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Radionuclides Notice of Data Availability Technical Support Document

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Preface

Under the authority of the Safe Drinking Water Act (SDWA) and its amendments, EPA sets drinking water standards for contaminants, which potentially pose a threat to public health via a public drinking water source. In 1976, EPA promulgated National Interim Primary Drinking Water Regulations (NIPDWR) for three categories of radionuclides: gross alpha emitters, radium 226 and 228 combined, and gross beta and photon emitters. The 1976 NIPDWRs for these radionuclides regulated:

1. gross alpha at 15 picoCuries per liter (pCi/L)(excluding radon and uranium);
2. radium-226 and radium-228 combined at 5 pCi/L;
3. and beta/photon emitters at a 4 millirem dose of radioactivity.

The 1986 reauthorization of the Safe Drinking Water Act (SDWA) declared the interim standards for these radionuclides to be final National Primary Drinking Water Regulations (NPDWR). The 1986 amendment also required standards to be set as close to the maximum contaminant level goal (MCLG; the health goal) as possible. At that time, radionuclides did not have an MCLG.

In 1991 (56 FR 33050), EPA <u>proposed</u>:

- A Maximum Contaminant Level Goal (MCLG) of zero for all radionuclides;
- A NPDWR for uranium at 20 ug/L;
- A NPDWR for radon of 300 pCi/L;
- To revise the standards for the combined radiums (226 and 228) from a MCL of 5 pCi/L to a MCL of 20 pCi/L each;
- To revise gross alpha to exclude not only uranium and radon but also radium 226 (while keeping the gross alpha MCL of 15 pCi/L);
- To revise the MCL for beta/photon emitters from 4 mrem to 4 mrem ede;
- Entry point sampling;
- A standardized monitoring framework; and
- To extend the applicability of the radionuclides NPDWRs to include Non-Transient Non-community Water Systems (NTNCWS).

Due largely to concerns by commenters and Congress over the most effective way to regulate radon, the proposed rule was never finalized. The 1996 reauthorization of the Safe Drinking Water Act withdrew the radon portion of the 1991 radionuclides proposal. Subsequent to the 1996 SDWA amendments, EPA was sued by an Oregon plaintiff (Bull Run Coalition) for failure to finalize the radionuclides rule. EPA entered into a court-stipulated agreement with the Oregon Bull Run Coalition to promulgate a final action for the non-radon portion of the 1991 proposal by November 21, 2000. This document contains additional information about the occurrence, health effects, analytical methods and treatment options for radionuclides in drinking water. This document also presents data concerning the costs and benefits of several regulatory options. This additional information and analyses will be taken into account as EPA is evaluates the 1991 proposal.

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I - Fundamentals of Radioactivity in Drinking Water

The following section provides a brief review of the fundamental concepts of radioactivity. The background information contained within this document is meant for those not familiar with the fundamentals of nuclear chemistry. It is written in broad and general terms and some statements may be simplified.

A. Fundamentals of Nuclear Structure and Radioactivity

An *atom* is the smallest unit of a chemical element. An atom is composed of three basic subatomic particles: *protons*, *neutrons* and *electrons*. As the graphic below depicts, an atom consists of a heavy concentration of mass at the center, which is called the nucleus. The primary constituents of the nucleus are *proton(s)* and *neutron(s)*. A proton carries a positive charge and the neutron carries no charge (i.e., the neutron is neutral). *Electrons* are located in shells or "orbits" that surround the nucleus. Orbital electrons have a negative charge and are approximately 1,836 times smaller in mass than the proton or the neutron. In un-ionized atoms, electrons are equal in number to the protons, making the atom neutral in overall charge.

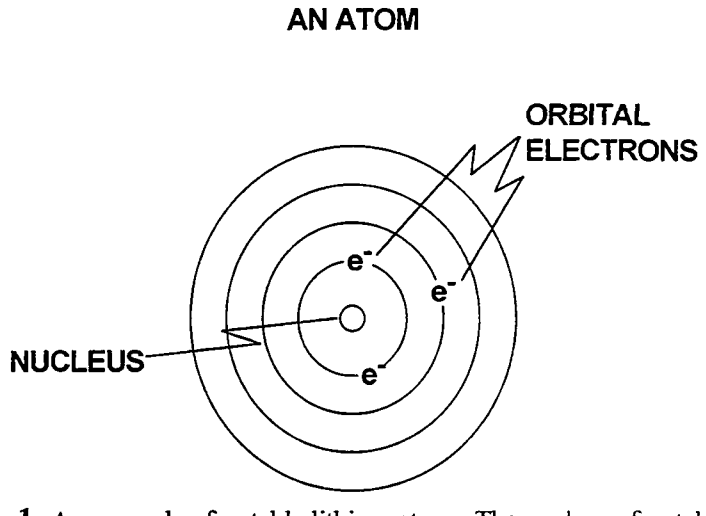


Figure I -1. An example of a stable lithium atom. The nucleus of a stable lithium atom consists of three protons and three neutrons (protons and neutrons not shown). The nucleus is orbited by three electrons.

The number of protons contained within the nucleus of an atom determines the element and the atomic number. The number of neutrons for a given element can vary. Atoms with the same number of protons but different number of neutrons are called *isotopes*. For example, the element radium has six isotopes; radium-223, radium-224, radium-225, radium-226, radium-227 and radium-228. While each of the radium isotopes contain 88 protons in their nucleus, the number of neutrons for each isotope is different. The variation in the number of neutrons does not change the element or its chemical properties, but it can affect the stability of the element.

This instability can result in *radioactive decay.*

Radioactive decay refers to the transformation of the nucleus of an unstable isotope (or radioactive nuclide). As a general rule, an unstable atom will naturally try to attain the lowest energy (or most stable) configuration. Stability can occur by electrons "falling" to the lower orbits, or by a nucleus emitting radioactive particles or radiation. The unstable isotope can decay into a different isotope or into the same isotope with a lower energy state. The principal types of radioactive decay addressed in this document include:

- alpha (α) particle emission;
- beta (β) particle emission;
- and gamma (γ) ray emission.

An *alpha particle* is a positively charged particle emitted during the decay of certain radioactive elements. An alpha particle consists of two protons and two neutrons and is indistinguishable from a helium nucleus (USEPA, 1991). An alpha particle is the heaviest form of nuclear radiation. The decay of polonium-210 to lead-206 (shown below) is an example of alpha decay. The isotope that decays is called the parent (e.g. ^{210}Po) and the resulting progeny is called the daughter (e.g. ^{206}Pb). During alpha decay the atomic mass decreases by 4 and the atomic number decreases by 2 (due to the loss of 2 protons).

$$^{210}Po \rightarrow {}^{206}Pb + {}^{4}He \ (\alpha \text{ particle})$$

A *beta particle* can be either a very energetic, negatively charged particle (electron or negatron) or a positively charged particle (positron), which is emitted from a nucleus during the decay of certain radioactive elements. A negative beta particle results from the emission of an electron from the nucleus during neutron decay (USEPA, 1991). During the process of neutron decay, the neutron decays into a proton, an electron and a neutrino. When a negative beta particle (or electron) is emitted from the nucleus of the parent isotope, a daughter product results that is higher by one atomic number. Beta decay can be described as a neutron in the nucleus being converted to a proton (USEPA, 1991). The decay of radium-228 to actinium-228 (shown below) is an example of negative beta particle or neutron decay. The atomic number for radium is 88 and the atomic number for actinium is 89.

$$^{228}Ra \rightarrow {}^{228}Ac + \beta^{-} \text{ particle (an electron)}$$

Gamma radiation (or rays) is high-energy, short-wavelength electromagnetic radiation emitted from nucleus. Other forms of electromagnetic radiation include light, radio waves, infrared rays, ultraviolet waves and x-rays (USEPA, 1991). Whereas the emission of an alpha or beta particle (from an atom) results in a different element, gamma decay changes neither the atomic number nor the element; it only involves the loss of energy (USEPA, 1981).

Alpha particles, beta particles, and gamma rays each have distinct masses and energies

and produce different effects when they interact with atoms or matter (USEPA, 1981). Each of these radioactive particles or rays are capable of knocking an electron from its orbit around the nucleus. This process of dislodging an electron from its orbit is know as *ionization*. Hence, alpha particles, beta particles and gamma rays are called *ionizing radiation*. Alpha particles are the least penetrating of the common forms of ionizing radiation. While an alpha particle can be stopped by a sheet of paper or the skin, they are still harmful if inhaled or ingested since they are in contact with internal organs (DOE, 1994). Because alpha particles can be stopped in relatively short distances, they deposit their energy within a small distance. Consequently, alpha particles cause more damage per unit volume than beta particles and gamma rays (USEPA, 1981). On the other hand, beta particles and gamma rays deposit their energy over longer distances. Beta particles can be stopped by a piece of wood, or by a thin sheet of metal such as aluminum foil. Like x-rays, gamma rays can pass through the human body (USEPA, 1981). Because gamma rays are highly penetrating, they are best shielded by dense materials like lead and thick concrete (DOE, 1994).

The process of *atomic fission* is the splitting of a heavy nucleus into two nuclei of lighter elements. The fission process is accompanied by the release of a relatively large amount of energy and frequently one or more neutrons. Fission can occur spontaneously, but it is usually caused by the absorption of gamma rays, neutrons and other particles (DOE, 1994). The atomic fission process that occurs in a nuclear reactor is triggered by adding neutrons to heavy nuclei. The classic examples of heavy nuclei that undergo the fission process in nuclear reactors are the isotopes of uranium (uranium-235) and plutonium (plutonium-239). When these heavy nuclei are bombarded with neutrons, the isotope break into two roughly equal parts. Each of the parts (called fission fragments) is itself a radioactive nucleus and decays through a sequence of isotopes by beta and gamma decay. Nuclear reactors can be a source of radioactivity contaminants in drinking water, if radioactive by-products are released. (USEPA, 1991)

The Concept of Half life

Not all atoms are equally stable and different isotopes decay from parent to daughter at characteristically different rates. The concept of *half life* is quantitatively used to describe these differences in decay rates. The half life of an isotope is the time required for one half (50%) of the parent atoms present to decay to daughter products (or progeny). For example, the a half life of radium-228 is 5.76 years. It takes approximately 5.76 years for one half of the original amount of radium-228 present to decay to the daughter product, actinium-228. After a period equal to 10 half-lives, the radioactivity has decreased to about 0.1% of its original value. In other words, it would take 57.6 years for radium-228 to decay to 0.1 % of the amount present. Half lives can range from billions of years or more to millionths of a second. Whereas the half life of uranium-238 is 4.5×10^9 years, the half life of polonium-214 is 164×10^{-6} seconds. Of importance to drinking water are isotopes whose half-lives are long enough to survive transport through water distribution systems. These isotopes have the greatest potential to reach faucets or other end uses, resulting in exposure to people and possibly causing health concerns.

B. Units of Radioactivity

When discussing drinking water, units such as milligrams/liter (mg/l), or parts-per-million (ppm) are commonly used to describe the concentration of pollutants, contaminants, and hazardous substances. However, certain unique properties of radioactive substances limit the utility of these units, and alternative units are used to directly compare the health effects of different radionuclides. Two important concepts are needed to describe radioactivity:

- How many nuclear transformations or disintegrations occur per second (i.e. the *activity*)?
- How much radiation or how much energy is imparted to the tissue from the particles emitted by the radioisotope (i.e. the *absorbed dose* and *the dose equivalent*)?

Activity Units

Potential effects from radionuclides depends on the number of radioactive particles or rays emitted (alpha, beta, or gamma) and not the mass of the radionuclides (USEPA, 1981). As such, it is essential to have a unit that describes the number of radioactive emissions per time period. The *activity* unit is used to describe the nuclear transformations or disintegrations of a radioactive substance, which occur over a specific time interval (USEPA, 1991). The activity is related to the half life; longer half lives mean lower activity. A special unit of activity called a *Curie* is equal to a nuclear transformation rate of 37 billion (3.7×10^{10}) disintegrations or decays per second. One *picoCurie* is equal to 10^{-12} curies, which is approximately 2 nuclear disintegrations per minute (or more specifically one disintegration every 27 seconds) (USEPA, 1991). Historically, by definition, one gram of radium is said to have 1 Curie (1 Ci) of activity. By comparison, 1 gm of uranium-238 has an activity of 0.36 millionth of a curie. Another unit of activity is the *Becquerel (Bq)*, which is a special unit of radioactivity in the International System of units (SI). One Becquerel is equal to one disintegration per second (USEPA, 1991). For drinking water the concentration units for radionuclides is reported as the activity per liter (usually pCi/L).

Dose Units

The effect of radioactivity depends not only on the activity (disintegrations/time) but also on the kind of radiation (alpha, beta, or gamma) and its energy. These two properties, activity and type of radiation, collectively determine the absorbed dose to tissue when decay occurs internally and the internal organs are the target (USEPA, 1991). A dose unit, *absorbed dose*, reflects the amount of radiation, or how much energy, was imparted to tissue. The total amount of energy imparted is related to the number of particles emitted by the radioisotope per second, and the respective energies of the particles (USEPA, 1991).

A common unit used to measure absorbed dose is a *rad*. One rad is equivalent to the

amount of ionizing radiation that deposits 100 ergs (metric unit of energy) in one gram of matter or tissue. For perspective on the size of an erg, 10 million erg/sec is equivalent to one watt. In general, rad units are quite large (USEPA, 1991). Because of the particle mass and charge, 1 rad deposited in tissue by alpha particles creates more concentrated biological damage than 1 rad of gamma rays.

To compensate for the difference in damage between different types of radiation particles and their subsequent effect, a new unit was created—the rem. *Rem* is the unit of measurement for the *dose equivalent* from ionizing radiation to the total body or any internal organ or organ system. It is equal to the absorbed dose in rads multiplied by a quality factor (to account for different radiation types). This relationship is written as follows:

Number of rems = [the number of rads] x [Q]

Q = 1, for beta particles and all electromagnetic radiation (gamma rays and x-rays);
Q = 10, for neutrons from spontaneous fission and protons; and
Q = 20, for alpha particles and fission fragments.

An example of dose equivalent: As discussed earlier, an alpha particle is a heavy form of radiation and, in a relatively short distance, imparts a great amount of energy to the human body. Alpha exposure produces approximately 20 times the effect of a beta particle, which is a smaller form of radiation and travels faster, consequently imparting its energy over a longer path. The difference in energy is accounted for by a quality factor (Q) and the result is expressed in rems or dose units. While dose equivalent is sometimes not an exact measurement, it nevertheless can be a useful administrative unit (USEPA, 1981). To reiterate, the absorbed dose is measured in rads and the dose equivalent is measured in rem (USEPA, 1991).

Two additional technical terms are also useful; a *rem ede* (effective dose equivalent) is a dose to organs adjusted for different radiation types and by an organ weighting factor to account for organ sensitivity to the effect of radiation (USEPA, 1991). A *sievert (Sv)* is the unit of dose equivalent in the International System of units (SI) from ionizing radiation to the total body or any internal organ or organ system. One Sievert equals 100 rem (USEPA, 1991).

C. Definitions

Some of the following definitions are taken directly from the 40 CFR 141.2 or the July 18, 1991 *Federal Register Notice* (USEPA 1991, 56 FR 33050). Others definitions cited are from other EPA glossaries or glossaries from other organizations (e.g. NIH, DOE or WHO)

Absorbed dose means the mean energy imparted by ionizing radiation to an irradiated medium per unit mass (NIH, 1994).

Activity means the number of nuclear transformations of a radioactive substance which occur in a specific time interval (USEPA, 1991).

Alpha particle means a positively charged particle emitted during decay of certain radioactive elements. Alpha particles are the heaviest, slowest, and least penetrating of the three common forms of ionizing radiation (alpha, beta, gamma).

Atom means the smallest unit of a chemical element (USEPA, 1994).

Atomic mass means the total number of protons and neutrons in the atomic nucleus (USEPA, 1981).

Atomic number means the number of protons in the nucleus; the atomic number identifies the element (USEPA,1981).

Background radiation means the amount of radiation to which a member of the population is exposed from natural sources, such as terrestrial radiation due to naturally occurring radionuclides in the soil, cosmic radiation originating in outer space, and naturally occurring radionuclides deposited in the human body (NIH, 1994).

Beta particle means a negatively charged particle emitted from a nucleus during decay of certain radioactive elements (as a result of neutron decay) (USEPA, 1991). A beta particle is identical to an electron. Beta particles are light and relatively fast (compared to alpha particles), and are easily stopped by a thin sheet of metal, such as aluminum foil, or several inches of wood. Exposure to high levels of beta radiation can cause skin burns (DOE, 1994).

Becquerel (Bq) is a special unit of radioactivity in the International System of units (SI). One Becquerel is equal to one disintegration per second (USEPA, 1991). [1 Becquerel is equal to 27 picoCuries (pCi)]

Cancer means a malignant tumor of potentially unlimited growth, capable of invading surrounding tissue and/or spreading to other parts of the body by metastasis (NIH, 1994).

Carcinogen means an agent that may cause cancer. Ionizing radiations are physical carcinogens; there are also chemical and biological carcinogens; biological carcinogens may be external (e.g., viruses) or internal (genetic defects) (NIH, 1994).

Consumer Confidence Report Rule means the public right-to-know rule which requires water suppliers to put annual drinking water quality reports into the hands of their customers, beginning in 1999 (USEPA, 1999).

Cortical (bone) means bone that is not rapidly remodeled (*i.e.*, deep bone compartment).

Curie means a special unit of activity equal to a nuclear transformation rate of 3.7×10^{10} (or 37 billion) disintegrations/second (USEPA, 1991).

Daughter product means the resulting isotope (of a different element) from radioactive decay of a *parent* isotope.

Decay, or *radioactive decay*, means the spontaneous radioactive transformation of one nuclide (or isotope) into a different nuclide or into lower energy state of the same nuclide (DOE, 1994). Radionuclides decay by emission of alpha and beta particles and gamma rays (USEPA, 1991).

Deterministic effects means effects for which the severity of the damage caused is proportional to the dose, and for which a threshold exists below which the effect does not occur (WHO, 1993). Deterministic effects may also be referred to as non-stochastic.

Dose equivalent means the product of the absorbed dose from ionizing radiation and such factors which account for differences in biological effectiveness due to the type of radiation and its distribution in the body as specified by the International Commission on Radiological Units and Measurements (ICRU) (40 CFR 141.2).

Effective dose equivalent (EDE) means the sum of the products of the dose equivalents in individual organs and the organ weighting factor (USEPA, 1991).

Fission means the splitting of a heavy nucleus into two roughly equal parts (which are nuclei of lighter elements), accompanied by the release of a relatively large amount of energy and frequently one or more neutrons.

Gamma radiation (or rays) means high-energy short-wavelength electromagnetic radiation emitted by a nucleus.

Genetic effect means a health effect that shows up in subsequent generations (USEPA, 1981).

Gross alpha particle activity means the total radioactivity due to alpha particle emission as inferred from measurement on a dry sample (40 CFR 141.2). As an MCL it refers to the total alpha particle count minus contributions from radon 222 and uranium.

Gross beta particle activity means the total radioactivity due to beta particle emissions measured in an aliquot of an evaporated water sample (40 CFR 141.2).

Ground water means the supply of fresh water found beneath the Earth's surface, usually in aquifers, which is often used for supplying wells and springs (USEPA, 1994).

Half-life means the length of time required for a radioactive substance to lose 50% of its activity by decay.

Hormesis means the theory that low-level exposure to radiation may result in beneficial or stimulatory health effects. The theory of hormesis generally runs counter to the Linear No-Threshold Risk Model for low-level exposures to radiation (BELLE 1998).

Ionization means the splitting or dissociation (separation) of molecules into negatively and positively charged ions (USEPA, 1981).

Ionizing radiation means radiation that is capable of removing one or more electrons from an atom (USEPA, 1981).

Irradiation means exposure to radiation of wavelengths shorter than those of visible light (gamma, X-ray, or ultraviolet).

Isotope means one of two or more atoms whose nuclei have the same number of protons but different numbers of neutrons (DOE, 1994). Said differently, isotopes are varieties of the same element that have with different masses from one another (USEPA, 1981).

Leukemia means a malignant disease of the white blood cells and their precursors NIH, 1994).

Linear non-threshold risk model (or *hypothesis*) means effects are linear to zero; all exposures to radiation result in adverse health effects. See also hormesis.

Man-made beta particle and photon emitters means all radionuclides emitting beta particles and/or photons listed in Maximum Permissible Body Burdens and Maximum Permissible Concentration of Radionuclides in Air or Water for Occupational Exposure, NBS Handbook 69, except the daughter products of thorium-232, uranium-235 and uranium-238 (40 CFR 141.2).

Maximum contaminant level (MCL) means the maximum permissible level of a contaminant in water which is delivered to any regulated user of a public water system (40 CFR 141.2).

Milli- means 1 of 1,000.

Millirem (mrem) means 1/1,000 of a rem (40 CFR 141.2).

Morbidity means the rate of disease or ill health in a population.

Mortality (rate) means the rate at which people die from disease (e.g., a specific type of cancer), often expressed as number of deaths per 100,000 per year (NIH 1994).

Mutagen means a substance that can change the structure of DNA, and thus change the basic blueprint for cell replication (USEPA, 1981).

Natural radioactive series means the sequence of elements resulting from the decay of naturally

occurring radioactive elements. Typically, the reference is to the decay series of natural uranium and thorium-232that exist naturally, and decay into each other in a serial fashion (USEPA, 1981).

Natural uranium means uranium composed of uranium-234 plus uranium-235 plus uranium-238 which has an isotopic composition typically of 0.006% uranium-234, 0.7% uranium-235, and 99.27% uranium-238 (USEPA, 1991).

Nuclear radiation means particulate and electromagnetic radiation emitted from nuclei (DOE, 1994).

Nucleus means the heavy concentration of mass at the center of an atom, which is surrounded by shells of electrons in different orbits. The primary constituents of the nucleus are neutrons and protons.

Nuclide means any species of atom that exists for a measurable length of time. The term is often used synonymously with isotope (DOE, 1994). It is characterized by the constitution of its nucleus, which is specified by: its atomic mass and atomic number (Z), or by its number of protons (Z), number of neutrons (N), and energy content (NIH, 1994).

Neutron radiation means an uncharged subatomic particle capable of producing ionization in matter by colliding with charged particles (NIH, 1994). Neutrons are a form of radiation that can penetrate living tissue, but can be stopped by using dense materials like lead, steel, or thick concrete (DOE, 1994).

Organ weighting factor means the ratio of the stochastic risk for that organ to the total risk when the whole body is irradiated uniformly (USEPA, 1991).

PicoCurie (pCi) means the quantity of radioactive material producing 2.22 nuclear transformations per minute (40 CFR 141.2). One *picoCurie* is equal to 10^{-12} curies, or approximately one disintegration every 27 seconds.

Photon means particles of light.

Plutonium means a radioactive element with an atomic number of 94. Its most important isotope is fissionable plutonium-239, produced by neutron irradiation of uranium-238 (DOE, 1994).

Progeny means the decay products resulting after a series of radioactive decay. Progeny can also be radioactive, and the chain continues until a stable nuclide is formed (NIH, 1994).

Quality factor means a factor that roughly approximates the relative differential damage that different types of ionizing radiation can do to tissue. Q=1 for beta particles and all electromagnetic radiation (gamma and x-rays); Q=10 for protons and neutrons from spontaneous fission; and Q=20 for alpha particles and fission fragments. (USEPA 1981, 1991).

Rad means a common unit of absorbed dose; one rad is equivalent to the amount of ionizing radiation that deposits 100 ergs (metric unit of energy) in one gram of matter or tissue (for perspective on the size at an erg, 10 million erg/sec is one watt). In general, these units are quite large (USEPA, 1991).

Radiation means the emitting of energy through matter or space in the form of waves (rays or particles) (DOE, 1994).

Radioactivity means the rate at which radioactive material is emitting radiation, given in terms of the number of nuclear disintegrations occurring in a unit of time (*e.g.* disintegrations per minute, or dpm). The common unit of radioactivity is the curie (Ci), which measures the number of disintegrations in one second of one gram of radium (DOE, 1994). *Artificial radioactivity* means man-made radioactivity produced by fission, fusion, particle bombardment, or electromagnetic irradiation. *Natural radioactivity* means the property of radioactivity exhibited by more that 50 naturally occurring radionuclides (NIH, 1994).

Radionuclide means any man-made or natural element which emits radiation in the form of alpha or beta particles, or as gamma rays (USEPA, 1994).

Rem means the unit of dose equivalent from ionizing radiation to total body or any internal organ or organ system (40 CFR 141.2). It is equal to the absorbed dose in rads multiplied by a quality factor to account for different types of radiation (USEPA, 1991).

A *rem ede* (effective dose equivalent) is a dose to organs adjusted for different radiation types and by an organ weighting factor to account for organ sensitivity to the effect of radiation (USEPA, 1991).

Risk means the potential for harm to people exposed to radioactive substances. In order for there to be risk, there must be hazard and there must be exposure (USEPA, 1997).

Safe Drinking Water Act, commonly referred to as SDWA. A law passed by the U.S. Congress in 1974. The SDWA and subsequent amendments establishes a cooperative program among local, state, and federal agencies to ensure safe drinking water for consumers(USEPA, 1994).

Scientific notation a means expressing numbers as 1 or more significant figures, raised by the power of 10.

Shielding means a material interposed between a source of radiation and humans for protection against the effects of radiation (DOE, 1994).

Sievert (Sv) means the unit of dose equivalent in the International System of units (SI) from ionizing radiation to the total body or any internal organ or organ system. One Sievert equals 100 rem.(USEPA, 1991)

Somatic effect means a health effect on an exposed body. With regard to ionizing radiation, somatic effects mainly refer to cancers and leukemias (USEPA, 1981).

Stochastic effects means effects for which the probability of occurrence is proportional to dose, but not the severity of effect, and it is assumed that there is no threshold below which they do not occur (WHO, 1993).

Non-stochastic describes effects whose severity is a function of dose; for these, a threshold may occur. Examples of non-stochastic somatic effects are cataract induction, nonmalignant damage to the skin, hematological deficiencies, and impairment of fertility (NIH, 1994). Non-stochastic may also be referred to as deterministic effects.

Stochastic means random events leading to effects whose probability of occurrence in a exposed population (rather than severity in an affected individual) is a direct function of dose; these effects are commonly regarded as having no threshold; heredity effects are regarded as being stochastic; some somatic effects, especially carcinogens, are regarded as being stochastic (NIH, 1994).

Teratogenic effect means a health effect to a fetus (USEPA, 1981).

Trabecular (bone) means bone that turns over rapidly, or soft bone. It is accessible to blood compartment.

Transuranic Elements means elements with an atomic number greater than 92, including neptunium, plutonium, americium, and curium (DOE, 1994).

Treatment technique means a specific treatment method required by EPA to be used to control the level of a contaminant in drinking water. In specific cases where EPA has determined it is not technically or economically feasible to establish an MCL, EPA can instead specify a treatment technique (WOT, EPA 815-K-97-002, 1997).

Uranium means a naturally-occurring radioactive element with an atomic number of 92 and an atomic weight of 238. The two principle isotopes are the fissionable uranium-235 (0.7% of natural uranium) and uranium 238 (99.3% of natural uranium) (DOE, 1994).

D. References

BELLE, 1998. "Toxicological and Societal Implications of Hormesis - Part 2." BELLE Newsletter: Biological Effects of Low Level Exposures. Volume 7, Number 1. May, 1998. ISSN 1092-4736. http://www.belleonline.com/

DOE, 1994. "Appendix C - Glossary." Committed to Results: DOE's Environmental Management Program, An Introduction (EM Primer). Department of Energy. DOE/EM-0152P.

April, 1994. http://www.em.doe.gov/emprimer/appc.html

NIH, 1994 "Radiation Glossary." What We Know About Radiation. National Institutes of Health. May, 1994. http://www.nih.gov/health/chip/od/radiation/

USEPA, 1981. Radioactivity in Drinking Water. "Glossary." U. S. Environmental Protection Agency, Health Effects Branch, Criteria and Standards Division, Office of Drinking Water. January, 1981. EPA 570/9-81-002.

USEPA, 1991. "National Primary Drinking Water Regulations; Radionuclides; Proposed Rule." Appendix A Fundamentals of Radioactivity in Drinking Water. Federal Register. Vol. 56, No. 138, July 18, 1991, p. 33050.

USEPA, 1994. Drinking Water Glossary: A Dictionary of Technical and Legal Terms Related to Drinking Water. U.S. Environmental Protection Agency, EPA 810-B-94-006. June, 1994. http://www.epa.gov/watrhome/pubs/gloss2.html

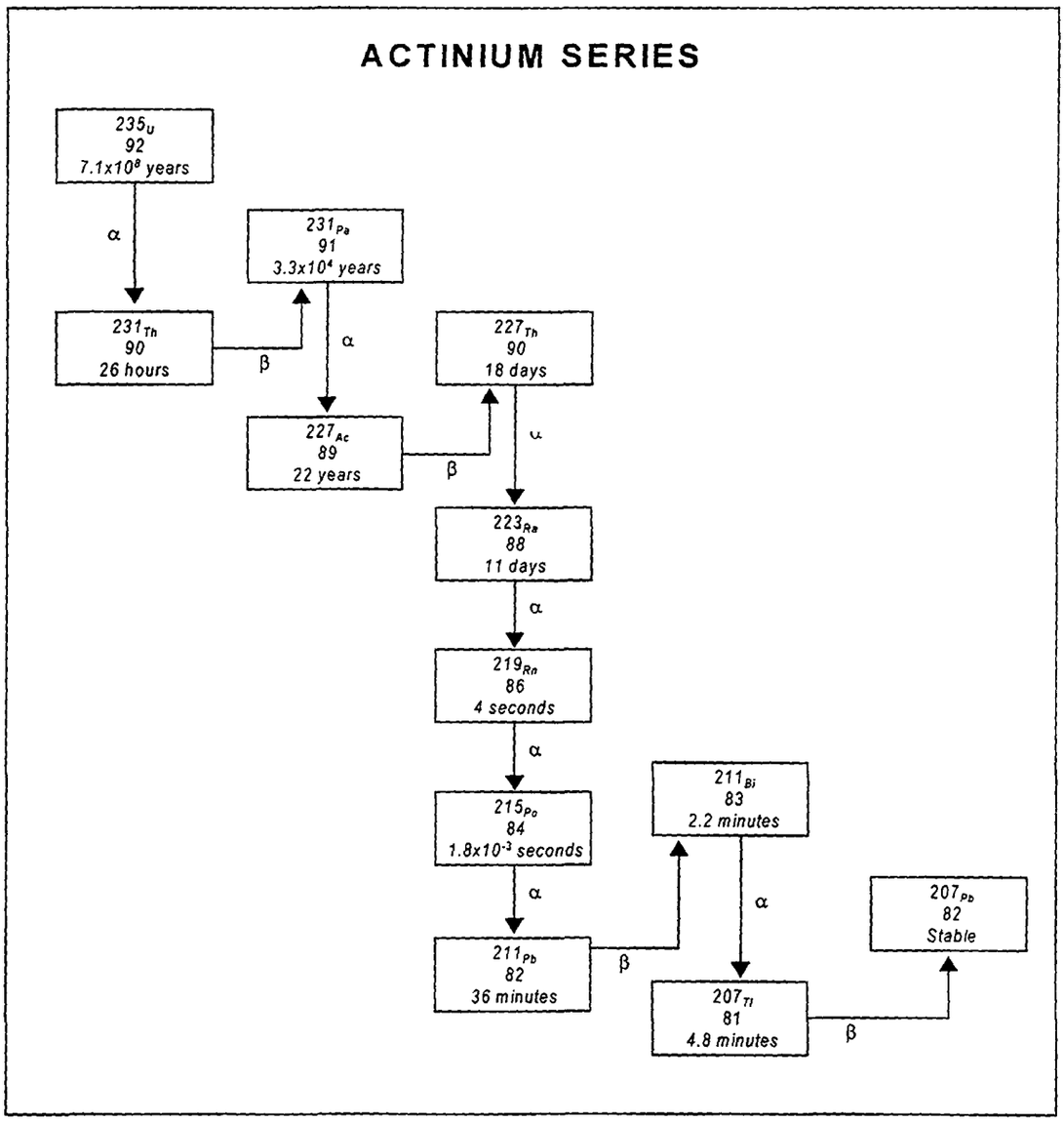
USEPA, 1997. "Appendix D - Glossary." Water on Tap: A Consumer's Guide to the Nation's Drinking Water. U. S. Environmental Protection Agency, EPA 815-K-97-002. July, 1997. http://www.pueblo.gsa.gov/cic_text/health/watertap/appd.html

USEPA, 1999. "Public Drinking Water System Programs." Environmental Protection Agency. Office of Ground Water and Drinking Water. 1999. http://www.epa.gov/safewater/pws/pwss.html

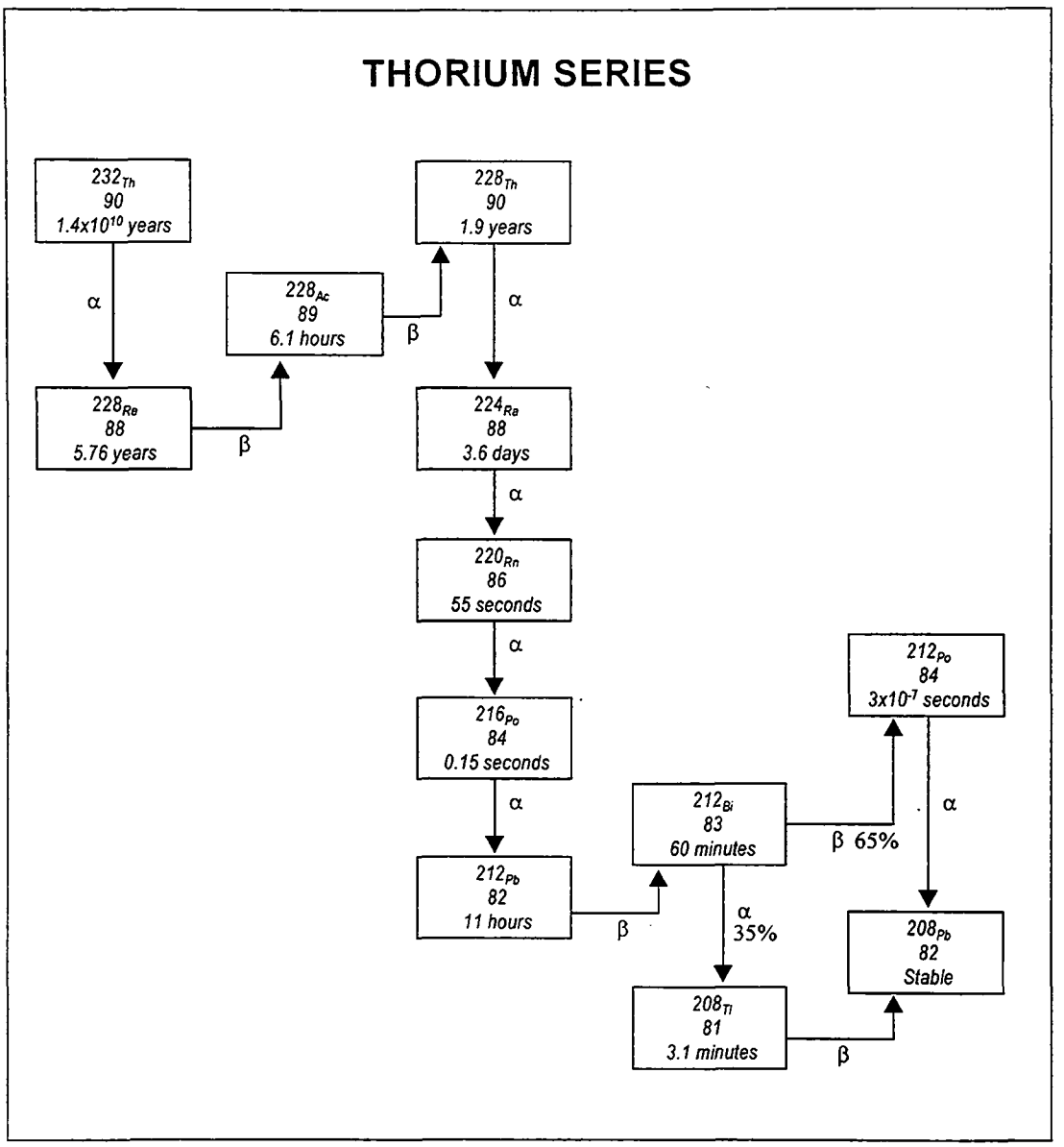
WHO, 1993. "Radiological Aspects." Guidelines For Drinking Water Quality: Recommendations. Second Edition, Volume 1. World Health Organization.

