weapons tests can exist in a variety of organic and inorganic forms. Humans are exposed to radioiodine in the environment by inhalation and by ingestion of food and water.

Distribution and Retention of Iodine in the Human Body

Essentially all of the iodine that enters the lungs and the gastrointestinal tract is assumed to be rapidly absorbed in body fluids (ICRP, 1979; ICRP 1989). This assumption is based on experiments with mice, rats, dogs and sheep (Willard and Bair, 1961; Bair et al., 1963). In all age groups of humans, with the exception of newborn infants up to 5 days, the fraction transferred to the thyroid is 0.3. Fractional uptake by the thyroid gland of newborns is about 0.7, declining to values at or below that observed for adults by five days after birth (ICRP, 1989). Iodine that does not reach the thyroid is assumed to go directly to excretion (ICRP, 1979).

The thyroid secretes organic iodine in the form of hormones into the blood. The iodine is metabolized in tissues and returned to the plasma pool as inorganic iodide; therefore, some iodine will be recycled to the thyroid. About 20 percent of the organic iodide is excreted in the feces. Most of the inorganic iodine is excreted in the urine (ICRP, 1989).

The turnover rate (rate of depletion) of natural iodine in the thyroid gland and the rest of the body decreases with increasing age as shown in Table 5-2. These biokinetic data are used in the ICRP model (ICRP, 1989) to calculate age-dependent internal doses from radioiodine. Also shown in Table 5-2 are standardized masses of the thyroid and body for various age groups.

Radiation Carcinogenesis of the Thyroid Gland (BEIR, 1990)

The possible radiological damage to the thyroid gland from human exposure to radioiodine can manifest itself in the form of immediate physical damage (cell death from very high doses) or an increased incidence of cancer. Doses high enough to cause immediate physical damage would rarely be encountered except in the intentional administration of radioiodine by medical practitioners for therapeutic purposes. Of more concern are radiogenic thyroid abnormalities that might be associated with unusual environmental exposure from inhalation and ingestion of food and water.

Table 5-2. Biokinetic data and standard organ masses for radioiodine dose calculations

Age	Fractional Uptake by Thyroid	Biological Half-time, days		Apparent Half-time, days	Standard Mass, g	
		Thyroid	Body	Thyroid	Thyroid	Body
3 mos	0.3	11.2	1.12	15	1.29	6,000
1 yr	0.3	15.0	1.50	20	1.78	9,800
5 yrs	0.3	23.0	2.30	30	3.45	19,000
10 yrs	0.3	58.0	5.80	70	7.93	32,000
15 yrs	0.3	67.0	6.70	80	12.40	55,000
Adult	0.3	80.0	12.00	91	20.00	70,000

An increased incidence of thyroid cancer from ionizing radiation has been observed in a number of groups including infants exposed to therapeutic x-rays, Japanese A-bomb survivors, and Marshall Islanders exposed to radioactive fallout from nuclear weapons tests. Studies of these groups indicate that the greatest susceptibility to radiation-induced thyroid cancer occurs during infancy and early childhood — especially during the first five years. In those individuals exposed before puberty, the tumors usually do not become apparent until after sexual maturation. The risk of radiation-induced cancer in adults is only one-half, or less, of that in children. Females are roughly three times more sensitive than males to radiogenic thyroid cancer.

Diagnostic procedures to evaluate thyroid function commonly use ^{131}I. In one large study of 35,000 people undergoing such tests, the mean thyroid dose was 50 rad. The results of the study suggest that diagnostic doses of ^{131}I do not significantly increase the risk of thyroid cancer.

Accepted criteria for the definitions of various types of thyroid neoplasms have been developed and promulgated by the World Health Organization. This classification system divides thyroid cancers into follicular, papillary, squamous cell, undifferentiated (anaplastic), and medullary types. Medical studies have indicated that radiogenic thyroid cancers are generally papillary growths; relatively few are follicular or mixed pathology. Radiogenic thyroid cancer causes death in only about 10 percent of all cases. The more malignant types of thyroid cancers have not been found to have a radiogenic origin.

Iodine Dosimetry

The inhalation and ingestion of all common compounds of radioiodine results in 100 percent transfer into the blood. Approximately 30 percent of the intake translocates to the thyroid, except in newborn infants. The ICRP (ICRP, 1989) has adopted a three-compartment model to describe the biokinetics of iodine after entry into the blood. The compartments are (1) blood, (2) thyroid, and (3) rest of the body.

This model allows for recycling of radioiodine released from the thyroid in the form of organic iodine compounds (hormones). The consideration of recycling is only necessary for long-lived isotopes of iodine. Because iodine is recycled through the thyroid, monitoring the gland after an intake of radioiodine will not reveal a single exponential clearance, but would be approximately described by the sum of two exponentials

$$R(t) = Ae^{-(0.693t/T1)} + (1-A)e^{-(0.693t/T2)}$$

where

$R(t)$ = fractional retention in the thyroid gland as a function of time, t

A = initial uptake by the thyroid gland

T1 and T2 = observed half-times for clearance

Retention data are largely determined using ^{131}I whose 8-day radiological half-life is too short to observe a two-exponential clearance. For dosimetric purposes, a single mean apparent half-time of 91 days for stable iodine in adults has been adopted (ICRP 1989) to replace the two-exponential biological half-times. Values for other age groups are: 3 months - 15 days, 1 year - 20 days, 5 years - 30 days, 10 years - 70 days, and 15 years - 80 days. These apparent half-times were used to calculate the biological half-times shown in Table 5-2. In previous ICRP internal dose calculations (ICRP, 1979), a biological half-time of 120 days was assumed for the adult thyroid.

The newer age-specific internal dose factors in ICRP 1989 represent dose commitments from the age of intake until age 70. Thus, the integrating period varies with each age group. However, all radioiodine isotopes are effectively eliminated from the body in periods ranging from a few days to a few years after intake.

The iodine dose conversion factors used in the late 1980s by DOE were calculated with 1979 ICRP methodology. A comparison of those factors with the values calculated using 1989 ICRP methodology is shown in Table 5-3.

The ICRP 1989 internal dose factors for adults in Table 5-3 are lower than corresponding DOE 1988 dose factors. This difference is the result of the shorter biological half-life used in ICRP 1989. The differences are small for isotopes with short radiological half-lives (^{131}I and ^{132}I), but more significant for the long-lived ^{129}I.

ICRP 1989 limited its consideration of age-dependent dose factors to the most radiologically significant isotopes that might be released to the environment due to various human activities. Thus, age-dependent ICRP 1989 dose factors were only provided for three radioisotopes of iodine: ^{129}I, ^{131}I, and ^{132}I. The DOE dose factors include all iodine isotopes with radiological half-lives long enough to result in a significant internal dose; DOE dose factors are limited to adults.

Table 5-3. Internal dose factors for iodine isotopes

Isotope	Radio-logical Half-life	Age Group	Intake Mode	Dose Factor, mrem/pCi				Reference
				EDE*	Stomach Wall	Thyroid	Lungs	
^{125}I	60.14 d	adult	ingestion	3.8×10^{-05}	–	1.3×10^{-03}	–	DOE 1988
		adult	inhalation	2.4×10^{-05}	–	8.1×10^{-04}	–	DOE 1988
^{126}I	13.02 d	adult	ingestion	7.1×10^{-05}	–	2.4×10^{-03}	–	DOE 1988
		adult	inhalation	4.3×10^{-05}	–	1.4×10^{-03}	–	DOE 1988
^{128}I	24.99m	adult	ingestion	8.5×10^{-08}	1.2×10^{-06}	4.1×10^{-04}	–	DOE 1988
		adult	inhalation	4.5×10^{-08}	1.2×10^{-07}	2.0×10^{-07}	2.7×10^{-07}	DOE 1988
^{129}I	1.57×10^{7} y	3 mos	ingestion	4.1×10^{-04}	4.4×10^{-06}	1.4×10^{-02}	–	ICRP 1989
			inhalation	2.6×10^{-04}	1.1×10^{-06}	8.5×10^{-03}	8.1×10^{-07}	ICRP 1989
		1 yr	ingestion	4.8×10^{-04}	2.6×10^{-06}	1.6×10^{-02}	–	ICRP 1989
			inhalation	3.0×10^{-04}	6.7×10^{-07}	1.0×10^{-02}	1.3×10^{-06}	ICRP 1989
		5 yrs	ingestion	3.7×10^{-04}	1.4×10^{-06}	1.3×10^{-02}	–	ICRP 1989
			inhalation	2.4×10^{-04}	4.1×10^{-07}	8.1×10^{-03}	7.8×10^{-07}	ICRP 1989
		10 yrs	ingestion	4.1×10^{-04}	1.0×10^{-06}	1.4×10^{-02}	–	ICRP 1989
			inhalation	2.7×10^{-04}	3.7×10^{-07}	8.9×10^{-03}	8.5×10^{-07}	ICRP 1989
		15 yrs	ingestion	3.1×10^{-04}	7.0×10^{-07}	1.0×10^{-02}	–	ICRP 1989
			inhalation	2.0×10^{-04}	2.6×10^{-07}	6.7×10^{-03}	4.8×10^{-07}	ICRP 1989
		adult	ingestion	2.4×10^{-04}	7.4×10^{-07}	7.8×10^{-03}	–	ICRP 1989
			ingestion	2.8×10^{-04}	–	9.3×10^{-03}	–	DOE 1988
			inhalation	1.5×10^{-04}	3.1×10^{-07}	4.8×10^{-03}	5.9×10^{-07}	ICRP 1989
			inhalation	1.8×10^{-04}	–	5.9×10^{-03}	–	DOE 1988
^{130}I	12.36 h	adult	ingestion	4.3×10^{-06}	–	1.4×10^{-04}	–	DOE 1988
			inhalation	2.5×10^{-06}	–	7.4×10^{-05}	2.2×10^{-06}	DOE 1988
^{131}I	8.04 d	3 mos	ingestion	4.1×10^{-04}	1.3×10^{-05}	1.4×10^{-02}	–	ICRP 1989
			inhalation	2.6×10^{-04}	2.8×10^{-06}	8.5×10^{-03}	3.3×10^{-06}	ICRP 1989
		1 yr	ingestion	4.1×10^{-04}	7.4×10^{-06}	1.3×10^{-02}	–	ICRP 1989
			inhalation	2.5×10^{-04}	1.7×10^{-06}	8.1×10^{-03}	2.4×10^{-06}	ICRP 1989
		5 yrs	ingestion	2.3×10^{-04}	3.6×10^{-06}	7.8×10^{-03}	–	ICRP 1989
			inhalation	1.4×10^{-04}	8.9×10^{-07}	4.8×10^{-03}	1.4×10^{-06}	ICRP 1989
		10 yrs	ingestion	1.2×10^{-04}	2.1×10^{-06}	4.1×10^{-03}	–	ICRP 1989
			inhalation	7.4×10^{-05}	5.2×10^{-07}	2.4×10^{-03}	8.9×10^{-07}	ICRP 1989
		15 yrs	ingestion	7.8×10^{-05}	1.4×10^{-06}	2.6×10^{-03}	–	ICRP 1989
			inhalation	4.8×10^{-05}	3.4×10^{-07}	1.6×10^{-03}	5.2×10^{-07}	ICRP 1989
		adult	ingestion	4.8×10^{-05}	1.1×10^{-06}	1.6×10^{-03}	–	ICRP 1989
			ingestion	5.3×10^{-05}	–	1.8×10^{-03}	–	DOE 1988
			inhalation	3.0×10^{-05}	2.6×10^{-07}	1.0×10^{-03}	4.4×10^{-07}	ICRP 1989
			inhalation	3.2×10^{-05}	–	1.1×10^{-03}	–	DOE 1988

*EDE = effective dose equivalent.

Table 5-3. **Internal dose factors for iodine isotopes (continued)**

Isotope	Radio-logical Half-life	Age Group	Intake Mode	Dose Factor, mrem/pCi EDE*	Stomach Wall	Thyroid	Lungs	Reference
^{132}I	2.3 h	3 mos	ingestion	7.0×10^{-06}	2.5×10^{-05}	1.5×10^{-04}	--	ICRP 1989
			inhalation	3.7×10^{-06}	3.7×10^{-06}	6.7×10^{-05}	1.1×10^{-06}	ICRP 1989
		1 yr	ingestion	5.6×10^{-06}	1.4×10^{-05}	1.3×10^{-04}	--	ICRP 1989
			inhalation	3.0×10^{-06}	2.1×10^{-06}	5.9×10^{-05}	7.4×10^{-07}	ICRP 1989
		5 yrs	ingestion	2.8×10^{-06}	7.0×10^{-06}	7.0×10^{-05}	--	ICRP 1989
			inhalation	1.6×10^{-06}	1.1×10^{-06}	3.2×10^{-05}	3.7×10^{-07}	ICRP 1989
		10 yrs	ingestion	1.4×10^{-06}	4.1×10^{-06}	3.1×10^{-05}	--	ICRP 1989
			inhalation	8.1×10^{-07}	6.7×10^{-07}	1.4×10^{-05}	2.5×10^{-07}	ICRP 1989
		15 yrs	ingestion	9.3×10^{-07}	3.0×10^{-06}	2.1×10^{-05}	--	ICRP 1989
			inhalation	5.6×10^{-07}	4.4×10^{-07}	9.3×10^{-06}	1.6×10^{-07}	ICRP 1989
		adult	ingestion	6.3×10^{-07}	2.3×10^{-06}	1.3×10^{-05}	--	ICRP 1989
			ingestion	1.0×10^{-06}	2.3×10^{-06}	1.4×10^{-05}	--	DOE 1988
			inhalation	3.7×10^{-07}	3.6×10^{-07}	5.9×10^{-06}	1.3×10^{-07}	ICRP 1989
			inhalation	3.3×10^{-07}	3.7×10^{-07}	6.3×10^{-06}	1.0×10^{-06}	DOE 1988
^{133}I	20.8 h	adult	ingestion	1.0×10^{-05}	--	3.4×10^{-04}	--	DOE 1988
			inhalation	5.4×10^{-06}	--	1.8×10^{-04}	--	DOE 1988
^{134}I	52.6 m	adult	ingestion	1.9×10^{-07}	2.0×10^{-06}	2.3×10^{-06}	--	DOE 1988
			inhalation	1.1×10^{-07}	2.6×10^{-07}	1.1×10^{-06}	5.2×10^{-07}	DOE 1988
^{135}I	6.61 h	adult	ingestion	2.0×10^{-06}	--	6.7×10^{-05}	--	DOE 1988
			inhalation	1.1×10^{-06}	--	3.1×10^{-05}	1.6×10^{-06}	DOE 1988

*EDE = effective dose equivalent

The ICRP 1989 inhalation dose factors shown in Table 5-3 were calculated from the lung model of ICRP 1979. The lung model does not make any allowance for age-dependent changes in deposition or clearance of inhaled material. Therefore, the age-dependent inhalation dose factors in the table are considered to be interim values. An age-dependent lung model for inhalation doses being developed by ICRP was not yet available for use in this report.