E. The Actinium, Thorium and Uranium Decay Series

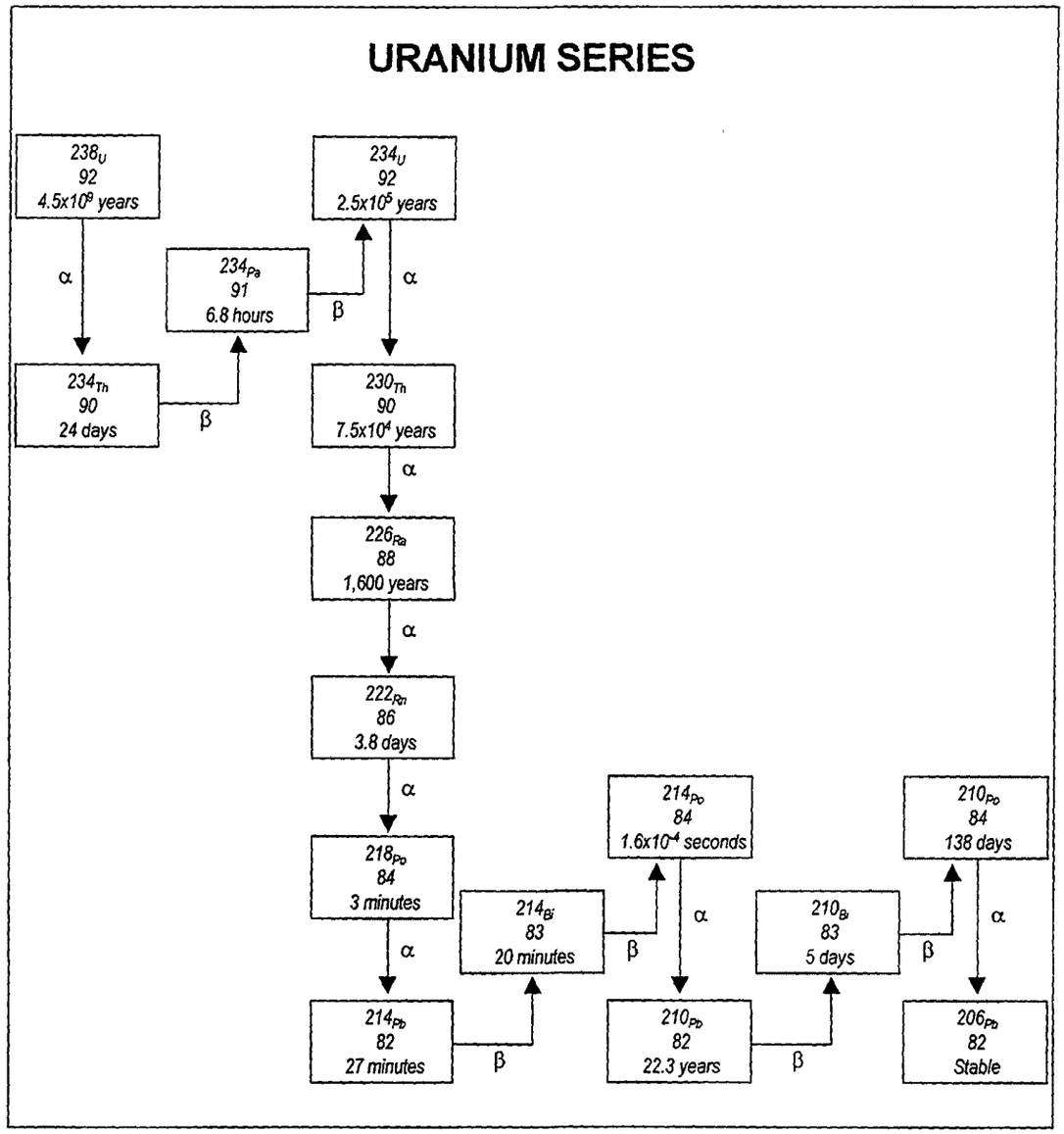
The following three graphics are used to explain the decay series for actinium, thorium and uranium. The number located at the bottom of each box indicates the length of time—in years, days, minutes, or seconds—that a particular element (or isotope) in the decay series takes to lose half of its activity. At this point, it either transforms into a lower energy state of the same element, or transforms into a different element. Alpha and beta radiation given off during the decay series is indicated by the alpha and beta symbols appearing next to the lines between boxes.



E. The Actinium, Thorium and Uranium Decay Series (Continued)



F. The Actinium, Thorium and Uranium Decay Series (Continued)



II - Occurrence

A. Introduction

In order to estimate the total national costs and benefits of potential revisions of the MCLs for radionuclides, it is necessary to develop updated national estimates of the occurrence and exposure to these radionuclide contaminants in drinking water. Occurrence data and associated analyses provide indications of the number of public water supply systems with concentration of radionuclides above the revised MCL as well as the population served by these systems. Monitoring and treatment costs can be estimated from the occurrence data. Parsa (1998) outlined some of the impacts that short-lived radionuclides, such as radium-224, may have on the Safe Drinking Water Act.

To help provide a better understanding of radiation and any possible revisions to the MCLs, a brief discussion of the fundamentals of radiation is presented. There are approximately 2,000 known radionuclides which are all are isotopes of elements that break down by emitting radiation. Radioactive decay occurs through emission of alpha particles, beta particles, or gamma rays. Radionuclides can be categorized:

- As naturally occurring or man-made;
- By type of radioactive decay (alpha, beta or gamma emission); and
- By radioactive decay series.

The natural radionuclides include the primordial elements that were incorporated into the Earth's crust during its formation, the radioactive decay products of these primordial elements, and radionuclides that are formed in the atmosphere by cosmic ray interactions. Man-made radionuclides are produced through the use of nuclear fuels, radio pharmaceuticals, and other nuclear industry activities. Man-made radionuclides have also been released into the atmosphere as the result of atmospheric testing of nuclear weapons and in rare cases, accidents at nuclear fuel stations, and discharge of radio pharmaceuticals.

The two types of radioactive decay that carry the most health risks due to ingestion of water discussed in this document are alpha emitters and beta/photon emitters. Many radionuclides are mixed emitters with each radionuclide having a primary mode of disintegration. The naturally-occurring radionuclides are largely alpha emitters, though many of the short-lived daughter products emit beta particles. The alpha emitting radionuclides, discussed in this document, include radiums, uranium, and polonium.

Man-made radionuclides are predominantly beta/photon emitters and include those that are released to the environment as the result of activities of the nuclear industry, but also include releases of alpha-emitting plutonium from nuclear weapon and nuclear reactor facilities (Ibrahim and Culp, 1989). Appendix B of the 1976 National Interim Primary Drinking Water Regulations lists the 168 man-made beta/photon emitters specifically regulated. One naturally occurring beta

emitter, lead-210, is of potential concern as a contaminant in drinking water due to the low concentration at which it causes the 4 millirems per year (mrem/yr) dose equivalent, which is the aggregate MCL for beta/photon emitters.

The natural radionuclides involve three decay series, which start with uranium-238, thorium-232, and uranium-235, and are known collectively as the uranium, thorium and actinium series. These decay series are shown in Section I.E of this document. Each series decays through stages of various nuclides which emit either an alpha or a beta particle as they decay and terminates with a stable isotope of lead. Some of the radionuclides also emit gamma radiation, which accompany the alpha or beta decay. The uranium series contains uranium-238 and-234, radium-226, lead-210, and polonium-210. The thorium series contains radium-228 and radium-224. The actinium series contains uranium-235.

B. Previous Sources of Radionuclide Occurrence Data

In 1985, the EPA released a nationwide occurrence study of radon and other naturally occurring radionuclides in public water supplies called the National Inorganic and Radionuclides Survey (NIRS). Longtin (1988) summarized NIRS and compared it with data from other studies that existed at that time. Among those data compared to NIRS was that of Hess et al., (1985) who had described the occurrence of radioactivity in public water supples in the United States. The objective of NIRS was to characterize the occurrence of a variety of constituents present in community ground-water supplies (finished water) in the United States, and its territories. The gross alpha- and gross beta-particle activities were surveyed to define the potential combined contribution to radioactivity in drinking water from the naturally occurring and man-made radioisotopes. The survey included a random sample from 990 collection sites stratified into four categories, and based on population served which represented about 2 percent of drinking water supplies in each category. Results of NIRS were used to develop the proposed radionuclide rule in 1991 (Federal Register Vol. 56, No. 138). There has not been a comparable national survey for radionuclides since. The analyses quantified by NIRS include radium-226, radium-228, radon-222, uranium (mass analysis), and gross alpha-particle, and gross beta-particle activities.

The major source of data for man-made radionuclides is the Environmental Radiation Ambient Monitoring System (ERAMS), which is published quarterly in the Environmental Radiation Data (ERD) reports. The ERD reports provide concentration data on gross beta-particle activity, tritium, strontium-90, and iodine-131 for 78 surface-water sites that are either near major population centers or near selected nuclear facility environs. Other sources of information include the EPA's Safe Drinking Water System (SDWIS) and data collected from the USGS National Stream Water Quality Accounting Network (NASQAN; Alexander and others, 1996). The EPA SDWIS database is used for compliance purposes and is therefore limited to information on violations; typically insufficient for national occurrence assessments. The USGS NASQAN data were collected from surface-water sources that may, or may not, be used for drinking water purposes (Zapecza and Szabo,1987).

C. Occurrence of Radium

Radium isotopes occur as decay products in each of the three decay series previously mentioned. The two isotopes with a MCL include radium-226, a decay product in the uranium series, and radium-228, a decay product in the thorium series. Uranium and thorium are ubiquitous components of rocks and soils therefore, radium radionuclides are also ubiquitous trace elements in rocks and soils.

The occurrence of radionuclides in ground water depends first on the presence and solubility of the parent products. Each radioactive decay product has its own unique chemical characteristics that differ from the radionuclide parent. Consequently, the occurrence and distribution of a parent radionuclide in solution does not necessarily indicate the presence of a daughter radionuclide in solution. For example, uranium (parent of radium-226) tends to be the least mobile in oxygen-poor ground water, and tends to be strongly adsorbed onto humic substances. Conversely, radium tends to be most mobile in reducing ground water that is chloride-rich with high concentrations of total dissolved solids (Zapecza and Szabo, 1986, Kramer and Reid, 1984). Radium behaves similarly to other divalent cations such as calcium, strontium, and barium. Therefore, in aquifers with limited sorption sites, radium solubility can be enhanced by the common ion effect in which competing cations are present in abundance and occupy sorption sites keeping the radium in solution. Recently, high concentrations of radium were found to be associated with ground water that was geochemically affected by agricultural practices in the recharge areas by strongly enriching the water with competing ions such as hydrogen, calcium, and magnesium (Szabo and dePaul, 1998). Radium-228 was detected in about equivalent concentrations as radium-226 in the aquifer study in New Jersey (Szabo and dePaul, 1998).

The process of "alpha recoil" of alpha-emitting radionuclides also enhances their solubility. When an alpha particle is ejected from the nucleus of a radionuclide during decay, the newly created progeny radionuclide recoils in the opposite direction. The energy associated with this recoil is 10^4 to 10^6 times larger than typical chemical bond energies (Cothern and Rebers, 1990) and can cause atoms on the surface of a grain to be recoiled directly into the water in pore spaces. To date, measured radium in surface waters has been traced back to discharge from ground water (Elsinger and Moore, 1983).

1. Occurrence of Radium-226

Radium-226 is the fifth member of the uranium-238 series, has a half-life of approximately 1,622 years, and decays by alpha-particle emission. Uranium forms soluble complexes under oxygen-rich conditions, particularly with carbonates. Uranium precipitates from ground water under oxygen-poor conditions and can be concentrated in secondary deposits (Cothern and Rebers, 1990). Consequently, uranium can be expected to range widely with different geologic environments. Because the chemical behavior of uranium and radium are vastly different, the degree of mobilization of the parent and product are different in most

chemical environments. Szabo and Zapecza (1991) detail the differences in the occurrence of uranium and radium-226 in oxygen-rich and oxygen-poor areas of aquifers.

The most significant radium-226 occurrence in ground water in the nation is concentrated in the North-central states, including southern Minnesota, Wisconsin, northern Illinois, Iowa, and Missouri (Zapecza and Szabo, 1987). The drinking-water wells tap deep aquifers of Cambrian and Ordovician sandstones and dolomites and Cretaceous sandstones. These aquifers tend to have limited sorption sites and radium solubility is enhanced by the common ion effect wherever total dissolved solids are high (Cothern and Rebers, 1990).

Radium-226 is also found in high concentrations in water derived from aquifers straddling the Fall Line of the southeastern states from Georgia to New Jersey (King and others, 1982, Szabo and dePaul, 1998). These aquifers are composed of sands which contain fragments of uranium-bearing minerals derived from the granites of the Blue Ridge and Piedmont provinces. Sands of the Coastal Plain where uranium and radium concentrations tend to be the highest (Zapecza and Szabo, 1986) were directly derived from these granites as fluvial deposits. Furthermore, at least in northern Illinois, reduction of sulfate limits co-precipitation of radium-226 with barium sulfate, another mechanism which would tend to limit dissolved radium-226 (Gilkeson, and others, 1983).

Activities of radium-226 equaled or exceeded the minimum reporting level of 0.18 pCi/L in 40.2 percent of the sites surveyed for NIRS (Longtin, 1988). The national median activity (of positives) was 0.39 pCi/L and the maximum was 15.1 pCi/L. Approximately 1 percent of the samples were above 5 pCi/L.

2. Occurrence of Radium-228

Radium-228 is the second member of the thorium-232 series, has a half-life of about 5.7 years, and decays by beta-particle emission. Thorium is extremely insoluble (Cothern and Rebers, 1990) and thus is not subject to mobilization in most ground-water environments. The relatively short half life of this isotope limits the potential for transport of unsupported radium-228 relative to that of the longer lived radium-226 isotope. Consequently, although radium-228 is chemically similar to radium-226, its occurrence distribution can be different. Michel and Cothern (1986) developed a national model to depict the occurrence of radium-228. There was little available occurrence data on radium-228; the model and associated data showed, however, that radium-228 activities tend to be the highest in arkosic sand and sandstone aquifers. Generally, the areas associated with the highest potential for radium-228 include the Coastal Plain aquifers that straddle the Fall Line from Georgia to Pennsylvania. Large areas of northern Illinois, Iowa, Minnesota, and Wisconsin were also ranked as having a high potential for radium-228 occurrence due to the sandstone aquifers and presence of high total dissolved solids. In contrast, aquifers that were mostly alluvial or glacial sand and gravel aquifers have a low potential for radium-228 in those same states. Other parts of the country such as areas in Colorado, Montana, and California are underlain by granitic rock that have the prerequisite

geochemical characteristics for radium-228 occurrence but the aquifers are not used extensively for water supplies. Other areas such as the High Plain aquifers and other locations where alluvial valley and sandstone aquifer material was derived from feldspathic minerals are considered to have medium potential for radium-228 occurrence.

The minimum reporting level (1 pCi/L) for radium-228 was exceeded at 11.9 percent of the sites sampled for NIRS (Longtin, 1988). The median (of positives) activity of radium-228 was 1.47 pCi/L and the maximum was 12.1 pCi/L. About 90 percent of the samples were below the minimum reporting level and about 1.7 percent exceeded 3.0 pCi/L.

3. Occurrence of Radium-224

Radium-224 is the fifth member of the thorium-232 decay series, has a half life of about 3.6 days, and decays by alpha-particle emission. There has not been a national survey or model for the occurrence of radium-224. Radium-224 has been determined to be present in ground water in the South Carolina Coastal Plain in investigations using radium to trace ground-water discharge (Elsinger and Moore, 1983). The relatively short half life of radium-224 limits the time frame that samples must be screened for gross alpha emissions. The U.S. Geological Survey working in cooperation with the New Jersey Department of Environmental Protection, has recently shown that a number of public water systems in the New Jersey would be out of compliance with gross alpha-particle activity standard if analyses were completed within 48 hours to account for the presence of radium-224 (Szabo and others, 1998). In response to these findings, the U.S. Geological Survey worked with EPA and the American Water Works Association to collect preliminary occurrence data on radium-224 and several other radionuclides as part of a reconnaissance study of selected radionuclides in the United States (as described in more detail below).

D. Occurrence of Alpha Emitters

The alpha-emitting radionuclides of concern to drinking water standards include those with half lives that are considered long enough to travel through distributions systems. Radionuclides having a half-life of 1 hour or less were not considered in the group proposed for regulation in the 1991 proposed rule (Federal Register Vol. 56, No. 138) because they most are likely to decay prior to consumption of drinking water. Alpha emitting radionuclides with a half-life longer than 1 hour include those shown in Table II-1.

Table II-1 Alpha-Particle Emitters from Actinium, Uranium and Thorium Decay

Decay series	*Alpha-particle emitting radionuclides with half-life longer than one hour*
Actinium	U-235, Po-231, Th-227, Ra-223
Uranium	U-238, U-234, Th-230, Ra-226, Rn-222, Po-210
Thorium	Th-232, Th-228, Ra-224, Pb-212

The crustal abundance of actinium (parent of U-235) is very low (less than 1% of either uranium or thorium) in comparison with the other decay series. Therefore, the actinium series is not considered an important source of radionuclides in drinking water.

The geochemistry and occurrence of most of the alpha emitters has been described in previous sections with the exceptions of radon-222 (Rn-222), lead-212 (Pb-212), and polonium-210 (Po-210). There is little or no occurrence information on the isotopes lead-212 and polonium-210. The concentrations of polonium-210 that can cause health concerns is quite low (see Section III). Although polonium-210 is very particle reactive and is not expected to be found in high concentrations in natural waters (Harada, et al., 1989), it has been found in relatively high concentrations in a shallow aquifer in west central Florida (Harada, et al., 1989). The aquifer is characterized by uranium-enriched phosphatic strata and in acidic water that is sulfide bearing, and relatively high in radon-222. Such water is geochemically rare in the environment. Thus, it is likely that occurrence of polonium-210 is limited to unusual geochemical environments that are not commonly used as sources of public water supplies but this hypothesis has not been tested. The U.S. Geological Survey included polonium-210 in the recent reconnaissance study (see below). The short half-life of lead-212 (10.6 hours) has precluded its analysis in this study. The national distribution of radon-222 in ground water is detailed elsewhere (Lassovszky, written comm., 1998).

E. Occurrence of Beta/Photon Emitters

There are two major sources of information on beta/photon emitters in drinking water supplies. The ERAMS (Environmental Radiation Ambient Monitoring System) collects and publishes quarterly data on gross beta, tritium, strontium-90, and iodine-131 in surface water and ground water; however, the sampling is not correlated to public water supplies. Gross beta concentrations ranged from 0.3 to 17.8 pCi/L, with an average value of 3 pCi/L over a three year period, well below the 50 pCi gross beta-particle activity screening level that would require testing of individual isotope concentrations. There were no instances in the ERAMS database where the gross beta concentration of a surface water supply exceeded the 50 pCi/L level. Anthropogenic sources of beta/photon-emitting radionuclide contamination can include Department of Energy nuclear facilities, commercial nuclear power plants, institutional sources (research facilities, hospitals, and universities) and industrial sources (laboratories, pharmaceutical companies). Atmospheric fallout is not currently a major source of contamination due to declining levels of bomb-generated beta-emitting radionuclides as the result of the atmospheric testing moratorium.

Nationwide gross beta-particle activity for groundwater were determined from the NIRS (National Inorganics and Radionuclides Survey; Longtin, 1988). Of the 990 public water supplies sampled, 9 had gross beta-particle activities above the 50 pCi/L screening level. The specific radionuclide(s) responsible for these high gross beta-particle activities is unknown, although an analysis of the data indicates it is not radium-228. This result is consistent with the

findings of Welch and others (1997) who also found that radium-228 was a major contributor to gross beta-particle activity. Based on this data, it can be projected that approximately 1% of PWSs using ground water would be required to do further testing (Longtin, 1988).

There are 4 man-made beta/photon emitters that can exceed the 4 mrem/year MCL at concentrations less than the 50 pCi/L screening level, including strontium-90, rubidium-106, iodine-129 and iodine-131. A naturally occurring beta/photon emitter, lead-210, is currently not regulated, also has a concentration exceeding 4 mrem the screening level. An EPA study (NFIC, 1974) of central Florida ground waters found concentrations from 0.0 to 0.6 pCi/L lead-210 in 25 municipal water supplies. A concentration of 7.6 pCi/L was found in one private well. Cothern and others (1986) reported an average population-weighted concentration of lead-210 in U.S. community water supplies (surface and ground water supplies) of <0.11 pCi/L. Lead-210 concentrations in surface water are expected to be very low due to sorption onto aquifer sediments. Lead tends to sorb more strongly to sediment than most other trace elements (Cothern and Rebers 1990).

F. Occurrence of Uranium

Uranium is predominately found in ground water in the Colorado Plateau, the Western Central Platform, the Rocky Mountain System, Basin and Range, and the Pacific Mountain System (Zapecza and Szabo, 1987; Hess and others, 1985). Concentrations of uranium in ground water in the eastern United States is typically low. Uranium is found in concentrated amounts in granite, metamorphic rocks, lignites, monazite sand and phosphate deposits, as well as in the uranium-rich minerals of uraninite, carnotite, and pitchblende. Uranium must be oxidized before it is transported into groundwater but once in solution, it can travel great distances. Elevated concentrations of uranium can be found in both surface and ground water (Zielinski and others, 1997).

Natural uranium in rock contains three isotopes: uranium-234 (0.006% by weight), uranium-235 (0.72 %) and uranium-238 (99.27 %). The activity to mass ratio of the sum of the three radioisotopes in rock is 0.68 pCi/ug. The crustal abundances of uranium are not duplicated in ground water. Uranium-234 is enriched in water relative to rock when standardized to uranium-238 in the water. Uranium-234 activity to uranium mass ratio in water varies from that in rock. In order to convert ug/l to pCi/L a ratio of U-234/U-238 of 0.9 to 1.3 is typically observed to account for the excess alpha-particle activity from uranium-234 in the water as opposed to the expected activity if crustal abundance were preserved. This uranium isotopic ratio may vary regionally, as well as seasonally.

The NIRS database contains uranium concentrations reported as mass for 990 public water supplies. Using a conversion factor of 1.3, derived from values reported in the "Nationwide Survey of Radon and Other Natural Radioactivity in Public Water Supplies (EPA 1985), 1% of the systems exceeded a level of 30 pCi/L, 3% exceeded 10pCi/L, and 72% exceeded 0.8 pCi/L. Based on an EPA study (EPA/ORNL, 1981) a range of 0.01 - 582.4 pCi/L

uranium was detected in surface water samples, with an average of 1.06 pCi/L. Domestic ground-water sources ranged from 0.07 to 653 pCi/L, with an average of 1.73 pCi/L. The population weighted average concentration or uranium in U.S. domestic waters was estimated to be 0.8 pCi/L. Most of the water supplies with high uranium concentrations are in the very small system size category, those serving 25-500 people. In the NIRS data 0.20% of smallest systems in the population range of 25-100 people and 0.10% in the 101-500 size systems had concentration between 50-60 μg/L. Only 0.10%, a total of one sample had concentration between 80 and 90 μg/L.

G. SDWIS Data

The SDWIS/FED data, which reports violations of the standards, provides some insights on the magnitude of the radionuclides violations in finished public drinking water and, by virtue of the states in which the violations occur, a sense of the distribution of radionuclides. However, as a violations report, data are lacking on sub-MCL levels which would have had some utility in evaluation of national impacts of, for instance, an option of a radium-228 limit at 3 pCi/l.

Additionally, because there have been widespread inaccuracies and/or incompleteness in reporting, coupled with, or brought about by the general confusion over the MCL since the 1991 proposal, these data do not provide a useful quantification of radionuclide occurrence. But these data do provide a pattern of occurrence, which helped USGS and EPA target the recent reconnaissance survey of select radionuclides discussed later in this section. Table II-2 below summarizes the violations data since the 1976 radionuclides rule. Because Table II-2 summarizes the major radionuclides groups and includes some information on several additional non-regulatory categories not depicted on the table, some columns do not add to the reported total. Figure II-1 presents a graphical display of the violations data reported in Table II-2.

In 1996, fifteen states reported violations of the radium-226 and 228 MCL in mostly very small to small systems and occasional violations in medium to large systems in Kansas, and South Dakota. In all, the following states had some occurrence of radium and reported violations of radium-226 and radium-228, alone or in combination: Arkansas; Arizona; Connecticut; Florida; Illinois; Indiana; Kansas; Missouri; North Carolina; Nebraska; Pennsylvania; South Dakota; Texas; Virginia; Wisconsin.

Figure II- 1 - Community Water Systems with MCL Violations from 1976 to 1998

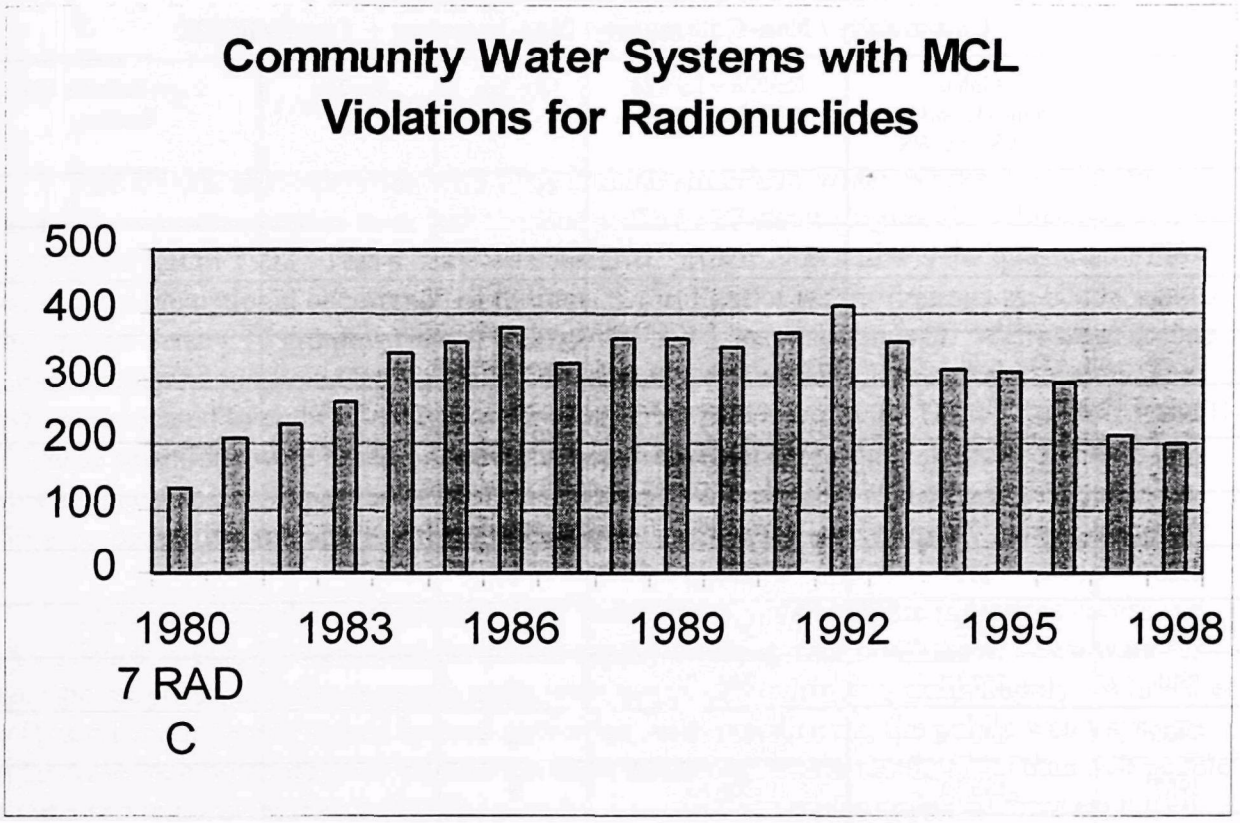


Table II-2. Number of Systems Exceeding Standards 1976-1998

Numbers of Systems Exceeding Standards Community / Non-Community: Non-transient + Transient					
Year	Alphas (minus U and Rn) CWS / NCWS	Ra-226 + Ra-228 CWS / NCWS	Ra-226	Ra-228	Gross Beta Particle Emitters
1976	1				
1977	1				
1978	1	5			
1979	20	49 / 1	4		
1980	81	242 / 3	4		
1981	157	282 / 1	4		1
1982	149	261/1	4		
1983	171	291 / 1	4		
1984	205	357 / 3	0		
1985	232 / 2	344 / 3	3		
1986	210 / 1	361 / 4	3		1
1987	186 / 1	306 / 5	3		
1988	227 / 3	350 / 3	4		
1989	243 / 20	317 / 7	11 / 2	2	2
1990	340 / 6	241 / 28	14 / 5	5 / 6	1
1991	253 / 28	289 / 4	20 / 5	9 / 6	1
1992	308 / 31	308 / 3	33 / 5	19 / 6	1
1993	271 / 28	240 / 3	33 / 5		1
1994	242 / 13	234 / 1	30 / 3	24 / 5	1
1995	220 / 28	252 / 3	27 / 0	20 / 0	
1996	200 / 6	300 / 2	26 / 0	20 / 0	1
1997	129 / 1	205 / 1	8 / 8	9 / 9	
1998	125 / 0	195 / 1			
SUMMARY: ALL EXCEEDENCES OF RADIONUCLIDES BY SYSTEM SIZE					
Very Small	Small	Medium	Large	Very Large	TOTAL / CWS / NCWS
5,167	3,074	889	719	31	10,159 / 9,980 / 279

H. USGS/EPA Preliminary Reconnaissance of Selected Radionuclides

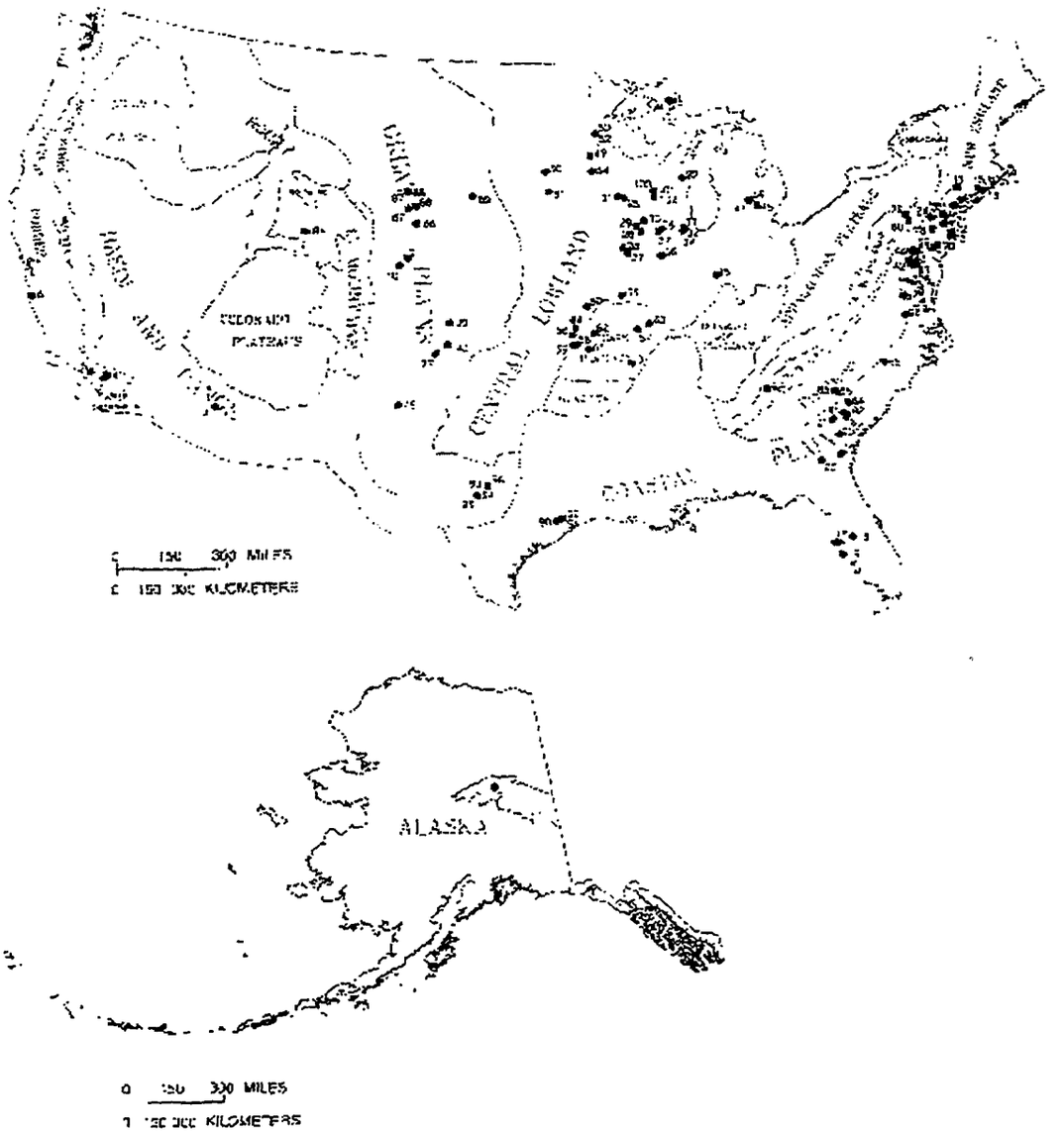
1. Approach

a. The Radium Survey

The USGS, in cooperation with EPA and the American Water Works Association, analyzed raw-water samples from 103 targeted wells in 27 states (Figure II-2) during the late summer and fall of 1998. This survey was designed primarily to address the question of the potential for the national occurrence of radium-224 in light of recent findings by USGS and the state of New Jersey, in ground-water resources of New Jersey (see above). Wells were targeted in areas known or suspected to have high concentrations of radium in the ground water. The wells were located in eight of the major physiographic provinces in the United States (Figure II-2). These selections were based on previous occurrence data, geologic maps, and known or inferred geochemical properties of aquifers. The AWWA assisted in the final well selection efforts by contacting members in the targeted areas and requesting voluntary participation.

The sampled wells were located in 27 states and 8 physiographic provinces across the U.S. Of these wells, 95 were used for public supply drinking water withdrawal, two were for domestic purposes, and two research wells were not used for drinking water supply. While the exact numbers of people served by each individual well is unknown, the public water systems surveyed serve a broad range of population sizes including several serving less than 500 people and one serving greater than 1,000,000 people. Raw-water samples collected from each well were analyzed for radium-224, radium-226, radium-228, polonium-210, and lead-210 (Table II-3). Comparisons of analytical methods and quality assurance/quality control analyses will be documented elsewhere in a peer-reviewed publication. The radium-226 and radium-228 analyses were performed chiefly to evaluate co-occurrence with radium-224. The polonium-210 and lead-210 analyses were done because of the general lack of data on these radionuclides and their associated health risks even at low concentrations (see Section III for health risks).

Figure II-2. Map of the physiographic provinces and the locations of the wells with completed radium-224 analysis.



Major physiographic provinces of the United States and locations of wells with completed radium-224 analysis.

Table II-3. Radionuclides Analyzed in the USGS Survey, the Associated Analytical Method, and Number of Samples Analyzed by Each

	Ra-224	Ra-226	Ra-228	Po-210	Pb-210
Analytical Methods (number of samples)	Alpha spectroscopy (90), Gamma spectroscopy (12), Split with both (15), Split with both in duplicate (36)	Alpha spectroscopy (90), Gamma spectroscopy (12), Split with both (15), Split with both in duplicate (36)	Beta counting (90), Gamma spectroscopy (12), Split with both (15), Split with both in duplicate (36)	Alpha spectroscopy (113)	Column extraction and beta counting (109)

b. The Gross-Beta Particle Data Collection Effort

An additional data collection effort was completed in the summer of 1999 to analyze targeted beta-particle emitting radionuclides from a small number of public water systems that had shown relatively high levels of beta/photon emitters during the original NIRS survey. The NIRS database includes gross beta-particle activities for each system sampled. In some cases the gross beta-particle activity was relatively high (exceeding 25 pCi/L); however, tests were never run to determine which radionuclides were contributing to the beta-particle activity. Not all the public water systems with high gross beta-particle activities participated in this effort. Of the 26 pubic water systems contacted for this effort none could ascertain which wells in their systems were originally sampled as part of NIRS. Consequently, although all efforts were made to include as many of the original systems as possible, it is presently unknown if the wells sampled match those in NIRS. The radionuclide analyses for this data collection effort included; short-term (48 hour) gross beta-particle and gross alpha-particle activities, long-term (30 days) gross beta-particle and gross alpha-particle activities, tritium, strontium-89, strontium-90, cesium-134, cesium-137, iodine-131, uranium-234, uranium-235, uranium-238, radium-228, radium-226, lead-210, and cobalt-60.

2. Results of the Radium Survey

a. Radium-224

Concentrations of radium-224 in samples analyzed for the survey ranged from less than 1 pCi/L to 73.6 pCi/L. The maximum concentration of radium-224 (73.6 pCi/L) was found in a Coastal Plain aquifer in Maryland. Most of the samples analyzed contained radium-224 in concentrations less than 1 pCi/L with half of the concentrations less than 0.3 pCi/L (Table II-4). Nine percent of the samples exceeded 7 pCi/L of radium-224 (Table II-5). Concentrations of radium-224 and radium-228 were highly correlated (Table II-6; Figure II-4). Both radium-224 and radium-228 are derived from the thorium-232 decay series thus the high correlation is

expected. The correlation of radium-226 (derived from the U-238 series) with radium-224 is not as high as the correlation with radium-228 (Table II-6; Figure II-5).

The concentration of radium-224 plotted against those of the corresponding radium-228 concentrations group tightly about the 1:1 line (Figure II-4) particularly above 1 pCi/L. The exception is the few samples from the Coastal Plain with extremely high radium-224 concentrations. This radium-224 rich water samples, from Maryland and South Carolina, are from highly acidic water (pH < 4). The concentration of radium-224 does correlate with that of radium-226, but there is much more scatter around the 1:1 line than with radium-228 (Figures II-4 and II-5).

Table II-4. Summary Statistics for the USGS Survey

Statistic	Ra-224 (all samples)	Ra-226 (all samples)	Ra-228 (all samples)	Po-210 (all samples)	Pb-210 (all samples)
Number of Sample	104	104	104	95	96
Arithmetic Mean	3.2	1.6	2.1	0.1	0.6
Median	0.3	0.4	0.5	0.01	0.5
Standard Deviation	10.1	2.8	7.9	0.5	0.5
Maximum	73.6	16.9	72.3	4.9	4.1

Table II- 5. Percent of Samples Exceeding Specified Concentrations

Radionuclide	Total number of samples	Percent of samples exceeding given concentration (pCi/L)					
		1	2	3	5	7	10
Ra-224	104	30	26	20	15	9	5
Ra-226	104	33	22	17	10	5	2
Ra-228	104	26	18	14	9	2	2
Po-210	95	3	1	1	1	0	0
Pb-210	96	10	3	1	1	0	0

Table II-6. Spearman Correlation Coefficients (significant above 0.05) of Selected Radium Isotope Concentrations

	Ra-224	Ra-226	Ra-228
Ra-224	1.0	0.69	0.82
Ra-226	---	1.0	0.63
Ra-228	---	---	1.0

Figure II-4 Correlation Between Radium-224 and Radium-228

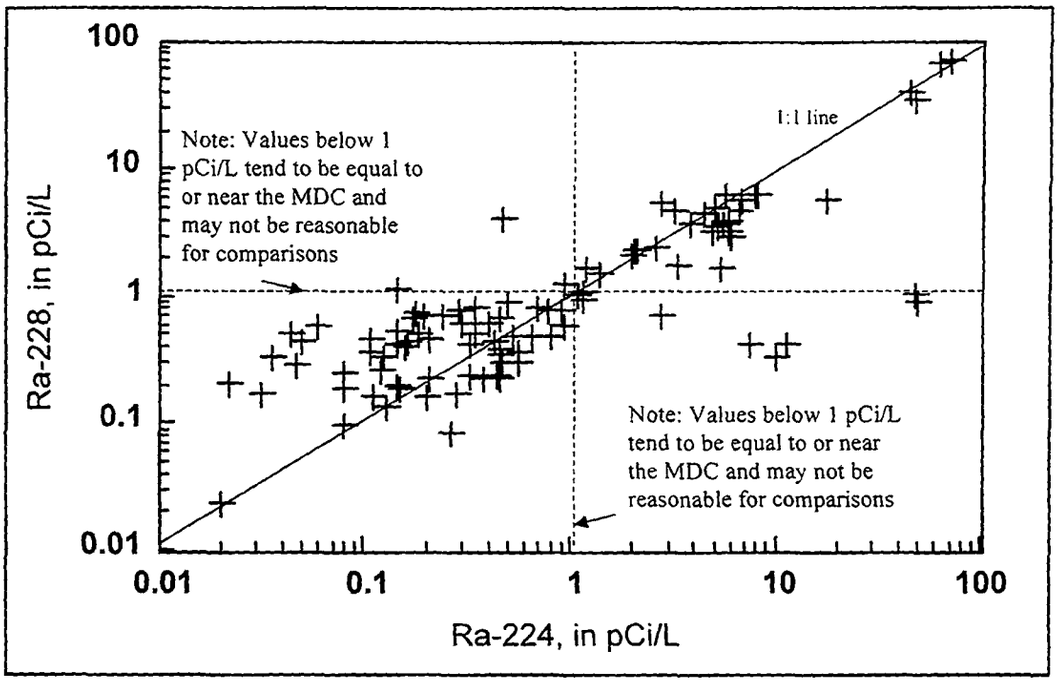


Figure II-5 Correlation Between Radium-224 and Radium-226

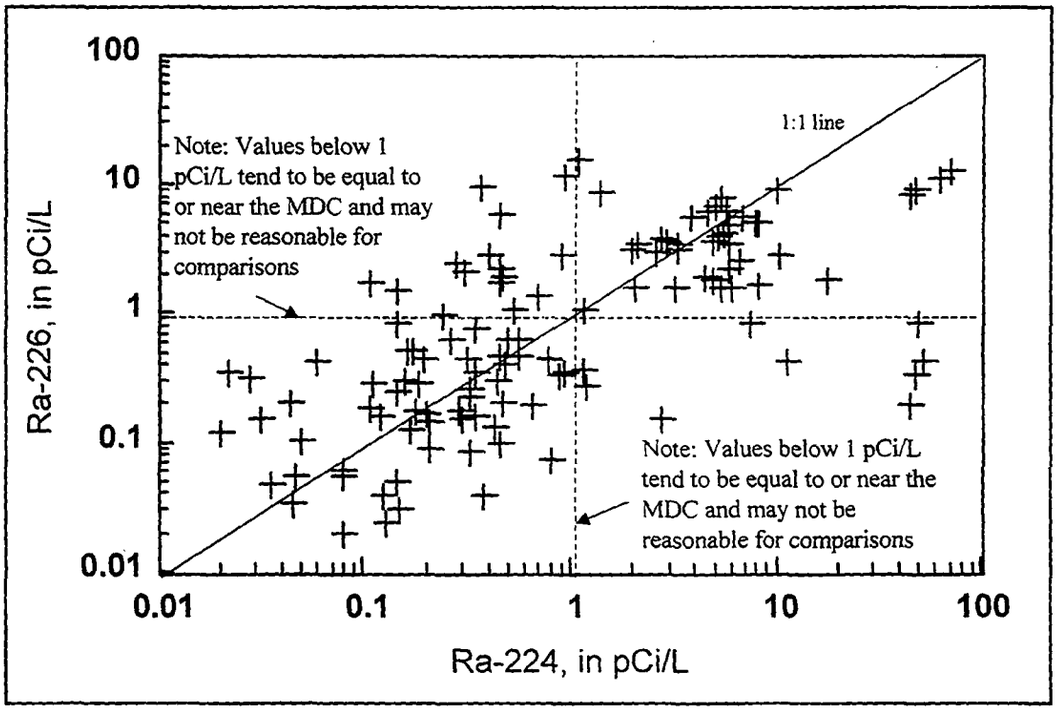
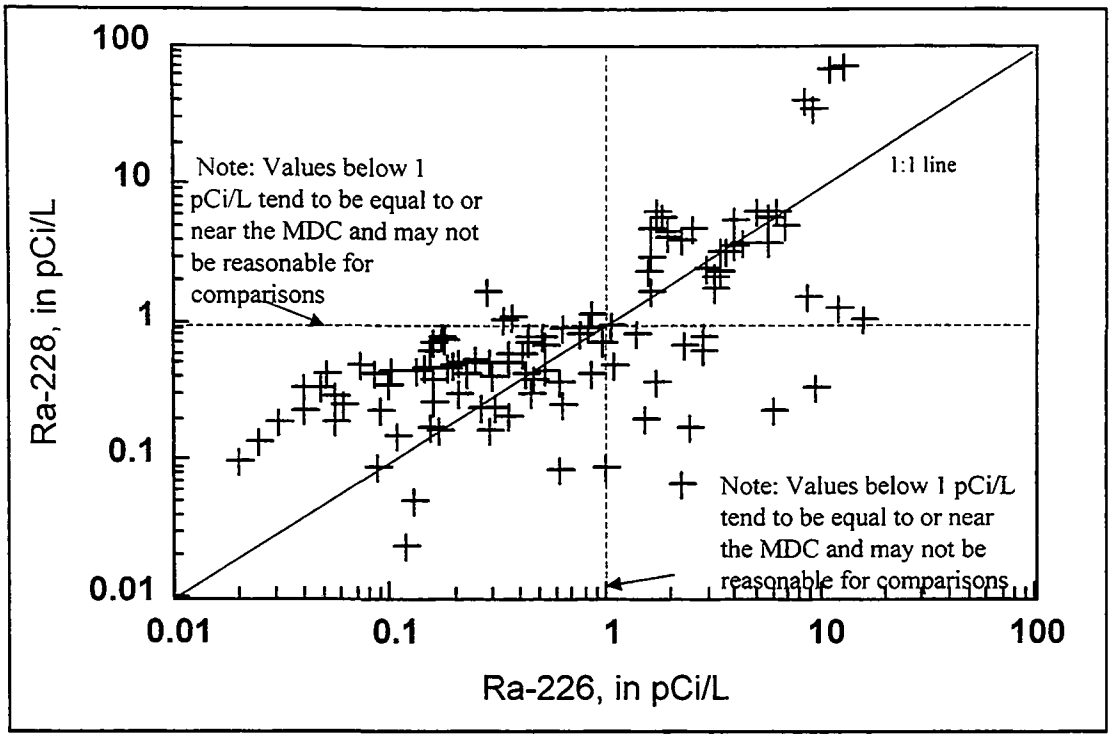


Figure II-6 Correlation Between Radium-226 and Radium-228



b. Radium-226 and Radium-228

Concentrations of radium-228 in samples analyzed for the survey ranged from less than 1 pCi/L to 72.3 pCi/L (Table II-4). The maximum concentration of radium-228 (72.3 pCi/L) was found in the same Coastal Plain well in Maryland with the maximum radium-224 concentration. The radium-228 concentrations correspond to radium-224 concentrations closely (Figure II-4). Most samples contained less than 1 pCi/L of radium-228. About 16% of the samples exceeded 3 pCi/L (Table II-5).

The maximum concentration of radium-226 was 16.9 pCi/L from a well in Iowa. Most of the concentrations of radium-226 were less than 1 pCi/L (median = 0.4 pCi/L). The rank correlation coefficients of radium-226 with radium-228 and radium-224 was 0.63 and 0.69 respectively (Table II-6; Figure II-5 and II-6).

The results, while not necessarily useful for revision of previous occurrence estimates of radium-226 and radium-228, provide some new information regarding the co-occurrence of these two isotopes with radium-224. Concentrations of radium-226 and radium-228 do correlate (Table II-6), but there is much scatter among the corresponding samples (Figure II-6). Many samples deviate from the 1:1 ratio. The radium-228 and radium-226 concentration data in this study are thus consistent with numerous other studies completed in the past few years which all

indicate that radium-228 and radium-226 are not found in a 1:1 ratio in ground water (Longtin, 1988).

3. Polonium-210 and Lead-210

The maximum concentration of polonium-210 was 4.9 pCi/L (Table II-4) in a sample collected from a private homeowner's well in the Coastal Plain of Virginia and exceeded 1 pCi/L (1.4) in only one other well. Preliminary assessments by the USGS Virginia District of the water chemistry in the well with the high polonium-210 concentration indicates that the radium isotope and lead-210 concentrations were all less than one, the pH was 7.5, the specific conductance was 259 microsiemens/cm, the turbidity was 0.53 NTU, the alkalinity was 119 mg/L as $CaCO_3$, and the dissolved oxygen concentration was 0.22 mg/L (David Nelms, USGS, person. comm. 1999). Thus low pH does not appear to control the mobility of the polonium in this aquifer (as in the Florida study described above). The geochemical characteristics that mobilize polonium remain unknown at this time.

The maximum concentration of lead-210 was 4.1 pCi/L in a sample collected from the Appalachian Province from Connecticut (Table II-4). Approximately 10% of the samples exceeded a lead-210 concentration of 1 pCi/L (Table II-45) but only one exceeded 3 pCi/L. The greatest frequency of detection was for samples from the Appalachian Physiographic Province, especially centered in the northeastern United States.

4. Results of the Gross Beta-Particle Emitters in Drinking Water Data Collection Effort

The concentrations of beta-particle emitting radionuclides and gross beta- and alpha-particle activities in 27 public-supply well-water (or in 2 cases, mixed surface or spring water and ground water) samples were determined in accordance with the USEPA accepted techniques outlined by Krieger and Whittaker (1980). Short- and long-term gross beta-particle activity exceeded 20pCi/L in only one sample (Table 6; 32.1 pCi/L within 48 hours of sampling, 21.3 pCi/L after 30 days). Two other samples contained gross beta-particle activity greater than 10 pCi/L (both very near 15 pCi/L within 48 hours and 30 days). Laboratory split samples of 2 of these 3 well waters produced nearly identical results, confirming the presence of the slightly elevated gross beta-particle radioactivity.

The concentration of a suite of the most common manmade and naturally occurring beta-particle activity emitting radionuclides was analyzed for in order to determine the source of the gross beta-particle activity. The manmade radionuclides determined were tritium (H-3), strontium (both Sr-89, Sr-90), cesium (both Cs-134, Cs-137), iodine (I-131), and cobalt (Co-60). The naturally occurring radionuclides sought were potassium (K-40), uranium (U-238, U-235, and U-234, with all three U radionuclides being alpha-particle emitters, but uranium-238 decays to beta-particle emitting Th-234 and Pa-234; and U-235 decays to beta-particle emitting Th-231), radium (Ra-228), and lead (Pb-210). The concentration of gross alpha-particle activity and that

of radium-226 were also analyzed to gain additional information regarding the possible presence of these naturally occurring radionuclide contaminants as well.

No sample contained detectable concentrations of any manmade radionuclide. Six samples had an analytical result for concentration of strontium-89 greater than 1 pCi/L, but the MDC of each of these samples was even higher than the analytical result indicating that because the laboratory did not use robust analytical techniques to minimize the MDC, a detectable concentration could not actually be verified. The MDC for tritium was 130 pCi/L and no concentration in excess of this value was determined; because the energy of the beta-particle emitted by tritium is very low, it is doubtful that the beta-particle emission of tritium is detected by a standard gross beta-particle analysis.

These 26 samples show no evidence of large amounts of beta-particle ingrowth (i.e. the gross-beta particle activities did not change much after 30 days). Ingrowth of beta-particle activity directly into the sample bottle after sample collection has been proposed by Welch et al., 1995, for ground waters containing significant amounts of dissolved uranium. The concentrations of uranium in the samples are low. Only one sample, that from Pennsylvania, contained a concentration of uranium-238 of about 3 pCi/L. A small net increase of just over 1 pCi/L was noted for the gross beta-particle activity 30 days after sample collection relative to the gross beta-particle activity determined within 48 hours. This difference, however, is less than the analytical precision and thus is not considered to represent an actual reproducible difference. For 19 of the 26 samples analyzed, the uranium-238 activity was less than 1 pCi/L; thus even if a beta-particle-emitting product of uranium-238 had ingrown into the sample, the amount of ingrowth would be so small that it could not be distinguished by the gross beta-particle activity determination when considering the poor resolution or precision of the method; i.e. the analytical error is greater than 1 pCi/L for every analysis of every one of the samples.

The sample with short-term gross beta-particle activity of 32.1 pCi/L (collected at a public water system in Massachusetts) had large contributions from potassium-40 (28.4 pCi/L). The gross beta-particle activity actually decayed to 21.3 pCi/L after 30 days in that sample. This result indicates that a potential, uncharacterized, short-lived radionuclide may be present in the water. Further radionuclide analysis of the water would be required to determine the nature of the radionuclide. The presence of naturally occurring potassium-40 is a significant source of beta-particle activity in the majority of the 26 samples analyzed – a result that is consistent with the five-region survey of sources of beta-particle activity in ground water presented by Welch et al., 1995. The concentration of potassium-40 exceeded 3 pCi/L in 9 of the 26 samples.

Radium-228 concentration exceeded the MDC in 17 of the 26 samples and were greater than 1 pCi/L in 4 of the samples with a maximum concentration of 3.1 pCi/L. For a select number of ground-water samples, therefore, naturally occurring radium-228 may be a significant source of beta-particle activity. Welch et al., 1995, demonstrate that the significance of radium-228 as a source of beta-particle activity in ground water can vary sharply from geologic region to geologic region. The 4 samples with the highest radium-228 concentration among these 26

samples originate from states with known naturally occurring radium-228 dissolved in aquifers: Illinois, Iowa, Minnesota, and Texas.

The lead-210 (Pb-210) concentrations were less than 1 pCi/L in all samples, though 2 of the samples did contain detectable lead-210 in concentrations just over 0.9 pCi/L. The concentration of lead-210 in most ground waters is generally less than one as confirmed by results of this initial nationwide survey. Thus, it is unlikely that lead-210 is a major source of gross beta-particle activity in typical ground water.

Other notable observations of results from this phase of sampling include the relatively high (greater than 10 pCi/L) gross alpha-particle activity in several (5 of 26, with 4 equal to or greater than the 15 pCi/L MCL for this constituent) samples. The potential contributions from short-lived alpha-particle emitters (e.g. gross alpha-particle activity decreased from greater than 20 pCi/L when analyzed within 48 hours to around 15 pCi/L after 30 days in a sample from Kendall County, Texas) are an indirect indication of the likely prevalence of these naturally occurring contaminants as well. The presence of short-lived radium-224 in ground water has been documented both in other studies (Szabo et al., 1998; Elsinger and Moore., 1983; Kraemer and Reid, 1984) and in this study via direct analysis of radium-224 concentration of 104 samples presented earlier in this document.

5. Summary of USGS Survey

The USGS survey was designed to target areas of known, or suspected, high concentrations of radium-224 as inferred by associated radium occurrence data, geologic maps, and other geochemical considerations. Thus, the survey is likely biased toward the extreme high end of the occurrence distribution for radium-224 and co-occurring contaminants such as radium-228. Approximately half of the samples were below the minimum reportable concentration of radium-226 and radium-228 in spite of the fact that public water systems were targeted in areas where high concentrations of radium were expected. Of the 104 samples, 21 exceeded the MCL for combined radium, and about 5 percent exceeded 10 pCi./L of radium-224, though several of these samples with pH less than 4.0 also contained detectable concentrations of thorium isotopes as well. Concentrations exceeded 1 pCi/L in about 10 percent of the samples analyzed for lead-210 and about 3 percent for polonium-210.

Radium-224 occurs in many of the wells sampled at concentrations that highlight the limitations of the present monitoring scheme for the gross alpha-particle standard. In addition, the contribution of radium-224 and its short-lived daughter products to gross alpha emissions was estimated with data from a concurrent study of ground-water supplies by the USGS in cooperation with the state of New Jersey (Szabo et al., 1998). In that study, gross alpha emissions were measured before the decay of radium-224 and after sufficient time had elapsed for radium-224 decay (about 18-22 days). In this way, the difference between the initial gross-alpha measurement and the final measurement is indicative of the contribution of radium-224 and all other alpha emitting isotopes that would decay within this time frame. The results

indicate that the contribution of radium-224 and its short-lived daughter products is approximately three times the concentration of radium-224 (Figure II-5). While this analysis was developed with a small data set in a restricted geographic range, it is based on a physical process and has important implications for such things as projections of radium-224 occurrence in association with gross-alpha concentrations. These results are also important in light of both the costliness and difficulty of the radium-224 analysis.

Concentrations of radium-228 were highly correlated with radium-224. Although this correlation was based on a limited number of data points, there is a physical basis to the correlation since both nuclides originate from the same decay chain. Therefore, there is potential for using radium-228 as a proxy indicator for the much shorter lived and infrequently sampled radium-224. In addition, the isotopic ratios of radium-226 to radium-228 were below 3:2 in many samples indicating that the gross alpha-particle screen that is currently used for combined radium (radium-226 + radium-228) compliance would be inadequate in many situations.

Polonium-210 and lead-210 are derived from the uranium-238 decay series; the decay series that produces radium-226. However, the survey was designed to assess radium-224; therefore results are possibly biased to areas that would more likely have isotopes in the Th-232 decay series. In addition, the correlations of radium-226 with radium-224 and radium-228 are only 0.51 and 0.61 respectively (table 4); consequently, the wells that were sampled may not be located in areas expected to have polonium-210 or lead-210. Within these constraints, the new data help to fill the gap in occurrence information that exists for these isotopes. Polonium-210 was found in concentrations exceeding 1 pCi/L in only two wells. At this time, these observations could not be associated with unique geochemical controls (as has been accomplished in a previous study in Florida; Harada et al., 1989) and further investigations would be necessary to infer anything more about the national distribution and occurrence of polonium-210.

Ten percent of the samples exceeded a lead-210 concentration of 1 pCi/L; however only one sample was greater than 3 pCi/L. The greatest frequency of detection was in the Appalachian Physiographic Province of the northeastern United States, especially in of Connecticut and Pennsylvania. The geochemical mechanism that controls lead-210 dissolution is also not well established and needs further study, though Pb is less soluble than Ra. In addition, lead-210, like polonium-210, is derived from a different decay chain than Radium-224 and it was therefore not considered in designing the study. One possible explanation for the frequent detection of lead-210 in concentrations greater than 1 pCi/L in the Appalachian region may be the high concentrations of radon-222 in ground water in this region (Zapecza and Szabo, 1987). As the radon in solution decays through a series of very short half-lived products to lead-210, a small fraction of the lead-210 may not be sorbed onto the aquifer matrix; thus, the higher the initial radon-222 concentration, the more likely measurable amounts of lead-210 would be found in the ground water. This hypothesis could not be tested however because radon-222 was not analyzed in this study.

Gross beta-particle activities were all below 50 pCi/L in water collected from public water systems that were sampled previously during the National Inorganics and Radionuclide Survey (NIRS) and had been found to contain gross beta-particle activity in excess of 20 pCi/L . It is uncertain if these samples were collected from the same wells as were sampled in the NIRS survey about a decade ago – because of insufficient data quality assurance, record keeping, and ongoing follow through, much of the data from NIRS could not be verified; further, even when the actual operator who had collected the sample for NIRS was located, with the passage of time, he/she had forgotten the location of the previously collected NIRS sample. To the extent possible, all samples were collected from the original public water systems surveyed for NIRS where gross beta-particle activities were 20 pCi/L or greater.

Though the number of samples was limited (26 samples), a few conclusions can be reached. Concentrations of gross beta-particle activities will rarely exceed 50 pCi/L in water collected from public water systems (and did not do so in this study). A significant percentage (15% or 4 samples) of the 26 samples analyzed, however, contained gross alpha-particle activities at or in excess of the 15 pCi/L MCL indicating that concern over the presence of elevated concentration of gross alpha-particle activity in ground water is justified. Long-term (30-day) gross beta-particle activity analyses did not indicate significant ingrowth of beta-particles in any of the samples, though this result is qualified by the absence of significant quantities of Uranium-238 in any of the samples collected. Naturally occurring K-40 and radium-228 are a significant source of gross beta-particle activity to many of the samples in agreement with results of Welch et al. (1995). Minor concentrations of naturally-occurring lead-210 are also detected occasionally. No manmade radionuclide was detected in concentration above the maximum detectible concentration (MDC) in any of the samples. The presence of naturally occurring beta-particle emitting radionuclides must be taken into account when evaluating the source of high gross beta-particle activity in ground water as first suggested by Welch et al., 1995.

I. References

Cothern, Richard C., and Paul A. Rebers, 1990. Radon, radium and uranium in drinking water. Lewis Publishers, Inc. Chelsea, Michigan. 286 p.

Elsinger, R.J., P.T. King, and W.S. Moore, 1982. Radium-224 in natural waters measured by gamma-ray spectrometry. Analytical Chemical Acta, vol. 144, p. 277-281.

Elsinger, R.J., and W.S. Moore, 1983. Radium-224, Ra-228, and Ra-226 in Winyah Bay and Delaware Bay. Earth and Planetary Science Letters, vol. 64, p. 430-436.

Harada, Kow, William C. Burnett, Paul A. LaRock, and James B. Cowart, 1989. Polonium in Florida groundwater and its possible relationship to the sulfur cycle and bacteria. Geochemical et Cosmochimica Acta Vol. 53, pp. 143-150.

Hess, C.T., J. Michel, T.R. Horton, H.M. Richard and W.A. Coniglio, 1985. The occurrence of radioactivity in public water supplies in the United States. Health Physics Vol. 48, No 5, p. 553-586.

Ibrahim, S., and T. Culp, 1989. Plutonium distribution and oxidation states in a reactor leaching pond system. Health Physics, vol. 57, no. 4, p. 607-614.

King, P.T., J. Michel, and W.S. Moore, 1982. Ground-water geochemistry of Ra-228, Ra-226, and Rn-222. Geochemical Cosmochimica Acta, vol. 46, P. 1173-1182.

Kramer, T.F., and D.F. Reid, 1984, The occurrence and behavior of radium in saline formation water of the U.S. Gulf Coast Region: Isotope Geoscience, vol. 2, p. 153-174.

Krieger, H.L., and Whitaker, E.L., 1980, Prescribed procedures for measurement of radioactivity in drinking water: U.S. Environmental Protection Agency Manual EPA-600/4-80-032, 111 p.

Longtin, J.P., 1988, Occurrence of radon, radium and uranium in groundwater: Journal of the American Water Works Association, pp. 84-93.

Lloyd, D.H., and E.N. Drake, 1989. An alternative method for radium determinations in water. Health Physics, vol. 57, no. 1, p. 71-77.

Michel, J., and C.R. Cothern, 1986. Predicting the occurrence of Ra-228 in ground water. Health Physics Vol. 51, No. 6, pp 715-721.

Parsa, B., 1998, Contribution of short-lived radionuclides to alpha-particle radioactivity in drinking water and their impact on the Safe Drinking Water Act Regulations. Radioactivity and Radiochemistry, vol. 9, no. 4, p. 41-50.

Szabo, Z., and V.T dePaul, 1998. Radium-228 and radium-228 in shallow ground water, southern New Jersey. U.S. Geological Survey Fact Sheet FS-062-98.

Szabo, Z., V.T. dePaul, and B. Parsa, 1998. Decrease in gross alpha-particle activity in water samples with time after collection from the Kirkwood Cohansy aquifer system in southern New Jersey: Implications for regulations. Drinking Water 63rd annual meeting American Water Works Association New Jersey Section. Atlantic City, NJ.

Szabo, Z., and O.S. Zapecza, 1991, Geologic and geochemical factors controlling uranium, radium-226, and radon-222 in ground water, Newark Basin, New Jersey: Gundersen, L.C.S. and Wanty, R.B., eds., Field studies of radon in rocks, soils, and water, U.S. Geological Survey Bulleting 1971, p. 243-266.

USEPA, 1985, Nationwide occurrence of radon and other natural radioactivity in public water

supplies. National Air and Environmental Radiation Laboratory. EPA 550-5-85-008.

Welch, A.H., Szabo, Z., Parkhurst, D.L., Van Meter. P.C., and Mullin, A.H., 1995, Gross-beta activity in ground water: natural sources and artifacts of sampling and laboratory analysis: Applied Geochemistry, v. 10, no. 5, p. 491-504.

Zapecza, O. S., and Z. Szabo, 1986. Natural radioactivity in ground water- a review. U.S. Geological Survey National Water Summary 1986, Ground-Water Quality: Hydrologic Conditions and Events, U. S. Geological Survey Water Supply Paper 2325. pp. 50-57.

III - HEALTH EFFECTS

A. Past Basis for Estimating Health Effects from Radionuclides in Drinking Water in 1976 and 1991

This section reviews the historical bases for setting MCLs for radionuclides in drinking water under both the current regulations and the 1991 proposed revisions to these regulations. Particular focus is placed on the estimated risks of adverse health effects associated with the lifetime ingestion of radionuclides in drinking water at their MCL concentrations and on the scientific studies, methods, and models used to calculate these risks. This review provides the background information needed to understand the reasons and approaches EPA used to set the current MCLs and the 1991 proposed revised MCLs. This information also provides the knowledge needed to evaluate the new information EPA is currently considering in its effort to establish final regulations for radionuclides in drinking water.

1. General Considerations

Whenever EPA establishes or revises MCLs for drinking water contaminants, including radionuclides, the law requires the Agency to first consider health effects, and then other factors. Specifically, the Safe Drinking Water Act directs EPA to first set an MCLG "at the level at which no known or anticipated adverse effects on the health of persons occur and which allows an adequate margin of safety." In turn, each maximum contaminant level (MCL) is specified "which is as close to the maximum contaminant level goal as feasible...with the use of the best technology, treatment techniques or other means which, ...after examination for efficacy under field conditions, and not solely under laboratory conditions, are available (taking costs into consideration)." In addition to these factors, EPA considers data on the natural and man-made sources and concentrations of contaminants in drinking water; intake of contaminants by other exposure pathways such as food ingestion, inhalation of contaminated air, etc.; and co-occurrences of contaminants in drinking water.

MCLGs for specific contaminants are set either on the evidence of carcinogenicity from drinking water exposure or the Agency's reference dose (RfD) for non-carcinogens. MCLGs are non-enforceable health goals based only on health effects and exposure information. Consistent with the SDWA, MCLGs for known or probable human carcinogens are typically set at zero because it is assumed, in the absence of conclusive data to the contrary, that there is no "safe" level of exposure. Since radionuclides emit ionizing radiation, which is known to be carcinogenic, all radionuclides are classified as human carcinogens.