To calculate the committed dose equivalent to specific organs and the committed effective dose equivalent (CEDE) to an individual, SRS dose codes access dose factor libraries. The libraries contain radionuclide-specific and exposure pathway-specific factors that express the 50-year committed dose that would result from the intake of a unit quantity of radioactivity.

Doses to the offsite population from SRS radioiodine releases, as presented later in this chapter, were calculated using both the DOE 1988 and the ICRP 1989 dose conversion factors (Table 5-3). The discussion of the results, however, focuses on the dose estimates generated with the DOE dose factors because they are generally higher and, therefore, more conservative.

Models of Radioiodine Transport and Dose

Concentrations of SRS radioiodine in the offsite environment as of the late 1980s are so low that they are not detectable by conventional monitoring techniques. Therefore, radiation doses to offsite individuals and populations are calculated with mathematical models. These models use known transport mechanisms for atmospheric and liquid releases and known major pathways of exposure to humans. Modeled atmospheric and aqueous dispersion are periodically verified using environmental measurements of tritium released during normal SRS operations.

Calculational Models

The first models used at SRS for calculating offsite doses were developed at SRL (Cooper, 1975). These models, MREM (atmospheric releases) and RIVDOSE (liquid releases), were first used in 1972. In 1982 MREM and RIVDOSE were replaced with more technologically advanced models.

SRS annual offsite doses were calculated with the transport and dose models developed for the commercial nuclear industry (NRC, 1977a; NRC, 1977b). The models were implemented at SRS in the following computer programs:

Atmospheric Releases

- MAXIGASP: calculates maximum and average doses to offsite individuals
- POPGASP: calculates offsite population doses

Liquid Releases

- LADTAP II: calculates maximum and average doses to offsite individuals as well as doses to offsite populations

MAXIGASP and POPGASP are SRL-modified versions of the Nuclear Regulatory Commission (NRC) programs XOQDOQ (Sagendorf et al., 1982) and GASPAR (Eckerman et al., 1980). The modifications were made to meet the requirements for input of physical and biological data which are specific to the SRS. The basic calculations in the XOQDOQ and GASPAR programs have not been modified. LADTAP II (Simpson and McGill, 1980) is an essentially unaltered version of the NRC code of the same name.

Modeling Atmospheric Dispersion of Radioactive Releases

The routine atmospheric transport of radioactive materials from SRS is evaluated on the basis of meteorological conditions measured continuously at eight meteorological towers, one of which is offsite. The towers relay wind speed and direction and atmospheric stability information at five-second intervals to SRL through the WIND (Weather Information and Display) system. A data base of this information containing the 60-minute average values for the period 1982 - 1986 is accessed by the dispersion code to determine site-specific atmospheric dispersion characteristics.

The dispersion of an atmospheric release from SRS is modeled using XOQDOQ, which computes concentrations in the plume as a function of downwind distance and compass sector. At the user's option, the plume can be depleted due to dry deposition or radioactive decay. Also at the user's option, plume concentrations can be reduced by taking into account the upward displacement of the plume resulting from thermal buoyancy or momentum effects. This option is not used at SRS—no credit is taken for plume rise.

The plume concentration information generated by XOQDOQ is then used by the dose modeling program GASPAR to estimate doses to offsite individuals and populations.

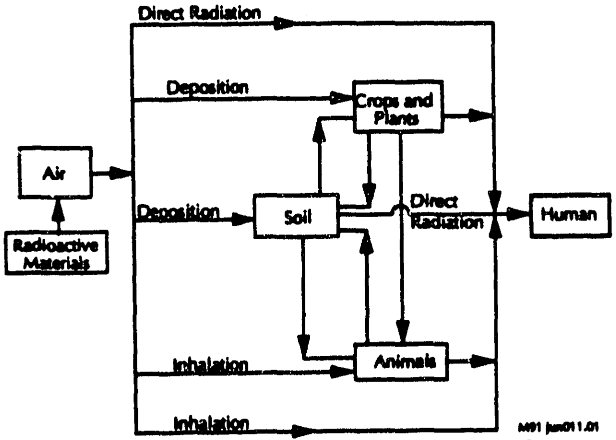

Figure 5–2. Simplified Pathways to Humans from Radioactive Materials Released to the Atmosphere

GASPAR estimates doses from a number of pathways which are illustrated in a general sense in Figure 5-2.

The doses estimated by GASPAR are reported on a pathway-specific basis as follows:

- Plume - external dose from radioactive materials suspended in the atmosphere
- Ground - external dose from radioactive materials deposited on the ground
- Inhalation - internal dose from inhalation of radioactive materials present in the plume
- Vegetation - internal dose from consumption of contaminated crops
- Milk - internal dose from consumption of milk produced in a contaminated area
- Meat - internal dose from consumption of meat produced in a contaminated area

The method of modeling the concentration of radioactivity as it moves through these pathways is fairly consistent from radionuclide to radionuclide. There is, however, a noteworthy modification made for radioiodine. For dose purposes it is assumed that all of the radioiodine deposited on vegetation is retained by the vegetation. This differs from the retention fraction of 0.2 that is used for most other radionuclides.

MAXIGASP. The calculations required by XOQDOQ and GASPAR to estimate maximum and average individual doses are performed at SRS using the computer program MAXIGASP. MAXIGASP calculates annual average ground-level air concentrations and 50-yr committed doses at a number of points along the site boundary in each of 16 compass sectors.

The main outputs from the program are the maximum dose equivalents to an individual along the SRS perimeter. The maximally-exposed individual is assumed to reside continuously at the location of highest exposure and to have living and eating habits that maximize his dose. These assumptions provide a ceiling on doses from atmospheric releases as no such individual is believed to exist.

The parameters used to calculate doses with MAXIGASP are presented in Table 5-4.

POPGASP. The calculations required by XOQDOQ and GASPAR to estimate population doses from atmospheric releases are performed at SRS using POPGASP. POPGASP calculates annual average ground-level air concentrations and annual doses for each of 160 regions (16 wind direction sectors at 10 distances per sector) within an 80-kilometer radius of the release location.

In addition to compass sector-specific meteorological information, POPGASP uses sector-specific data on population distribution and composition. Comparable data on milk, meat, and vegetable production and consumption are also used in the code. These data bases are quite extensive and are available for review in the SRS environmental reports published annually. With respect to the human parameters used in POPGASP, the key values are shown in Table 5-4.

Table 5-4. Site- and age-specific parameters for atmospheric releases

		Age Distribution, %		
Population Group	**Size**	**Children**	**Teen**	**Adult**
80-km radius	555,100	18.6	11.1	70.3
	Maximum Individual (MAXIGASP)			
Exposure Pathway	**Infant**	**Child**	**Teen**	**Adult**
Inhalation (m^3/yr)	1,400	3,700	8,000	8,000
Ingestion				
Cow's Milk (L/yr)	330	330	400	310
Meat (kg/yr)	0	41	65	110
Leafy vegetables (kg/yr)	0	26	42	64
Fruits, grains and other vegetables (kg/yr)	0	520	630	520
External exposure Transmission factor for shielding from buildings	0.7	0.7	0.7	0.7
	General Population (POPGASP)			
Exposure Pathway	**Infant**	**Child**	**Teen**	**Adult**
Inhalation (m^3/yr)	1,400	3,700	8,000	8,000
Ingestion				
Cow's Milk (L/yr)	170	170	200	110
Meat (kg/yr)	0	37	59	95
Leafy vegetables (kg/yr)	0	10	20	30
Fruits, grains and other vegetables (kg/yr)	0	200	240	190
External exposure Transmission factor for shielding from buildings	0.5	0.5	0.5	0.5

Modeling Doses from Liquid Releases

The consequences of liquid releases from SRS are modeled using LADTAP II. The potential pathways of exposure from liquid releases to the environment are shown in Figure 5-3. The pathway-specific doses calculated by LADTAP are grouped into the following five categories:

- Potable drinking water - internal dose from consuming drinking water of Savannah River origin
- Sports fish and commercial fish - internal dose from consuming fish of Savannah River origin
- Salt water invertebrates - internal dose from consuming shell fish from estuaries of the Savannah River
- Recreation - external dose from recreation activities in and along the Savannah River (boating, swimming, and shoreline activities)

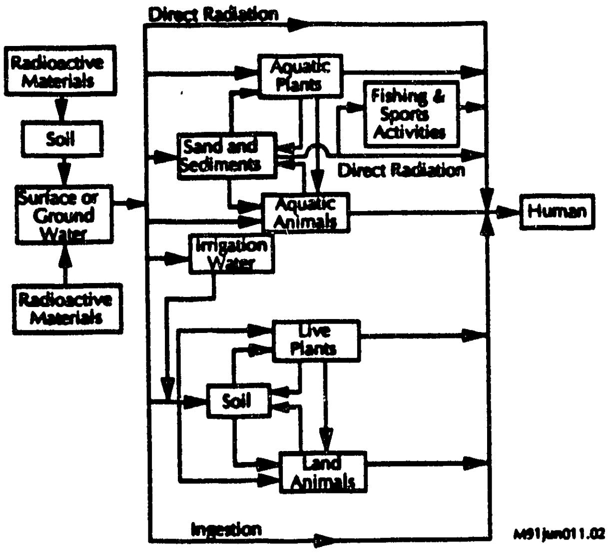

Figure 5–3. Simplified Pathways to Humans from Radioactive Materials Released to Surface or Ground Waters

- Irrigation - internal dose from foods produced by irrigation with Savannah River water (there are no known users of the river for this purpose)

LADTAP estimates individual and population doses at specific downstream locations. The only removal mechanism included in the transport model as it is used at SRS is radioactive decay. No credit is taken for adsorption of radioiodine on stream sediments.

The major assumption inherent in the application of LADTAP to SRS releases is that liquid discharges undergo complete mixing in the Savannah River before reaching potentially exposed populations. This assumption is supported by repeated tritium measurements indicating that complete mixing occurs in the river between the SRS and the Highway 301 sampling station (Cummins et al., 1990a).

LADTAP II generates maximum individual and population doses for all of the exposure pathways identified previously. Though standard input values are provided in LADTAP II, SRS calculations are performed with site-specific information to the extent that it is available. Summary tables of principal input values used in the SRS version of LADTAP II are shown in Table 5-5.

Table 5-5. Site- and age-specific parameters for liquid releases

Maximum Individual Dose Assessments (LADTAP II)

Site Parameters	
Savannah River flow rate, m^3/s	measured average or 170
Transit time from SRS to Savannah River, hr	24
Shore width factor	0.2

	Maximum Individual(a)			
Human Parameters	**Infant**	**Child**	**Teen**	**Adult**
Water consumption (L/yr)	260 (330)	260 (510)	260 (510)	370 (730)
Fish consumption (kg/yr)	0	11.2	25.9	34
Shell fish consumption (kg/yr)	0	1.7	3.8	5
Recreation (person-hr/yr)				
Shoreline activities	0	14	67	20
Swimming	0	10	10	10
Boating	0	60	60	60

(a) The values in parentheses are EPA parameters mandated for use when calculating maximum individual doses to Beaufort-Jasper and Port Wentworth water users.

Table 5-5. Site- and age-specific parameters for liquid releases (continued)

Population Dose Assessments (LADTAP II)

Site Parameters	
Savannah River flow rate, m^3/s	measured average or 280
Transit time from SRS to Savannah River, hr	24
Transit time from SRS to water treatment plant, hr	72
Retention time in water treatment system, hr	24
Shore width factor	0.2
River dilution in estuary	3
Aquatic food harvest (kg/yr)	
Sport fish	103,700
Commercial fish	31,800
Salt water invertebrates	299,400

		Age Distribution, %		
Population Group	Size	Child	Teen	Adult
80-km radius	555,100	18.6	11.1	70.3
Beaufort-Jasper water users	51,000	21.0	10.0	69.0
Port Wentworth water users	20,000	0.0	0.0	100.0

	Average Individual			
Human Parameters	Infant	Child	Teen	Adult
Water consumption (L/yr)	260	260	260	370
Fish consumption (kg/yr)	0	3.6	8.5	11.3
Shell fish consumption (kg/yr)	0	0.33	0.75	1.0

Human Parameters	80-km Population
Recreation (person-hr/yr)	
Shoreline activities	108,400
Swimming	8,465
Boating	232,000

Table 5-6. Maximum individual doses from atmospheric releases of ^{129}I (1955 - 1989)

Year	Releases (curies)	ICRP 30 Adult Dose (mrem)		ICRP 56 Adult Dose (mrem)		ICRP 56 Teen Dose (mrem)		ICRP 56 Child Dose (mrem)		ICRP 56 Infant Dose (mrem)	
		EDE	Thyroid	EDE	Thyroid	EDE	Thyroid	EDE	Thyroid	EDE	Thyroid
1955	2.12×10^{-01}	2.18×10^{-01}	7.21	1.87×10^{-01}	6.04	2.80×10^{-01}	8.95	2.69×10^{-01}	9.43	5.58×10^{-02}	1.83
1956	2.12×10^{-01}	2.18×10^{-01}	7.21	1.87×10^{-01}	6.04	2.80×10^{-01}	8.95	2.69×10^{-01}	9.43	5.58×10^{-02}	1.83
1957	2.12×10^{-01}	2.18×10^{-01}	7.21	1.87×10^{-01}	6.04	2.80×10^{-01}	8.95	2.69×10^{-01}	9.43	5.58×10^{-02}	1.83
1958	2.12×10^{-01}	2.18×10^{-01}	7.21	1.87×10^{-01}	6.04	2.80×10^{-01}	8.95	2.69×10^{-01}	9.43	5.58×10^{-02}	1.83
1959	2.12×10^{-01}	2.18×10^{-01}	7.21	1.87×10^{-01}	6.04	2.80×10^{-01}	8.95	2.69×10^{-01}	9.43	5.58×10^{-02}	1.83
1960	2.12×10^{-01}	2.18×10^{-01}	7.21	1.87×10^{-01}	6.04	2.80×10^{-01}	8.95	2.69×10^{-01}	9.43	5.58×10^{-02}	1.83
1961	2.12×10^{-01}	2.18×10^{-01}	7.21	1.87×10^{-01}	6.04	2.80×10^{-01}	8.95	2.69×10^{-01}	9.43	5.58×10^{-02}	1.83
1962	2.12×10^{-01}	2.18×10^{-01}	7.21	1.87×10^{-01}	6.04	2.80×10^{-01}	8.95	2.69×10^{-01}	9.43	5.58×10^{-02}	1.83
1963	2.12×10^{-01}	2.18×10^{-01}	7.21	1.87×10^{-01}	6.04	2.80×10^{-01}	8.95	2.69×10^{-01}	9.43	5.58×10^{-02}	1.83
1964	2.12×10^{-01}	2.18×10^{-01}	7.21	1.87×10^{-01}	6.04	2.80×10^{-01}	8.95	2.69×10^{-01}	9.43	5.58×10^{-02}	1.83
1965	2.12×10^{-01}	2.18×10^{-01}	7.21	1.87×10^{-01}	6.04	2.80×10^{-01}	8.95	2.69×10^{-01}	9.43	5.58×10^{-02}	1.83
1966	2.12×10^{-01}	2.18×10^{-01}	7.21	1.87×10^{-01}	6.04	2.80×10^{-01}	8.95	2.69×10^{-01}	9.43	5.58×10^{-02}	1.83
1967	2.12×10^{-01}	2.18×10^{-01}	7.21	1.87×10^{-01}	6.04	2.80×10^{-01}	8.95	2.69×10^{-01}	9.43	5.58×10^{-02}	1.83
1968	2.12×10^{-01}	2.18×10^{-01}	7.21	1.87×10^{-01}	6.04	2.80×10^{-01}	8.95	2.69×10^{-01}	9.43	5.58×10^{-02}	1.83
1969	2.12×10^{-01}	2.18×10^{-01}	7.21	1.87×10^{-01}	6.04	2.80×10^{-01}	8.95	2.69×10^{-01}	9.43	5.58×10^{-02}	1.83
1970	2.12×10^{-01}	2.18×10^{-01}	7.21	1.87×10^{-01}	6.04	2.80×10^{-01}	8.95	2.69×10^{-01}	9.43	5.58×10^{-02}	1.83
1971	2.12×10^{-01}	2.18×10^{-01}	7.21	1.87×10^{-01}	6.04	2.80×10^{-01}	8.95	2.69×10^{-01}	9.43	5.58×10^{-02}	1.83
1972	2.12×10^{-01}	2.18×10^{-01}	7.21	1.87×10^{-01}	6.04	2.80×10^{-01}	8.95	2.69×10^{-01}	9.43	5.58×10^{-02}	1.83
1973	2.12×10^{-01}	2.18×10^{-01}	7.21	1.87×10^{-01}	6.[illegible]	2.80×10^{-01}	8.95	2.69×10^{-01}	9.43	5.58×10^{-02}	1.83
1974	1.70×10^{-01}	1.75×10^{-01}	5.78	1.50×10^{-01}	4.85	2.24×10^{-01}	7.17	2.16×10^{-01}	7.57	4.47×10^{-02}	1.47
1975	1.40×10^{-01}	1.44×10^{-01}	4.76	1.23×10^{-01}	3.99	1.85×10^{-01}	5.91	1.78×10^{-01}	6.23	3.68×10^{-02}	1.21
1976	1.50×10^{-01}	1.55×10^{-01}	5.10	1.32×10^{-01}	4.28	1.98×10^{-01}	6.33	1.91×10^{-01}	6.68	3.95×10^{-02}	1.30
1977	1.40×10^{-01}	1.44×10^{-01}	4.76	1.23×10^{-01}	3.99	1.85×10^{-01}	5.91	1.78×10^{-01}	6.23	3.68×10^{-02}	1.21
1978	1.30×10^{-01}	1.34×10^{-01}	4.42	1.14×10^{-01}	3.71	1.72×10^{-01}	5.49	1.65×10^{-01}	5.79	3.42×10^{-02}	1.12
1979	1.30×10^{-01}	1.34×10^{-01}	4.42	1.14×10^{-01}	3.71	1.72×10^{-01}	5.49	1.65×10^{-01}	5.79	3.42×10^{-02}	1.12
1980	1.60×10^{-01}	1.65×10^{-01}	5.44	1.41×10^{-01}	4.56	2.11×10^{-01}	6.75	2.03×10^{-01}	7.12	4.21×10^{-02}	1.38
1981	1.60×10^{-01}	1.65×10^{-01}	5.44	1.41×10^{-01}	4.56	2.11×10^{-01}	6.75	2.03×10^{-01}	7.12	4.21×10^{-02}	1.38
1982	5.90×10^{-02}	6.08×10^{-02}	2.01	5.19×10^{-02}	1.68	7.79×10^{-02}	2.49	7.49×10^{-02}	2.63	1.55×10^{-02}	0.51
1983	4.10×10^{-02}	4.22×10^{-02}	1.39	3.61×10^{-02}	1.17	5.41×10^{-02}	1.73	5.21×10^{-02}	1.82	1.08×10^{-02}	0.35
1984	3.54×10^{-02}	3.65×10^{-02}	1.20	3.12×10^{-02}	1.01	4.67×10^{-02}	1.49	4.50×10^{-02}	1.58	9.31×10^{-03}	0.31
1985	6.50×10^{-02}	6.70×10^{-02}	2.21	5.72×10^{-02}	1.85	8.58×10^{-02}	2.74	8.26×10^{-02}	2.89	1.71×10^{-02}	0.56
1986	8.70×10^{-02}	8.96×10^{-02}	2.96	7.66×10^{-02}	2.48	1.15×10^{-01}	3.67	1.10×10^{-01}	3.87	2.29×10^{-02}	0.75
1987	7.20×10^{-02}	7.42×10^{-02}	2.45	6.34×10^{-02}	2.05	9.50×10^{-02}	3.04	9.14×10^{-02}	3.20	1.89×10^{-02}	0.62
1988	6.30×10^{-02}	6.49×10^{-02}	2.14	5.54×10^{-02}	1.80	8.32×10^{-02}	2.66	8.00×10^{-02}	2.80	1.66×10^{-02}	0.54
1989	5.19×10^{-02}	5.35×10^{-02}	1.76	4.57×10^{-02}	1.48	6.85×10^{-02}	2.19	6.59×10^{-02}	2.31	1.36×10^{-02}	0.45
Total	5.67	5.85	1.94×10^{2}	5.00	1.62×10^{2}	N/A		N/A		N/A	

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Doses for 1955 through 1973 reflect annual average releases for this period.
EDE = effective dose equivalent
N/A reflects the fact that an individual could not have remained in this age group throughout the 35-year period studied.

Table 5-7. Maximum individual doses from atmospheric releases of ^{131}I (1955 - 1989)

Year	Releases (curies)	ICRP 30 Adult Dose (mrem)		ICRP 56 Adult Dose (mrem)		ICRP 56 Teen Dose (mrem)		ICRP 56 Child Dose (mrem)		ICRP 56 Infant Dose (mrem)	
		EDE	Thyroid	EDE	Thyroid	EDE	Thyroid	EDE	Thyroid	EDE	Thyroid
1955	6.92×10^{01}	7.13×10^{-01}	2.42×10^{01}	6.46×10^{-01}	2.15×10^{01}	9.34×10^{-01}	3.10×10^{01}	1.99	6.75×10^{01}	1.99	6.31×10^{01}
1956	1.58×10^{03}	1.63×10^{01}	5.51×10^{02}	1.47×10^{01}	4.89×10^{02}	2.13×10^{01}	7.06×10^{02}	4.55×10^{01}	1.54×10^{03}	4.54×10^{01}	1.44×10^{03}
1957	2.92×10^{02}	3.01	1.02×10^{02}	2.73	9.05×10^{01}	3.94	1.31×10^{02}	8.41	2.85×10^{02}	8.41	2.66×10^{02}
1958	1.99×10^{01}	2.05×10^{-01}	6.96	1.86×10^{-01}	6.18	2.69×10^{-01}	8.93	5.74×10^{-01}	1.95×10^{01}	5.74×10^{-01}	1.82×10^{01}
1959	1.62×10^{02}	1.67	5.66×10^{01}	1.51	5.03×10^{01}	2.19	7.26×10^{01}	4.67	1.58×10^{02}	4.67	1.48×10^{02}
1960	7.03	7.24×10^{-02}	2.45	6.57×10^{-02}	2.18	9.49×10^{-02}	3.15	2.02×10^{-01}	6.86	2.02×10^{-01}	6.41
1961	1.62×10^{02}	1.67	5.65×10^{01}	1.51	5.02×10^{01}	2.19	7.25×10^{01}	4.66	1.58×10^{02}	4.66	1.48×10^{02}
1962	1.66×10^{01}	1.71×10^{-01}	5.81	1.55×10^{-01}	5.16	2.25×10^{-01}	7.45	4.79×10^{-01}	1.62×10^{01}	4.79×10^{-01}	1.52×10^{01}
1963	4.82	4.97×10^{-02}	1.68	4.50×10^{-02}	1.49	6.51×10^{-02}	2.16	1.39×10^{-01}	4.71	1.39×10^{-01}	4.40
1964	1.16×10^{01}	1.20×10^{-01}	4.05	1.08×10^{-01}	3.60	1.57×10^{-01}	5.20	3.34×10^{-01}	1.13×10^{01}	3.34×10^{-01}	1.06×10^{01}
1965	1.81×10^{01}	1.86×10^{-01}	6.31	1.69×10^{-01}	5.60	2.44×10^{-01}	8.09	5.20×10^{-01}	1.76×10^{01}	5.20×10^{-01}	1.65×10^{01}
1966	3.16×10^{01}	3.26×10^{-01}	1.10×10^{01}	2.95×10^{-01}	9.80	4.27×10^{-01}	1.42×10^{01}	9.11×10^{-01}	3.09×10^{01}	9.11×10^{-01}	2.88×10^{01}
1967	2.01×10^{01}	2.07×10^{-01}	7.01	1.88×10^{-01}	6.23	2.71×10^{-01}	9.00	5.79×10^{-01}	1.96×10^{01}	5.79×10^{-01}	1.83×10^{01}
1968	2.21×10^{01}	2.28×10^{-01}	7.72	2.07×10^{-01}	6.86	2.99×10^{-01}	9.91	6.37×10^{-01}	2.16×10^{01}	6.37×10^{-01}	2.02×10^{01}
1969	3.60×10^{01}	3.70×10^{-01}	1.25×10^{01}	3.36×10^{-01}	1.11×10^{01}	4.85×10^{-01}	1.61×10^{01}	1.04	3.51×10^{01}	1.04	3.28×10^{01}
1970	3.44×10^{01}	3.54×10^{-01}	1.20×10^{01}	3.21×10^{-01}	1.07×10^{01}	4.64×10^{-01}	1.54×10^{01}	9.90×10^{-01}	3.35×10^{01}	9.90×10^{-01}	3.13×10^{01}
1971	2.66×10^{01}	2.74×10^{-01}	9.29	2.49×10^{-01}	8.25	3.59×10^{-01}	1.19×10^{01}	7.67×10^{-01}	2.60×10^{01}	7.67×10^{-01}	2.43×10^{01}
1972	2.74	2.82×10^{-02}	9.57×10^{-01}	2.56×10^{-02}	8.50×10^{-01}	3.70×10^{-02}	1.23	7.90×10^{-02}	2.68	7.90×10^{-02}	2.50
1973	1.86	1.91×10^{-02}	6.48×10^{-01}	1.73×10^{-02}	5.75×10^{-01}	2.51×10^{-02}	8.31×10^{-01}	5.34×10^{-02}	1.81	5.34×10^{-02}	1.69
1974	1.91	1.97×10^{-02}	6.68×10^{-01}	1.79×10^{-02}	5.93×10^{-01}	2.58×10^{-02}	8.57×10^{-01}	5.51×10^{-02}	1.87	5.51×10^{-02}	1.75
1975	1.23×10^{-01}	1.27×10^{-03}	4.31×10^{-02}	1.15×10^{-03}	3.83×10^{-02}	1.67×10^{-03}	5.53×10^{-02}	3.56×10^{-03}	1.21×10^{-01}	3.56×10^{-03}	1.13×10^{-01}
1976	1.55×10^{-01}	1.60×10^{-03}	5.43×10^{-02}	1.43×10^{-03}	4.82×10^{-02}	2.10×10^{-03}	6.96×10^{-02}	4.48×10^{-03}	1.52×10^{-01}	4.48×10^{-03}	1.42×10^{-01}
1977	6.09×10^{-02}	6.27×10^{-04}	2.12×10^{-02}	5.69×10^{-04}	1.89×10^{-02}	8.22×10^{-04}	2.73×10^{-02}	1.75×10^{-03}	5.94×10^{-02}	1.75×10^{-03}	5.55×10^{-02}
1978	6.54×10^{-02}	6.74×10^{-04}	2.28×10^{-02}	6.11×10^{-04}	2.03×10^{-02}	8.83×10^{-04}	2.93×10^{-02}	1.88×10^{-03}	6.39×10^{-02}	1.88×10^{-03}	5.97×10^{-02}
1979	8.43×10^{-02}	8.69×10^{-04}	2.94×10^{-02}	7.88×10^{-04}	2.61×10^{-02}	1.14×10^{-03}	3.78×10^{-02}	2.43×10^{-03}	8.23×10^{-02}	2.43×10^{-03}	7.69×10^{-02}
1980	2.51×10^{-02}	2.59×10^{-04}	8.77×10^{-03}	2.35×10^{-04}	7.79×10^{-03}	3.39×10^{-04}	1.13×10^{-02}	7.23×10^{-04}	2.45×10^{-02}	7.23×10^{-04}	2.29×10^{-02}
1981	4.70×10^{-02}	4.84×10^{-04}	1.64×10^{-02}	[illegible]	1.46×10^{-02}	6.35×10^{-04}	2.11×10^{-02}	1.35×10^{-03}	4.59×10^{-02}	1.35×10^{-03}	4.29×10^{-02}
1982	1.06×10^{-01}	1.09×10^{-03}	3.70×10^{-02}	[illegible]	3.29×10^{-02}	1.43×10^{-03}	4.75×10^{-02}	3.05×10^{-03}	1.03×10^{-01}	3.05×10^{-03}	9.67×10^{-02}
1983	8.41×10^{-02}	8.66×10^{-04}	2.93×10^{-02}	[illegible]	2.61×10^{-02}	1.13×10^{-03}	3.77×10^{-02}	2.42×10^{-03}	8.20×10^{-02}	2.42×10^{-03}	7.67×10^{-02}
1984	2.82×10^{-01}	2.91×10^{-03}	9.86×10^{-02}	[illegible]	8.75×10^{-02}	3.81×10^{-03}	1.27×10^{-01}	8.13×10^{-03}	2.76×10^{-01}	8.13×10^{-03}	2.58×10^{-01}
1985	6.05×10^{-02}	6.23×10^{-04}	2.11×10^{-02}	5.65×10^{-04}	1.87×10^{-02}	8.16×10^{-04}	2.71×10^{-02}	1.74×10^{-03}	5.90×10^{-02}	1.74×10^{-03}	5.51×10^{-02}
1986	2.64×10^{-02}	2.72×10^{-04}	9.21×10^{-03}	2.47×10^{-04}	8.18×10^{-03}	3.56×10^{-04}	1.18×10^{-02}	7.60×10^{-04}	2.58×10^{-02}	7.60×10^{-04}	2.41×10^{-02}
1987	1.26×10^{-02}	1.30×10^{-04}	4.40×10^{-03}	1.18×10^{-04}	3.91×10^{-03}	1.70×10^{-04}	5.64×10^{-03}	3.63×10^{-04}	1.23×10^{-02}	3.63×10^{-04}	1.15×10^{-02}
1988	4.96×10^{-03}	5.11×10^{-05}	1.73×10^{-03}	4.63×10^{-05}	1.54×10^{-03}	6.70×10^{-05}	2.22×10^{-03}	1.43×10^{-04}	4.84×10^{-03}	1.43×10^{-04}	4.52×10^{-03}
1989	3.64×10^{-03}*	3.75×10^{-05}	1.27×10^{-03}	3.40×10^{-05}	1.13×10^{-03}	4.91×10^{-05}	1.63×10^{-03}	1.05×10^{-04}	3.55×10^{-03}	1.05×10^{-04}	3.32×10^{-03}
Total	2.52×10^{03}	2.60×10^{01}	8.80×10^{02}	2.35×10^{01}	7.81×10^{02}	N/A		N/A		N/A	