In estimating the health effects from radionuclides in drinking water, EPA subscribes to the linear, nonthreshold model which assumes that any exposure to ionizing radiation has a potential to produce deleterious effects on human health, and that the magnitude of the effects are directly proportional to the exposure level. The Agency further believes that the extent of such harm can be estimated by extrapolating effects on human health that have been observed at

higher doses and dose rates to those likely to be encountered from environmental sources of radiation. The risks associated with radiation exposure are extrapolated from a large base of human data. EPA recognizes the inherent uncertainties that exist in estimating health impact at the low levels of exposure and exposure rates expected to be present in the environment. EPA also recognizes that, at these levels, the actual health impact from ingested radionuclides will be difficult, if not impossible, to distinguish from natural disease incidences, even using very large epidemiological studies employing sophisticated statistical analyses. However, in the absence of other data, the Agency continues to support the use of the linear, nonthreshold model in assessing risks associated with all carcinogens.

2. Basis of the 1976 and 1991 Estimates of Health Risks

In 1976, EPA established national interim primary drinking water regulations for radium-226 and radium-228, gross alpha particle radioactivity, and man-made beta particle and photon radioactivity (41 FR 28402, July 9, 1976). Maximum contaminant levels for these categories of radionuclides are contained in the following sections of 40 CFR 141:

- Radium-226 and radium-228, section 141.15(a)
- Gross alpha particle activity, section 141.15(b)
- Man-made beta particle and photon radioactivity, sections 141.6(a) and (b).

The subsections below briefly summarize EPA's reasons for setting these limits, as well as the rationale used by the Agency in proposing revisions to them. For more detailed discussions, readers are referred to the Statement of Basis and Purpose in the 1975 Federal Register notice of proposed maximum contaminant levels for radioactivity (40 FR 34321, Aug. 14, 1975); the Response to Comments section of the 1976 regulations (41 FR 28402, July 9, 1976); Appendix B of the National Interim Primary Drinking Water Regulations, EPA-570/9-76-003 (EPA, 1976); and the Federal Register notice proposing revisions to the existing MCLs (56 FR 33050, July 18, 1991).

In 1991, EPA proposed revised MCLs for radium-226 and radium-228, gross alpha particle radioactivity, man-made beta particle and photon radioactivity, and new MCLs for radon and uranium. (The 1986 SDWA Amendments required EPA to set MCLs for these latter two radionuclides.) During the years since the publication of the 1976 interim regulations, the Agency obtained a great deal of additional data and a better understanding of the risks posed to human health by ingested radionuclides. Many of these new studies were presented and discussed in the Advance Notice of Proposed Rulemaking announcing EPA's intent to revise the MCLs (51 FR 34836, Sept. 20, 1986) and the supporting health criteria documents (EPA, 1991a; 1991b; 1991c; 1991d; 1991e).

Among the most important changes made by EPA in developing the 1991 revisions was the adoption of a common calculational framework, the RADRISK computer code, to estimate the risks posed by ingestion of radionuclides in drinking water. The RADRISK code consisted

of intake, metabolic, dosimetric, and risk models that integrated the results of a large number of studies on a variety of radioactive compounds and radiation exposure situations into an overall model to estimate risks for many different radionuclides. Radionuclide-specific parameters were based on the results of individual scientific studies of a specific radionuclide, such as radium; human epidemiological studies; or experimental animal studies of groups of chemically-similar radionuclides.

a. Radium-226 and Radium-228

i. Basis for the 1976 MCL for Radium-226 and Radium-228

In 1976, EPA set the MCL for radium-226 and radium-228 combined at 5 pCi/L. This limit was based on four considerations:

(1) *Health Risk.* EPA estimated that continuous consumption of drinking water containing a 5 pCi/L concentration of radium-226 and/or radium-228 may cause between 0.7 and 3 total (bone and soft tissue) cancers per year per million exposed persons, or between 0.5 to 2 x 10^{-4} total cancers over a 70-year lifetime. EPA assumed that most of these cancers would be fatal.

In estimating the health risks associated with radium ingestion in 1976, EPA relied primarily on data and models presented in reports by the Federal Radiation Council (FRC); the National Academy of Sciences (NAS) Committee on the Biological Effects of Ionizing Radiation (BEIR); the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR); and the International Commission on Radiological Protection (ICRP). Additional sources of information included studies by Evans (1966) and Dougherty and Mays (1969). In general, EPA followed FRC's recommendation that radium ingestion limits for the general population should be based on environmental studies and not the models used to establish occupational dose limits (FRC 1961). In setting the MCL, EPA considered bone cancer and other soft tissue cancers to be the principal health effects associated with radium ingestion.

To calculate body burdens, doses, and risks from ingestion of radium-226 and radium-228, EPA relied on data from the 1972 UNSCEAR report (UNSCEAR, 1972) and the 1972 NAS BEIR I Report (NAS 1972). Additional support was found in ICRP Publication 20 (ICRP, 1972), Evans (1966), and Dougherty and Mays (1969). These sources suggested that radium-228 was as toxic as radium-226, and possibly twice as toxic for bone cancers in dogs. Given this, EPA believed that it was prudent to assume that the adverse health effects due to chronically ingested radium-228 were at least as great as those from radium-226.

Assuming equal toxicity with radium-226, EPA reasoned that lifetime ingestion of only radium-228 at 5 pCi/L would yield lifetime total cancer risks equal to those for a lifetime ingestion of only radium-226 at the same concentration, i.e., between 0.5 to 2 x 10^{-4}. By setting the MCL at 5 pCi/L for radium-226 and radium-228 combined, rather than individually, EPA sought to limit the lifetime total cancer risk from the ingestion of both isotopes in drinking water to 2 x 10^{-4} or less.

(2) *Occurrence and Intake by Other Exposure Pathways*. Although occurrence data were limited in 1976, EPA observed that radium concentrations in public water systems ranged from less than 0.1 pCi/L to over 60 pCi/L, depending on local geographical conditions. Small quantities of radium are also present in many foods, thus, EPA considered it unrealistic to require public water supplies to limit radium concentrations to zero. EPA selected 0.5 pCi/L as the lowest radium concentration considered in health risk and cost evaluations. EPA concluded that, because the average daily intake of radium in food equals or exceeds 1 P.I., concentrations of radium-226 in drinking water less than 0.5 pCi/L were unlikely to significantly increase risk.

(3) *FRC Guidance*. In 1962, the Federal Radiation Council (FRC) recommended an intake limit for radium-226 of 20 P.I./day. EPA concluded that an MCL of 5 pCi/L for radium, consumed at a rate of two liters of water per day (L/day), was within the FRC limit (i.e., 5 pCi/L x 2 L/day = 10 P.I./day).

(4) *Number of Systems Impacted, National Cost of Radium Abatement*. Based on an Agency-sponsored study, EPA estimated that about 450 public water systems would have radium concentrations exceeding a 6 pCi/L limit, 500 public water systems would be potentially impacted by a 5 pCi/L limit, and 670 were likely to exceed a 4 pCi/L limit. Costs would be $5.6 million/year at 6 pCi/L, $8.8 million at 5 pCi/L and $14 million at 4 pCi/L. At these control levels, the deaths prevented were estimated to be 2.9, 3.7, and 5.5 respectively. By moving the MCL from 6 to 5 pCi/L, an additional 50 systems would be impacted, but an additional 0.8 deaths could be prevented at a cost of $3.2 million. Moving the MCL from 5 pCi/L to 4 pCi/L would affect an additional 170 systems and could prevent another 1.8 deaths at a cost of $14 million/year. The control level of 5 pCi/L was thought to offer the most favorable marginal effectiveness in terms of cost, numbers of systems impacted, and lives saved.

ii. Basis for the 1991 MCL for Radium-226 and Radium-228

In 1991, EPA proposed revised MCLs for radium-226 and radium-228 individually at 20 pCi/L each. The Agency believed at that time that the limit for each of these radium isotopes was with the Agency's acceptable risk range of 10^{-6} and 10^{-4}.

(1) *Human and Animal Health Effects Data Considered.* In 1991, EPA based its risk estimates for radium using information from two epidemiological study groups. The first group consisted of radium dial painters who had ingested considerable amounts of radium paint (containing various proportions of radium-226 and radium-228) by sharpening the point of the paint brush with the lips. The second group consisted of patients in Europe injected with a short-lived isotope of radium, radium-224, for treatment of spinal arthritis and tuberculous infection of the bone (NAS, 1988; EPA, 1991b). The results of these studies are described briefly below.

At high levels of exposure to radium, several non-cancer health effects were observed in radium dial painters, such as benign bone growths, osteoporosis, severe growth retardation, tooth breakage, kidney disease, liver disease, tissue necrosis, cataracts, anemia, immunological suppression and death (ATSDR, 1990). Thirty or more years after exposure, the incidence of bone necrosis in female radium dial painters, with an estimated total ingestion of radium-226 or radium-228 above 50 μCi was significantly higher than in unexposed controls (Keane et al., 1983). However, levels of exposure from naturally-occurring radium are much lower than this threshold, and so bone necrosis and other non-cancer health effects were not of concern for setting a limit for radium in drinking water (EPA, 1991b; 1990g; 1990n).

Exposed radium dial painters also exhibited elevated rates of two rare types of cancer, bone sarcomas (osteosarcomas, fibrosarcomas and chondrosarcomas) and carcinomas of head sinuses and mastoids (Evans et al., 1944; Sharpe, 1974). The incidence of head carcinomas was associated with exposure to radium-226, but not radium-228 (Rowland et al., 1978). This is because these latter cancers were due to an accumulation of radon gas (radon-222) in the mastoid air cells and paranasal sinuses caused by the escape of radon-222 into the air spaces. Radon-222 resulted from the radium-226 deposited in the bones of the head surrounding the air spaces. Radon-222 is a decay product of radium-226. The comparable decay product of radium-228, radon-220, decays to Radium-224 too quickly to be present in the air (NAS, 1988). No statistically significant increase in cancers other than bone sarcomas and head carcinomas were found in cohorts of radium dial painters (Stebbings et al., 1984). Patients medically treated with radium-224 also showed an increase in bone sarcomas, but not head carcinomas (Mays and Speiss, 1984). These data were consistent with a linear dose-response relationship (NAS, 1988).

Possible correlations between cancer rates and radium in drinking water were examined in three studies in the United States. Peterson et al. (1966) found

an elevated rate of fatalities from bone malignancies among residents of Iowa and Illinois with elevated radium-226 in drinking water. However, the statistical significance was marginal and confounding factors could not be ruled out (NAS, 1986). Bean et al. (1982) found an increased incidence of four out of 10 cancers investigated among Iowa residents of small communities with elevated radium-226 content of the water supply. However, confounding by radon exposure could not be ruled out and cancer sites (bladder and lung cancer for males and breast and lung cancer for females) were different from those observed in dial painters. Lyman et al. (1985) found a small consistent excess of leukemias in Florida counties with elevated radium-226 or radium-228 in private wells, but there was no evidence of a dose-response trend. Rates of colon, lung and breast cancer and lymphoma showed no consistent excess (Lyman and Lyman, 1986).

Animal studies have shown that exposure to radium causes bone sarcoma in mice, rats and dogs and leukemia in mice (ATSDR, 1990). In addition to bone sarcomas, other malignancies associated with radium exposure in animals included eye melanomas in beagle dogs injected with radium-226 or radium-228 (Taylor et al., 1972) and leukemias in mice injected with radium-224 (Humphreys et al., 1985; Muller et al., 1988).

(2) *Basis for Body Burden, Dose, and Risk Calculations.* Risk calculations for ingested radium were made using RADRISK based on annual dose rates. For this purpose, EPA computed dose rates for specific organs and tissues at specific ages for an annual unit intake of each radium isotope (Sullivan et al., 1980; Dunning et al., 1980; EPA, 1989). Calculation of body burdens was based on metabolic models derived from the radium dial painter studies. Calculations of absorbed doses in specific organs or tissues included cross irradiation from radium in all other organs. RADRISK included lifetime cancer risk estimates for high- and low-LET (linear energy transfer) radiation separately for leukemia, osteosarcomas, sinus tumors, and other solid tumors. These estimates were taken from the BEIR III (NAS, 1980) and BEIR IV (NAS, 1988) reports.

(3) *Comparison of the Derivation of 1976 and 1991 MCLs for Radium.* Table III-1 compares the methods used by EPA in 1976 and 1991 to calculate organ burdens, doses, and risks from radium ingestion. Bone doses calculated for Ra-226 in 1991 were about 33 percent lower than those assumed in 1976, and the soft tissue doses were about 40 percent lower. Risk estimates for bone per unit dose were about 65 percent lower in 1991 than in 1976, and the soft tissue risk estimates were about 9 percent lower.

Table III-1. Comparison of Derivation of 1976 and 1991 MCLs for Radium

Model	1976	1991
Organ and Tissue Burdens	Calculation of body burdens based on environmental studies and ratio of intakes	Calculation of body burdens based on toxicokinetic models derived from studies of patients injected with radium
Dosimetry	Calculation of absorbed dose based on organ and tissue burden	Calculation of absorbed dose based on organ or tissue burden and cross irradiation terms from all other organs
Risk Coefficients	Risk estimated using the geometric mean of the absolute and relative risk coefficients from the 1972 BEIR I report.	Risk estimated using the absolute risk coefficient from the 1980 BEIR III report.

b. Gross Alpha Particle Activity

i. Basis for the 1976 MCL for Gross Alpha Particle Activity

One of the main intentions of the 15 pCi/L MCL for gross alpha particle activity, which includes radium-226 but excludes uranium and radon, was to limit the concentration of other naturally-occurring and man-made alpha emitters relative to radium-226. Specifically, this limit was based on the following considerations:

(1) *Health Risk.* EPA estimated that continuous consumption of drinking water containing polonium-210, the next most radiotoxic alpha particle emitter in the radium-226 decay chain, at a concentration of 10 pCi/L might cause the total dose to bone to be equivalent to less than 6 pCi/L of radium-226.

The 15 pCi/L limit, which includes radium-226 but excludes uranium and radon, was based on the conservative assumption that if the radium concentration is limited to 5 pCi/L and the balance of the alpha particle activity (i.e., 10 pCi/L) is due to polonium-210, the total dose to bone would be less than that dose associated with an intake of 6 pCi/L of radium-226.

(2) *Capability of State Laboratories.* EPA established an MCL for gross alpha, rather than individual MCLs for each alpha-emitting radionuclide, because the Agency was concerned about analytical costs and the capabilities of State laboratories to perform the analyses.

The analytical costs associated with identifying all of the alpha-emitting radionuclides were considered to be prohibitive at that time. Establishment of a limit on gross alpha particle activity was also more in keeping with the existing capabilities of State laboratories.

ii. Basis for the 1991 MCL for Gross Alpha Particle Activity

In 1991, EPA proposed to retain the 15 pCi/L MCL for gross alpha particle activity, but modify it by excluding radium-226, as well as uranium and radon. The 1991 limit was intended to limit the lifetime cancer risk due to ingestion of naturally-occurring and man-made alpha particle emitters in drinking water to between 10^{-6} and 10^{-4}, the Agency's target risk range for carcinogens. Specifically, this limit was based on the following considerations:

(1) *Health Risk.* Using RADRISK modeling, EPA estimated that continuous consumption of 15 pCi/L of most alpha particle emitters in drinking water at 2 L/day would pose a lifetime cancer risk between 10^{-6} and 10^{-4}.

EPA performed the risk assessment for the alpha emitters using RADRISK (EPA, 1991a). The model was used to estimate radiation dose to organs, the dose was used to calculate risk to organs, and the risks to organs were summed to estimate overall risk. EPA used RADRISK to calculate concentrations of alpha emitters corresponding to lifetime mortality and incidence risks of 10^{-4}, assuming ingestion of two liters of drinking water daily, and presented those values in Appendix C of the 1991 proposed rule.

In determining the risks from ingestion of alpha emitters in drinking water, EPA was particularly interested in polonium-210 and isotopes of thorium and plutonium, because these radionuclides had been observed in water and may cause health effects at relatively low concentrations. For polonium, EPA reviewed the health effects data summarized in the BEIR IV report (NAS, 1988). In experimental animal studies, polonium was observed to cause lymphomas in mice and various soft tissue tumors in rats. In addition, a number of non-cancer effects were reported in test animals, including sclerotic changes in blood vessels, atrophy of the seminiferous epithelium and hyperplasia of the interstitial (Leydig) cells in the testes, and other effects. These occurred only at relatively high doses. In exposed people, noncancer effects included hematologic changes and impairment of the liver, kidney, and reproductive organs. However, the BEIR IV report concluded that there was no direct measure of risk for most polonium isotopes based on the human data, and suggested several possible means of estimating risk. EPA, as discussed, relied on RADRISK in assessing polonium risk. The model estimated that continuous ingestion of two liters per day of drinking water containing 14 pCi/L would pose a lifetime fatal cancer risk of 1×10^{-4}. EPA noted that several public water supplies and private wells had exceeded this value, although most reported polonium concentrations ranged from 1 to 10 pCi/L (EPA, 1991f).

EPA also consulted the BEIR IV report for available information on the adverse effects of thorium. Epidemiological studies of patients injected with

Thorotrast, a contrast agent consisting of ThO_2 and used in medical radiology from the 1920s to 1955, showed clear increases in liver cancer, as well as possible increases in leukemia and other cancers. However, the BEIR IV report discussed the limitations of these data for assessing the risk due to other forms of thorium that might have different metabolic behaviors and effects. Using RADRISK, EPA estimated that, at a lifetime fatal cancer risk level of 1×10^{-4}, derived drinking water concentrations for thorium isotopes ranged from 50 to 125 pCi/L, and noted that thorium concentrations in drinking water were generally near one pCi/L (EPA, 1991f).

EPA relied on the BEIR IV report for information on the health effects of plutonium isotopes and other transuranic radionuclides that were widely distributed in the environment in very low concentrations due to atmospheric testing of nuclear weapons from 1945 to 1963. The BEIR IV report concluded that plutonium exposures caused clear increases in cancers of the bone, liver, and lungs in animals, but not in humans. At that time, the limited available epidemiological studies had not demonstrated a clear association between plutonium exposure and the development of cancer in human exposure cases. The report recommended that assessing the risks of plutonium exposure should be based on analogy with other radionuclides and high-LET radiation exposure risks. Using RADRISK, EPA estimated that, at a lifetime fatal cancer risk level of 1×10^{-4}, derived drinking water concentrations for plutonium isotopes ranged from about 7 to 68 pCi/L, and noted that plutonium concentrations in drinking water were generally less than 0.1 pCi/L (EPA, 1991f).

(2) *Occurrence*. EPA observed that few public water systems had ever exceeded the gross alpha MCL as a result of anything but high radium-226 levels. EPA believed that a 15 pCi/L limit that excluded radium, radon, and uranium would provide an effective screening level for the most likely alpha emitters in drinking water (i.e., polonium, thorium, plutonium, and possibly americium) without requiring separate MCLs or separate monitoring requirements for individual radionuclides.

c. Man-Made Beta Particle and Photon Radioactivity

i. Basis for the 1976 MCL for Beta Particle and Photon Radioactivity

In setting the 1976 MCL for man-made beta particle and photon radioactivity at 4 millirem per year, EPA stated that the limit was chosen primarily on the basis of avoiding undesirable future contamination of public water supplies as a result of controllable human activities. Other considerations were:

(1) *Health Risk.* EPA estimated that continuous consumption of drinking water containing beta and photon emitting radioactivity yielding a 4 mrem/yr total body dose may cause an individual fatal cancer risk of 0.8×10^{-6} per year, or a lifetime cancer risk of 5.6×10^{-5}, assuming a 70-year lifetime.

In setting the MCL for man-made beta and photon emitters, EPA used cancer risk estimates from the BEIR I report for the U.S. population in the year 1967 (NAS, 1972). For an exposed group having the same age distribution as the U.S. 1967 population, the BEIR I report indicated that the individual risk of a fatal cancer from a lifetime total body dose rate of 4 mrem per year ranged from about 0.4 to 2×10^{-6} per year depending on whether an absolute or relative risk model was used. Using best estimates from both models for fatal cancer, EPA believed that an individual risk of 0.8×10^{-6} per year resulting from a 4 mrem annual total body dose was a reasonable estimate of the annual risk from a lifetime ingestion of drinking water. Over a 70-year period, the corresponding lifetime fatal cancer risk would be 5.6×10^{-5}, with the risk from the ingestion of water containing less amounts of radioactivity being proportionately smaller. EPA was aware that the estimated total health risk from radiation exceeded that due to fatal cancers alone. The BEIR I report projected that the incidence of non-fatal cancers would be about the same as fatal cancers. The incidence of genetic effects was more difficult to estimate, but the increase, expressed over several generations, was thought to be comparable to the increased incidence of fatal cancers.

EPA also compared the estimated risks of a fatal cancer due to a lifetime exposure to ionizing radiation to the risk of a fatal cancer occurring without radiation exposure. To do this, the Agency normalized the BEIR data for the 1967 population in terms of a single individual's exposure history. Based on 1967 U.S. Vital Statistics, the probability that an individual would die of cancer was about 0.19, and was thought to be increased by 0.1 percent from a lifetime dose equivalent rate of 15 mrem per year. Therefore, EPA calculated that the 4 mrem/yr MCL for man-made beta and photon emitters corresponded to a lifetime risk increase of 0.025 percent to exposed groups.

EPA knew that partial body irradiation was common for ingested radionuclides since they are, like radium, largely deposited in a particular organ. In such cases, EPA acknowledged that the risk per millirem varies depending on the radiosensitivity of the organs at risk. For example, EPA estimated that cancers due to the thyroid gland receiving 4 mrem per year continuously ranged from about 0.2 to 0.5 per year per million exposed persons (averaged over all age groups). Fatality due to thyroid cancer varies with age, from nearly zero for children and young adults to about 20 percent of the incidence for persons well past middle age. EPA noted that estimated fatalities from thyroid exposure were at least five times less than that from whole body exposure. However, the Agency

considered other factors bearing on the health impact to be significant.

(1) *Occurrence and Intake by Other Exposure Pathways.* EPA did not anticipate that the MCL for man-made beta particle or photon radioactivity would be exceeded except in extraordinary circumstances. Then-current levels of fallout radioactivity in public water supplies were declining and controls on releases of materials from the nuclear industry were in place.

In 1976, EPA stated that contamination of drinking water by man-made beta particles and photon radioactivity originated from two sources: (a) ubiquitous activity from deposited fallout and (b) releases of radioactive materials from medical and industrial applications, including effluents from energy generation. Ambient levels from fall-out were decreasing at that time due to the cessation of most atmospheric testing of nuclear devices. Industrial and medical uses of radioactive materials were rapidly increasing. However, the releases from the latter sources were controlled by a number of regulatory authorities so that the build-up of man-made radioactivity in public water systems was considered to be minimal.

Considering the sum of the deposited fallout radioactivity and the additional amounts due to releases from other sources existing at that time, EPA believed that the total dose equivalent from man-made radioactivity was not likely to result in a total body or organ dose to any individual that exceeded 4 mrem/yr. Consequently, EPA did not believe that the 4 mrem/yr standard would affect many public water systems, if any. At the same time, the Agency believed that an MCL set at this level would provide adequate public health protection.

(2) *Derived Activity Concentrations Yielding 4 mrem/yr.* Except for tritium and strontium-90, the 1976 MCL specified that the concentrations of man-made radionuclides causing 4 mrem/yr were to be calculated using data in NBS Handbook 69, assuming a 2 L/day drinking water intake rate.

The dose models used in preparing Handbook 69 (NBS, 1963) were taken from ICRP Publication 2 (ICRP, 1959) and were defined in terms of the annual dose equivalent to the critical organ which could be the total body or any internal organ.

When the total body is designated as the critical organ for a particular radionuclide, doses to a specific organ must be added to the total body dose. For example, EPA considered drinking water which had, on an annual basis, a strontium-90 concentration of 4 pCi/L and a tritium concentration of 15,000 pCi/L. The annual dose to bone marrow from strontium-90 would be 2 mrem. The total body dose from tritium would be 3 mrem annually. Even though the

annual concentration of each contaminant alone would be permissible (because they are less than 4 mrem/yr), the total dose to bone marrow (total body dose from tritium + bone marrow dose from strontium) would be 5 mrem annually, thereby exceeding the MCL.

It should be noted that EPA intended to express the MCL for man-made radionuclides in terms of the annual dose rate (millirem per year) from continuous ingestion. EPA considered, but rejected, the idea of specifying the MCL in terms of radioactivity concentrations (pCi/L) for individual beta and photon emitters in view of the short length of time such limits would be appropriate. The dose conversion factors for ingested radioactivity available at that time were becoming obsolete and the ICRP was developing new models. When appropriate models for doses due to environmental contamination became available, the Agency intended to revise the Interim Regulations to permit the use of newer data. However, before the Agency could act, the 1986 Amendments to the SDWA finalized the Interim Primary Drinking Water Regulations, making them National Primary Drinking Water Regulations.

(3) *Number of Systems Impacted and National Cost.* EPA estimated that 60 community water systems required monitoring, but thought that none required treatment. EPA estimated the annual monitoring cost at $330 per year per system, or $20,000 per year nationwide.

ii. Basis for the 1991 MCL for Beta Particle and Photon Radioactivity

In 1991, EPA proposed to retain the 4 mrem/yr MCL for beta particle and photon radioactivity. However, the Agency did modify the standard by basing the limit on the committed effective dose equivalent (EDE). (An effective dose equivalent approach adjusts the dose that an individual organ may receive based on its radiosensitivity. The less radiosensitive an organ is, the greater the allowable radiation dose.) The MCL was also modified to include naturally-occurring beta/photon emitters. The 1991 proposed standard was intended to limit the lifetime cancer risk due to ingestion of naturally-occurring and man-made beta particle and photon emitters in drinking water to between 10^{-6} and 10^{-4}, the Agency's target risk range for carcinogens. Specifically, this limit was based on the following considerations:

(1) *Health Risk.* Using RADRISK modeling, EPA estimated that continuous consumption of two liters per day of drinking water containing a concentration of beta particle or photon emitting radiation corresponding to 4 mrem EDE/yr would pose a lifetime cancer risk of about 10^{-4}.

Similar to the approach used for alpha particle emitters, EPA performed the risk assessment for the beta particle and photon emitting radionuclides using

RADRISK (EPA, 1991a). EPA used RADRISK to calculate concentrations of beta/photon emitters corresponding to 4 mrem EDE/yr, assuming ingestion of two liters of drinking water daily, and presented these values in Appendix B of the 1991 proposed rule.

(2) *Occurrence*. EPA observed that few public water systems had ever exceeded the beta/photon MCL for any other reason than high lead-210 levels. EPA believed that the 4 mrem EDE/yr limit would provide an effective screening level for the most likely beta/photon emitters in drinking water without requiring separate MCLs or separate monitoring requirements for individual radionuclides.

iii. Comparison of the 1976 Regulation and 1991 Proposed Regulation

In 1976, EPA based the MCL for beta particle and photon emitters on a target dose rate of 4 mrem/yr. The annual average activity concentration of individual radionuclides and mixtures of radionuclides resulting in a 4 mrem/yr dose to the total body or any internal organ was then calculated. This "critical organ dose" radiation protection philosophy was based on the recommendations of ICRP Publication 2 (ICRP, 1959). The Agency was aware that, when exposed to equal doses of radiation, different organs and tissues in the human will exhibit different cancer induction rates. Consequently, EPA knew that the lifetime cancer risks for individual radionuclides would vary widely (from about 10^{-7} to 5.6×10^{-5}) because the same dose equivalent would be applied to different critical organs, resulting in different cancer risks. However, at that time, EPA did not have an accepted method for equalizing risks.

This problem was solved in the 1991 proposed rule when EPA adopted the effective dose equivalent, or EDE, radiation protection philosophy recommended by ICRP (1977). The effective dose equivalent normalizes radiation doses and effects on a whole body basis for regulation of occupational exposures. The EDE is computed as the sum of the weighted organ-specific dose equivalent values, using weighting factors specified by the ICRP (1977, 1979). By changing to a limit of 4 mrem EDE/yr, EPA was able to derive activity concentrations for individual beta/photon emitters that corresponded to a more uniform level of risk. As a result of this change, as well as the use metabolically-based dose calculations, derived concentrations for most beta particle and photon emitters increased in 1991 compared to the values calculated in 1976, as shown in Table III-2.

Table III-2. Comparison of 1976 and 1991 Derived MCLs for Beta Particle and Photon Emitting Radionuclides (in pCi/L)

Radionuclide	1976 MCL*	1991 MCL**	Ratio of 1991:1976
H-3	20,000	60,900	3
C-14	2,000	3,200	2
S-35	500	12,900	26
Ca-45	10	1,730	173
Ca-47	80	846	11
Sc-47	300	2,440	8
Fe-55	2,000	9,250	5
Fe-59	200	844	4
Co-57	1,000	4,870	5
Co-58	300	1,590	5
Co-60	100	218	2
Ni-59	300	27,000	90
Ni-63	50	9,910	198
Zn-65	300	396	1.3
Se-75	900	5,740	6
Sr-89	20	599	30
Sr-90	8	42	5
Y-90	60	510	9
Zr-95	200	1,460	7
Nb-95	300	2,150	7
Tc-99	900	3,790	4
Tc-99m	20,000	89,600	4
Ru-103	200	1,810	9
Ru-106	30	203	7
Ag-110m	90	512	6
Sb-124	60	563	9
Sb-125	300	1,940	6
Te-125m	600	1,490	2
Te-127	900	7,920	9
Te-127m	200	663	3
Te-129	2,000	27,200	14
Te-129m	90	524	6
Te-131m	200	971	5
Te-132	90	580	6
I-125	30	151	5
I-129	1	21	21
I-131	3	108	36

Radionuclide	1976 MCL*	1991 MCL**	Ratio of 1991:1976
I-132	90	8,190	91
I-133	10	549	55
I-134	100	21,400	214
I-135	30	2,340	78
Cs-134	80	81	1.0
Cs-135	900	794	0.9
Cs-136	800	518	0.6
Cs-137	200	119	0.6
Ba-140	90	582	6
La-140	60	652	11
Ce-141	300	1,890	6
Ce-144	29	261	9
Pa-233	300	1,510	5

*1976 values taken from Appendix B in EPA-570/9-76-003.
**1991 values taken from Appendix B in 56 FR 33121.

d. Uranium

i. Basis for the 1991 Proposed MCL for Uranium

In 1991, EPA proposed an MCL of 20 micrograms/L (~30 pCi/L) for uranium isotopes based on a consideration of kidney toxicity and carcinogenicity. In setting the limit based on carcinogenicity, EPA considered several factors.

(1) *Health Risk.* Using RADRISK modeling, EPA estimated that uranium in water posed a cancer risk of 5.9×10^{-7} per picoCurie per liter (pCi/L), assuming continuous intake of water of two liters per day. Concentrations in water of 1.7 pCi/L, 17 pCi/L and 170 pCi/L corresponded to lifetime mortality risks of approximately 1×10^{-6}, 1×10^{-5} and 1×10^{-4}, respectively. A concentration of 30 pCi/L of uranium-238 was thought to be equivalent to about 20 micrograms/L, the level considered to be protective against kidney toxicity (the corresponding mortality risk was 5×10^{-5}).

In determining the MCL for uranium in 1991, EPA proposed to regulate uranium at a level that would be protective of both kidney toxicity, resulting from the element's chemical properties, and carcinogenic potential due to radioactivity. The carcinogenic effects of uranium were based on the effects of ionizing radiation generally, the similarity of uranium to isotopes of radium, and on the effects of high activity uranium. Studies showed that, like radium, uranium accumulates primarily in bone, and that bone sarcomas may result from radium

ingestion (EPA, 1991b; 1991e). The induction of bone sarcomas was regarded as a common property of both radium and uranium and results from alpha-particle emissions. Finally, studies of enriched and high activity isotopes of uranium were shown to be carcinogenic in animal studies.

Animal studies of exposure to natural uranium did not provide direct evidence of carcinogenic potential (EPA, 1991e). Malignant tumors were observed in mice following injection of uranium-232 or uranium-233 (at levels greater than 0.1 μCi/kg), but not following injection of natural uranium (Finkel, 1953). This may be because the radiation dose levels resulting from the injection of natural uranium were about 100-fold lower than the dose at which tumors were observed in mice injected with uranium-232 and uranium-233. Highly enriched uranium (i.e., uranium enriched in the uranium-235 and uranium-234 isotopes) has been shown to induce bone sarcomas in rats (NAS, 1988). Existing epidemiological data were also inadequate to assess the carcinogenicity of uranium ingested in drinking water (EPA, 1991e). Despite this data gap, EPA thought it was appropriate to consider the health effects of exposure to uranium in drinking water similar to the effects of radium ingestion and to protect public health accordingly.

(2) *Occurrence.* EPA was concerned about exposures to uranium isotopes because of their potential radiogenic and chemical effects and their ubiquitous occurrence in the environment, including water supplies.

B. Today's Methodology for Assessing Risks from Radionuclides in Drinking Water

1. Background

Since 1991, EPA has refined the way in which it estimates potential adverse health effects associated with ingestion of radionuclides in drinking water. The Agency's new health effects model uses state-of-the-art methods, models and data that are based on more recent scientific knowledge. Compared with the approaches used in 1976 and 1991, the revised methodology includes several substantial refinements. Specifically, the new risk-assessment methodology:

- accounts for age- and gender-specific water-consumption rates and radionuclide intakes, and physiological and anatomical changes with age in quantifying costs and benefits;
- uses Blue Book (EPA, 1994) for estimating radiogenic risk: ICRP dosimetry model, 1990 vital statistics of 1980;
- uses the most recent age-dependent biokinetic and dosimetric models recommended by the ICRP;
- incorporates the latest information on radiogenic human health effects summarized by the National Academy of Sciences and other national and

international radiation-protection advisory committees;

- includes updated life tables based on data from the National Center for Health Statistics that are used to adjust radionuclide risk estimates for competing causes of death; and
- uses an improved computer program to handle the complex calculations of radiation doses and risks.

Overall, EPA believes that these refinements significantly strengthen the scientific and technical bases for estimating risks, and consequently, for deriving MCLs for radionuclides. A brief overview of this new methodology is summarized below. Interested individuals are referred to two EPA publications—*Estimating Radiogenic Cancer Risks* (EPA, 1994) and *Federal Guidance Report No. 13* (EPA, 1998)—for detailed discussions on the revised risk assessment methodology for radionuclides. Electronic copies of both documents are available for downloading at EPA's web site (http://www.epa.gov/radiation/rpdpubs.htm).