EDE = *effective dose equivalent*

* The reported 1989 release was reevaluated to have been 3.46×10^{-04}.

N/A reflects the fact that an individual could not have remained in this age group throughout the 35-year period studied.

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Table 5-8. Population doses from atmospheric radioiodine releases (1955 - 1989)

Year	$^{129}I^{(a)}$ Releases (curies)	^{129}I Person-rem ICRP 30 EDE	^{129}I Person-rem ICRP 30 Thyroid	^{129}I Person-rem ICRP 56 EDE	^{129}I Person-rem ICRP 56 Thyroid	^{131}I Releases (curies)	^{131}I Person-rem ICRP 30 EDE	^{131}I Person-rem ICRP 30 Thyroid	^{131}I Person-rem ICRP 56 EDE	^{131}I Person-rem ICRP 56 Thyroid
1955	2.12×10^{-01}	7.72	2.48×10^{02}	7.55	2.44×10^{02}	6.92×10^{01}	4.71×10^{01}	1.59×10^{03}	7.68×10^{01}	2.58×10^{03}
1956	2.12×10^{-01}	7.72	2.48×10^{02}	7.55	2.44×10^{02}	1.58×10^{03}	1.07×10^{03}	3.62×10^{04}	1.75×10^{03}	5.88×10^{04}
1957	2.12×10^{-01}	7.72	2.48×10^{02}	7.55	2.44×10^{02}	2.92×10^{02}	1.99×10^{02}	6.71×10^{03}	3.24×10^{02}	1.09×10^{04}
1958	2.12×10^{-01}	7.72	2.48×10^{02}	7.55	2.44×10^{02}	1.99×10^{01}	1.36×10^{01}	4.58×10^{02}	2.21×10^{01}	7.43×10^{02}
1959	2.12×10^{-01}	7.72	2.48×10^{02}	7.55	2.44×10^{02}	1.62×10^{02}	1.10×10^{02}	3.73×10^{03}	1.80×10^{02}	6.05×10^{03}
1960	2.12×10^{-01}	7.72	2.48×10^{02}	7.55	2.44×10^{02}	7.03	4.78	1.62×10^{02}	7.80	2.62×10^{02}
1961	2.12×10^{-01}	7.72	2.48×10^{02}	7.55	2.44×10^{02}	1.62×10^{02}	1.10×10^{02}	3.72×10^{03}	1.80×10^{02}	6.04×10^{03}
1962	2.12×10^{-01}	7.72	2.48×10^{02}	7.55	2.44×10^{02}	1.66×10^{01}	1.13×10^{01}	3.82×10^{02}	1.85×10^{01}	6.21×10^{02}
1963	2.12×10^{-01}	7.72	2.48×10^{02}	7.55	2.44×10^{02}	4.82	3.28	1.11×10^{02}	5.35	1.80×10^{02}
1964	2.12×10^{-01}	7.72	2.48×10^{02}	7.55	2.44×10^{02}	1.16×10^{01}	7.89	2.67×10^{02}	1.29×10^{01}	4.33×10^{02}
1965	2.12×10^{-01}	7.72	2.48×10^{02}	7.55	2.44×10^{02}	1.81×10^{01}	1.23×10^{01}	4.15×10^{02}	2.01×10^{01}	6.74×10^{02}
1966	2.12×10^{-01}	7.72	2.48×10^{02}	7.55	2.44×10^{02}	3.16×10^{01}	2.15×10^{01}	7.27×10^{02}	3.51×10^{01}	1.18×10^{03}
1967	2.12×10^{-01}	7.72	2.48×10^{02}	7.55	2.44×10^{02}	2.01×10^{01}	1.37×10^{01}	4.62×10^{02}	2.23×10^{01}	7.50×10^{02}
1968	2.12×10^{-01}	7.72	2.48×10^{02}	7.55	2.44×10^{02}	2.21×10^{01}	1.50×10^{01}	5.08×10^{02}	2.46×10^{01}	8.25×10^{02}
1969	2.12×10^{-01}	7.72	2.48×10^{02}	7.55	2.44×10^{02}	3.60×10^{01}	2.44×10^{01}	8.26×10^{02}	3.99×10^{01}	1.34×10^{03}
1970	2.12×10^{-01}	7.72	2.48×10^{02}	7.55	2.44×10^{02}	3.44×10^{01}	2.34×10^{01}	7.90×10^{02}	3.81×10^{01}	1.28×10^{03}
1971	2.12×10^{-01}	7.72	2.48×10^{02}	7.55	2.44×10^{02}	2.66×10^{01}	1.81×10^{01}	6.12×10^{02}	2.95×10^{01}	9.93×10^{02}
1972	2.12×10^{-01}	7.72	2.48×10^{02}	7.55	2.44×10^{02}	2.74	1.86	6.30×10^{01}	3.04	1.02×10^{02}
1973	2.12×10^{-01}	7.72	2.48×10^{02}	7.55	2.44×10^{02}	1.86	1.26	4.26×10^{01}	2.06	6.92×10^{01}
1974	1.70×10^{-01}	6.19	1.99×10^{02}	6.05	1.96×10^{02}	1.91	1.30	4.40×10^{01}	2.12	7.14×10^{01}
1975	1.40×10^{-01}	5.10	1.64×10^{02}	4.98	1.61×10^{02}	1.23×10^{-01}	8.40×10^{-02}	2.84	1.37×10^{-01}	4.61
1976	1.50×10^{-01}	5.46	1.76×10^{02}	5.34	1.73×10^{02}	1.55×10^{-01}	1.06×10^{-01}	3.57	1.73×10^{-01}	5.80
1977	1.40×10^{-01}	5.10	1.64×10^{02}	4.98	1.61×10^{02}	6.09×10^{-02}	4.14×10^{-02}	1.40	6.76×10^{-02}	2.27
1978	1.30×10^{-01}	4.73	1.52×10^{02}	4.63	1.50×10^{02}	6.54×10^{-02}	4.45×10^{-02}	1.50	7.26×10^{-02}	2.44
1979	1.30×10^{-01}	4.73	1.52×10^{02}	4.63	1.50×10^{02}	8.43×10^{-02}	5.74×10^{-02}	1.94	9.36×10^{-02}	3.15
1980	1.60×10^{-01}	5.83	1.87×10^{02}	5.70	1.84×10^{02}	2.51×10^{-02}	1.71×10^{-02}	5.77×10^{-01}	2.79×10^{-02}	9.37×10^{-01}
1981	1.60×10^{-01}	5.83	1.87×10^{02}	5.70	1.84×10^{02}	4.70×10^{-02}	3.20×10^{-02}	1.08	5.22×10^{-02}	1.75
1982	5.90×10^{-02}	2.15	6.91×10^{01}	2.10	6.79×10^{01}	1.06×10^{-01}	7.21×10^{-02}	2.44	1.18×10^{-01}	3.95
1983	4.10×10^{-02}	1.49	4.80×10^{01}	1.46	4.72×10^{01}	8.41×10^{-02}	5.72×10^{-02}	1.93	9.33×10^{-02}	3.14
1984	3.54×10^{-02}	1.29	4.14×10^{01}	1.26	4.07×10^{01}	2.82×10^{-01}	1.92×10^{-01}	6.49	3.13×10^{-01}	1.05×10^{01}
1985	6.50×10^{-02}	2.37	7.61×10^{01}	2.31	7.48×10^{01}	6.05×10^{-02}	4.11×10^{-02}	1.39	6.71×10^{-02}	2.26
1986	8.70×10^{-02}	3.17	1.02×10^{02}	3.10	1.00×10^{02}	2.64×10^{-02}	1.80×10^{-02}	6.07×10^{-01}	2.93×10^{-02}	9.85×10^{-01}
1987	7.20×10^{-02}	2.62	8.43×10^{01}	2.56	8.28×10^{01}	1.26×10^{-02}	8.57×10^{-03}	2.90×10^{-01}	1.40×10^{-02}	4.70×10^{-01}
1988	6.30×10^{-02}	2.29	7.37×10^{01}	2.24	7.25×10^{01}	4.96×10^{-03}	3.37×10^{-03}	1.14×10^{-01}	5.51×10^{-03}	1.85×10^{-01}
1989	5.19×10^{-02}	1.89	6.07×10^{01}	1.85	5.97×10^{01}	3.64×10^{-04}(b)	2.48×10^{-04}	8.36×10^{-03}	4.04×10^{-04}	1.36×10^{-02}
Total	5.68	2.07×10^{02}	6.65×10^{03}	2.02×10^{02}	6.53×10^{03}	2.52×10^{03}	1.71×10^{03}	5.79×10^{04}	2.79×10^{03}	9.39×10^{04}

(a) Doses for 1955 through 1973 reflect annual average releases for this period.

(b) The reported 1989 release was reevaluated to have been 3.45×10^{-4}.

EDE= effective dose equivalent

Verification of Models Using Measurement Data

Models are verified with environmental measurements of tritium. Tritium is routinely detected offsite using conventional measuring techniques.

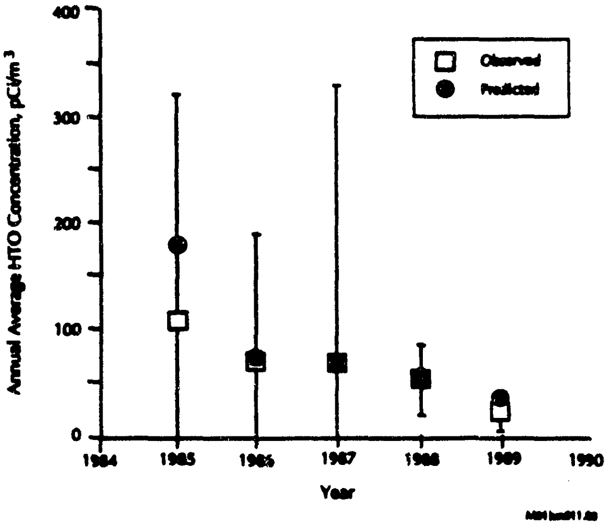

Figure 5–4. Comparison of Predicted and Measured Tritium Concentration in Air at the Site Perimeter (±2 Standard Deviations at the 95 Percent Confidence Level)

Atmospheric Releases

The radionuclide concentrations predicted by XOQDOQ are routinely compared with measured values of tritium concentrations in air to evaluate the performance of the code. Figure 5-4 shows a comparison of calculated and measured tritium oxide concentrations in air at the site perimeter. As seen in the figure, predicted values tend to exceed observed values, yet not to a degree that would indicate an excessively conservative approach.

Other comparisons of predicted and measured concentrations have been made (Marter, 1984) and have exhibited similar results. The available data suggest that calculated concentrations of tritium in air are generally conservative estimates of actual offsite values.

Liquid Releases

Throughout the period 1954-1989, drinking water doses from routine SRS releases have been dominated by tritium. Measured, rather than estimated, tritium concentrations are therefore used for the downriver locations that are most important in terms of dose calculations: (1) just below SRS, (2) at the Beaufort-Jasper water treatment plant, and (3) at the Port Wentworth water treatment plant. In addition to supplying measured tritium concentrations, such data also provide Savannah River dilution factors for these locations. The availability of accurate estimates of river dilution characteristics significantly enhances LADTAP's ability to predict downriver concentrations of other radionuclides such as radioiodine.

Impact of SRS Radioiodine Releases on the Offsite Population

The computer codes MAXIGASP, POPGASP, and LADTAP II are used to calculate effective and thyroid dose equivalents that have been theoretically received by segments of the offsite population from radioiodine (^{129}I, ^{131}I) exposure over the course of site operations. The results are shown are in tables 5-6 through 5-8 for doses attributable to atmospheric releases and tables 5-9 through 5-11 for doses from liquid releases to streams. Atmospheric radioiodine release levels have consistently exceeded liquid release levels to streams, and this trend is reflected in the calculated doses to the offsite populations.

Doses from Atmospheric Releases

Doses due to atmospheric releases of radioiodine are discussed in terms of routine releases and the 1961 acute release incident.

Routine Releases

Doses to individuals from ^{129}I and ^{131}I are shown in tables 5-6 and 5-7, respectively. As shown in Table 5-7, the largest annual effective dose equivalent theoretically received by an adult occurred in 1956, the year atmospheric ^{131}I releases peaked. The effective and thyroid dose equivalents from ^{131}I to the "maximum" adult at the site perimeter in 1956 were 16 and 551 mrem, respectively. The doses to other age groups, calculated with ICRP Publication 56 methodology (ICRP, 1989) were higher because of biokinetic and dietary differences. The age group most impacted was children. The effective and thyroid dose equivalents from ^{131}I to the "maximum" child at the site perimeter in 1956 were 46 and 1540 mrem, respectively.

"Maximum" individuals are hypothetical persons who lived at the SRS boundary and subsisted on diets of locally produced milk, meat, and vegetables. No such individuals are known to exist. Nevertheless, if one examines the case of the maximum adult individual living continuously at the site perimeter throughout the period 1955-1989, the cumulative effective dose equivalent from atmospheric ^{129}I and ^{131}I releases is estimated as 32 mrem (tables 5-6 and 5-7). This value is believed to represent the bounding case for an adult's iodine dose and would make a minor contribution to the overall dose received during that time period.

A person who was an infant or a child in the 1950s would have cumulative effective and thyroid dose equivalents somewhat higher than a person who was an adult in the 1950s. These doses would depend on the age of the individual at the time of the maximum releases. The maximum possible cumulative dose equivalents would be received by a person who was a child in 1955 and who lived at the point along the site boundary having the maximum concentration throughout the 35-year period 1955-1989. The cumulative effective and thyroid dose equivalents to this hypothetical person from ^{129}I and ^{131}I releases would be 77 and 2,610 mrem, respectively.

A person living in the Central Savannah River Area would have received an effective dose of approximately 10,300 mrem from exposure to natural sources of radioactivity and an additional 2,300 mrem from medical practices and various consumer products during this 35-year period (Cummins et al., 1990b). Therefore, the SRS atmospheric radioiodine releases contributed less than an estimated 0.6 percent of the total cumulative dose to this individual even if the individual was a child in 1955.

Because the contribution of SRS radioiodine releases to any one individual's total radiation dose is so small, it is necessary to pool the radiation exposures from a given population if an assessment of potential health risks is desired. The population dose within an 80-kilometer radius is a figure-of-merit frequently used to make such an assessment.

The population doses reported in Table 5-8 are based on 1980 census data (555,100 people within 80 km) and the most recently available meteorological and dose factor data. If it is assumed that this population has lived in the SRS vicinity throughout the period of site operation, the total collective effective dose received by that population would be 3,000 person-rem, due to SRS radioiodine releases through 1989. As indicated in the table, this value is dominated by the ^{131}I component of the dose, 2,790 person-rem.

The risks associated with this collective dose are quite small. The ICRP nominal risk estimate for the number of excess fatal cancers potentially induced by a collective dose of 3,000 person-rem is 1.5. Conversely, in that small population, at the current fatal cancer frequency of 16 percent (EPA, 1989), there will be approximately 90,000 spontaneous fatal cancers from all other causes. Therefore, it is very unlikely that a relationship between any of the cancer deaths occurring in this population and the releases of radioiodine to the atmosphere from the SRS will be encountered.

Acute ^{131}I Release Incident of 1961

Of the 162 curies of ^{131}I released in 1961, 153 curies were released from the F-Area chemical separations facility during May 29 - June 23, 1961, as a result of the unintentional reprocessing of very short-cooled material (Marter, 1963). About 67 percent (103 curies) of the ^{131}I was released during the first eight days. Prevailing southwest winds and atmospheric inversions dispersed the radioiodine mainly to the northeast of the site where it was detected in air, vegetation, and food.

Serial sampling showed that the effective half-time of ^{131}I in vegetation and milk was 4 to 5 days. The maximum concentration of ^{131}I in milk from sampling 48 farms and dairies during the peak period of June 5 and 6 was 5,451 pCi/L. The maximum sample was obtained from a farm approximately 3 km northeast of the site perimeter.

The only other fresh, locally produced foods available during this period in early June were eggs and peaches. Fresh eggs from the farm where the maximum milk concentration was found contained an average of 33 pCi/egg. Peaches from an area of high deposition showed no evidence of radioiodine.

The greatest concentrations of ^{131}I in air for the period of maximum release (May 30 - June 7) ranged from 1 to 5 pCi/m^3.

From the concentration data, it was obvious that the milk pathway would be the major source of offsite exposure. To avoid underestimating the dose from the ingestion of milk, original dose calculations were made with an assumed 8-day half-time of ^{131}I in milk instead of the 5-day half-time actually observed (Marter, 1963). The original dose calculations indicated the maximum probable dose of 1,200 mrem to a child's thyroid and 140 mrem to an adult's thyroid.

Dose calculations for food consumption were based on the average concentration of ^{131}I in eggs. Using a consumption rate of 2 eggs per day and an assumed 5-day half-time in eggs, the dose to a child and adult thyroid would have been 7.1 and 0.7 mrem, respectively.

Immersion (plume) thyroid doses from one week of exposure to air contaminated with 1 to 5 pCi/m^3 of ^{131}I would have ranged from 4x10^{-5} to 2x10^{-4} mrem. Thyroid doses from inhalation of this air would have ranged from 1.4 to 6.9 mrem for a child and 0.3 to 1.5 for an adult.

Based on this discussion of ^{131}I doses from various pathways, greater than 95 percent of the thyroid dose for a child and an adult would have resulted from the milk pathway.

Thyroid doses from the milk pathway were recalculated (Marter, 1990) with a more recent internal dosimetry method (ICRP, 1989) and the observed half-time of ^{131}I in milk of 5 days. Maximum doses were reduced to 277 mrem to a child thyroid and 53 mrem to an adult thyroid. In addition, doses of 497 mrem to an infant thyroid and 112 mrem to a teen thyroid were calculated with the current methodology.

Regardless of the methodology, the calculated thyroid doses from the 1961 acute release incident are less than the thyroid doses from routine releases in 1956, the year of maximum ^{131}I releases.

Doses from Liquid Releases

Dose equivalents potentially received by downstream consumers of Savannah River water and fish are shown in tables 5-9, -10, and -11. The dose calculations are based on the total measured curies of ^{131}I released directly to onsite streams.

Dose equivalents have been calculated for a "maximum" individual living just below the SRS. This is an adult who subsists on a diet of untreated Savannah River water and fish of Savannah River origin. The individual partakes in recreational activities in and along the river just below SRS. Doses to this adult are believed to represent the bounding case for liquid radioiodine releases (Table 5-9). Drinking water doses have also been calculated for age-specific maximum individuals at the downriver water treatment plants (Table 5-10) and for the populations served by the water treatment plants (Table 5-11).

Dose contributions from ^{129}I have not been included in the tables. The total amount of ^{129}I that migrated to streams throughout the 35-year period has been estimated as 0.2 Ci and is therefore not a significant contributor to dose.

Table 5-9 shows that the maximum doses occurred in 1957, a year of relatively high ^{131}I releases coupled with low river flow rates. The effective and thyroid dose equivalents to the maximum individual for that year are estimated as 0.4 and 13.7 mrem, respectively. If the hypothetical "maximum" exposure conditions are used as the bounding case for the 35-year period of site operation considered here, the cumulative effective and thyroid doses to such an individual would be on the order of 1.4 and 47.6 mrem, respectively—none of which was delivered after 1978.

Because this individual's dose from non-SRS sources of radiation for that same time period would have exceeded 12,000 mrem, it may be concluded that the contribution to downstream individuals' doses by SRS radioiodine releases is insignificant. This conclusion is further evidenced by the fact that most of the age-specific annual effective doses reported in tables 5-9 and 5-10 do not exceed 1 mrem. An effective dose equivalent of < 1 mrem/yr. has been termed a Negligible Individual Risk Level by the National Council on Radiation Protection and Measurements (NCRP, 1987). In three cases effective dose equivalent does exceed 1 mrem. The greatest value is 1.2 mrem to an infant in 1957.

With respect to the effect of liquid ^{131}I releases on the populations downriver from SRS, drinking water doses for users of the Beaufort-Jasper, SC, and Port Wentworth, GA, water treatment plants have also been estimated. Different terminology is used to describe the two populations to reflect the difference in their compositions. The Beaufort-Jasper plant (51,000 customers) services residential areas and therefore provides full-scale domestic water service. The Port Wentworth facility (20,000 effective consumers) serves a commercial complex in which contact with treated Savannah River water is limited to industrial workers who consume tap water. In past years, however, a locally-operated soft drink bottling facility may have presented an additional pathway of exposure.

If the cumulative effective doses received by both of these water-consuming populations are summed, the collective dose equivalent would be 13 person-rem. Using the ICRP nominal risk factor, the predicted impact of this collective dose is an estimated 0.007 excess fatal cancers in a population of 71,000 people — 11,400 of whom, at the current fatal cancer rate, are projected to succumb to cancer from all other sources.

The maximum individual exposure is to the water consumer who was an infant in 1957. If that person continued to consume treated Savannah River water, the cumulative effective and thyroid doses would be 3.7 and 123 mrem, respectively — none of which was delivered after 1978.