2. Current Methodology for Computing Risk Coefficients for Ingestion of Radionuclides in Tap Water

Federal Guidance Report No. 13 (EPA, 1998) presents the current methods, models, and calculational framework EPA uses to estimate the lifetime excess risk of cancer induction following intake or external exposure to radionuclides in environmental media. The report presents compilations of risk coefficients that may be used to estimate excess cancer morbidity (cancer incidence) and mortality (fatal cancer) risks resulting from exposure to radionuclides through various pathways. The risk coefficients for internal exposure represent the incremental probability of radiogenic cancer morbidity or mortality occurring per unit of radioactivity inhaled or ingested. For most radionuclides, *Federal Guidance Report No. 13* presents risk coefficients for seven exposure pathways: inhalation, ingestion of food, ingestion of tap water, ingestion of milk, external exposure from submersion in air, external exposure from the ground surface, and external exposure from soil contaminated to an infinite depth. For some radionuclides, however, only external exposure pathways are considered; these include noble gases and the short-lived decay products of radionuclides addressed in the internal exposure scenarios.

The method used by EPA to compute a risk coefficient is shown schematically in Figure III-1 for the case of internal exposure to a radionuclide resulting from ingestion of contaminated drinking water. The main steps in the computation are shown in the numbered boxes; a detailed explanation of each of these steps can be found in the *Technical Support Document: Radionuclides in Drinking Water; Notice of Data Availability* (EPA, XX).

The 1996 Safe Drinking Water Act Amendments direct EPA to provide the same or greater protection against potential adverse human health effects whenever the Agency proposes to modify an existing MCL for any contaminant. This requirement was not applied in 1991, and now compels EPA to reevaluate the levels of protection afforded by the 1991 proposed MCLs for radionuclides and to compare these levels with those achieved by the current MCLs.

As a part of its efforts to use best available science to comply with the 1996 Amendments regarding protection, EPA calculated the lifetime excess cancer risk associated each radionuclide at its 1976 and 1991 MCL limits using the new radionuclide risk coefficients contained in *Federal Guidance Report No. 13* (EPA, 1998). Using this approach, EPA determined whether the 1991 proposed MCL for a given radionuclide was more protective or less protective than its existing level. EPA also determined whether or not the risks from the proposed or current MCLs for a given radionuclide fall within the Agency's acceptable risk range of 10^{-6} to 10^{-4}, when recalculated using the most recent methodology.

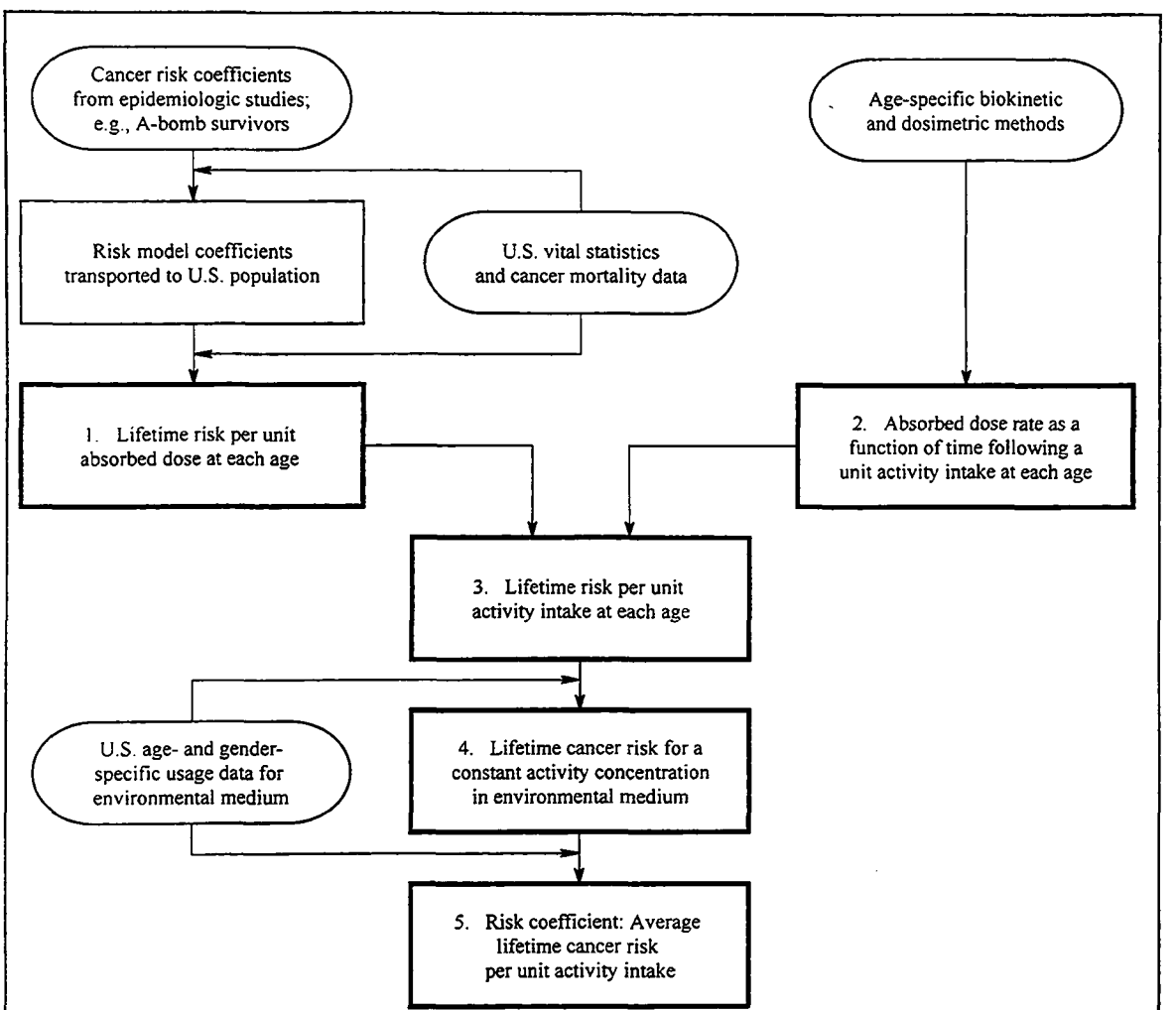


Fig. III-1. Components of the computation of risk coefficients. (The numbers identify the key steps described in the text.)

3. Impact on Drinking Water Regulations

The following subsections describe the quantitative evaluation of current and 1991

proposed MCLs and the radionuclide-specific drinking water concentrations which, when ingested over a lifetime, result in carcinogenic risks at predetermined risk levels. The mortality and morbidity risk coefficients used in this evaluation are published in the interim version of *Federal Guidance Report No. 13* (EPA, 1998). For this evaluation, EPA assumed a daily intake rate of two liters of drinking water, 365 days per year, for 70 years. To derive MCLs for individual radionuclides in terms of activity concentration (i.e., pCi/L) corresponding to a specified target risk level, the following equation was used:

$$MCL\ (pCi/L) = TR/[(RF * 0.037) * 2 * 365 * 70)]$$

where:

TR = target risk level
RF = risk coefficient from *Federal Guidance Report No. 13* (in risk per Becquerel (Bq))
0.037 = factor to convert from Bq to pCi/L

a. Beta- and Photon-Emitting Radionuclides

As discussed previously in Section III.A.2.c, EPA set the current (1976) MCL for man-made beta particle- and photon-emitting radionuclides, individually and combined, at the annual average concentration in drinking water which yielded a dose rate of 4 millirem per year to any internal organ or total body. Following the methods, assumptions, and dose conversion factors from NBS Handbook 69 prescribed in 40 CFR 141.16(b), the Agency derived activity concentrations for individual beta and photon emitters corresponding to the annual dose limit. These derived concentrations were published in Tables IV-2A and IV-2B of EPA-570/9-76-003 and are reproduced in Table III-3 below. EPA estimated that a 70-year continuous intake of each beta/photon emitter at its derived concentration would, pose a maximum lifetime excess fatal cancer risk of about 5.6×10^{-5} for whole body dosers, with the regaining radionuclides yielding a lower level of risk.

In 1991, EPA proposed a similar, but different limit of 4 millirem per year based on the concept of effective dose equivalent, or EDE, which normalized the radiation dose and risk from ingested beta and photon emitters on a total body basis. As a result of the adoption of the 4 mrem/yr EDE dose limit and the use of updated dose-conversion factors, EPA derived new activity concentrations for individual beta/photon emitters (see Table III-3) and estimated that, for most of these radionuclides, the corresponding lifetime fatal cancer risk would be 1×10^{-4}, about twice as high as the risk level estimated in 1976, but still within EPA's acceptable range of 10^{-6} to 10^{-4}.

Table III-3. Comparison of Derived MCLs for Beta Particle and Photon Emitting Radionuclides (in pCi/L)

Nuclide (*half-life of 24 hours or less)	1976 limits based on critical organ at 4 mrem/yr	1976 Risks	1991 proposed limits at 4 ede mrem/yr	1991 risks	Comments (HB69 means National Bureauof Standards Handbook-69)	Corrected limits based on 4 mrem/yr critical	Risk at corrected limits
H-3 (HTO)	20,000	3.57e-05	60,900	1.09e-04			
Be-7	6,000	1.60e-05	43,500	1.16e-04			
C-11	NC		99,200	1.75e-04			
N-13			152,000				
C-14	2,000	1.09e-04	3,200	1.75e-04			
C-15			6,690,000				
O-15			495,000				
F-18 *	2,000	8.32e-06	39,500	1.64e-04			
Na-22	400	1.36e-04	466	1.59e-04			
Na-24			3,350	1.60e-04	Not in 1976, but in HB69	60	2.87e-06
Si-31 *	3,000	5.96e-05	10,200	2.02e-04			
P-33			1,870				
P-32	30	9.53e-06	641	2.04e-04			
S-35 (Inorg)	500	8.39e-06	12,900	2.16e-04			
Cl-36	700	7.86e-05	1,850	2.08e-04			
Cl-38 *	1,000	8.41e-06	21,200	1.78e-04			
K-42 *	900	4.08e-05	3,900	1.77e-04			
Ca-45	10	8.96e-07	1,730	1.55e-04			
Ca-47	80	1.80e-05	846	1.90e-04			
Sc-46	1,000	1.95e-05	863	1.68e-04	Error in 1976 Calculation	100	1.95e-05
Sc-47	300	2.97e-05	2,440	2.42e-04			
Sc-48	80	1.71e-05	766	1.64e-04			
V-48	90	2.16e-05	644	1.55e-04			
Cr-51	6,000	3.26e-05	38,000	2.06e-04			
Mn-52	90	1.77e-05	733	1.44e-04			
Mn-54	300	2.23e-05	2,010	1.50e-04			
Mn-56 *	300	9.64e-06	5,640	1.81e-04			
Fe-55	2,000	6.84e-05	9,250	3.17e-04			
Fe-59	200	5.14e-05	844	2.17e-04			
Co-57	1,000	3.21e-05	4,870	1.57e-04			
Co-58	9,000	8.80e-04	1,590	1.57e-04	MCL switched with Co-58m	300	2.96e-05
Co-58m	300	1.18e-06	64,900	2.56e-04	MCL switched with Co-58	9000	3.55e-05
Co-60	100	5.20e-05	218	1.13e-04			
Ni-59	300	2.52e-06	27,000	2.27e-04			

Nuclide (*half-life of 24 hours or less)	1976 limits based on critical organ at 4 mrem/yr	1976 Risks	1991 proposed limits at 4 ede mrem/yr	1991 risks	Comments (HB69 means National Bureauof Standards Handbook-69)	Corrected limits based on 4 mrem/yr critical	Risk at corrected limits
Ni-63	50	1.02e-06	9,910	2.02e-04			
Ni-65 *	300	6.52e-06	8,810	1.92e-04			
Cu-64 *	900	1.70e-05	11,900	2.25e-04			
Zn-65	300	1.23e-04	396	1.62e-04			
Zn-69 *	6,000	1.62e-05	63,100	1.71e-04			
Zn-69m *	200	1.09e-05	4,220	2.30e-04			
Ga-67	NC		7,020	2.10e-04			
Ga-72 *	100	1.62e-05	1,190	1.93e-04			
Ge-71	6,000	1.13e-05	436,000	8.19e-04			
As-73	1,000	4.52e-05	7,850	3.55e-04			
As-74	100	1.97e-05	1,410	2.77e-04			
As-76	60	1.67e-05	1,060	2.95e-04			
As-77	200	1.44e-05	4,330	3.11e-04			
Se-75	900	2.65e-04	574	1.69e-04			
Br-82	100	5.86e-06	3,150	1.85e-04			
Rb-82			436,000				
Rb-86	600	2.06e-04	485	1.67e-04			
Rb-87	300	5.41e-05	501	9.04e-05			
Rb-88	NC		29,100	1.83e-04			
Rb-89	NC		52,700	1.81e-04			
Sr-82	NC		241	2.29e-04			
Sr-85	21,000	1.75e-03	2,830	2.36e-04	Wrong critical organ selected	900	7.49e-05
Sr-85m	900	5.66e-07	237,000	1.49e-04	Wrong critical organ selected	21000	1.32e-05
Sr-89	20	1.66e-06	599	2.38e-04			
Sr-90	8	2.03e-05	42	1.06e-04			
Sr-91 *	200	1.90e-05	2,160	2.05e-04			
Sr-92	200	1.31e-05	3,100	2.03e-04			
Y-90	60	3.06e-05	510	2.60e-04			
Y-91	90	4.07e-05	576	2.60e-04			
Y-91m *	9,000	1.07e-05	132,000	1.57e-04			
Y-92 *	200	1.48e-05	2,870	2.13e-04			
Y-93	90	1.85e-05	1,200	2.47e-04			
Zr-93	2,000	8.55e-05	5,090	2.17e-04			
Zr-95	200	2.68e-05	1,460	1.96e-04			
Zr-97 *	60	2.14e-05	650	2.32e-04			
Nb-93m	1,000	2.29e-05	10,500	2.40e-04			

Nuclide (*half-life of 24 hours or less)	1976 limits based on critical organ at 4 mrem/yr	1976 Risks	1991 proposed limits at 4 ede mrem/yr	1991 risks	Comments (HB69 means National Bureauof Standards Handbook-69)	Corrected limits based on 4 mrem/yr critical	Risk at corrected limits
Nb-94	NC		707	1.63e-04			
Nb-95	300	2.16e-05	2,150	1.55e-04			
Nb-95m	NC		2,390	2.48e-04			
Nb-97 *	3,000	2.04e-05	23,500	1.60e-04			
Nb-97m			1,370,000				
Mo-99	600	3.54e-05	1,830	1.08e-04			
Tc-95	NC		69,700	1.22e-03			
Tc-95m	NC		3,120	1.75e-04			
Tc-96	300	3.17e-05	2,050	2.17e-04			
Tc-96m *	30,000	3.44e-05	176,000	2.02e-04			
Tc-97	6,000	4.82e-05	32,500	2.61e-04			
Tc-97m	1,000	6.94e-05	4,450	3.09e-04			
Tc-99	900	7.28e-05	3,790	3.07e-04			
Tc-99m	20,000	4.61e-05	89,600	2.07e-04			
Ru-97	1,000	1.86e-05	7,960	1.48e-04			
Ru-103	200	2.22e-05	1,810	2.01e-04			
Ru-105 *	NC		4,990	2.13e-04	Error in 1976, listed as Rh-105	300	1.28e-05
Rh-105m			5,551,000				
Ru-106	30	3.66e-05	203	2.48e-04			
Rh-103m *	30,000	1.03e-05	471,000	1.62e-04			
Rh-105 *	300	2.00e-05	3,720	2.48e-04	Error: should be listed as Ru-105		
Rh-106	NC		1,240,000	1.97e-04			
Pd-100	NC		1,300	1.53e-04			
Pd-101	NC		13,400	1.67e-04			
Pd-103	900	3.18e-05	6,940	2.45e-04			
Pd-107	NC		36,600	2.59e-04			
Pd-109	300	2.99e-05	2,120	2.12e-04			
Ag-105	300	1.63e-05	2,700	1.47e-04			
Ag-108			626,000				
Ag-108m	NC		723	1.94e-04			
Ag-109m			16,700,000				
Ag-110			1,840,000				
Ag-110m	90	2.86e-05	512	1.63e-04			
Ag-111	100	2.34e-05	1,080	2.53e-04			
Cd-109	600	9.81e-05	227	3.71e-05			
Cd-115	90	2.21e-05	958	2.35e-04			

Nuclide (*half-life of 24 hours or less)	1976 limits based on critical organ at 4 mrem/yr	1976 Risks	1991 proposed limits at 4 ede mrem/yr	1991 risks	Comments (HB69 means National Bureauof Standards Handbook-69)	Corrected limits based on 4 mrem/yr critical	Risk at corrected limits
Cd-115m	90	4.46e-05	339	1.68e-04			
In-113m *	3,000	9.36e-06	52,400	1.63e-04			
In-114			976,000				
In-114m *	60	4.37e-05	323	2.35e-04			
In-115	300	4.46e-04	35	5.22e-05			
In-115m *	1,000	1.30e-05	16,400	2.14e-04			
Sn-113	300	3.72e-05	1,740	2.16e-04			
Sn-121	NC		6,060	2.58e-04			
Sn-121m	NC		2,260	1.53e-04			
Sn-125	60	3.41e-05	446	2.54e-04			
Sn-126	NC		293	2.19e-04			
Sb-122	90	2.72e-05	810	2.45e-04			
Sb-124	60	2.27e-05	563	2.13e-04			
Sb-125	300	4.12e-05	1,940	2.67e-04			
Sb-126	NC		544	1.77e-04			
Sb-126m	NC		58,500	1.61e-04			
Sb-127	NC		818	2.35e-04			
Sb-129	NC		3,090	1.99e-04			
Te-125m	600	6.15e-05	1,490	1.53e-04			
Te-127	900	2.62e-05	7,920	2.31e-04			
Te-127m	200	5.71e-05	663	1.89e-04			
Te-129	2,000	1.21e-05	27,200	1.65e-04			
Te-129m	90	4.07e-05	524	2.37e-04			
Te-131m	NC		26,800	4.58e-03			
Te-131	200	7.87e-07	971	3.82e-06			
Te-132	90	3.30e-05	580	2.13e-04			
I-122			211,000				
I-123	NC		10,700	2.13e-04			
I-125			151	1.10e-04	Not in 1976 list,but in HB69		
I-126	3	7.50e-06	81	2.02e-04			
I-129	1	4.22e-06	21	8.87e-05			
I-130	NC		1,190	2.17e-04			
I-131	3	3.91e-06	108	1.41e-04			
I-132 *	90	2.17e-06	8,190	1.98e-04			
I-133 *	10	4.13e-06	549	2.27e-04			
I-134 *	100	7.16e-07	21,400	1.53e-04			

Nuclide (*half-life of 24 hours or less)	1976 limits based on critical organ at 4 mrem/yr	1976 Risks	1991 proposed limits at 4 ede mrem/yr	1991 risks	Comments (HB69 means National Bureauof Standards Handbook-69)	Corrected limits based on 4 mrem/yr critical	Risk at corrected limits
I-135 *	30	2.62e-06	2,340	2.04e-04			
Cs-131	20,000	1.29e-04	22,800	1.47e-04			
Cs-134	20,000	3.22e-02	81	1.22e-04	Wrong critical organ selected	80	1.29e-04
Cs-134m *	80	1.41e-07	101,000	1.78e-04	Wrong critical organ selected	20,000	3.52e-05
Cs-135	900	1.48e-04	794	1.31e-04			
Cs-136	800	2.42e-04	518	1.57e-04			
Cs-137	200	2.14e-04	119	1.27e-04			
Cs-138	NC		25,600	1.75e-04			
Ba-131	600	3.57e-05	2,950	1.76e-04			
Ba-133			1,520				
Ba-133m			2,620				
Ba-137m			2,150,000				
Ba-139	NC		13,800	1.74e-04			
Ba-140	90	3.91e-05	582	2.53e-04			
La-140	60	1.89e-05	652	2.06e-04			
Ce-141	300	3.93e-05	1,890	2.48e-04			
Ce-143	100	2.02e-05	1,210	2.45e-04			
Ce-144	NC		261	2.60e-04	Not in 1976 list, but in HB69	30	3.22e-05
Pr-142 *	90	2.20e-05	1,040	2.54e-04			
Pr-143	100	2.23e-05	1,170	2.61e-04			
Pr-144	NC		47,000	1.67e-04			
Pr-144m			112,000				
Nd-147 *	NC		1,250	2.64e-04	Not in 1976 list	200	4.23e-05
Nd-149 *	900	1.51e-05	11,700	1.97e-04			
Pm-147	NC		5,240	2.71e-04	Not in 1976 list, but in HB69		
Pm-148	NC		605	2.95e-04			
Pm-148m	NC		575	1.34e-04			
Pm-149	100	1.88e-05	1,380	2.60e-04			
Sm-151	1,000	1.60e-05	14,100	2.26e-04			
Sm-153	200	2.74e-05	1,830	2.51e-04			
Eu-152 *	60	1.16e-05	841	1.62e-04	Reclassified as Eu-154m	200	1.84e-05
Eu-154	200	6.46e-05	573	1.85e-04	MCL switched with Eu-152	60	1.94e-05
Eu-155	600	3.27e-05	3,590	1.95e-04			
Eu-156	NC		600	2.17e-04			

Nuclide (*half-life of 24 hours or less)	1976 limits based on critical organ at 4 mrem/yr	1976 Risks	1991 proposed limits at 4 ede mrem/yr	1991 risks	Comments (HB69 means National Bureauof Standards Handbook-69)	Corrected limits based on 4 mrem/yr critical	Risk at corrected limits
Gd-153	600	2.62e-05	4,680	2.04e-04			
Gd-159 *	200	1.82e-05	2,760	2.50e-04			
Tb-158	NC		1,250	1.81e-04			
Tb-160	100	2.50e-05	815	2.03e-04			
Dy-165 *	1,000	1.29e-05	15,100	1.95e-04			
Dy-166	100	3.14e-05	830	2.61e-04			
Ho-166	90	2.35e-05	981	2.56e-04			
Er-169	300	2.14e-05	3,640	2.60e-04			
Er-171 *	300	1.76e-05	3,800	2.23e-04			
Tm-170	100	2.53e-05	1,030	2.61e-04			
Tm-171	1,000	1.99e-05	12,700	2.52e-04			
Yb-169	NC		1,830	2.09e-04			
Yb-175	300	2.44e-05	3,110	2.53e-04			
Lu-177	300	2.99e-05	2,550	2.54e-04			
Hf-181	200	3.64e-05	1,170	2.13e-04			
Ta-182	100	2.29e-05	842	1.93e-04			
W-181	1,000	1.15e-05	19,000	2.18e-04			
W-185	300	2.50e-05	3,440	2.86e-04			
W-187 *	200	2.11e-05	2,660	2.80e-04			
Re-183	2,000		5,400		Unknown risk		
Re-186	300	4.69e-05	1,880	2.94e-04			
Re-187	9,000	4.83e-06	582,000	3.13e-04			
Re-188 *	200	2.56e-05	1,790	2.29e-04			
Os-185	200	1.15e-05	2,460	1.42e-04			
Os-191	600	6.19e-05	2,380	2.46e-04			
Os-191m *	9,000	1.57e-04	14,300	2.49e-04			
Os-193	200	3.00e-05	1,690	2.54e-04			
Ir-190	600	9.88e-05	1,010	1.66e-04			
Ir-192	100	2.12e-05	957	2.03e-04			
Ir-194 *	90	2.21e-05	1,040	2.56e-04			
Pt-191	300	1.51e-05	3,810	1.92e-04			
Pt-193	3,000	1.79e-05	46,100	2.75e-04			
Pt-193m	3,000	2.58e-04	3,020	2.59e-04			
Pt-197	300	2.23e-05	3,400	2.53e-04			
Pt-197m *	3,000	3.63e-05	17,500	2.12e-04			
Au-196	600		3,660		Unknown risk		
Au-198	100	1.79e-05	1,310	2.35e-04			

Nuclide (*half-life of 24 hours or less)	1976 limits based on critical organ at 4 mrem/yr	1976 Risks	1991 proposed limits at 4 ede mrem/yr	1991 risks	Comments (HB69 means National Bureauof Standards Handbook-69)	Corrected limits based on 4 mrem/yr critical	Risk at corrected limits
Au-199	NC				Not in 1976 list, but in HB69	600	5.10e-05
Hg-197	NC		5,760	2.52e-04	Not in 1976 list, but in HB69	880	3.85e-05
Hg-197m	NC					600	5.51e-05
Hg-198							
Hg-203	NC		2,390	2.27e-03	Not in 1976 list, but in HB69	60	5.7e-04
Tl-200	NC				Not in 1976 list, but in HB69	1200	2.7e+00
Tl-201	NC				Not in 1976 list, but in HB69	880	1.11e-05
Tl-202	300	1.50e-05	3,840	1.92e-04			
Tl-204	300	5.43e-05	1,680	3.04e-04			
Tl-207			400,000				
Tl-208			283,000				
Tl-209			358,000				
Pb-203	1,000	3.04e-05	5,060	1.54e-04			
Pb-209	NC		25,300	1.88e-04			
Pb-210	NC		1	3.34e-05			
Pb-211	NC		12,800	2.03e-04			
Pb-212	NC		123	9.81e-05			
Pb-214	NC		11,800	1.52e-04			
Bi-206	100	2.29e-05	656	1.50e-04			
Bi-207	200	3.31e-05	1,010	1.67e-04			
Bi-212			5,200				
Bi-213	NC		15,000	2.79e-04			
Bi-214	NC		18,900	1.55e-04			
Fr-223	NC		3,410	8.51e-04			
Ra-225	NC		9	3.80e-05			
Ra-228			7.85				
Ac-227	NC		1	1.06e-05			
Ac-228	NC		3,270	1.92e-04			
Th-231	NC		4,070	2.55e-04			
Th-234	NC		401	2.62e-04			
Pa-233	300	4.73e-05	1,510	2.38e-04			
Pa-234	NC		2,560	1.94e-04			
Pa-234m			930,000				

Nuclide (*half-life of 24 hours or less)	1976 limits based on critical organ at 4 mrem/yr	1976 Risks	1991 proposed limits at 4 ede mrem/yr	1991 risks	Comments (HB69 means National Bureauof Standards Handbook-69)	Corrected limits based on 4 mrem/yr critical	Risk at corrected limits
U-237	NC		1,780	2.46e-04			
U-240	NC		1,540	3.09e-04			
Np-236			5,960				
Np-238	NC		1,390	2.14e-04			
Np-239	NC		1,680	2.45e-04			
Np-240	NC		23,100	1.83e-04			
Np-240m			174,000				
Pu-241	NC		63	4.66e-06			
Pu-243	NC		16,400	2.27e-04			
Am-242m	NC		1	3.53e-06			
Bk-249					Not in 1976 list, but in HB69	1800	6.67e-05

Additional Notes:
1. 1976 values taken from Appendix B in EPA-570/9-76-003.
2. NC = not calculated.
3. 1991 values taken from Appendix B in 56 FR 33121.
4. Calculated using tap water mortality (fatal) and morbidity (total) risk coefficients from Table 2.2 of FGR-13, assuming lifetime daily intake of 2 liters of drinking water.

b. Alpha-Emitting Radionuclides

Both the current and 1991 proposed MCLs for alpha-emitting radionuclides permit up to 15 pCi/L of alpha particle radioactivity in drinking water from individual and multiple alpha emitters. EPA based the current gross alpha MCL of 15 pCi/L (including radium-226 and excluding radon and uranium) on a consideration of the risk posed by polonium-210, which the Agency believed was the most toxic alpha emitter likely to be found in drinking water besides radium-226. At that time, EPA thought that exposure to 10 pCi/L of polonium-210 posed a lifetime fatal cancer risk comparable to that from continuous lifetime ingestion of about 5 pCi/L of radium-226, that is, between 0.5 and 2 x 10^{-4}. In 1991, EPA based the revised, adjusted gross alpha MCL on revised dose and risk calculations which indicated that the 15 pCi/L limit posed a lifetime cancer risk for most alpha emitters that fell within EPA's acceptable risk range of between 10^{-6} and 10^{-4}.

Table III-4 compares the 15 pCi/L limit for individual alpha emitters with derived activity concentrations at the two reference risk levels — 5 x 10^{-5} lifetime fatal cancer risk and 1 x 10^{-4} lifetime total cancer risk.

Table III-4. Comparison of MCLs for Alpha-Emitting Radionuclides (in pCi/L)

Radionuclide	1976 MCL*	1991 MCL*	Comparable MCLs derived using current risk models** based on	
			Lifetime fatal cancer risk = 5×10^{-5}	Lifetime total cancer risk = 1×10^{-4}
Sm-146	15	15	30	48
Sm-147	15	15	33	52
Gd-148	15	15	31	46
Gd-151	15	15	1,545	1,722
Gd-152	15	15	44	66
Pt-186	15	15	4,208	4,986
Bi-210	15	15	197	219
Bi-212	15	15	1,958	2,753
Po-210	15	15	0.7	1.1
At-207	15	15	2,049	2,811
At-211	15	15	43	58
Ra-223	15	15	7	8
Ra-224	15	15	10	12
Ac-224	15	15	398	444
Ac-225	15	15	11	13
Ac-226	15	15	26	28
Th-226	15	15	1,823	2,936
Th-227	15	15	37	41
Th-228	15	15	15	18
Th-229	15	15	6	9
Th-230	15	15	16	21
Th-232	15	15	14	19
Pa-227	15	15	1,321	2,017
Pa-228	15	15	474	540
Pa-230	15	15	442	508
Pa-231	15	15	6	8
Pa-232	15	15	494	559
Np-237	15	15	24	32
Pu-234	15	15	2,017	2,278
Pu-235	15	15	288,506	447,917
Pu-236	15	15	18	26

Table III-4. Comparison of MCLs for Alpha-Emitting Radionuclides (in pCi/L)
(continued)

Radionuclide	1976 MCL*	1991 MCL*	Comparable MCLs derived using current risk models** based on	
			Lifetime fatal cancer risk = 5×10^{-5}	Lifetime total cancer risk = 1×10^{-4}
Pu-237	15	15	3,027	3,388
Pu-238	15	15	10	15
Pu-239	15	15	9	15
Pu-240	15	15	9	14
Pu-242	15	15	10	15
Pu-244	15	15	9	14
Pu-245	15	15	392	437
Pu-246	15	15	102	113
Am-237	15	15	28,539	38,580
Am-238	15	15	15,545	20,329
Am-239	15	15	1,258	1,417
Am-240	15	15	661	756
Am-241	15	15	13	19
Am-242	15	15	975	1,094
Am-243	15	15	13	19
Cm-240	15	15	49	56
Cm-242	15	15	43	51
Cm-243	15	15	15	21
Cm-244	15	15	17	23
Cm-245	15	15	13	19
Cm-246	15	15	13	19
Cm-247	15	15	14	20
Cm-248	15	15	15	22
Cm-250	15	15	43	62
Cf-252	15	15	29	39

*Assuming no radium-226, radon, or uranium.

**Calculated using tap water mortality (fatal) and morbidity (total) risk coefficients from Table 2.2 of FGR-13, assuming lifetime daily intake of 2 liters of drinking water.

c. Radium

As discussed previously in Section III.A.2.b, EPA set the current MCL of 5 pCi/L for radium-226 and radium-228, combined, based on limiting the lifetime excess total cancer risk to between 0.5 and 2 x 10^{-4}. In 1991, EPA proposed separate, and revised, MCLs for radium-226 and radium-228 of 20 pCi/L for each. At that time, EPA believed that the revised MCLs corresponded to lifetime excess fatal cancer risks of 1 x 10^{-4} each, or 2 x 10^{-4} combined, assuming lifetime ingestion.

Table III-5 compares the current and 1991 proposed MCLs for radium-226 and radium-228 with derived activity concentrations at the two reference risk levels used to evaluate the MCLs for alpha emitters and the beta/photon emitters. As shown in this table, comparable MCLs for radium-226 and radium-228 calculated using current risk models at the specified target risk limits are less than those established in 1976 and 1991.

Table III-5. Comparison of MCLs for Radium-226 and Radium-228 (in pCi/L)

Radionuclide	1976 MCL	1991 Proposed MCL	Comparable MCLs derived using current risk models* based on	
			Lifetime fatal cancer risk = 5 x 10^{-5}	Lifetime total cancer risk = 1 x 10^{-4}
Ra-226 only	5	20	3.4	5
Ra-228 only	5	20	1.2	2

*Calculated using tap water mortality (fatal) and morbidity (total) risk coefficients from Table 2.2 of FGR-13, assuming lifetime daily intake of 2 liters of drinking water.

d. Uranium

In 1991, EPA proposed an MCL of 20 μg/L for uranium based on kidney toxicity and a corresponding limit of 30 pCi/L based on cancer risk. The MCLG was proposed at zero because of the carcinogenicity of uranium, and the MCL was proposed at the most sensitive endpoint, kidney toxicity. The MCL was based on kidney effects seen in the 30 day study in rats by Maynard and Hodge, 1949.

Since the 1991 proposal, a number of new studies have been published in peer-reviewed journals. A literature search was conducted and covered the time period between January1991 to July, 1998. Databases searched were TOXLINE, MEDLINE, EMBASE, BIOSIS, TSCATS and

Current Contents. The results of the literature search were reviewed and articles were identified, retrieved and reviewed and analyzed.

i. Health Effects in Animals

The potential toxic effects of uranium following oral exposures have been evaluated in recent animal studies (Gilman et al. 1998 a, b; McDonald-Taylor et al. 1992). In a 28-day range-finding study, male and female Sprague-Dawley rats (15/sex/group) were administered concentrations of 0, 0.96, 4.8, 24, 124, or 600 mg uranyl nitrate/L (UN/L) in drinking water for a period of 28 days (Gilman et al. 1998a). Results of the study showed no significant dose-related effects on body weight gain, food intake, fluid consumption, clinical signs, or hematological parameters of treated animals when compared with control animals. Histologic examinations indicated no statistically significant differences in the incidence of a particular lesion in animals in the 600 mg UN/L treatment group when compared with animals in the control group. However, a slight increase in the number of affected animals in the 600 mg UN/L group was observed, when compared with the control group.