Table 5-9. Liquid ^{131}I releases and doses to the SRS "maximum" individual[a]

Year	Estimated Flow Rate (m^3/s)[b]	Releases To Streams (Ci)[c]	ICRP 30 Adult Dose (mrem)		ICRP 56 Adult Dose (mrem)	
			EDE	Thyroid	EDE	Thyroid
1955	171	–	–	–	–	–
1956	168	–	–	–	–	–
1957	205	6.50×10^{01}	4.03×10^{-01}	1.37×10^{01}	3.65×10^{-01}	1.22×10^{01}
1958	354	–	–	–	–	–
1959	222	–	–	–	–	–
1960	388	2.43×10^{01}	7.95×10^{-02}	2.70	7.20×10^{-02}	2.40
1961	276	2.88×10^{01}	1.32×10^{-01}	4.49	1.20×10^{-01}	3.99
1962	289	8.71×10^{01}	3.82×10^{-01}	1.30×10^{01}	3.46×10^{-01}	1.16×10^{01}
1963	298	4.65×10^{01}	1.98×10^{-01}	6.73	1.79×10^{-01}	5.98
1964	517	8.70	2.14×10^{-02}	7.25×10^{-01}	1.94×10^{-02}	6.44×10^{-01}
1965	403	5.39	1.70×10^{-02}	5.76×10^{-01}	1.54×10^{-02}	5.12×10^{-01}
1966	296	3.02	1.29×10^{-02}	4.39×10^{-01}	1.17×10^{-02}	3.90×10^{-01}
1967	261	1.36×10^{01}	6.62×10^{-02}	2.24	6.00×10^{-02}	1.99
1968	282	9.49	4.27×10^{-02}	1.45	3.87×10^{-02}	1.29
1969	306	5.02	2.08×10^{-02}	7.07×10^{-01}	1.88×10^{-02}	6.28×10^{-01}
1970	219	2.17	1.26×10^{-02}	4.26×10^{-01}	1.14×10^{-02}	3.79×10^{-01}
1971	270	1.53	7.19×10^{-03}	2.44×10^{-01}	6.51×10^{-03}	2.17×10^{-01}
1972	319	9.30×10^{-01}	3.70×10^{-03}	1.26×10^{-01}	3.35×10^{-03}	1.12×10^{-01}
1973	411	1.55×10^{-01}	4.78×10^{-04}	1.62×10^{-02}	4.33×10^{-04}	1.44×10^{-02}
1974	306	1.40×10^{-02}	5.81×10^{-05}	1.97×10^{-03}	5.26×10^{-05}	1.75×10^{-03}
1975	380	–	–	–	–	–
1976	377	2.00×10^{-03}	6.73×10^{-06}	2.28×10^{-04}	6.10×10^{-06}	2.03×10^{-04}
1977	344	1.24×10^{-02}	4.58×10^{-05}	1.55×10^{-03}	4.15×10^{-05}	1.38×10^{-03}
1978	240	7.26×10^{-04}	2.88×10^{-06}	9.76×10^{-05}	2.61×10^{-06}	8.68×10^{-05}
Totals		3.02×10^{02}	1.40	4.76×10^{01}	1.27	4.23×10^{01}

[a] Throughout 1955-1989, approximately 0.2 Ci of ^{129}I migrated to streams. The ^{129}I did not significantly contribute to offsite doses.

[b] Flow rate data based on USGS measurements at Augusta, GA (1954-1972) and the SRS boat dock (1973-1978).

[c] No direct ^{131}I releases to streams after 1978. Total does not include 1 curie from heat exchangers, leaks, and sumps.

EDE = effective dose equivalent

Table 5-10. Liquid ^{131}I doses to "maximum" individuals at the downriver water treatment plants[a]

Year	ICRP 30 Adult Dose (mrem)		ICRP 56 Individual Doses (mrem)							
			Adult		Teen		Child		Infant	
	CEDE	Thyroid	CEDE	Thyroid	CEDE	Thyroid	CEDE	Thyroid	CEDE	Thyroid
1955	--	--	--	--	--	--	--	--	--	--
1956	--	--	--	--	--	--	--	--	--	--
1957	3.41×10^{-01}	1.16×10^{01}	3.10×10^{-01}	1.03×10^{01}	3.51×10^{-01}	1.17×10^{01}	1.03	3.51×10^{01}	1.19	3.79×10^{01}
1958	--	--	--	--	--	--	--	--	--	--
1959	--	--	--	--	--	--	--	--	--	--
1960	6.74×10^{-02}	2.29	6.12×10^{-02}	2.04	6.93×10^{-02}	2.31	2.04×10^{-01}	6.93	2.36×10^{-01}	7.48
1961	1.12×10^{-01}	3.81	1.02×10^{-01}	3.40	1.16×10^{-01}	3.84	3.40×10^{-01}	1.16×10^{01}	3.93×10^{-01}	1.25×10^{01}
1962	3.24×10^{-01}	1.10×10^{01}	2.94×10^{-01}	9.82	3.34×10^{-01}	1.11×10^{01}	9.82×10^{-01}	3.34×10^{01}	1.14	3.60×10^{01}
1963	1.68×10^{-01}	5.70	1.52×10^{-01}	5.08	1.73×10^{-01}	5.74	5.08×10^{-01}	1.73×10^{01}	5.88×10^{-01}	1.86×10^{01}
1964	1.81×10^{-02}	6.15×10^{-01}	1.64×10^{-02}	5.48×10^{-01}	1.86×10^{-02}	6.20×10^{-01}	5.48×10^{-02}	1.86	6.34×10^{-02}	2.01
1965	1.44×10^{-02}	4.89×10^{-01}	1.31×10^{-02}	4.36×10^{-01}	1.48×10^{-02}	4.92×10^{-01}	4.36×10^{-02}	1.48	5.04×10^{-02}	1.60
1966	1.10×10^{-02}	3.73×10^{-01}	9.97×10^{-03}	3.32×10^{-01}	1.13×10^{-02}	3.76×10^{-01}	3.32×10^{-02}	1.13	3.84×10^{-02}	1.22
1967	5.61×10^{-02}	1.90	5.09×10^{-02}	1.70	5.77×10^{-02}	1.92	1.70×10^{-01}	5.77	1.96×10^{-01}	6.23
1968	3.62×10^{-02}	1.23	3.29×10^{-02}	1.10	3.73×10^{-02}	1.24	1.10×10^{-01}	3.73	1.27×10^{-01}	4.02
1969	1.77×10^{-02}	5.99×10^{-01}	1.60×10^{-02}	5.34×10^{-01}	1.82×10^{-02}	6.04×10^{-01}	5.34×10^{-02}	1.82	6.18×10^{-02}	1.96
1970	1.07×10^{-02}	3.62×10^{-01}	9.68×10^{-03}	3.23×10^{-01}	1.10×10^{-02}	3.65×10^{-01}	3.23×10^{-02}	1.10	3.73×10^{-02}	1.18
1971	6.10×10^{-03}	2.07×10^{-01}	5.54×10^{-03}	1.85×10^{-01}	6.27×10^{-03}	2.09×10^{-01}	1.85×10^{-02}	6.27×10^{-01}	2.13×10^{-02}	6.77×10^{-01}
1972	3.14×10^{-03}	1.07×10^{-01}	2.85×10^{-03}	9.49×10^{-02}	3.23×10^{-03}	1.07×10^{-01}	9.49×10^{-03}	3.23×10^{-01}	1.10×10^{-02}	3.48×10^{-01}
1973	4.06×10^{-04}	1.38×10^{-02}	3.68×10^{-04}	1.23×10^{-02}	4.18×10^{-04}	1.39×10^{-02}	1.23×10^{-03}	4.18×10^{-02}	1.42×10^{-03}	4.51×10^{-02}
1974	4.92×10^{-05}	1.67×10^{-03}	4.47×10^{-05}	1.49×10^{-03}	5.07×10^{-05}	1.68×10^{-03}	1.49×10^{-04}	5.07×10^{-03}	1.72×10^{-04}	5.47×10^{-03}
1975	--	--	--	--	--	--	--	--	--	--
1976	5.71×10^{-06}	1.94×10^{-04}	5.18×10^{-06}	1.73×10^{-04}	5.87×10^{-06}	1.95×10^{-04}	1.73×10^{-05}	5.87×10^{-04}	2.00×10^{-05}	6.34×10^{-04}
1977	3.88×10^{-05}	1.32×10^{-03}	3.52×10^{-05}	1.17×10^{-03}	3.99×10^{-05}	1.33×10^{-03}	1.17×10^{-04}	3.99×10^{-03}	1.36×10^{-04}	4.31×10^{-03}
1978	2.44×10^{-06}	8.29×10^{-05}	2.22×10^{-06}	7.39×10^{-05}	2.51×10^{-06}	8.35×10^{-05}	7.39×10^{-06}	2.51×10^{-04}	8.55×10^{-06}	2.71×10^{-04}
Total[b]	1.19	4.03×10^{1}	1.08	3.59×10^{1}	N/A		N/A		N/A	

[a] Throughout 1955-1989, approximately 0.2 Ci of ^{129}I migrated to streams. The ^{129}I did not significantly contribute to offsite doses.

[b] No direct ^{131}I releases to streams after 1978.

CEDE = committed effective dose equivalent.

N/A reflects the fact that an individual could not have remained in this age group throughout the 35-year period studied.

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Table 5–11. Liquid ^{131}I doses to populations served by downriver water treatment plants[a]

Year	Population Dose, (person-rem)							
	Beaufort-Jasper, SC[b]				Port Wentworth, GA			
	ICRP 30		ICRP 56		ICRP 30		ICRP 56	
	CEDE	Thyroid	CEDE	Thyroid	CEDE	Thyroid	CEDE	Thyroid
1955	–	–	–	–	–	–	–	–
1956	–	–	–	–	–	–	–	–
1957	–	–	–	–	2.80	9.53×10^{01}	2.53	8.42×10^{01}
1958	–	–	–	–	–	–	–	–
1959	–	–	–	–	–	–	–	–
1960	–	–	–	–	5.52×10^{-01}	1.88×10^{01}	5.00×10^{-01}	1.66×10^{01}
1961	–	–	–	–	9.18×10^{-01}	3.13×10^{01}	8.33×10^{-01}	2.77×10^{01}
1962	–	–	–	–	2.65	9.05×10^{01}	2.41	8.01×10^{01}
1963	–	–	–	–	1.38	4.69×10^{01}	1.25	4.15×10^{01}
1964	–	–	–	–	1.48×10^{-01}	5.06	1.34×10^{-01}	4.47
1965	3.00×10^{-01}	1.02×10^{01}	4.09×10^{-01}	1.38×10^{01}	1.18×10^{-01}	4.01	1.07×10^{-01}	3.55
1966	2.29×10^{-01}	7.77	3.12×10^{-01}	1.05×10^{01}	8.98×10^{-02}	3.06	8.15×10^{-02}	2.71
1967	1.17×10^{00}	3.97×10^{01}	1.59	5.39×10^{01}	4.59×10^{-01}	1.57×10^{01}	4.16×10^{-01}	1.38×10^{01}
1968	7.57×10^{-01}	2.57×10^{01}	1.03	3.48×10^{01}	2.97×10^{-01}	1.01×10^{01}	2.69×10^{-01}	8.94
1969	3.69×10^{-01}	1.25×10^{01}	5.02×10^{-01}	1.70×10^{01}	1.45×10^{-01}	4.93	1.31×10^{-01}	4.36
1970	2.22×10^{-01}	7.55	3.03×10^{-01}	1.02×10^{01}	8.72×10^{-02}	2.97	7.91×10^{-02}	2.63
1971	1.27×10^{-01}	4.32	1.73×10^{-01}	5.86	4.99×10^{-02}	1.70	4.53×10^{-02}	1.51
1972	6.55×10^{-02}	2.22	8.92×10^{-02}	3.01	2.57×10^{-02}	8.75×10^{-01}	2.33×10^{-02}	7.74×10^{-01}
1973	8.46×10^{-03}	2.87×10^{-01}	1.15×10^{-02}	3.90×10^{-01}	3.32×10^{-03}	1.13×10^{-01}	3.01×10^{-03}	1.00×10^{-01}
1974	1.03×10^{-03}	3.49×10^{-02}	1.40×10^{-03}	4.73×10^{-02}	4.03×10^{-04}	1.37×10^{-02}	3.65×10^{-04}	1.22×10^{-02}
1975	–	–	–	–	–	–	–	–
1976	1.19×10^{-04}	4.04×10^{-03}	1.62×10^{-04}	5.48×10^{-03}	4.67×10^{-05}	1.59×10^{-03}	4.24×10^{-05}	1.41×10^{-03}
1977	8.10×10^{-04}	2.75×10^{-02}	1.10×10^{-03}	3.73×10^{-02}	3.18×10^{-04}	1.08×10^{-02}	2.88×10^{-04}	9.58×10^{-03}
1978	5.09×10^{-05}	1.73×10^{-03}	6.94×10^{-05}	2.35×10^{-03}	2.00×10^{-05}	6.68×10^{-04}	1.81×10^{-05}	6.03×10^{-04}
Totals[c]	3.25	1.10×10^{02}	4.42	1.50×10^{02}	9.72	3.31×10^{02}	8.81	2.93×10^{02}

(a) Throughout 1955-1989, approximately 0.2 Ci of ^{129}I migrated to streams. The ^{129}I did not significantly contribute to offsite doses.

(b) The Beaufort-Jasper water treatment plant did not operate prior to 1965.

(c) No direct ^{131}I releases to streams after 1978.

CEDE = committed effective dose equivalent

Comparisons of Radioiodine Concentrations and Doses with Applicable Regulations

EPA and DOE standards are the regulations by which concentrations and doses due to SRS releases are compared. The standards are applicable to air and drinking water.

Concentrations and Doses in Air

Atmospheric concentrations at the site perimeter are modelled with computer codes because ^{131}I has not been routinely detectable since 1963 and routine analysis of ^{129}I was not performed. For 1989, the average annual perimeter concentrations of ^{129}I and ^{131}I, as predicted by XOQDOQ, were 1.4×10^{-5} and 3.8×10^{-7} pCi/m^3, respectively. These values represent less than one millionth of the DOE and EPA derived air concentration guides for these radionuclides (see Table 5-12).

In 1956, the year of greatest atmospheric release of ^{131}I, the average annual perimeter concentration would have been 0.4 pCi/m^3. This greatest concentration is only 0.1 per cent of the DOE guide in effect during 1989. However, these concentration guides only ensure that the annual effective dose equivalents to members of the public from the inhalation of radioiodine are <100 mrem/yr. As all reasonably achievable steps are taken to ensure that environmental doses are only small fractions of the dose limit, it is reasonable to expect offsite air concentrations to be quite small fractions of the allowable concentration limits.

The highest hypothetical effective dose received by the maximally exposed individual due to atmospheric releases of radioiodine from SRS was 16.3 mrem in 1956 (Table 5-7). The current annual limits on doses to members of the public recommended by the ICRP and the DOE are 500 and 100 mrem, respectively. However, a 1989 regulation established a 10 mrem/year limit on effective dose equivalents to members of the public (EPA, 1989). This value was developed to establish a baseline for risk assessment work and does not imply that a higher dose is unacceptable. With the exception of 1956, annual doses to offsite individuals from SRS radioiodine releases have not exceeded 10 mrem/year.

Concentrations in Groundwater

The standards for ^{129}I and ^{131}I in drinking water are 1 and 3 pCi/L, respectively (EPA, 1977). Short-lived ^{131}I decays to immeasurably small concentrations during the migration time for contaminated groundwater to reach surface outcrops; however, the inventory of ^{129}I is not significantly depleted by decay.

In 1989 the McBean Member of the Santee Formation was found to be contaminated with ^{129}I near the F-Area seepage basins; the maximum concentration detected was 140 pCi/L. At other locations on the site, the Santee Formation is a minor source of drinking water. The portion of the formation tapped for drinking water is not contaminated by ^{129}I as evidenced by the lack of elevated tritium (Chapter 4).

The maximum ^{129}I concentration observed for groundwater outcrops is 410 pCi/L near the F-Area seepage basin system. This value exceeds the applicable drinking water standard, but before this water reaches downriver water treatment plants, it is diluted by Four Mile Creek and the Savannah River to concentrations to well below the EPA drinking water standards.

Concentrations and Doses in Offsite Surface and Potable Waters

There are two water treatment plants downriver from SRS that use Savannah River water for domestic and commercial service. The intake canals for the plants are located approximately 170 kilometers below the site. The raw and finished water supplies of the Beaufort-Jasper, SC, and Port Wentworth, GA, plants are routinely sampled for radioactivity. Concentrations of ^{131}I at both plants are similar to the river concentrations. They have been below the limits of conventional detection since the late 1960s. Estimates of ^{129}I concentrations determined with the computer code LADTAP II are shown in Table 5-13 for the year 1989. The value of 0.0029 pCi/L reported in the table is 0.29 percent of the EPA drinking water standard. Concentrations of ^{131}I have not been included because after 1978 ^{131}I was not routinely discharged to site streams that flow to the Savannah River.

No committed effective dose equivalent from SRS radioiodine releases to the Savannah River exceeds 1.2 mrem as shown in tables 5-9 and 5-10. Doses from this exposure pathway were calculated under extremely conservative conditions and are much lower than the DOE dose standard of 4-mrem effective dose equivalent.

It is important to note that the dose standard of 4 mrem is meant to be applied to all releases of radioactivity — not just ^{129}I and ^{131}I. When the calculations are expanded to include all radionuclide releases, the relationship between SRS doses and the various dose limits described above remains valid. The SRS, based on radioiodine or total releases, is consistently a minor contributor to radioactivity levels in the SRS environment.

Table 5-12. Comparison of airborne ^{129}I and ^{131}I concentrations with regulatory guidelines, pCi/m^3

Parameter	^{129}I	^{131}I
Derived Air Concentration Guide (EPA 1988)	4,000	20,000
Derived Concentration Guide for Air (DOE 1990)	70	400
Average Annual Perimeter Concentration for 1989$^{(a)}$	0.000014	0.00000038

(a) Concentrations estimated using the XOQDOQ transport and dispersion code.

Table 5-13. Comparison of estimated ^{129}I concentrations in surface waters with the regulatory guidelines, pCi/L.

Parameter	Downriver Water Treatment Plant	
	Beaufort-Jasper, SC	Port Wentworth, GA
Derived Concentration Guide for Water (DOE 1990)$^{(a)}$	500	500
Drinking Water Standard (EPA 1977)	1	1
Average Annual Concentration for 1989$^{(b)}$	0.0029	0.0028

(a) The DOE guideline is shown solely for purposes of comparison. The 500 pCi/L limit does not apply beyond the operational boundaries of a DOE facility.

(b) Concentrations estimated using the LADTAPII transport and dispersion code.

Summary of Dosimetric Impacts

The overall radiological impact of SRS radioiodine releases (1955–1989) on the offsite maximum adult can be characterized by total doses of 32 mrem (atmospheric) and 1 mrem (liquid). For the maximum individual who was a child in 1955, the total dose due to atmospheric releases is 77 mrem. For the maximum individual who was an infant in 1957 and a consumer of treated Savannah River water, the total dose due to aqueous releases is 3.7 mrem. However, an individual would have received a dose of approximately 12,600 mrem from other sources of ionizing radiation during 1955 - 1989.

The impact of SRS radioiodine releases on offsite populations is minor. The total collective dose from atmospheric radioiodine releases (1955–1989) is estimated as 3,000 person–rem distributed among 555,100 individuals within 80 km of SRS. The dose results in a lifetime risk of 1.5 fatal cancer cases in that population; the lifetime risk from all other causes of cancer is 90,000 fatalities. The total collective dose from liquid radioiodine releases for the same period is 13 person–rem distributed among 71,000 consumers of treated Savannah River water. The dose results in a lifetime risk of 0.007 fatal cancer cases; the lifetime risk from all other causes of cancer is 11,000 fatalities.

Radioiodine releases from SRS have decreased dramatically over the course of site operations. Measured and predicted ^{129}I and ^{131}I concentration data clearly reflect this trend. Based on data published in the annual environmental reports, routine ^{129}I and ^{131}I releases from SRS present a negligible risk to the offsite environment and the population it supports.

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Appendix A—Physical, Chemical and Biological Properties of Iodine

The element iodine has only one stable isotope, but numerous radioactive isotopes. Stable and radioactive isotopes of iodine show the same chemical and biological behavior. The physical properties of some of the radioactive isotopes will be described in this appendix along with the chemical and biological properties of the element iodine. A brief description of iodine in nuclear reactor accidents is also included.

Physical Properties

Up to seventeen radioactive isotopes of iodine have been present at SRS due to site operations. These isotopes originate from three sources: (1) nuclear fission, (2) activation reactions (neutron capture or (n,2n) reactions), or (3) commercially purchased products. Typically, only the longer-lived ^{131}I and ^{129}I isotopes have been detected in releases to the environment from SRS facilities.

An atom of ^{131}I decays by emitting a beta particle that belongs to one of several characteristic energy groups. The most abundant group is emitted in 86 percent of decays. Within each energy group, the beta particles have a distribution of energies. For the most abundant group, the maximum beta particle energy is 610 keV. Several gamma rays are also emitted in ^{131}I decay. The most abundant gamma ray has an energy of 364 keV and is emitted in 82 percent of decays. The half-life of ^{131}I atoms is 8.0 days.

The decay of ^{131}I produces ^{131}Xe in both the ground and metastable states. More than 99 percent of the decays of ^{131}I produce the stable ground state ^{131}Xe without formation of the metastable state. The ^{131}Xe^{m} that is formed decays with a 12-day half-life to the stable ground state ^{131}Xe by isomeric transition emitting a 164-keV gamma ray in 2 percent of the transitions.

An atom of ^{129}I decays by emitting a beta particle to form stable ^{129}Xe. The emitted beta particle belongs to only one characteristic energy group. The maximum beta particle energy is 150 keV. Several low energy photons are also emitted. One, a gamma ray with an energy of 39.5 keV, is emitted in only 7 percent of decays. More abundant are the xenon x-rays which are emitted in approximately 60 percent of decays. The x-rays have an energy of approximately 30 keV. The half-life of ^{129}I atoms is 16 million years. The combination of very long half-life and low-abundance, low-energy photons make the routine detection of ^{129}I at environmental concentrations difficult.

Chemical and Biological Properties

With oxidation states of -1, 0, +1, +3, +5, and +7, iodine can exist in numerous organic or inorganic chemical forms, some of which are volatile at ambient temperatures. The chemical and biological behavior of a specific chemical form of iodine is the same whether the iodine is stable or radioactive.

The distribution of chemical forms of radioiodine released from nuclear facilities may differ from the distribution of chemical forms of stable iodine found in the environment. As a result, the environmental behavior of radioiodine and stable iodine may differ. The environmental behavior determines the pathway to humans. A brief description of the environmental behavior of iodine follows. A more thorough description can be found in the references Cohen, 1985, and Whitehead, 1984.

Iodine becomes redistributed about the earth by physical and chemical processes in the terrestrial environment, the oceans, and the atmosphere. Much of the iodine in soil originates from the weathering of rocks. Iodine can be transferred from the terrestrial environment to the atmosphere by the suspension of iodine-containing soil particles, the formation of volatile iodine compounds in vegetation and soil, and the combustion of coal.

Surface water transport of dissolved iodine and iodine-containing soil particles can redistribute iodine from the terrestrial environment to the oceans. Iodine can be transferred from bodies of water to the atmosphere by the formation of volatile iodine compounds in the water and the formation of iodine-containing particles in spray.

Once airborne, iodine undergoes exchange between gaseous and particulate forms. In the atmosphere, gaseous iodine can exchange to the particulate form by reacting with particulate matter. Gaseous or particulate forms of iodine can undergo decomposition reactions due to physical influences such as sunlight, or chemical influences such as ozone. An example of this cycle is methyl iodide which is not very reactive with particulate surfaces. However, decomposition and oxidation of methyl iodide produces elemental iodine which is very reactive with particulate surfaces.

Airborne iodine is transported with the wind. Iodine is removed from the atmosphere by wet and dry deposition. When ^{129}I that has been released to the atmosphere from a nuclear fuel reprocessing facility is deposited over land, it is not readily transported downward in the vertical soil column. This has been demonstrated by measurements of ^{129}I near nuclear fuel reprocessing facilities that experience different climatic and soil conditions. These facilities are SRS, Hanford, and Nuclear Fuels Services in the United States, the Karlsruhe Reprocessing Plant in Germany, the Sellafield Works in England, and the Power Reactor and Nuclear Fuel Development Corporation in Japan (Anderson, 1978; Kantelo et al., 1982; Rickard et al., 1974; Brauer and Strebin, 1982; Schuttelkopf and Pimpl, 1982; Robens et al., 1989; Stewart and Wilkins, 1985; Muramatsu and Ohmomo, 1986).

Deposited iodine can be removed from soil by resuspension, surface water runoff, volatilization, and uptake into the root system of plants. Plants also receive iodine by direct deposition onto leafy surfaces. Vegetation is a step in the major pathway for the uptake of ^{131}I by humans who consume milk from cows that graze on vegetation contaminated with fallout radioiodine.

About 90 percent of the 0.011 grams of iodine in adult humans is found in the thyroid gland. Iodine in the thyroid undergoes a series of biochemical reactions to form hormones, which are important to metabolism in most body tissues such as the heart, skeletal muscles, liver, and kidney. The removal of iodine from the thyroid is characterized by the biological half-time of iodine.

Properties in Nuclear Reactor Accidents

Considerable information about the chemical properties of iodine in a nuclear reactor accident was learned from analysis of the Three Mile Island accident, which occurred in 1979. One of the surprising aspects of the accident was the small release of ^{131}I to the atmosphere. The 15 curies of ^{131}I that was released to the atmosphere represented less than one part per million of the inventory of ^{131}I in the reactor core (Voilleque, 1986). About 40 percent of the ^{131}I inventory was dispersed to the inside of the containment and auxiliary buildings (Paquette et al., 1986). Most of the iodine condensed as cesium iodide on walls, ductwork, etc.

The Chernobyl reactor accident in 1986 also released considerably less radioiodine to the environment than was contained in the reactor core. As serious as this accident was, only 20 percent of the inventory of ^{131}I was released to the atmosphere (USSR,1987).

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Appendix B—Environmental Sampling and Analysis

Over the history of SRS, the routine environmental sampling and analysis program has evolved into a comprehensive network of sampling stations in the local environment. Similarly, the site's capability to respond to emergency conditions has evolved to the point where mobile laboratories provide near real-time measurement results for environmental samples. The SRS capabilities for routine and emergency monitoring of radioiodine are described in this appendix.

Routine Site Operating Conditions

The routine analysis of environmental samples such as air, rain, milk, water, vegetation, wildlife, soil, and sediments has been directed towards the detection of ^{131}I. Detection of ^{129}I at the very low levels found in the environment is technically difficult—the measurement technique is not amendable to routine analyses of the magnitude of the ^{131}I sampling program. However, since 1989, selected groundwater samples have been routinely analyzed for ^{129}I.

The environment is sampled by a continuous collection for a specified time, such as a 7-day collection of water, or by an instantaneous collection, such as a grab sample of water. The reported concentration for the instantaneous sample represents the concentration at the time the sample was collected; the concentration at measurement time is corrected for radioactive decay during the time between sample collection and sample measurement. For the continuous collection, the measured concentration is corrected for radioactive decay during the elapsed time since either the start or the midpoint of the collection; one or the other correction has been used during the history of SRS.

Air

Atmospheric sampling locations near the site perimeter increased from 8 in 1955 to 14 in 1989. Correspondingly, at the 40-km radius, sampling locations increased from 3 to 12. Figure B-1 shows the locations within the 40-km radius.

Locations at greater distances were added in 1962 to help distinguish SRS-released radioiodine from that due to fallout, which was more generally distributed in the environment. Four locations (Columbia, SC, Greenville, SC, Macon, GA, and Savannah, GA) were added at the 160-km radius to aid this effort.