Concentration-related increases in the glomerular basement membrane thickness were reported in rabbits that received 0, 24, or 600 mg uranyl nitrate/L in the drinking water for 91 days (McDonald-Taylor et al. 1992). In a 91-day study by Gilman et al. (1998a), rats received uranyl nitrate in the drinking water at concentrations that corresponded to doses of 0, 0.09, 0.42, 2.01, 9.98 and 53.56 mg/kg/day for female rats and 0, 0.06, 0.31, 1.52, 7.54 and 36.73 mg/kg/day for male rats. Results of the histologic examination revealed statistically-significant dose-related increases in the incidence of renal lesions in treated rats, when compared with controls. In the kidneys of the high-dose males, these changes included dilation of tubules, apical displacement, vesiculation of tubular nuclei, and cytoplasmic vacuolation and degranulation. The changes observed in high-dose females included sclerosis of glomerular capsules and reticulin sclerosis of tubular basement membranes and interstitial scarring. However, there were no statistically-significant dose-related changes in any clinical chemistry indicators of renal function. Renal lesions similar to the ones reported in rats by Gilman et al. (1998a) were also observed in the kidneys of rabbits that received uranyl nitrate in the drinking water at concentrations that corresponded to intakes of 0, 0.05, 0.20, 0.88, 4.82 and 28.70 mg U/kg/day in male rabbits and 0, 0.49, 1.32 and 43.02 mg U/kg/day in female rabbits (Gilman et al. (1998b).

Novikov and Yudina (1970) evaluated the long-term effects of exposure to low-levels of uranium in drinking water. Female rabbits and male albino rats were exposed to 0, 0.02, 0.2, and 1 mg/kg uranyl nitrate for 12 months or 0.05, 0.6, 6, and 60 mg/L uranyl nitrate for 11 months, respectively. Results of the study indicated a decrease in acid phosphatase activity in the spleens of rabbits in the 1 mg/kg group, but not in rats, when compared to controls. A statistically-significant ($p<0.05$) increase in serum alkaline phosphatase activity was observed by the eleventh month of exposure in rats in the 6 and 60 mg/L groups, when compared with controls. A statistically-significant decrease in the content of nucleic acids in the renal and hepatic tissues was observed in rats in the 60 mg/L group and in rabbits in the 1 mg/kg group, when compared

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with controls.

ii. *Health Effects in Humans*

Recent epidemiological studies have evaluated the effects observed in humans exposed to uranium in the drinking water (Mao et al. 1995; Limson Zamora et al. 1998). Mao et al. (1995) conducted an epidemiological study in which the relationship between uranium levels in the drinking water and urine albumin, an indicator of renal dysfunction, was evaluated. Three sites were selected for the controls (site 1) and the exposed groups (sites 2 and 3), with mean uranium water levels of 0.71, 19.6 and 14.7 μg/L reported for sites 1, 2 and 3, respectively. An index of uranium exposure was estimated for each study participant by multiplying the uranium concentration in the water supply by the average number of cups consumed at each residence and the total number of years at that residence. Based on the results of a linear regression analysis, which included terms for age, diabetes, sex, smoking, and the use of water filters and softeners, a statistically-significant association was reported for cumulative exposure to uranium and urine albumin levels. However, the authors noted that for most of the study participants, the urine albumin levels were within the range of normal values.

Limson Zamora et al. (1998) evaluated the renal effects following chronic exposure to uranium in the drinking water in a cohort from a village in Nova Scotia. Two groups were evaluated, a low exposure group (uranium levels $< 1\mu$g/L) and a high exposure group (uranium levels $\geq 1\mu$g/L). Twenty-four hour and 8-hour urine samples were collected and evaluated for uranium, creatinine, glucose, protein, β_2-microglobulin (BMG), alkaline phosphatase (ALP), gamma glutamyl transferase (GGT), lactate dehydrogenase (LDH), and N-acetyl-β-D-glucosaminidase (NAG). Statistically significant positive correlations were reported with uranium intake for glucose (males, females and pooled data), ALP (pooled data) and BMG (pooled data). No other statistically significant differences were reported. Based on these results, the authors concluded that the proximal tubule was the site of uranium nephrotoxicity.

In June 1998, a workshop was held by the USEPA to discuss issues associated with assessing the risk associated with uranium exposure and updating the RfD and MCLG for uranium. The numerous technical issues associated with the development of a risk assessment for uranium in drinking water were discussed. Based on these discussions, it was apparent that there is a range of values for each factor used in the development of the RfD and MCL for uranium. However, based upon the input received at the workshop and the most current information, EPA believes that the LOAEL for renal effects in male rats of 0.06 mg U/kg/day reported by Gilman et al. (1998a) could be used for the development of an RfD for uranium. The RSC was revised to 80 percent (0.8). The total uncertainty factor was determined to be about 100 (about 3 for animal to human extrapolation, about 10 for intraspecies differences, about 1 for a less than lifetime study, and about 3 for the use of a LOAEL), with the body weight of 70 kilograms (kg) and daily water consumption of two liters used in the calculation. These assumptions are consistent with the data presented at the workshop and appear to be reasonable and justifiable. EPA believes these factors allow for the calculation of a safe level of uranium in

drinking water (in terms of chemical toxicity).

The application of the total uncertainty factor of 100 to the LOAEL of 0.06 mg/kg/day reported by Gilman et al. (1998a) results in an RfD of 0.6 μg uranium/kg/day. The RfD can be used to determine the MCL by multiplying the RfD by body weight (70 kg) and RSC (0.8) and dividing by water consumption (2 L), resulting in a value of 17 μg uranium/L, which can be rounded off to 20 μg/L.

4. Consideration of Sensitive Sub-populations

a. Children's Environmental Health

In compliance with Executive Order 13045 "Protection of Children from Environmental Health Risks and Safety Risks" (62 F. R. 19885, April 23, 1997) and EPA "Policy on Evaluating Health Risks to Children" (October, 1995), risks to children from radionuclides have been considered. There is evidence that children are more sensitive to radiation than adults, the risk per unit exposure in children being greater than in adults.

Risk coefficients used by the Agency for radiation risk assessment explicitly account for these factors. The age-specific, organ-specific risk per unit dose coefficients used in the lifetime risk model apply the appropriate age-specific sensitivities throughout the model. The model also includes age-specific changes in organ mass and metabolism. The risk estimate at any age is the best estimate for that age. In developing the lifetime risks, the model uses the life table for a stationary population. Use of the life table allows the model to account for competing causes of death and age-specific survival. These adjustments make the lifetime risk estimate more realistic.

At the same time, consumption rates of food, water and air are different between adults and children. The lifetime risk estimates for radionuclides in water use age-specific water intake rates derived from average national consumption rates when calculating the risk per unit intake. Since the intake by children is usually less than the intake by adults, it tends to partially mitigate the greater risk in children compared to adults when evaluating lifetime risk.

While radiation protection organizations have developed the concept of committed dose, the dose to an organ or tissue from time of intake to end of life, there is no equivalent for risk. If we define "committed risk" as the lifetime risk from a given intake, then it will be easier to compare the risks of intakes at different times of life. In Table III-6, the "committed risk" is given for 5 isotopes and 5 periods of life and continuous lifetime exposure. If the radionuclide concentration in the water is kept constant, the fraction of the lifetime risk committed during any age interval will also remain constant. Unless the intake is restricted in an age-specific manner, the fraction of the lifetime risk contributed by any age interval is a constant.

Table III-6 Lifetime Risks and Fractions of Lifetime Risk Per Age Group

Lifetime Risk for Intake of Water containing 1 Bq/L during several different age intervals						
Age (yrs)	**0-6**	**6-18**	**18-30**	**30-70**	**70-110**	**0-110**
Ra-224	2.3e-05	3.3e-05	1.1e-05	1.5e-05	9.8e-07	8.4e-05
Ra-226	2.9e-05	8.6e-05	5.0e-05	5.1e-05	2.9e-06	2.2e-04
Ra-228	1.1e-04	2.6e-04	1.2e-04	1.1e-04	5.1e-06	6.1e-04
U-238	6.7e-06	1.2e-05	6.1e-06	9.8e-06	3.7e-07	3.4e-05
H-3	3.9e-09	8.5e-09	6.2e-09	9.6e-09	6.7e-10	2.9e-08
Percentage of Lifetime Risk Committed for Water Intake during the Age Interval						
Age (yrs)	**0-6**	**6-18**	**18-30**	**30-70**	**70-110**	**0-110**
Ra-224	28%	40%	13%	18%	1%	100%
Ra-226	13%	39%	23%	23%	1%	100%
Ra-228	17%	43%	20%	19%	1%	100%
U-238	19%	33%	18%	28%	1%	100%
H-3	13%	29%	21%	33%	2%	100%

b. Subgroups Potentially at Greater Risk

i. Risk to Children

As noted above, the age-specific, sex-specific models used by the EPA for estimating risk from ionizing radiation explicitly provide for risks in children, and in women. The computer program suite, DCAL (ref), uses age-specific metabolic models to calculate the dose from a unit intake of a radioisotope during each year of life from birth to 120 years of age. Age-specific organ masses are used for all ages up to adult, and for adult males and adult females. Risk coefficients are given by age and sex for each year of life from birth to 120 years of age[1]. The risk is then calculated by combining calculated doses and age-sex-specific risk coefficients with age-sex-specific intake data and age-sex-specific survival data.

These lifetime risk factors[2] refer to the accumulated risk incurred over a lifetime of water ingestion. The ingestion rates are gender specific and are assumed to change over the course of the lifetime of an individual, as does the ratio of the volume of water ingested to the average individual body mass (liters ingested per kilogram of body mass). The lifetime risk is additive

[1] It should be noted that the contribution to lifetime risk incurred between the ages of 70 and 120 years is only around one percent of the lifetime risk.

[2] These risks are generally expressed in terms of the activity of the radionuclide of interest, i.e., as the risk per pCi/L.

with time, so it increases as time passes, but the rate of risk accumulation is not constant due to ingestion rate and body mass changes. For example, as can be seen in Table III-6, more than 60% of the lifetime risk occurs in the first 18 years of life. On a per annum basis, this translates to average annual risks that are over ten times greater during the first 18 years of life relative to the over-all average annual risk. The general risk assessment previously described for community water systems takes this increased sensitivity for children into account in its estimation of lifetime risks. However, since some non-transient non-community water systems (NTNCWSs) serve populations comprised mostly of children (e.g., schools), lifetime risk estimates for NTNCWSs must be estimated in a separate analysis[3]. Risks to school children are discussed separately in the "Non-transient Non-Community Water Systems" in Section VI of this document.

ii. Risk to Elderly

The risk factors, as outlined above for children, explicitly address the risk to the elderly. Age-specific intake, dose and risk parameters in the model are provided for all ages from birth to 120 years of age. The model provides the best estimate of risk for all ages, including the elderly, that the Agency can make at this time. The elderly do not appear to face disproportionate risks and thus were not assessed as a sensitive sub-population.

iii. Risk to Other Sensitive Sub-populations

Chromosomal aberrations are characteristic features of cancer cells. Some investigators have proposed that individuals with hereditary chromosome breakage syndromes are at greater risk of developing cancer because they are more prone to develop chromosomal aberrations. Chromosomal fragility syndromes, including Bloom's syndrome, Fanconi's anemia, Ataxia telangiectasia, Xeroderma pigmentosa, Werner's syndrome, and Nijmegen breakage syndrome, are associated with increased cancer risk. Some of these diseases, including Ataxia telangiectasia and Nijmegen breakage syndrome, result in hypersensitivity to ionizing radiation. While the model described above does not address this potential sensitive sub-population explicitly, it does address the sub-population implicitly. The National vital statistics data used in the modeling, particularly the relative risk model, reflect the proportional contribution of these sensitive sub-populations to the age-specific mortality rates.

C. REFERENCES

1. References for the 1976 Regulations

Dougherty, T.F. and Mays, C.W. Bone Cancer Induced by Internally Deposited Emitters in Beagles. In: Radium Induced Cancer, IAEA-SM-118/3. International Atomic Energy Agency,

[3] There are other reasons that a separate analysis is necessary for NTNCWSs, e.g., different water ingestion assumptions, different numbers of years of exposure over a lifetime, etc.

Vienna, Austria (1969). [Dougherty and Mays, 1969]

Evans, R. D. Radium and Mesothorium Poisoning and Dosimetry and Instrumentation Techniques in Applied Radioactivity. MIT-952-3. Division of Technical Information, ORNL, Oak Ridge, Tennessee (1966). [Evans, 1966]

Federal Radiation Council. Background Material for the Development of Radiation Protection Standards. Report #2, U.S. Department of Health, Education and Welfare, USPHS, Washington, D.C. (September 1961). [FRC, 1961]

Federal Register. Vol. 40 No. 158. Interim Primary Drinking Water Regulations. Proposed Maximum Contaminant Levels for Radioactivity. (August 14, 1975) pp. 34321-34328 [40 FR 34321, Aug. 14, 1975]

Federal Register. Vol. 41 No. 133. National Interim Primary Drinking Water Regulations; Final Rule. (July 9, 1976) pp.28402- 28409 [41 FR 28402, July 9, 1976]

International Commission on Radiological Protection (ICRP). Report of Committee II on Permissible Dose for Internal Radiation. ICRP Publication 2. Pergamon Press, New York, N.Y. (1959). [ICRP, 1959]

International Commission on Radiological Protection (ICRP). A Review of the Radio sensitivity of the Tissues in Bone. ICRP Publication 11. Pergamon Press, New York, N.Y. (1968). [ICRP, 1968]

International Commission on Radiological Protection (ICRP). Alkaline Earth Metabolism in Adult Man. ICRP Publication 20. Pergamon Press, New York, N.Y. (1972). [ICRP, 1972]

National Academy of Sciences. National Research Council. The Effects on Populations of Exposure to Low Levels of Ionizing Radiation. Division of Medical Sciences. Washington, D.C. (November 1972). [NAS, 1972]

National Bureau of Standards (NBS). Maximum Permissible Body Burdens and Maximum Permissible Concentrations of Radionuclides in Air and Water for Occupational Exposure. NBS Handbook 69. Department of Commerce. Washington, D.C. (revised 1963). [NBS, 1963]

United Nations Scientific Committee on the Effect of Atomic Radiation (UNSCEAR). Ionizing Radiation Levels and Effects, Vol. I. United Nations Publication E.72.IX.17. New York, N.Y. (1972). [UNSCEAR, 1972]

U.S. Environmental Protection Agency. National Interim Primary Drinking Water Regulations. Office of Water Supply. EPA-570/9-76-003 (September 1976). [EPA, 1976]

2. References for the 1991 Proposed Regulations

Agency for Toxic Substances and Disease Registry (ATSDR). Toxicological Profile for Radium. Atlanta, GA: Agency for Toxic Substances and Disease Registry, U.S. Public Health Service. (December 1990). [ATSDR, 1990]

Bean, J.A. et al. Drinking water and cancer incidence in Iowa. II. Radioactivity in drinking water. Am. J. Epidemiol. 11;6:924-932 (1982). [Bean et al., 1982]

Dunning, D.E. et al. A Combined Methodology for Estimating Health Effects from Exposure to Radioactive Pollutants (draft). Prepared under Interagency Agreement EPA-78-D-X0231 by Oak Ridge National Laboratory (December 1980). [Dunning et al. 1980]

Evans, R.D. et al. Radium metabolism in rats and the production of osteogenic sarcoma by experimental radium poisoning. Am. J. Roetgenol. Radium Therapy. 52:353-373 (1944). [Evans et al., 1944]

Federal Register. Vol. 51 No. 189. Water pollution Control; National Primary Drinking Water Regulations; Radionuclides; Advance Notice of Proposed Rulemaking. (Sept. 30, 1986) pp. 34836-34862 [51 FR 34836, Sept. 20, 1986]

Federal Register. Vol. 56 No. 138. National Primary Drinking Water Regulations; Radionuclides; Proposed Rule. (July 18, 1991) pp.33050- 33127 [56 FR 33050, July 18, 1991]

Finkel, M.P. Relative biological effectiveness of radium and other alpha emitters in CF No. 1 female mice. Proc. Soc. Exp. Biol. Med. 83:494-498. (1953). [Finkel, 1953]

Finkel, M.P. et al. Toxicity of radium-226 in mice. In: Erickson A., ed. Radiation induced cancer. Vienna, Austria: International Atomic Energy Agency, pp. 369-391, (1969). [Finkel et al., 1969]

Humphreys et al. The induction by Ra-224 of myeloid leukemia and osteosarcoma in male CBA mice. Int. J. Radiat. Biol. 47:239-247 (1985). Humphreys et al., 1985]

International Commission on Radiological Protection (ICRP). Recommendations of the International Commission on Radiological Protection. ICRP Publication 2. Report of Committee II on Permissible Dose for Internal Radiation. Oxford: Pergamon Press (1959). [ICRP, 1959]

International Commission on Radiological Protection (ICRP). Recommendations of the International Commission on Radiological Protection. ICRP Publication 26. Oxford: Pergamon Press (1977). [ICRP, 1977]

International Commission on Radiological Protection (ICRP). Limits for Intakes of Radionuclides by Workers. ICRP Publication 30, Part 1. Annals of the ICRP Vol. 2. (3/4) Oxford: Pergamon Press (1979). [ICRP, 1979]

Keane, A.T., et al. Non-stochastic effects of Ra-226 and Ra-228 in the human skeleton. In: Biological effects of low-level radiation. Vienna, Austria: International Atomic Energy Agency. 329-350. (1983). [Keane et al., 1983]

Lyman, G.H. et al. Association of leukemia with radium groundwater contamination. JAMA 254:621-626 (1985). [Lyman et al., 1985]

Lyman, G.H. and Lyman, C.G.. Leukemia and groundwater contamination. JAMA 256:2676-2677 (1986). [Lyman and Lyman, 1986]

Maletskos, C.J., et al. The metabolism of intravenously administered radium and thorium in human beings and the relative absorption from the human gastrointestinal tract of radium and thorium in simulated radium dial paints. In: Radium and mesothorium poisoning and dosimetry and instrumentation techniques in applied radioactivity. Cambridge, MA: MIT Physics Department. 202-317 (1966). [Maletskos, et al. 1966]

Maletskos, C.J., et al. Retention and absorption of Ra-224 and Th-234 and some dosimetric consideration of Ra-224 in human beings. In: Mays, C.E. et al., eds. Delayed effects of bone-seeking radionuclides. Salt Lake City, UT: University of Utah Press. 29-49 (1969). [Maletskos et al., 1969]

Maynard EA, Hodge HC. 1949. Study of toxicity of various uranium compounds when fed to experimental animals. In: Voegtlin C, Hodge HC, eds. Pharmacology and Toxicology of Uranium Compounds. National Nuclear Energy Series, Div. VI, Vol. 1. New York: McGraw Hill, pp. 309-376.

Mays, C.W. and Speiss, H. Bone sarcomas inpatients given radium-224. In: Boice, J.D., Fraumeni, J.F. eds. Radiation carcinogenesis; epidemiology and biological significance. New York: Raven Press. 241-252 (1984). [Mays and Spiess, 1984]

Mays, C.W., et al. Cancer Risk from the Lifetime of Ra and U Isotopes. Health Physics Vol. 48, No. 5 (May 1985). [Mays et al., 1985]

Muller J. et al. Early induction of leukemia (malignant lymphoma) in mice by protracted low alpha-doses. Health Physics (1988) 54:461-463. [Muller et al., 1988]

National Academy of Sciences, National Research Council. The Effects on Populations of Exposure to Low Levels of Ionizing Radiation (BEIR III) (1980). [NAS, 1980]

National Academy of Sciences, National Research Council. Health Risks of Radon and Other Internally Deposited Alpha-Emitters (BEIR IV) (1988). [NAS, 1988]

Norris, W.P., et al. Studies of the metabolism of radium in man. Am, J. Roentg. Rad. Therapy Nuclear Med. 73:785-802 (1955). [Norris et al., 1955]

Peterson, N.J. et al. An epidemiologic approach to low-level radium-226 exposure. Public Health Rep. 81:805-814 (1966). [Peterson et al., 1966]

Raabe, O.G., et al. Dose-response relationships for bone tumors in beagles exposed to Ra-226 and Sr-90. Health Phys. (1981), 40:863-880. [Raabe et al., 1981]

Rowland et al. Dose-response relationships for female radium dial workers. Radiation Research. Vol. 76 (1978), 366-383. [Rowland et al., 1978]

Rundo, J. et al. Current (1984) status of the study of Ra-226 and Ra-228 in humans at the Center for Human Radiobiology. In: Gossner W., Gerber, G.B., Hagen U., Luz A., eds. The Radiobiology of Radium and Thorotrast. Baltimore: Urban and Schwarzenberg, Baltimore (1986). pp 14-21. [Rundo et al., 1986]

Sharpe, W.D. Chronic radium intoxication: Clinical and autopsy findings in long-term New Jersey survivors. Environ. Res. 8:243-383 (1974). [Sharpe, 1974]

Stebbings, J.H., et al. Mortality from cancers of major sites in female radium dial workers. Am. J. Ind. Med. Vol. 5 (1984) 435-459. [Stebbings et al., 1984]

Sullivan, R.E. et al. Estimates of Health Risk from Exposure to Radioactive Pollutants (draft). Prepared under Interagency Agreement 40-1073-80 by Oak Ridge National Laboratory (November, 1981). ORNL/TM-7745. [Sullivan et al., 1981]

Taylor, G.N. et al. Radium induced eye melanomas in dogs. Radiat. Res. 51:361-373. (1972). [Taylor et al., 1972]

U.S. Environmental Protection Agency. Risk assessments Methodology. Environmental Impact Statement. NESHAPS for Radionuclides. Background Information Document—Vol. 1. Office of Radiation Programs. EPA/520/1-89-005 (September 1989). [EPA, 1989a]

U.S. Environmental Protection Agency. Occurrence and Exposure Assessment for Radium-226 in Public Drinking Water Supplies (Draft). Prepared by Wade Miller Associates, Inc. (April 30, 1990). [EPA, 1990g]

U.S. Environmental Protection Agency. Occurrence and Exposure Assessment for Radium-228 in Public Drinking Water Supplies (Draft). Prepared by Wade Miller Associates, Inc. (November

30, 1990). [EPA, 1990n]

U.S. Environmental Protection Agency. Drinking Water Criteria Document for Gross Alpha Emitters (Draft). Prepared by Clement International Corp. (June 1991). [EPA, 1991a]

U.S. Environmental Protection Agency. Drinking Water Criteria Document for Radium in Drinking Water (Draft). Prepared by Life Systems, Inc.. (June 1991). [EPA, 1991b]

U.S. Environmental Protection Agency. Drinking Water Criteria Document for Man-Made Radionuclides in Drinking (Draft). Prepared by Clement International Corp. (June 1991). [EPA, 1991d]

U.S. Environmental Protection Agency. Drinking Water Criteria Document for Uranium (Draft). Prepared by Clement International Corp. (June 1991). [EPA, 1991e]

U.S. Environmental Protection Agency. Occurrence and Exposure Document for Gross Alpha Radiation (Draft). Prepared by Wade Miller Associates, Inc. (1991). [EPA, 1991f]

U.S. Environmental Protection Agency. Health Effects Assessment Summary Tables. Annual FY-1991. OERR 9200.6-303 (91-1) NTIS PB91-921199. [EPA, 1991p]

3. References for the 2000 NODA

Gilman AP, Villeneuve DC, Secours VE, Yagminas AP, Tracy BL, Quinn JM, Valli VE, Willes RJ, Moss MA. 1998a. Uranyl nitrate: 28-day and 91-day toxicity studies studiers in the Sprague-Dawley Rats. Toxicol. Sci. 41:117-128.

Gilman AP, Villeneuve DC, Secours VE, Yagminas AP, Tracy BL, Quinn JM, Valli VE, Moss MA. 1998b Uranyl nitrate: 91-day toxicity studies studiers in the New Zealand white rabbit. Toxicol. Sci. 41:117-128.

International Commission on Radiological Protection, Limits for Intakes of Radionuclides by Workers, ICRP Publication 30, Part 1, Ann. ICRP, 2 (3/4), Pergamon Press, New York (1979). [ICRP, 1979]

International Commission on Radiological Protection, Age-Dependent Doses to Members of the Public from Intake of Radionuclides: Part 1, ICRP Publication 56, Oxford: Pergamon Press (1989). [ICRP, 1989]

International Commission on Radiological Protection, 1990 Recommendations of the International Commission on Radiological Protection. Annals of the ICRP, 21 (1-3). Oxford: Pergamon Press (1991). [ICRP, 1991]

International Commission on Radiological Protection, Age-Dependent Doses to Members of the Public from Intake of Radionuclides: Part 2, ICRP Publication 67, Oxford: Pergamon Press (1993). [ICRP, 1993]

International Commission on Radiological Protection, Human Respiratory Tract Model for Radiological Protection, ICRP Publication 66, Oxford: Pergamon Press (1994). [ICRP, 1994]

International Commission on Radiological Protection, Age-Dependent Doses to Members of the Public from Intake of Radionuclides: Part 3, ICRP Publication 69, Oxford: Pergamon Press (1995). [ICRP, 1995a]

International Commission on Radiological Protection, Age-Dependent Doses to Members of the Public from Intake of Radionuclides: Part 4, ICRP Publication 71, Oxford: Pergamon Press (1995). [ICRP, 1995b]

International Commission on Radiological Protection, Age-Dependent Doses to Members of the Public from Intake of Radionuclides: Part 5, ICRP Publication 72, Oxford: Pergamon Press (1996). [ICRP, 1996]

Linsom Zamora M, Tracy BL, Zielinski JM, Meyerhof DP, Moss MA. 1998. Chronic ingestion of uranium in drinking water: a study of kidney bioeffects in humans. Toxicol. Sci. 43:68-77.

Mao Y, Desmeules M, Schaubel D, Berube D, Dyck R; Brule, Thomas B. 1995. Inorganic components of drinking water and microalbuminuria. Environ. Res. 71: 135-140.

McDonald-Taylor CK, Bhatnagar MK, Gilman A, Yagminas A, Singh A. 1992. Uranyl nitrate-induced glomerular basement membrane alterations in rabbits: A quantitative analysis. Bull. Env iron. Contam. Toxicol. 48: 367-373.

National Center for Health Statistics, Vital Statistics Mortality Data, Detail, 1979, PB82-132340, U.S. Department of Health and Human Services, Public Health Service, National Center for Health Statistics, Hyattsville, MD (1982). [NCHS, 1982]

National Center for Health Statistics, Vital Statistics Mortality Data, Detail, 1980, PB83-261545, U.S. Department of Health and Human Services, Public Health Service, National Center for Health Statistics, Hyattsville, MD (1983). [NCHS, 1983]

National Center for Health Statistics, Vital Statistics Mortality Data, Detail, 1981, PB84-213008, U.S. Department of Health and Human Services, Public Health Service, National Center for Health Statistics, Hyattsville, MD (1984). [NCHS, 1984]

National Center for Health Statistics, *U.S. Decennial Life Tables for 1989-1991, Vol. 1 No. 1, United States Life Tables*, (PHS) 85-1150-1, U.S. Department of Health and Human Services,

Public Health Service, National Center for Health Statistics, Hyattsville, MD (1997). [NCHS, 1997]

U.S. Environmental Protection Agency. Estimating Radiogenic Cancer Risks. EPA 402-R-93-076, J.S. Puskin and C. B. Nelson, U. S. Environmental Protection Agency, Washington, DC. (June 1994). [EPA, 1994]

U.S. Environmental Protection Agency. Health Effects from Low-Level Environmental Exposure to Radionuclides: Federal Guidance Report No. 13 - Part I: Interim Version. EPA 402-R-97-014, K.F.Eckerman, R.W. Leggett, C. B. Nelson, J.S. Puskin, and A.C.B. Richardson. Ridge National Laboratory, Oak Ridge, TN, and U. S. Environmental Protection Agency, Washington, DC (January 1998). [EPA, 1998]

Uranium Issues Workshop - Sponsored by United States Environmental Protection Agency, Washington, DC ; June 23 -24, 1998

IV - ANALYTICAL METHODS

A. Introduction

The Safe Drinking Water Act, as amended in 1996, directs EPA to promulgate national primary drinking water regulations which specify either maximum contaminant levels or treatment techniques for drinking water contaminants (42 USC 300g-1). According to the SDWA, EPA is required to set an MCL "if, in the judgment of the Administrator, it is economically and technologically feasible to ascertain the level of a contaminant in water in public water systems." (SDWA section 1401[1][C][i]). Alternatively, "if, in the judgment of the Administrator, it is not economically or technologically feasible to so ascertain the level of such contaminant," the Administrator may identify known treatment techniques, which sufficiently reduce the contaminant in drinking water, in lieu of an MCL (SDWA section 1401[1][C][ii]). In addition, the National Primary Drinking Water Regulations are to include "criteria and procedures to assure a supply of drinking water which dependably complies with such maximum contaminant levels; including accepted methods for quality control and testing procedures to insure compliance with such levels." (SDWA section 1401 [1][D]). This section provides updated information about the various testing procedures, hich have been identified by EPA as techniques that provide reliable compliance monitoring of radionuclides in drinking water. Table IV-1 briefly summarizes the regulatory events associated with:

- The testing procedures for regulated radionuclides approved in 1976;
- Major analytical additions or changes proposed or discussed in the 1991 radionuclides rule;
- Testing procedures and protocols approved in the March 5, 1997.radionuclides methods rule (62 FR 10168, cited 40 CFR 141.25; and
- Items which will be discussed in NODA.

B. Updates to the Analytical Techniques

In 1976, EPA published interim standards for radionuclides in drinking water and approved radiochemical methods to analyze for gross alpha-particle activity, radium-226, total gross radium alpha, gross beta-particle activity, strontium-89 and strontium-90, cesium-134, and uranium. (These interim standards were declared to be final National Primary Drinking Water Regulations in the Safe Drinking Water Amendments of 1986.) On July 18, 1991 (56 FR 33050), EPA proposed to approve fifty-six additional methods for the measurement of radionuclides in drinking water (excluding radon). Of the fifty-six that were proposed, fifty-four were actually approved in the March 5, 1997 final methods rule (62 FR 10168). In response to public comments from the 1991 proposed rule, EPA also evaluated and approved (on March 5, 1997) twelve techniques submitted by the public . In total, EPA approved 66 radiochemical methods on March 5, 1997 (62 FR 10168). Currently, 89 radiochemical methods are approved for compliance monitoring of radionuclides in drinking water. Table IV-2 of this of document the methods listed in 40 CFR 141.25.

Table IV-1. Brief Summary of the Regulatory Events Associated with Radiochemical Methods

1976 National Primary Drinking Water Regulations	July 18, 1991 Radionuclides Proposed Rule	March 5, 1997 Radionuclide Methods Final Rule	The 2000 Notice of Data Availability
The 1976 NPDWR approved: * Radiochemical methods to analyze for gross alpha-particle activity, radium-226, total radium, gross beta-particle activity, strontium-89 and -90, cesium-134 and uranium. * Defined the detection limit (DL) as the required measure of sensitivity and listed the required DL for each regulated radionuclide.	The July 18, 1991 radionuclides rule proposed: * Fifty-six additional methods for compliance monitoring of radionuclides. * Guidance for sample handling, preservation and holding times that were cited in the 1990 U.S.EPA "Manual for the Certification of Laboratories Analyzing Drinking Water." *The use of practical quantitation limits (PQLs) and acceptance limits as the measures of sensitivity for radiochemical analysis.	The March 5, 1997 final rule for radionuclide methods: *Approved 66 additional radionuclide techniques for gross alpha-particle activity, radium-226, radium-228, uranium, cesium-134, iodine-131, and strontium-90. * Responded to comments regarding the analytical methods (excluding radon) received from the July 18, 1991 proposed radionuclides rule.	Updates the public on changes that have occurred regarding radiochemical methods of analysis since the 1991 proposed rule. The updates that will be discussed in the 2000 NODA include: * A brief discussion of the analytical methods updates which were promulgated by the Agency on July 18, 1997 final rule. * Guidance for sample handling, preservation and holding times listed in the 1997 U.S.EPA "Manual for the Certification of Laboratories Analyzing Drinking Water." * Recommendations for the analysis of short-lived, alpha-emitting radioisotopes (i.e., radium-224). * Revised cost estimates for radiochemical analysis. * The Agency's intent to continue to use the detection limits defined in the 1976 rule as the required measures of sensitivity. * Response to some of the comments on the 1991 proposed radionuclides. * The externalization of the Performance Evaluation Program. * The Agency's plans to implement a Performance Based Measurement System.

Table IV-2. Analytical Methods Approved by EPA for Radionuclide Monitoring (40 CFR 141.25)

Radioactive Contaminant	Source	Radiochemical Methodology	Reference Method and/or Page Number								
			EPA [1]	EPA [2]	EPA [3]	EPA [4]	SM [5]	ASTM [6]	USGS [7]	DOE [8]	Other
Gross alpha[11] and beta	Natural	Evaporation	900.0	p. 1	00-01	p. 1	302, 7110 B		R-1120-76		
Gross alpha[11]	Natural	Co-precipitation	.....		00-02		7110 C				
Radium-226	Natural	Radon emanation	903.1	p. 16 p. 13	Ra-04 Ra-03	p. 19	7500-Ra C... 304, 305	D 3454--91 D 2460-90	R-1141-76	Ra-05	N.Y. [9]
		Radiochemical	903.0				7500-Ra B		R-1140-76		
Radium-228	Natural	Radiochemical	904.0	p. 24	Ra-05	p. 19	304, 7500-Ra D		R-1142-76		N.Y. [9] N.J. [10]
Uranium[12]	Natural	Radiochemical	908.0				7500-U B...				
		Fluorometric	908.1				7500-U C (17th Ed.)	D2907-91	R-1180-76 R-1181-76	U-04	
		Alpha spectrometry	.....		00-07	p. 33	7500-U C (18th or 19th Ed.)	D 3972-90	R-1182-76	U-02	
		Laser phosphorimetry	.....				.....	D 5174-91			
Radioactive Cesium-134	Man-Made	Radiochemical	901.0	p. 4			7500-Cs B...	D 2459-72	R-1111-76		
		Gamma ray spectrometry	901.1			p. 92	7120 (19th Ed.)	D 3649-91	R-1110-76	4.5.2.3	
Radioactive Iodine-131	Man-Made	Radiochemical	902.0	p. 6 p. 9			7500-I B.... 7500-I C.... 7500-I D....	D3649-91		4.5.2.3	
		Gamma ray spectrometry	901.1			p. 92	7120 (19th Ed.)	D 4785-88		4.5.2.3	
Radioactive Strontium -89, -90	Man-Made	Radiochemical	905.0	p. 29	Sr-04	p. 65	303, 7500-Sr B		R-1160-76	Sr-01 Sr-02	
Tritium - ^{3}H	Man-Made	Liquid scintillation	906.0	p. 34	H-02	p. 87	306, 7500-3H B	D 4107-91	R-1171-76		

Table IV-2. Analytical Methods Approved by EPA for Radionuclide Monitoring (40 CFR 141.25) (continued)

Radioactive Contaminant	Source	Radiochemical Methodology	Reference Method and/or Page Number								
			EPA [1]	EPA [2]	EPA [3]	EPA [4]	SM [5]	ASTM [6]	USGS [7]	DOE [8]	Other
Gamma emitters		Gamma ray spectrometry	901.1			p. 92	7120 (19th Ed.)	D 3649-91	R-1110-76	4.5.2.3	
		Radiochemical	902.0 901.0				7500-Cs B.... 7500-I B	D 4785-88			

1 "Prescribed Procedures for Measurement of Radioactivity in Drinking Water," EPA 600/4-80-032 , August 1980. Available at U.S. Department of Commerce, National Technical Information Service, 5285 Port Royal Road, Springfield, VA 22161 (Telephone 800-553-6847), PB 80-224744.