The air samplers for radioiodine continuously pass a filtered air stream through a cartridge of activated charcoal impregnated to 5 percent triethylenediamine (TEDA). The charcoal cartridge and particulate filter are changed weekly and analyzed by gamma spectrometry. (The sampling material used until the mid 1960s was a filter paper coated with silver nitrate.)

Rain

Some of the locations for air monitoring also have the capability to collect rainwater. Rainwater collects in stainless steel pans 60 cm by 60 cm and flows through an anion-cation exchange column beneath the pan and then into a collection bottle for volume measurement. The ion exchange column is changed every fourth week except at the 160-km locations, where it is changed quarterly. The ion exchange columns are analyzed by gamma spectrometry.

Because the rainwater collection pans are continually exposed to the atmosphere, they also collect dry deposition. Rainwater can leach radioiodine from the dry deposition; this radioiodine is trapped on the ion exchange column.

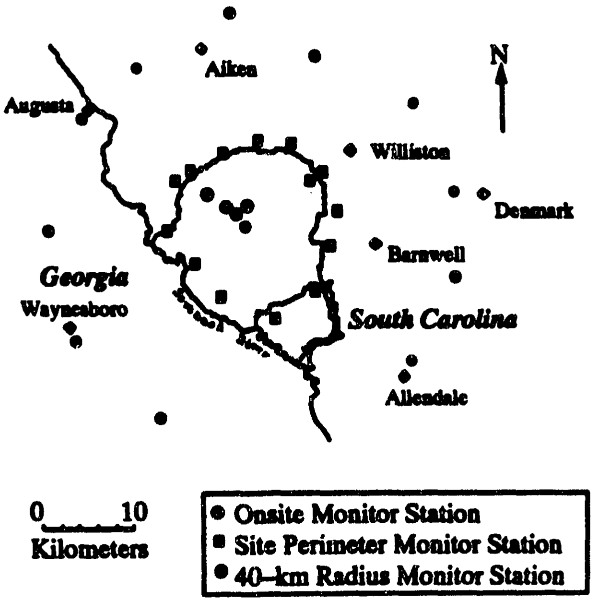

Figure B–1. Air Monitoring Locations within 40 km of SRS

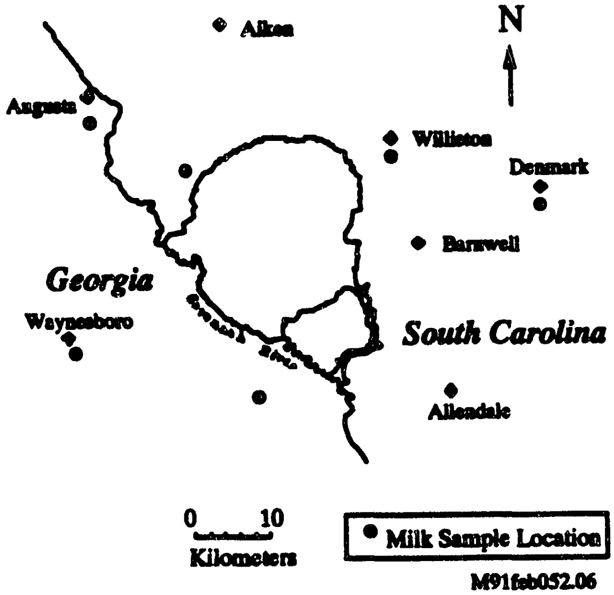

Figure B-2. Milk Monitoring Locations within 40 km of SRS

Milk

Milk samples are collected every two weeks from approximately five dairies and one major distributor within the 40-km radius of SRS. The locations of the dairies sampled in 1989 are shown in Figure B-2. The dairy in Williston, SC, ceased operation in 1989. Its replacement for sampling purposes is a dairy in Denmark, SC. During the history of the site, various dairies and farms out to 80 km have been sampled.

Four liters of milk are passed through an anion exchange column. The ion exchange column is analyzed by gamma spectrometry.

Groundwater

Routine monitoring of SRS wells for ^{131}I began in 1956. In 1989 approximately 140 wells were sampled for ^{131}I once during the year. Samples are collected either by pumping or bailing the wells. A 1-liter aliquot is analyzed by gamma spectrometry.

The routine analysis for ^{129}I in groundwater samples began in 1989. Iodine in quarterly samples is chemically separated, purified, and analyzed by low-energy photon spectrometry.

Surface Water

Sampled surface waters consist of process area effluent outfalls, site streams, and the Savannah River. Sampling locations are shown in Figure B-3. A greater number of sampling locations exist at outfalls from process areas than can be shown on this figure.

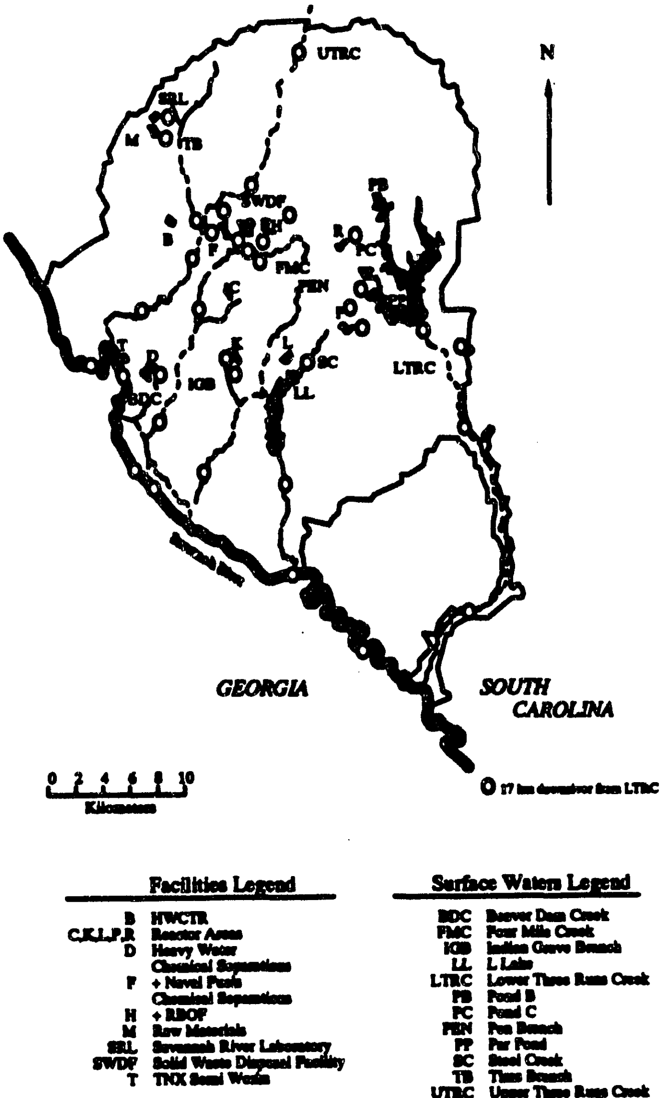

Figure B–3. Surface Water Sampling Locations Onsite and Offsite

Effluent outfalls from process areas

Samples are generally collected from the ditch or canal connecting the process outfall with a body of surface water. Various composite samplers are used, the principal ones being the paddle wheel sampler and the Brailsford pump sampler.

Routine water samples are retrieved weekly. A 1-liter aliquot is analyzed by gamma spectrometry.

Site Streams

Samplers are similar to those described for effluent outfalls. Routine water samplers are retrieved weekly, and a 1-liter aliquot is analyzed by gamma spectrometry.

Savannah River

The Savannah River has been sampled at the locations shown in Figure B-3. Water samples are collected by a paddle wheel sampler which provides a continuous sample. Samples are retrieved weekly, and a grab sample is also collected at the upriver sampling location .

Initially, iodine was chemically separated from the water by solvent extraction. This process was changed in 1963 to passing water through an anion-cation exchange column. The ion exchange column is analyzed by gamma spectrometry.

Other Environmental Media

Periodic samples of vegetation, wildlife, soil, sediment, and food other than milk are collected for radioactivity measurement.

Emergency Response Conditions

Site resources available for emergency response to unplanned releases of radioactivity include the SRS Operations Center, the Weather Information and Display System (WIND), two mobile laboratories, and field sample collection teams. The SRS Operations Center is the command center for coordination of the site's response to emergency conditions.

Weather Information and Display System

Material released from SRS to the atmosphere moves through the atmosphere as a plume. The WIND System is a computerized emergency response system that predicts the location of the plume as a function of time after an unplanned release. It also predicts concentrations of specific isotopes in the plume and the dose consequences of the release. The WIND System is linked to the Weather Center Analysis Laboratory in SRL and to the SRS Operations Center.

The WIND system uses a variety of real-time environmental data as input to its meteorological transport and dispersion computer codes. The inputs include meteorological data from seven instrumented towers onsite and one tower offsite, stack monitoring data, and perimeter monitoring data.

Predicted concentrations and doses from an unplanned release provide a preliminary assessment of the consequences to public health in the path of the plume. This information is communicated to the SRS Operations Center. WIND predictions are also used to position the site's field resources for emergency response —the two mobile laboratories and the sample collection teams—at ideal locations for sample collection.

Radiological Assistance Program Laboratory

The Radiological Assistance Program mobile laboratory became operational in 1986 to provide near real-time assessment of any radiological emergency in the southeastern states. It has numerous capabilities for this purpose, but only those capabilities applicable to a radioiodine release will be described here.

Grab samples of soil, water, milk, vegetation, and air can be collected. The air samples are obtained with portable sampling kits that pump air through silver zeolite cartridges. The on-board measurement instrumentation used to assess these environmental samples for isotopes of iodine include sodium iodide and germanium detectors for gamma spectrometric analysis. However, ^{129}I is not detectable by this technique. In the case of a radiological emergency at SRS, results of gamma spectrometric analyses performed in the field are communicated to the SRS Operations Center.

Tracking Radioactive Atmospheric Contaminants Laboratory

The Tracking Radioactive Atmospheric Contaminants (TRAC) mobile laboratory became operational in 1984. It was principally designed to assess the radiological impact on the general public of an atmospheric release from SRS. It has the capability to continuously monitor the environment for radionuclides in real-time and to perform specialized analyses on-board in near real-time. Only those capabilities applicable to a radioiodine release will be described here.

As a plume of material released from SRS moves through the atmosphere, the location of the plume can be continuously monitored by TRAC using scintillation spectrometers, provided the plume contains gamma-emitting nuclides, such as ^{131}I. This monitoring can occur while TRAC is parked or moving. Results of these real-time analyses to locate the plume and to monitor its movement are communicated to the Weather Center Analysis Laboratory where the information is used to refine WIND predictions.

To determine the atmospheric concentrations of radioiodine, samples of air are obtained with an on-board high volume sampler that pumps air at 25 m^3 per minute. Airborne particulate forms of iodine are trapped on a high efficiency particulate air (HEPA) filter. Volatile forms of iodine are trapped on a TEDA-impregnated charcoal filter. To maximize the absorption of iodine on the charcoal, only a portion of the HEPA-filtered air is passed through the charcoal. This flow rate is limited to 1 m^3 per minute. This sampling can occur while TRAC is parked or moving.

The on-board measurement instrumentation used to assay air samples for radioiodine include high resolution germanium detectors for gamma spectrometric analysis (^{129}I is not detectable by this technique). Typically, within 30 minutes of commencing sample collection, radioiodine concentration results are available for communication to the Weather Center Analysis Laboratory. These near real-time results provide measured concentrations to compare with computer-predicted concentrations. Interpretations of the results in terms of dose assessment are then communicated from the Weather Center Analysis Laboratory to the SRS Operations Center.

In addition to monitoring the atmosphere, TRAC has the capability to monitor the road for deposited radioiodine. This real-time analysis is done with another set of scintillation spectrometers. Environmental samples such as soil, water, milk, and vegetation can be collected in the field and analyzed for their radioiodine content using the on-board high resolution germanium detectors. Results of these additional analyses are communicated to the Weather Center Analysis Laboratory as they become available.

Field Sample Collection Teams

Sample collection teams are deployed to appropriate locations in the field to collect a variety of environmental samples. The samples are returned to environmental laboratories at SRS for subsequent analysis. These environmental samples provide greater comprehensive coverage of the consequences of a release than the coverage provided by the mobile laboratories. However, the availability of analytical results is not near real-time. Examples of this type of sample collection and analysis were presented in Chapter 4 where the 1961 acute ^{131}I release incident was discussed.

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A. Compilations of the history of radioactive releases:

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B. Reports on the entire environmental monitoring program at SRS. Report titles repeat in succeeding years until revised.

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DPST 71-302	Jan-Dec 1970	
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DPSPU 75-302	Jan-Dec 1974	

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DPSPU 85-302	Jan-Dec 1984	
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DPSPU 87-30-1	Jan-Dec 1986	
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WSRC-IM-90-60	Jan-Dec 1989	

(DEL) signifies reissue of report from which classified information was deleted.

C. Reports on the environmental monitoring program that involved principally the offsite environment. Report titles repeat in succeeding years until revised.

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DPSPU 63-30-32	Jan-Jun 1963	
DPSPU 64-30-1	Jul-Dec 1963	
DPSPU 64-30-2	Jan-Jun 1964	
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DPST 65-30-2	Jan-Jun 1965	
DPST 66-30-1	Jul-Dec 1965	
DPST 66-30-2	Jan-Jun 1966	
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DPST-85-704	F-Area Seepage Basins
DPST-85-706	H-Area Seepage Basins
DPST-85-707	Reactor Seepage Basins

II. Other reports in the series on radionuclides in the Savannah River Site environment:

WSRC-RP-90-424-1	"Tritium in the Savannah River Site Environment"

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