2 "Interim Radiochemical Methodology for Drinking Water," EPA 600/4-75-008 (revised), March 1976. Available at NTIS, *ibid.* PB 253258.

3 "Radiochemistry Procedures Manual", EPA 520/5-84-006, December 1987. Available at NTIS, *ibid.* PB 84-215581.

4 "Radiochemical Analytical Procedures for Analysis of Environmental Samples," U.S. Department of Energy, March 1979. Available at NTIS, *ibid.* EMSL LV 053917.

5 "Standard Methods for the Examination of Water and Wastewater," 13th, 17th, 18th, and 19th Editions, 1971, 1989, 1992, 1995. Available at American Public Health Association, 1015 Fifteenth Street, N.W., Washington, D.C. 20005. All methods are in the 17th, 18th and 19th editions except 7500-U C Fluorometric Uranium. This method was discontinued after the 17th Edition. 7120 Gamma Emitters is only in the 19th Edition, and 302, 303, 304, 305 and 306 are only in the 13th Edition.

6 Annual Book of ASTM Standards, Vol. 11.02, 1994. Available at American Society for Testing and Materials, 100 Barr Harbor Drive, West Conshohocken, PA 19428.

7 "Methods for Determination of Radioactive Substances in Water and Fluvial Sediments," Chapter A5 in Book 5 of Techniques of Water-Resources Investigations of the United States Geological Survey, 1977. Available at U.S. Geological Survey Information Services, Box 25286, Federal Center, Denver, CO 80225-0425.

8 "EML Procedures Manual," 27th Edition, Volume 1, 1990. Available at Environmental Measurements Laboratory, U.S. Department of Energy, 376 Hudson Street, New York, NY 10014-3621.

9 "Determination of Ra-226 and Ra-228 (Ra-02)," January 1980; Revised June 1982. Available at Radiological Sciences Institute Center for Laboratories and Research, New York State Department of Health, Empire State Plaza, Albany, NY 12201.

10 "Determination of Radium 228 in Drinking Water," August 1980. Available at State of New Jersey, Department of Environmental Protection, Division of Environmental Quality, Bureau of Radiation and Inorganic Analytical Services, 9 Ewing Street, Trenton, NJ 08625.

11 Natural uranium and thorium-230 are approved as gross alpha-particle activity calibration standards for the gross alpha co-precipitation and evaporation methods; americium-241 is approved for use with the gross alpha co-precipitation methods.

12 If uranium (U) is determined by mass-type methods (i.e., fluorometric or laser phosphorimetry), a 0.67 pCi/μg uranium conversion factor must be used. This conversion factor is conservative and is based on the 1:1 activity ratio of U-234 to U-238 that is characteristic of naturally-occurring uranium in rock.

The analytical methods listed in Table IV-2 of this document are technically and economically feasible for the monitoring radionuclides in drinking water at the current MCLs and detection limits. The reliability of these methods has been demonstrated by many years of use by State, Federal, and private laboratories for both drinking water samples and performance evaluation samples. Most of the methods have undergone either an inter-laboratory collaborative study or have been subjected to single laboratory tests. The majority of the validation studies were either performed or sponsored by EPA. Validations performed by accredited standard-setting bodies such as the American Society for Testing and Materials (ASTM) were reviewed by EPA and determined to be acceptable. Copies of the complete methods listed in Table IV-2 are available from sources listed in the table references. Copies of the EPA methods are available from the National Technical Information Service (NTIS), U.S. Department of Commerce, 5285 Port Royal Road, Springfield, Virginia 22161. The NTIS may also be reached at 1-800-553-6847. EPA refers readers to the references for information on precision, detection limits, accuracy, counting efficiency, background determination, sample and source preparations, interferences, and calibration information about the analytical methods.

Since the proposed 1991 proposed rule (56 FR 33050) and the March 5, 1997 final radionuclides method rule (62 FR 10168), there have been updates to the sample collection, handling, and preservation procedures, as well as updates to specific radiochemical methods. Below is a brief description of these updates.

1. Updates to Sample Collection, Handling, and Preservation

In 1991, EPA provided sample collection, handling, and preservation procedures for samples being analyzed for regulated radionuclides. EPA stated that sample collection for gross alpha, gross beta, radium and uranium analysis should be performed using the methods for inorganic contaminant monitoring described in EPA's "Manual for the Certification of Laboratories Analyzing Drinking Water" (EPA, 1990b). A new version of the certification manual was published in 1997 (EPA 815-B-97-001). Table IV-3 outlines the 1997 updates to sample handling, preservation, holding times, and instrumentation that appeared in this manual. Table IV-3 also includes additional recommendations for radiochemical instrumentation (footnoted by the number 6).

Table IV-3. Sample Handling, Preservation, Holding Times and Instrumentation

Parameter	Preservative [1]	Container [2]	Maximum Holding Time [3]	Instrumentation [4]
Gross Alpha	Concentrated HCl or HNO_3 to pH < 2[5]	P or G	6 months	A, B or G
Gross Beta	Concentrated HCl or HNO_3 to pH < 2[5]	P or G	6 months	A or G
Radium-226	Concentrated HCl or HNO_3 to pH < 2	P or G	6 months	A, B, C[6], D or G
Radium-228	Concentrated HCl or HNO_3 to pH < 2	P or G	6 months	A, B[6], C[6] or G
Uranium natural	Concentrated HCl or HNO_3 to pH < 2	P or G	6 months	A[6], F, G[6], or O
Cesium-134	Concentrated HCl to pH < 2	P or G	6 months	A, C or G
Strontium-89 and -90	Concentrated HCl or HNO_3 to pH < 2	P or G	6 months	A or G
Radioactive Iodine-131	None	P or G	8 days	A, C or G
Tritium	None	G	6 months	E
Gamma/Photon Emitters	Concentrated HCl or HNO_3 to pH < 2	P or G	6 months	C

1 It is recommended that the preservative be added to the sample at the time of collection. It is recommended that samples be filtered if suspended or settleable solids are present at any level observable to the eye prior to adding preservative. This should be done at the time of collection. If the sample has to be shipped to a laboratory or storage area, however, acidification of the sample (in its original container) may be delayed for a period not to exceed 5 days. A minimum of 16 hours must elapse between acidification and start of analysis.

2 P = Plastic, hard or soft; G = Glass, hard or soft.

3 Holding time is defined as the period from time of sampling to time of analysis. In all cases, samples should be analyzed as soon after collection as possible. If a composite sample is prepared, a holding time cannot exceed 12 months.

4 A = Low background proportional system; B = Alpha and beta scintillation system; C = Gamma spectrometer [Ge(Hp) or Ge (Li)]; D = Scintillation cell system; E = Liquid scintillation system; F = Fluorometer; G = Low background alpha and beta counting system other than gas-flow proportional; O = Other approved methods (e.g., laser phosphorimetry and alpha spectrometry for uranium).

5 If HCl is used to acidify samples which are to be analyzed for gross alpha or gross beta activities, the acid salts must be converted to nitrate salts before transfer of the samples to planchets.

6 Additional instrumentation that was not listed in the USEPA 1997 "Manual for the Certification of Laboratories Analyzing Drinking Water."

2. Updates Regarding Specific Analytical Techniques

On July 18, 1991 (56 FR 33050), EPA proposed to approve 56 additional methods for the monitoring of radionuclides in drinking water. In response to the proposed rule, commenters asked for the approval of 12 methods that were not included in the proposal. Of these 68 radionuclide methods, EPA approved 66 of these 68 methods and modifications to calibration standards on March 5, 1997 (62 FR 10168). Below is a brief description of these updates. For more details, EPA refers readers to the aforementioned *Federal Register* notices. Also discussed below is the Agency's recommendations for determining the presence of the short-lived, alpha-emitting radium-224 isotope.

a. Gross Alpha-Particle and Gross Beta-Particle Activity Methods

Gross alpha-particle and gross beta-particle activity methods are straightforward radioanalytical techniques. The simplest method, evaporation of a sample, involves no separation between the nuclides and the bulk of the dissolved solids in the sample. Using the evaporation method, a sample of preserved drinking water is evaporated to a small volume and then transferred to a stainless steel planchet, where the final remaining volume is evaporated to dryness. The alpha and beta particles within the dried residue are counted using one of the gross alpha or gross beta counting methods cited in Table IV-2. While evaporation methods are relatively simple techniques, sensitivity can be improved by using methods which employ chemical separation of a group of target nuclides or of a specific nuclide.

i. Gross Alpha by Co-precipitation

In 1997, the Agency approved the use of the gross alpha co-precipitation technique to address the effect of dissolved solids on the gross alpha-particle measurement. When dissolved solids in water are dried to a residue on the planchet, the alpha-particle emissions may be attenuated due to incorporation of a significant fraction of the alpha-emitting parent into the internal volume of the solid mass of the crystals. Alpha particles travel very short distances in matter before being absorbed. The inability of the alpha particles to penetrate beyond the dried residue reduces the sensitivity of the measurement by reducing the overall efficiency (i.e., the ratio of measured counts per minute (cpm) to the calculated number of transformations per minute in a standard). As more of the alpha-particle emitting radionuclide is incorporated into the volume of the crystals, less of the nuclide is on the surface to emit the alpha particles towards the detector. In some cases, the net reduction of emitted alphas can require an inordinately long (>200 minutes) counting time to achieve the required detection limit (3 ± 3 pCi/L) or the emission rate is so low that the detection limit cannot be achieved at all.

The gross alpha co-precipitation techniques reduce the effect of dissolved solids on the alpha emission rate by isolating some or all of the nuclides of interest from the bulk of dissolved solids in a sample. Alpha particle-emitting radium and uranium are separated (precipitated) from the bulk of non-radioactive dissolved solids and other interfering nuclides in the sample with

small masses of non-radioactive reagents. These co-precipitation techniques increase the sensitivity for the target alpha particle-emitting radionuclides (e.g., radium and uranium) because a larger sample size can be used and the mass of the precipitate remains essentially constant as the sample size increases. For samples for which the dissolved solids significantly reduce the counting efficiency to 10 percent or less and the resulting sample volume is less than 150 milliliters (mL), the Agency recommends, as a general guideline, the use of the gross alpha co-precipitation technique to improve method sensitivity by allowing the use of larger sample volumes.

ii. Radiochemical Methods Used for Screening

In addition to being used to determine compliance with the MCLs, gross alpha-particle and gross beta-particle methods can be used as screening procedures to determine if additional analyses for other radionuclides are necessary. The gross alpha-particle method, method 900.0, can be used to screen for total gross alpha-particle activity, radium-226, uranium, or any other alpha-particle emitting nuclide. The gross radium alpha-particle method, 903.0, can be used to screen for radium-226 or other nuclides with radium-like chemistry which precipitate as a barium or lead sulfate from a strong acid medium.

In 1991, EPA proposed an MCL of 20 pCi/L each for radium-226 and radium-228 and recommended the use of the gross beta-particle method to screen for the presence of radium-228. This recommendation was appropriate given the proposed increase in the MCLs. However, the current MCL of 5 pCi/L of radium-226 and radium-228 combined needs a different approach. The portion of radium-228 allowable under the existing MCL is significantly lower than the detection limit of the gross beta-particle activity method and other non-radium, beta-emitting nuclides may likely result in interference. As a consequence, screening for radium-228, using the gross beta method, is not feasible for determining compliance with the existing limits and cannot be recommended. Recommended radioanalytical methods specific to radium-228 are discussed in section IV.B.2 below.

iii. Calibration Standards

For gross alpha-particle methods used to screen for radium-226 and uranium, calibration of the counting system using a suitable calibration standard is necessary. In 1997, the Agency approved the use of either natural uranium (Unat) or thorium-230 as calibration standards for the gross alpha-particle evaporation method. For the gross alpha-particle co-precipitation method, either natural uranium, thorium-230, or americium-241 can be used as a calibration standard. Thorium-230 was selected for several reasons. First, the energy of the alphas from thorium-230 was comparable to the alpha energy for radium-226 and uranium. Second, thorium is subject to the same interference by solids as radium and uranium. Third, thorium will reliably concentrate in either the evaporation gross alpha method or the listed co-precipitation methods.

For gross beta-particle activity methods, either strontium-90 or cesium-137 can be

used as gross beta standards. As with any method listed in Table IV-2, the recommended calibration standards are intended for determining compliance of drinking water containing naturally-occurring radionuclides and/or potential long-lived or abundant fission-generated beta/gamma emitting nuclides. Specific radionuclides may be substituted for the recommended alpha or beta standards should the radiochemist desire to screen for other radionuclides suspected to be present.

iv. Calibration Curves to Correct for Self Absorption

In 1991, the Agency proposed that laboratories must also generate calibration curves for their counters demonstrating the change in counting efficiency versus the total solids in the water sample when gross alpha-particle methods are used to screen for radium and uranium. The Agency still feels it is necessary to generate and routinely validate calibration curves when the gross alpha-particle method is used for screening purposes. These standard curves are used to determine the counting efficiency for samples containing variable concentrations of dissolved solids. If the calibration curves are not initially accurate and routinely validated, the gross alpha-particle technique cannot be used as a valid screening method for determining radium-226 and uranium compliance.

Reliable gross beta-particle measurements can be determined with waters having a higher concentration of dissolved solids. Although beta-counting efficiencies do not change dramatically with dissolved solids in water samples, generation of calibration curves to correct for self absorption is still necessary.

b. Specific Radium Analytical Methods

In March 1997, EPA approved the use of several radiochemical methods for the specific analysis of radium-226 and radium-228. For radium-226, the gross radium alpha (e.g., 903.0) and the radon emanation (e.g., 903.1) methods can be used. In contrast to evaporative methods (like EPA 900.0), method 903.0 (and similar methods) are more sensitive for the analysis of alpha-emitting radium isotopes in water samples containing high dissolved solids. For the co-precipitation method (e.g. 903.0), both barium and lead sulfate precipitate radium adequately, but how much thorium is co-precipitated remains unresolved. The co-precipitation of thorium is not a factor for routine water analysis in which its concentration is unmeasurable.

However, quantitative precipitation of thorium could be a problem in future gross alpha performance testing samples because the chemistry of the non-evaporative methods are radium specific and not optimized for the collection of thorium. Performance evaluation samples containing radium-226, natural uranium, and radium-228 have been used historically in the EPA's Radiochemistry Performance Evaluation Program to represent the likely combined presence of these natural nuclides in drinking water. If appreciable thorium-228 is present in the radium-228 standard and the thorium-228 does not quantitatively follow the radium, then a negative bias will result in any measurement based on the alpha or gamma measurement. For

drinking water samples and the historical EPA performance evaluation samples, radium-228 standards were prepared to ensure that thorium-228 did not contribute to an interference.

While analysis of radium-226 by radon emanation is more time consuming than gross radium alpha methods and requires special equipment and a trained staff, the radon emanation method is specific for radium-226. EPA believes the radon emanation method is necessary for radium-226 analysis when the gross radium alpha method is inconclusive or greater sensitivity is required to determine compliance of a sample near the MCL. Examples of inconclusive results from gross radium alpha analysis include situations in which the combined sum of the count rate and associated uncertainty makes it difficult to determine compliance with the MCL or if a lower level of detection is necessary to demonstrate compliance with the combined radium MCL. The gross radium alpha technique for radium-226 separates the radium from the dissolved solids in the sample and reduces interferences. In most cases, an appropriately conducted gross alpha screen and the gross radium alpha-emission method (903.0) should eliminate the need for a specific radium-226 analysis using radon emanation in a substantial number of cases.

Because the presence of radium-228 cannot be reliably ascertained by the gross beta-particle activity technique at 5 pCi/L or less, it is necessary to use a method that is specific for radium-228. Naturally-occurring beta-emitting radioisotopes (e.g., potassium-40) and beta-emitting progeny from both radium-226 and natural uranium can cause interference and make it difficult to detect the presence of radium-228. Also, the relatively high beta background of proportional counters potentially reduces the overall sensitivity. Therefore, analysis for radium-228 using radiochemical methods specific for this isotope are necessary to demonstrate compliance with the combined radium MCL.

Demonstrating compliance with the combined radium MCL may involve analyzing the contribution of radium-228 at very low levels. The amount of uncertainty associated with radium-228 analysis at these low levels (< 2 pCi/L) can be very high.

c. Uranium Methods

In March 1997 (62 FR 10168), the Agency approved several additional methods for the analysis of uranium. Specific analysis for uranium can be performed by radiochemical methods, alpha spectrometry, fluorometric (mass), or laser phosphorimetry (mass) (see Table IV-2). The radio-chemical method separates and concentrates uranium from potentially-interfering radionuclides and non-radioactive sample constituents. The resulting concentrate, depending on the method, can then be counted by gas flow proportional counting, alpha scintillation, or alpha spectrometry. Results from proportional counting or alpha scintillation counting accurately determine the alpha emission rate from total uranium in the sample; however, the uranium isotope ratio (uranium-234/uranium-238) cannot be determined and the uranium mass cannot be estimated unless an empirical conversion factor is applied to the measured count rate. The use of alpha spectrometry allows for the determination of individual isotopes of uranium and the accurate calculation of the mass of uranium-238 present in the sample. Additionally, the

concentration of uranium-234 can be accurately measured, if necessary to assess the radiotoxicity of this isotope.

Both the fluorometric and the laser phosphorimetry methods measure the mass of uranium-238 present in the sample; a conversion factor must be used to convert the mass measurement to an approximate radioactivity concentration in picoCuries. The computed radioactivity is only approximate because the ratio of uranium isotopes must be assumed. The use of mass-type methods is acceptable provided a conversion factor of 0.67 pCi/μg is used to convert the fluorometric or laser phosphorimetry uranium-238 mass result from micrograms to picoCuries. This conversion factor is conservative and is based on a 1:1 ratio of uranium-234 to uranium-238 in uranium-bearing minerals. The scientific literature indicates that the activity ratio varies in ground water from region to region.

EPA recognizes that the mass conversion factor is conservative in that the calculated uranium alpha emission rate based on the mass measurement may be biased low (i.e., underestimated). The use of this conversion factor may result in a larger net gross alpha (gross alpha less the calculated uranium gross alpha contribution), which may require additional testing to resolve. Conversely, the calculated mass of uranium based on gross alpha is biased high and is overestimated, which may require additional testing to resolve. Both situations are protective in that the bias requires additional testing to resolve when the uranium concentration in a sample is near the proposed MCL regardless of which method is used to measure the uranium.

d. Recommendations for Determining the Presence of Radium-224

To determine the presence of the short-lived radium-224 isotope (half life ~3.66 days), the several options can be considered:

i Radium-224 by Gamma Spectrometry and Alpha Spectrometry

(1) Gamma Spectrometry

Radium-224 can be specifically determined by gamma spectrometry using a suitably prepared sample. In this method a precipitate in which the radium isotopes are concentrated is gamma counted. The primary advantage of this technique is specificity for radium gamma rays, radium-224 included. Other advantages of this method include:

- a simple sample preparation were radium isotopes are concentrated from samples 1 liter or larger;
- specificity for the radium-224 isotope based on a unique gamma energy;
- optimal accuracy and precision if the sample is counted within 72 hours of collection (40 hours is recommended);
- and is cost competitive with the gross methods because a single count rather that three counts (see the gross methods discussed below) is necessary to measure the

radium-224 in a routine sample.

A gamma spectrometry method by Standard Methods is currently pending but for now the reader is referred to the method used by Parsa. (Parsa, 1998).

(2) Alpha Spectrometry

The alpha spectrometry method measures alphas emitted by radium-224 and its alpha emitting daughters. The alpha spectrometry method, used for the USGS occurrence survey (Section II), was a slight modification of Sills and Olson (1970) and Sills et. al. (1979). Using an appropriate tracer (e.g. Ba -133), barium and radium isotopes are separated from other radionuclides and interferences using cation ion exchange chromatography. A prepared sample, counted for approximately 100 minutes using alpha spectrometry, can be used to measure the radium-224 in the sample and is capable of good accuracy and precision. Other alpha spectrometry techniques, similar to the modified method used for the USGS occurrence survey, should be sufficient for the detection of radium-224. It is cost competitive with the gross methods (discussed below) because a single count rather than three (for gross methods) is sufficient to for measurement of radium-224.

ii. Gross Radium Alpha (Co-precipitation) Within 72 Hours

The presence of radium-224 can be determined indirectly using the radium-224 half-life decay and the gross radium alpha technique. Gross radium co-precipitation methods, like EPA 903.0, concentrate radium isotopes by co-precipitation, separating radium and radium-like isotopes from potential interferences. Relative to evaporative methods, the co-precipitation technique can be used for larger (> 1 L) sample sizes with a resulting increase in the method sensitivity. Initial analysis within 72 hours after sample collection (40 hours recommended for optimal data quality) using the co-precipitation methods yield results, reflecting both alpha-emitting radium isotopes (radium-224 and radium-226). For these to produce unambiguous results, radium-224 must be the dominant isotope present, i. e. the ratio of radium-224 to radium-226 must be three or greater. If this is the prevailing composition, the estimated contribution of radium-224 to the overall value can be ascertained by recounting the sample at 4 or 8 days intervals and calculating the change in the measured activity. The noted change will show a decrease with a 4 day half-life indicative of Ra-224. Formulas are available to calculate the initial radium-224 concentration present in the sample when collected. The advantages of this technique include:

- enhanced sensitivity (≥ 1 L samples);
- it does not require additional analyst training;
- it is specific for radium isotopes; and
- the resulting precipitate can be measured by a number of techniques, including proportional counting, alpha scintillation counting, or gamma counting.

iii. Evaporative Gross Alpha-Particle Analysis Within 72 Hours

The radium-224 isotope, when in equilibrium with its decay progeny, emits four alpha particles. Three of these alpha particles equilibrate almost immediately (within 5 minutes) after sample preparation and add to or amplify the sample count rate. This count rate amplification can be exploited for the measurement of radium-224 in a sample at low concentration (< 15 pCi/L). The presence of the radium-224 radioisotope in drinking water may be ascertained by performing an initial evaporative gross alpha-particle analysis within 72 hours (40 hours recommended) after sample collection. In the absence of any other alpha-emitting nuclide (e.g., uranium or radium-226) and if the gross alpha-particle value is above the MCL, the sample may be re-counted at 4- and 8-day intervals to determine if the observed decrease in activity follows the 3.66 day half-life of radium-224. A decrease in the gross alpha value with a 4-day decay rate indicates the likely presence of radium-224. Formulas are available to calculate the concentration of radium-224 in the initial sample. The advantages of this option include:

- the method is similar to the general method for evaporative gross alpha;
- it requires no special training of the analyst; and
- it can be a definitive test if other alpha-emitting nuclides are known to be absent.

Performing an analysis within the 72-hour time frame may create difficulties in shipping and handling and may increase the price of the analysis.

C. Updates to the 1991 Cost Estimates for Radiochemical Analysis

In the July 18, 1991 proposed rule (56 FR 33050), EPA cited the approximate costs of the different radiochemical analyses outlined in Table IV-2. At that time, the Agency stated that these were preliminary cost estimates and several factors may affect the actual cost of performing radiochemical analysis. These factors included:

- Laboratory-to-laboratory variation in price;
- The specific radiochemical technique being performed;
- The volume of samples being analyzed by a laboratory (i.e., the number of samples analyzed per unit time);
- The quality-control efforts and the quality-assurance programs of individual laboratories; and
- The time required to count the sample.

In the 1991 proposed rule, the Agency also stated that few commercial laboratories existed that could perform certain analyses and hence it was difficult to determine the cost for those analyses. Since the publication of the 1991 proposed rule, the costs of radiochemical analyses have been re-evaluated. The 1999 cost estimates are shown in Table IV-4, along with the estimated costs from 1991.

Table IV-4. The 1991 and 1999 Estimated Costs of Analyses for Radionuclides

Radionuclides	Approximate Cost $ (1991)[1]	Approximate Cost $ (1999)[2]
Gross Alpha and beta	35	45
Gross alpha - coprecip.	35	45
Radium-226	85	90
Radium-228	100	110
Uranium (total)	45	48 (LP)
Uranium (isotopic)	125	128 (AS)
Radioactive Cesium (-134)	100	125
Radioactive Strontium	105	144
Total Strontium (-89 and -90)	.....	153
Radioactive Iodine-131	100	131
Tritium	50	60
Gamma/Photon Emitters	110	142

Source: [1] 56 FR 33050; July 18, 1991
[2] Revised Cost Estimates of Radiochemical Analysis, USEPA, 1999
Abbreviations: LP = laser phosphorimetry; AS = alpha spectrometry
Note: Estimated costs are on a per-sample basis; analysis of multiple samples may have a lower cost.

Due to resource limitations, on July 18, 1996 (61 FR 37464) EPA proposed options for the externalization of the Performance Evaluation (PE) studies program (now referred to as the Proficiency Testing or PT program). After evaluating public comment, in the June 12, 1997 final notice (62 FR 32112) EPA:

> *"decided on a program where EPA would issue standards for the operation of the program, the National Institute of Standards and Technology (NIST) would develop standards for private sector PE* (PT) *suppliers and would evaluate and accredit PE suppliers, and the private sector would develop and manufacture PE* (PT) *materials and conduct PE* (PT) *studies. In addition, as part of the program, the PE* (PT) *providers would report the results of the studies to the study participants and to those organizations that have responsibility for administering programs supported by the studies."*

EPA has addressed this topic in public stakeholders meetings and in some recent publications, including the aforementioned *Federal Register* notices.

D. Updates to the Measures of Sensitivity for Radiochemical Techniques

In 1976, the National Primary Drinking Water Regulations defined the *detection limit* (DL) as "the concentration which can be counted with a precision of plus or minus 100 percent at the 95 percent confidence level (1.96 σ, where σ is the standard deviation of the net counting rate of the sample)." Table IV-5 cites the detection limits or the required sensitivity for the specific radioanalyses that were listed in the 1976 rule and are also cited in 40 CFR 141.25.

In 1991, EPA proposed using the *method detection limit* (MDL) and the *practical quantitation level* (PQL) as measures of performance for specific radioanalytical methods (56 FR 33096). EPA also proposed acceptance limits based on the PQLs that were derived from performance evaluation studies. The use of acceptance limits and the relationship to the actual method performance was not clear and confusing to some commenters. With the exception of uranium, the Agency may not go forward with the proposed acceptance limits, PQL, or MDL. Because uranium has never been regulated, it did not have a detection limit in the CFR and one has never been proposed. In 1991, EPA did propose a PQL of 5 pCi/L with an acceptance limit of +/- 30%. Although it is believed that a detection limit for uranium would be very similar to the PQL, because one has never been proposed, the Agency may have to adopt the PQL for uranium until a detection limit is proposed. For the other radionuclides, which are regulated, the Agency will most likely maintain the current 1976 rule (which uses the detection limit as the measure of sensitivity). The existing definition of the detection limit takes into account the influence of the various factors (efficiency, volume, recovery yield, background, counting time) that typically vary from sample to sample. Furthermore, the detection limit is computed for each individual sample and does not represent an idealized set of measurement parameters. Therefore, the detection limit reflects the expected random uncertainty for a given sample analysis.

Table IV-5. Required Regulatory Detection Limits for the Various Radiochemical Contaminants (40 CFR 141.26)

Contaminant	Detection Limit (pCi/L)
Gross Alpha	3
Gross Beta	4
Radium-226	1
Radium-228	1
Cesium-134	10
Strontium-89	10
Strontium-90	2
Iodine-131	1
Tritium	1,000
Other Radionuclides and Photon/Gamma Emitters	1/10th of the rule * NIPDWR 1976 table IV-2A and 2B

E. References

Sill, C.W., and Olson, D.G., 1970, Sources and prevention of recoil contamination of solid-state alpha detectors: Analytical Chemistry, v. 42, no. 13, pp. 1596-1607.

Sill, C.W., Hindman, F.D., and Anderson, J.I., 1979, Simultaneous determination of alpha-emitting radionuclides of Radium through Californium in large environmental and biological samples: Analytical Chemistry, v. 51, no. 8, pp. 1307-1314.

USEPA. "National Primary Drinking Water Regulations; Radionuclides; Proposed Rule." Federal Register. Vol. 56, No. 138, p. 33050. July 18, 1991.

USEPA. "National Primary Drinking Water Regulations; Analytical Methods for Radionuclides; Final Rule and Proposed Rule." Vol. 62, No. 43, p. 10168. March 5, 1997.

USEPA. "Manual for the Certification of Laboratories Analyzing Drinking Water." EPA 815-B-97-001. 1997.

USEPA. "Revised Cost Estimates of Radiochemical Analysis". Prepared by EPA and Science Applications International Corporation (SAIC). April 1999. (EPA 1999)

V - TREATMENT TECHNOLOGIES AND COSTS

A. Treatment Technologies Feasibility Update

1. Updates on Performance of Technologies for Removal of Regulated Radionuclides and Uranium

As discussed in the 1992 and 1999 radionuclides T&C documents (EPA 1992 and EPA 1999a) , the applicable treatment technologies (e.g., ion exchange softening, lime softening, reverse osmosis, and green sand filtration) have maximum removal efficiencies of around 80% up to greater than 95%, depending upon site specific conditions, e.g., water quality. Since most systems in violation of an MCL would require lower removals to achieve compliance (< 50 %), treatment equipment is often designed to achieve high removals, but to treat only part of the flow. This process is called blending and typically involves the post-treatment mixing of the finished water with by-passed untreated water. While blending results in lower removal efficiencies, it also results in lower capital costs and lower operations and maintenance costs than full-flow treatment. Technology descriptions, design criteria, removal efficiencies, and blending are described in detail in the T&C documents already cited.

Although the 1999 T&C document concludes that the peer-reviewed literature describes no new technologies since the 1992 T&C document was completed, there have been some developments that are significant. In particular, both package plant technologies, including those equipped with remote control/communication capabilities, and point-of-entry (POE)/point-of-use (POU) versions of existing technologies have become more widely applicable for use for compliance. This is true both because of improvements in these technologies themselves (NRC 1997) and since the 1996 SDWA explicitly allows package plants and POE/POU devices to be used as compliance technologies for small systems (Section 1412.b.4.E). Issues relevant to package plants and POE/POU devices follow.

a. Package Plant Treatment Units

Package plant designs for treatment technologies have improved and become more widely used by small systems. The principle advantages of package plants are lower engineering costs, simpler operation, and hence, lower operator demand and labor costs. Potentially, package plant versions allow small systems to use technologies that would otherwise be too complex in operation or cost prohibitive (NRC 1997, Goodrich et al. 1992).

Cost savings can be reduced if extensive pilot testing is required, which can significantly increase package treatment start-up costs. Since it is, of course, essential to confirm that installed treatment is working properly, the need for site specific pilot testing can not be entirely eliminated, but it can be reduced (EPA 1993). To this end, verification of package plant performance by an independent party would reduce costs by reducing in degree the need for pilot testing. It is important to keep in mind that pilot testing needs are specific both to the technology

in question and to source water quality. For a discussion of pilot testing needs for individual technologies, see, e.g., NRC's Safe Water from Every Tap (NRC 1997).

This need for package plant performance verification is currently being addressed by EPA in cooperation with National Sanitation Foundation (NSF) International through an EPA Environmental Technology Verification project. Together they are cooperatively organizing and developing a pilot project (Package Drinking Water Treatment Systems Pilot Project) to allow for verification testing of package plant units. The aims of the project include developing verification testing protocols and test plans, independently testing and validating package plant equipment, conveying and supporting government/industry partnerships to obtain credible cost and treatment performance data, and preparing product verification reports for public use. It is expected that the program will become financially self-sustaining through user fees and leveraged resources. More information regarding this program may be found on the NSF Internet website at "http://www.nsf.org/verification/html/overview.html". Other sources contain useful information regarding the advantages and limitations of package plant technologies, e.g., NRC 1997, EPA 1998b, EPA/AWWA 1992a and 1992b, and Goodrich et al. 1992.

b. Point-of-Entry/Point-of-Use Treatment Units

For very small water systems (those serving fewer than ca. 500 persons), point-of-entry/point-of-use (POE/POU) may provide a low-cost alternative to centralized treatment. However, POE/POU devices have disadvantages to their use and so are generally used only when a centralized treatment facility is not technically or financially feasible for a water system. Regarding POU devices, if one is placed under the kitchen sink to treat food for drinking and cooking, only that water is potable; water from a bathroom tap would be untreated, and thus exposure to contaminants during teeth brushing, etc. is possible (NRC 1997). POE treatment units also have limitations. For example, some POE filters require backwashing or regeneration to maintain a reasonable filter bed/resin lifetime (number of bed volumes before the filter bed or resin needs replacement). A proper backwashing/regeneration frequency assures good removal performance and minimizes radioactive residue build-up on the treatment medium. If the backwash requires special disposal methods, costs may increase prohibitively. The legal requirements for the use of POE/U devices as compliance technologies are discussed in Section 1.4 of EPA's "Small System Compliance Technology List for the Non-Microbial Contaminants Regulated Before 1996" (EPA 1998b).

Effective operation, monitoring, and maintenance are vital to the successful use of POE/U treatment units for compliance purposes (Goodrich et al. 1992, NRC 1997). POE/U units are required by the SDWA to be "owned, controlled, and maintained by the public water system or by a person under contract with the public water system to ensure proper operation and maintenance and compliance with the MCL or treatment technique and equipped with mechanical warnings to ensure that customers are automatically notified of operational problems". Proper installation is of course essential. After installation, the PWS needs a well-defined operation and maintenance program to assure a continuous supply of safe drinking water.

The PWS also needs a routine monitoring program to confirm that the operation and maintenance program is adequate and that the treated water meets drinking water standards.

Technologies that are amenable to POE/U scale treatment include reverse osmosis, ion exchange, and activated alumina. POE/U reverse osmosis and ion exchange are applicable to the removal of radium, gross alpha emitters, total beta emitters, and uranium. POE/U activated alumina may be considered for the removal of uranium and for other radionuclides on a case-by-case basis. The advantages and limitations of POE/U technologies are discussed in more detail elsewhere (e.g., NRC 1997, Goodrich et al. 1992, EPA 1998b, and references cited within).

B. Waste Treatment, Handling and Disposal Guidance

In the proposed radionuclides rule of July 1991, EPA referenced a 1990 EPA draft report entitled "Suggested Guidelines for Disposal of Drinking Water Treatment Wastes Containing Naturally-Occurring Radionuclides" (EPA 1990). That 1990 report offered guidance to system managers, engineers, and State agencies responsible for the safe handling and disposal of treatment wastes that, in many cases, were not specifically addressed by any statute. EPA was aware that concern existed about the safe disposition of treatment wastes containing concentrated, naturally-occurring radioactive isotopes. Furthermore, a wide range of existing disposal practices and regulatory strictures may make waste disposal a particularly difficult issue, technically and politically. Indeed, by the mid-1980s, some State regulatory agencies (e.g., Wisconsin and Illinois) had crafted their own guidance on this issue. It soon became evident that revised national primary drinking water standards for radionuclides were going to create a need for accompanying waste disposal and handling guidance.

The guidance provided information on the following: (1) background on water treatment processes and characteristics of wastes generated; (2) rationale for radiation protection, including citation of programs and regulations affecting other sources of such waste; (3) guidelines for several methods of disposal of solid and liquid type wastes containing the subject radionuclides; and, (4) the specification of practical guidance to protect workers and others who may handle or be exposed to water-treatment wastes containing radiation above background levels.

It was evident from comments received on the proposal that some of EPA's stakeholders were pleased with the Agency's issuance of guidance for disposal of this type of waste, yet they were also concerned that decisions on selecting best treatment for particular sites may be driven by waste-handling and associated expenses. Specifically, the American Water Works Association (AWWA), in its comment of October 15, 1991, referred to the potential for radium and/or uranium bearing wastes to be classified as "low level radioactive wastes," for which the United States has few available repositories. EPA understands and accepts this as an important concern. However, the Agency feels application of EPA's guidance will decrease the likelihood that low level radioactive wastes (i.e., those covered under existing laws on the handling, transport and safe disposal of such waste) will be generated. The guidance attempts to guide decision makers *away from* treatment decisions that could in any manner pose a radiation threat

to the public. As of this date, EPA has not been appraised of a low level waste problem at public water supply systems following the Agency's guidance.

Other important concerns mentioned by the AWWA included: (1) the need for EPA to address the possibility of water treatment generating "mixed wastes" (i.e., hazardous and radioactive) that would prove difficult to dispose; (2) the need for EPA to address and resolve waste disposal issues before the final regulation is set (for radium and other radionuclides); (3) the necessity of allowing for State discretion in regard to waste disposal issues; and (4) the possibility that land application of wastes, such as lime softening wastes, containing radium might be considered a viable disposal option. EPA agrees with the first three of these concerns and in subsequent drafts of the subject guidance (discussed below) did address them. Regarding land application of wastes, EPA studied the risks involved and available protective guidance, and in subsequent guidance revisions specifically *did not recommend* open land application of radium-bearing treatment wastes. The Agency took this position due to a concern for long-term control and monitoring at such sites, a general lack of data on radiation exposure through plant, animal and human uptake from land application of such wastes, and the potential long-term radon inhalation risks (see the June 1994 draft suggested guidelines, as discussed below, for greater detail on the land application issue, EPA 1994). With regard to the "mixed waste" issue, EPA has addressed the possibility of such wastes being generated and has referred to a joint EPA/Nuclear Regulatory Commission (NRC) guidance on identifying such wastes (see the June 1994 draft guidelines). Again, EPA is not aware of an actual event at a treatment facility that has triggered the use of that guidance or caused any related problems.

The 1990 EPA draft guidance has received additional substantive review since the radionuclides proposal in 1991. In 1992, EPA's Science Advisory Board (SAB) reviewed the 1990 guidance and offered several recommendations (see referenced letter from Loehr and Nygaard). EPA has acted upon the most important SAB recommendations. In response to the SAB, the Agency has integrated quantitative information on radioactivity in water-treatment wastes within the document to facilitate decision-making processes. The Agency has also selected and provided a rationale for the occupational protection guidelines for protection of water-treatment plant workers contained in the document. In addition, EPA has addressed other SAB comments, including the need to (1) discuss non-radioactive hazardous substances and the possibility of generation of "mixed wastes"; (2) re-examine the reference to NRC regulations as guidance for underground injection and sanitary sewer disposal of natural radioactivity in wastes; (3) produce guidance on waste-sampling procedures for determination of radioactivity levels; and (4) more forcefully urge State agencies with authority over water supplies to consult with State radiation control programs as the most informed source of radiation protection guidance.

EPA's response to the recommendations of the SAB (see referenced letter and attached detailed responses from William K. Reilly to SAB's Loehr and Nygaard) reflected the approach the Agency intended to take in strengthening the technical aspects of the draft document to assist in guiding waste management. At the same time, in its response, EPA made clear that the subject guidance was not intended as a vehicle for undertaking detailed risk analyses and for estimating a

national risk associated with drinking water treatment wastes containing radioactivity. EPA does not intend the guidance to be used as a basis for revising existing radiation policies at the Federal or State levels.

In response to the SAB's recommendation to incorporate more quantitative information within the subject guidance on wastes generated, EPA developed a spreadsheet program for computing waste volumes and radioactivity levels for several pertinent water-treatment processes, along with a companion user's guide. These should aid in the planning and pre-engineering stages of waste management in which managers and reviewers will need information on the types of waste that may be generated at new facilities. The spreadsheet program was developed by EPA in August 1993 and is entitled "Spreadsheet Program to Ascertain Residual Radionuclide Concentrations" (SPAARC). The program and accompanying user's guide can be obtained by contacting: ERIC Clearinghouse, 1929 Kenny Road, Columbus OH 43210, (614) 292-6717, or the National Technical Information Service (NTIS), 5285 Port Royal Rd., Springfield, VA 22161, (703) 487-4650. It should be noted that EPA is currently testing this spreadsheet program and will revise it as necessary prior to a final radionuclides in drinking water rule.

In response to SAB comments on the occupational guidelines presented, the section of the guidance dealing with the protection of workers from radiation hazards has been substantially revised by EPA's Office of Radiation and Indoor Air. The Agency found that the radiation doses likely to be received by personnel in water-treatment facilities would be much lower than the upper bound of doses allowed for workers in radiation facilities. In fact, it appears reasonable to keep treatment plant workers' doses to well within the levels recommended for the general public, i.e., below 100 mrem/yr or a fraction thereof. However, EPA does not believe it is necessary to set a specific limit below that level due to the practical considerations of measuring lower levels of radiation exposures and the paucity of data available. Consequently, the guidance recommends 100 mrem/yr as the maximum administrative control level in water treatment facilities. The document also contains more guidance on seeking State assistance and/or approval in the design and implementation of radiation safeguards.

Finally, two other significant reviews of the draft guidance have occurred since the 1991 proposal. The Guide was reviewed internally in 1992 and updated to reflect recent Federal regulations and/or guidance addressing various radioactive wastes (see two memoranda authored by Dr. Ricardo Gonzalez Mendez). In addition, EPA found that the use of anion exchange as a treatment for uranium removal required some caution in limiting the time of service of the exchange unit between regeneration cycles and over the full service life. By so doing, uranium in the resin does not become a difficult to manage "source material" as defined by the Atomic Energy Act of 1954 as amended, per 10 CFR 20.

The US Department of Energy (DOE) also provided comments on the proposed rule. (See referenced letter, with attachment, from R. Pelletier of DOE, to J. Elder of EPA.) DOE expressed its concerns about water treatment wastes and potential risks to workers. DOE offered

a Brookhaven National Laboratory (BNL) report on the national exposure and risk estimates associated with the implementation of the proposed radionuclides in drinking water regulations. The BNL report estimated that "a year of implementing the rule for uranium could result in risks to workers and the public of from 0.16 to 0.49 lifetime health effects." These effects presumably include cancers and other non-cancer effects. EPA examined the BNL report and found that the study used extremely conservative engineering and other assumptions which were to some extent not realistic, and consequently skewed the results. EPA found that the BNL study overestimated the numbers of water treatment workers affected; applied more risky treatment options that would generate the most concentrated and difficult wastes to handle and dispose; and placed water treatment workers in very close proximity to treatment vessels, in effect maximizing radiation dose to their bodies. BNL also calculated the numbers of presumed accidents on the job, a factor not typically considered as a primary effect of a drinking water regulation.

The subject guidance, "Suggested Guidelines for Disposal of Drinking Water Treatment Wastes Containing Radioactivity," was issued by EPA to State agencies in 1994, with revisions as discussed above. The updated guidelines are summarized pictorially in two figures described below. For details, please see the referenced draft 1994 guidelines (EPA 1994).

Figure V-1 is a flow chart summarizing the guidelines for disposal of the radioactive liquid wastes of water treatment processes. Three options exist for waste disposal as illustrated in Figure V-1. Selecting a treatment option depends on the level of radioactivity in the residual to be disposed.

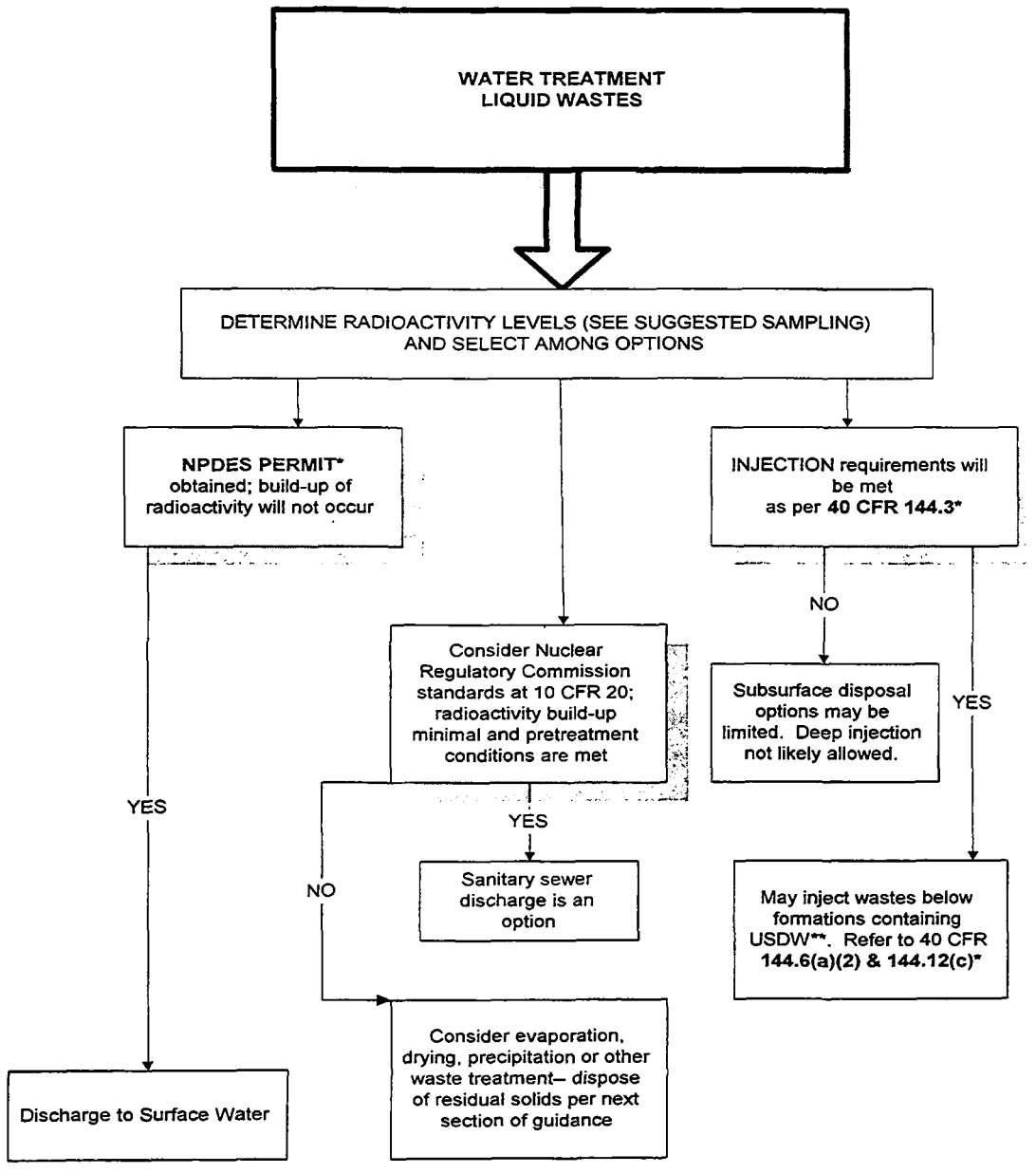
NPDES Permit. The Clean Water Act requires anyone discharging pollutants into U.S. waters to obtain a National Pollutant Discharge Elimination Permit (NPDES). This permit contains, at a minimum, technology-based effluent limitations. Sufficient flow and channel geometry must be available to prevent a buildup of radionuclides in the surface water or sediments to within appropriate limits established by the regulator.

Sanitary Sewers. Pretreatment regulations prohibit any discharge into sanitary sewers that would cause a municipal wastewater treatment plant to violate a NPDES permit. The NRC also limits the discharge of wastes containing radioactive materials into sanitary sewers.

Injection Wells. A drinking water treatment plant owner interested in disposing of wastes containing radionuclides into an injection well in a Primacy State should consult the appropriate State agency prior to doing so. State regulations may be more stringent than the Federal requirements and may ban such practice. The plant owner in other States should consult with the appropriate Underground Injection Control (UIC) Regional Branch office of the EPA before deciding to dispose of wastes containing radionuclides into an injection well.

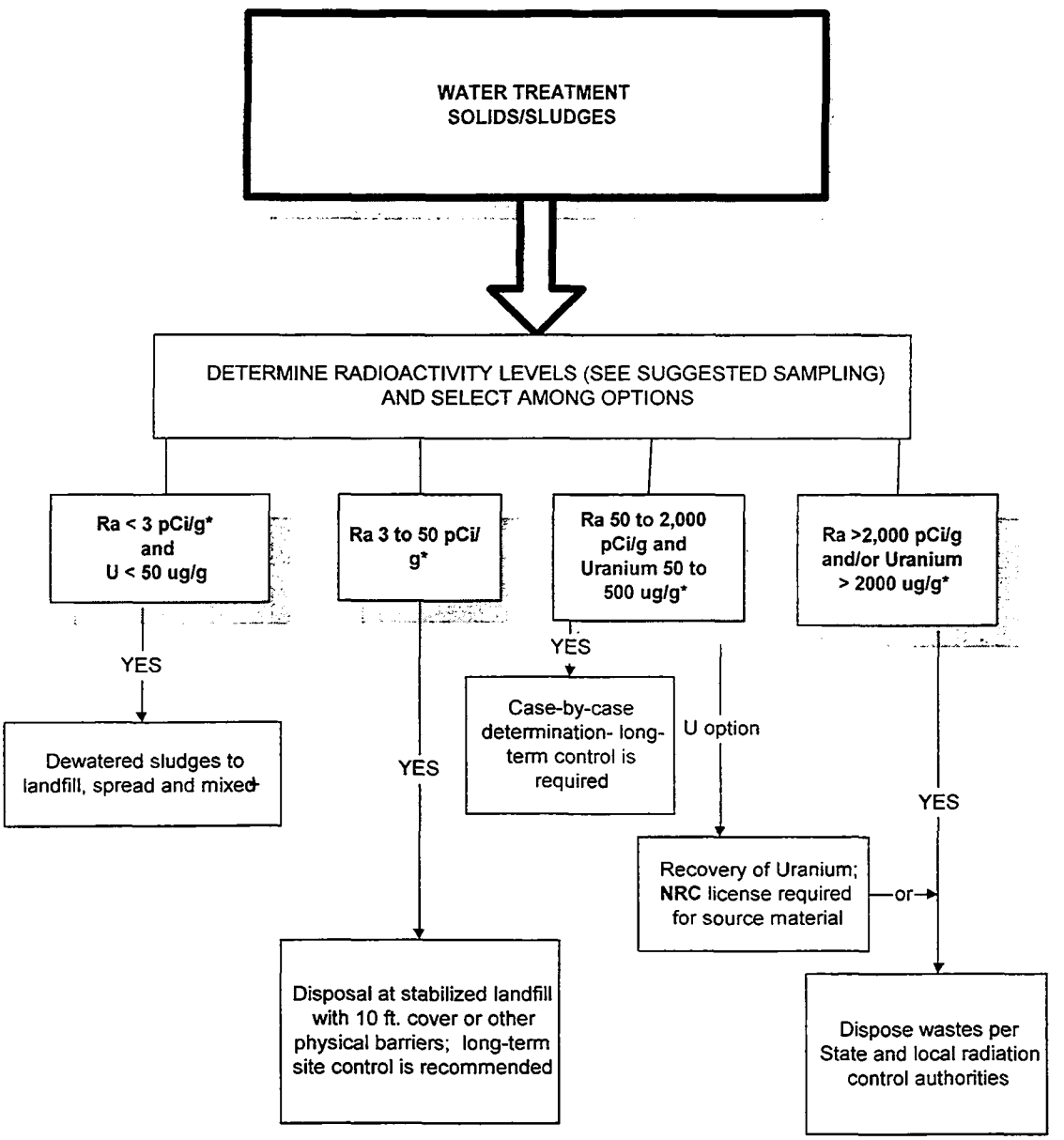
Figure V-2 is a flowchart summarizing the guidelines for disposal of radioactive solids/sludges. Wastes containing less than 3 pCi/g (dry) of radium and less than 50 μg/g (dry) of uranium may be disposed of in a municipal landfill if the wastes are first dewatered and then spread and mixed with other materials when emplaced. Solids/sludges containing 3 to 50 pCi/g (dry) of radium should be disposed of with a physical barrier that would protect against radon release and isolate the wastes; disposal sites should be provided with institutional controls designed to avoid inappropriate uses of the site. The disposal method for solids/sludges containing 50 to 2,000 pCi/g (dry) of radium should be determined on a case-by-case basis. Recovery of the uranium resource may be considered for solids containing 50 to 500 μg/g (dry) of uranium. Wastes containing more than 2,000 pCi/g (dry) of radium or more than 500 μg/g (dry) of uranium should be disposed of only as permitted by State regulations.

Figure V-1: Summary of Suggested Disposal Alternatives For Liquid Water Treatment Wastes



* **Bold type** indicates actual regulatory requirements; other options are EPA suggested disposal methods.
**USDW is underground source of drinking water

Figure V-2. Summary of Suggested Disposal Alternatives For Solids/Sludge Wastes From Water Treatment*



+ Provisions under 40 CFR 257, 258 and 260 thru 266 may apply.
*** Bold type indicates Federal regulatory requirements; other options are EPA suggested disposal methods.**

Table V-3 is a summary of estimated waste volumes, radioactivity levels, and types of waste generated by several drinking water treatment processes. These estimates are based on standard engineering assumptions, and average water production of 1.0 million gallons per day (MGD). Actual radionuclide levels are highly dependent on site-specific data.

Table V-3: Characteristics of Selected Water Treatment Wastes (Computed Levels)*

Treatment Method	Raw/ Finished Conc. (pCi/L)	Waste Characteristics: Vol. and Radioactivity	Types of waste generated
Coagulation/ Filtration	50/10 (Uranium)	U Sludge volume: 10 kgpd U sludge: 1,770 pCi/g dry	Iron or alum sludges from the contact and settling basins and from the filter backwash. Supernatant from this sludge is a liquid waste. Additional liquid waste is generated when the sludge is concentrated prior to disposal.
Lime Softening	25/2.5 (Radium)	Ra sludge volume: 22 kgpd Ra sludge: 1,769 pCi/L wet or 45 pCi/g dry	Settling tank sludge, which is precipitated during the softening process. Filter backwash (both solids and supernatant) and the sludge supernatant are other wastes.
Ion Exchange	25/5 (Radium)	Ra waste flow: 97 kgpd Ra wastes: 207 pCi/L	Liquid waste containing brine, rinse and backwash water, and contaminants stripped off the resin. In addition, the resin itself is a solid waste containing the contaminant exchanged. Assumes 150 bed volumes to bed exhaustion.
Reverse Osmosis	50/15 (Uranium)	U waste flow: 130 kgpd U wastes: 320 pCi/L	Reject streams which are continuously generated during the treatment process.

* Levels of radioactive components were computed using EPA's "Spreadsheet Program to Ascertain Residuals Radionuclide Concentrations" (SPAARC) of August 1993.

C. Unit Treatment Cost Updates

Unit treatment and other compliance costs are discussed in Section VII, "Supporting Material for Economics and Impacts Analysis".

D. References

American Water Works Association. Written comments of John H. Sullivan in reference to EPA proposed rulemaking on radionuclides in drinking water, dated October 15, 1991.

Gonzalez Mendez, R. Memorandum to A. Gera and M. Parrotta regarding, "New Levels for the Disposal of Radioactive Waste into Sanitary Sewer From 10 CFR 20." June 17, 1992.

Gonzalez Mendez, R. Memorandum to M. Parrotta and B. Smith regarding "Recommendations for Changes to the Guidelines for Disposal of Radioactive Wastes From Water Treatment Plants." July 23, 1992.

Goodrich, J.A., Adams, J.Q., Lykins, B.W., Jr., and Clark, R.M. "Safe Drinking Water from Small Systems: Treatment Options." JAWWA. 84(4). p. 49. May 1992.

Loehr, R.C. and Nygaard, O.F. Letter from Science Advisory Board to William K. Reilly, EPA Administrator, September 30, 1992.

National Research Council (NRC). Safe Water From Every Tap: Improving Water Service to Small Communities. National Academy Press. Washington, DC. 1997.

Reilly, William, US EPA Administrator. Letters (two) to Dr. R.C. Loehr and Dr. O.F. Nygaard, Science Advisory Board, November 18, 1992.

US Department of Energy, Office of Environmental Guidance. Letter of May 14, 1993 from Raymond Pelletier to James Elder, USEPA Office of Ground Water and Drinking Water, with attached report produced under contract by the Brookhaven National Laboratory, "National Radiation Exposures and Risks Caused by Implementing EPA's Proposed Revised National Primary Drinking Water Regulations (April 1993)."

USEPA. Office of Drinking Water. "Suggested Guidelines for Disposal of Drinking Water Treatment Wastes Containing Naturally-Occurring Radionuclides (July 1990 draft)."

USEPA. "National Primary Drinking Water Regulations; Radionuclides; Proposed Rule". Federal Register. Vol. 56, No. 138, p. 33050. July 18, 1991.

USEPA and the American Water Works Association (AWWA). "Package Water Treatment Plant Operation and Field Data: A Field Study". Draft. 1992. (EPA/AWWA 1992a).

USEPA and the American Water Works Association (AWWA). "Package Water Treatment Selection Guidance for Small and Rural Drinking Water Systems". Draft. 1992. (EPA/AWWA 1992b).

USEPA. "Technologies and Costs for the Removal of Radionuclides from Potable Water Supplies." Prepared by Malcolm Pirnie, Inc. July 1992.

USEPA. "Technical and Economic Capacity of States and Public Water Systems to Implement Drinking Water Regulations: Report to Congress." Office of Water. EPA 810-R-93-001. September 1993.

USEPA. Office of Ground Water and Drinking Water. "Suggested Guidelines for Disposal of Drinking Water Treatment Wastes Containing Radioactivity (June 1994 draft)."

USEPA. "Announcement of Small Systems Compliance Technology Lists for Existing National Primary Drinking Water Regulations and Findings Concerning Variance Technologies." Federal Register. Vol. 63, No. 151, p. 42032. August 6, 1998. (EPA 1998a).

USEPA. "Small System Compliance Technology List for the Non-Microbial Contaminants Regulated Before 1996." EPA-815-R-98-002. September 1998. (EPA 1998b).

USEPA. "Actual Cost for Compliance with the Safe Drinking Water Act Standard for Radium-226 and Radium-228." Final Report. Prepared by International Consultants, Inc. July 1998. (EPA 1998c).

USEPA. "Technologies and Costs for the Removal of Radionuclides from Potable Water Supplies." Draft. Prepared by International Consultants, Inc. April 1999. (EPA 1999a).

USEPA. "Small System Compliance Technology List for the Radionuclides Rule". Prepared by International Consultants, Inc. Draft. April 1999. (EPA 1999b).

VI. NON-TRANSIENT, NON-COMMUNITY CANCER RISK ANALYSIS

In order to calculate the number of cancer cases avoided, EPA developed a Monte-Carlo based risk model. This document provides a more detailed description of the risk analysis, including the assumptions and calculations used in the analysis. The following sections explain how risk reductions for populations exposed to radionuclides in excess of the lowest MCL levels being considered for each radionuclide were calculated. First, the data used in the analysis will be presented. Second, the calculations used in the analysis will be explained.

A. Data Inputs

Modeling an individual's risk associated with exposure to NTNC water requires information on a number of factors:

- Daily quantity of drinking water consumed
- Frequency of system utilization
- Lifetime risk per pCi consumed for each age group
- Occurrence level in water
- Estimates of distribution of consumer overlap across service area categories

In addition, characterization of the distribution of risk across the population of NTNC users requires knowledge of the number of people served by the various systems. The following sections describe the data used for each of these factors.

B. Water Consumption

EPA recently updated its estimates of personal (per capita) daily average water consumption (*Estimated per Capita Water Consumption in the United States*, EPA 2000). The estimates use data from the combined 1994, 1995, and 1996 Continuing Survey of Food Intakes by Individuals (CSFII), conducted by the U.S. Department of Agriculture (USDA). The CSFII is a complex, multistage area probability sample of the entire U.S. and is conducted to survey the food and beverage intake of the U.S. Estimates of water consumed include "Community Water" and "Total Water." Community Water only includes water known to have come from a community water supply. "Total" includes community water plus bottled water and other water of indeterminant or other origins.

Water consumption estimates broken down by age and sex were used in the computation of cancer cases avoided for the radionuclides. The age groupings used in the study are those provided in the original study and are shown in Tables VI.1 and VI.2. Because respondents were sometimes uncertain as to the origin of water they would drink, the Agency has used the two distributions to provide upper and lower bound estimates of drinking water consumption.

Table VI.1
Lower Bound
Drinking Water Consumption Groups
Consumption in milliliters per day

Age Group	Male Mean	Male 90th %ile	Female Mean	Female 90th %ile
Less than 1	298	868	384	904
1 to 10	406	894	394	915
11 to 19	772	1658	590	1307
20 & older	1162	2337	1039	2126
All ages	975	2115	880	1941

Table VI.2
Upper Bound
Drinking Water Consumption Groups
Consumption in milliliters per day

Age Group	Male Mean	Male 90th %ile	Female Mean	Female 90th %ile
Less than 1	549	1121	577	950
1 to 10	536	1024	528	993
11 to 19	1001	1898	830	1652
20 & older	1549	2740	1389	2416
All ages	1300	2483	1185	2221

C. Frequency of System Utilization & Customers Served

There are approximately 20,000 Non-Transient Non-Community (NTNC) water systems regulated under the Safe Drinking Water Act. By definition, these systems do not serve over 25 people on a continuous basis. However, they must serve at least 25 people for over six months out of the year, or they would be classified as Transient Non-Community (TNC) water systems. It is generally an important distinction since the Agency has not applied regulations for contaminants with chronic health effects to TNC water systems, while it often has applied them to NTNC systems. In the case of the radionuclides subject to this rule, the existing regulations do not apply to NTNC systems.

In the past, the Agency has directly used SDWIS population estimates for assessing the risks posed to users of NTNC water systems. It was assumed that the same person received the exposure on a year round basis. Under that approach, NTNC users were characterized as exposed for 270 days out of the year and as receiving fifty percent of their daily consumption from these systems. TNC users were assumed to use the system for only ten days per year.

The Agency now has a taken a closer look at NCWS and believes these characterizations can be improved. As described in the recent Agency report (*Geometries and Characteristics of Public Water Systems*, draft report, EPA, 1999), service areas reported for NCWS in SDWIS were tallied and supplemented by a random sample of non-specified service area systems[4]. The resultant categories of NTNC water systems are shown in Table VI.3.

One important characteristic to note about NTNC systems is that they provide water in due course as part of operating another line of business. Many systems are classified as NTNC, rather than TNC, water systems solely because they employ sufficient workers to trigger the "25 persons served for over six months out of the year" requirement. The overwhelming majority of users of these systems actually have exposure which is more similar to exposure in TNC water systems. For instance, it is fairly implausible that highway rest areas along interstate highways serve the same population on a consistent basis (with the exception of long distance truckers). Nevertheless, there are highway rest areas in both NTNC and TNC system inventories. By contacting State Agencies and retrieving information from the Internet, it has been determined that population figures reported in SDWIS which have been used for past risk assessments generally appear to reflect the number of workers in the establishment coupled with peak day customer utilization[5].

Under these conditions, the traditional use of the SDWIS figures for population could greatly overestimates the actual individual chronic exposure risk for most of the exposed population and also underestimates the number of people exposed to NTNC water[6]. Adequately characterizing individual and population risks necessitates some adjustments to the SDWIS population figures. For chronic contaminants, such as the radionuclides, health data reflect the

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[4]Service areas were assigned to the non-specified systems through the use of a Delphi Panel approach.

[5]To a more limited extent, there are some systems among the NCWS where reported population actually reflects total annual usage of the supply.

[6]For example, airports constitute only about a hundred of the NTNC water systems. Washington's Reagan National and Dulles, Dallas/Fort Worth, Seattle/Tacoma, and Pittsburgh airports are the five largest of the airports. SDWIS reports that these five airports serve about 300,000 people. In actuality, Bureau of Transportation Statistics suggest that they serve about eleven million passengers per year. Examination of this information and other BTS statistics suggests that these airports serve closer to seven million unique individuals over the course of a year and that exposure occurs on an average of ten times per year per individual customer, not 270 times.

consequences of a lifetime of exposure. Consequently, risk assessment requires the estimation of the portion of total lifetime drinking water consumption that any one individual would receive from a particular type of water system. In turn, one needs to estimate the appropriate portions for daily, day per year, and year per lifetime consumption. These estimates need to be prepared for both the workers at the facility and the "customers" of the facility.

This adjustment was accomplished through a comprehensive review of government and trade association statistics on entity utilization by SIC code. These figures, coupled with SDWIS information relating to the portion of a particular industry served by non-community water systems, made possible the development of two estimates needed for the risk assessment: customer cycles per year and worker per population served per day. These numbers are required to distinguish the more frequent and longer duration exposure of workers from that of system customers. The end points of the analysis are to determine what portion of worker and customer lifetime exposure could potentially come from an NTNC water system and to also determine the total numbers of customers served.

Customer cycles per year[7] is the term used to describe the number of independent customer groups which are served by a given NCWS. For example, NTNC hotels and motels are generally open year round. They do not serve the same customers every day, but by determining the portion of the population which uses overnight lodging in a given year and dividing into the total number of rooms utilized (and adjusting for multiple occupancy in the case of recreational travel), one is able to estimate the average days of use per year by an individual (frequency of utilization). If one then incorporates information on vacancy rates, it is possible to estimate the number of customer cycles per year. Coupled with statistics on employment in the hospitality industry, it is possible to determine the portion of reported SDWIS populations which represent employees and what portion represent customers. The number of customers times the customer cycles provide the total consumer population served by NTNC hotels and motels. All of this required information is available on the Internet through the Economic Census or hospitality industry statistics. With these information, one can calculate:

$$TC_c = (P_c * CC_c) * (1 - WP_c)$$

$$TW_c = P_c * WP_c$$

where:

TC=	total number of customers
TW=	total number of workers
P=	SDWIS population
WP=	workers per person per day
CC=	number of customer cycles per year
c=	NTNC service category

[7]the number of times each year the customer base turns over. For example, if this parameter equals one, then the same customer's are served each day. If the value is seven, then seven sets of customers use the facility.

and:

$$PWLE_c = \frac{PWDC_c * DW_c * YW_c}{365 * 70}$$

$$PCLE_c = \frac{PCDC_c * DC_c * YC_c}{365 * 70}$$

where;

PWLE	=percent of worker lifetime exposure
PCLE	=percent of customer lifetime exposure
PWDC	=percentage of workers daily consumption
PCDC	=percentage of customers daily consumption
DW	=worker days per year
DC	=customer days per year
YW	=worker years
YC	=customer years

Table VI.3 presents the resultant utilization patterns which were derived for the service areas identified. In developing these estimates, exposure assumptions were selected to maximize individual risk. This approach was taken because it was felt that if these assumptions still lead to the conclusion that there did not exist a meaningful opportunity for risk reduction, the case would only be more so when future studies further refined exposure scenarios. For example, workers were assumed to stay in the same job for 45 years and to have perfect attendance at that job. Likewise, school children were assumed to perfectly attend the same school for 12 years[8]. Daily exposure was generally apportioned based on a portion of waking hours basis with the exception of eating and drinking establishments which were assumed to provide one fourth of daily consumption on any given day of use. For the average adult male, one fourth of daily use corresponds to about thirteen ounces. For the 90th percentile male, it would amount to twenty-three ounces for every day of use.

D. Lifetime Risk per pCi Consumed

Earlier sections of the document describe the derivation of carcinogenic risk profiles for each of the radionuclides. Because risk varies with the age of exposure for the radionuclides, however, evaluation of NTNC control benefits requires a slightly different approach. In NTNC systems, the age of exposure is not constant over the lifetime. Consequently, it is important to know the ages at which exposure occurs. For example, day care kids would primarily be less

[8]This is a conservative assumption because three schools would generally be involved over this span of years. There is a small likelihoood that all three would be NTNC systems and, even more so that they all would have the same extremely high occurrence levels.

than five years of age. Likewise, workers would predominantly be between twenty and sixty-four. Table 8.4 presents the assumed age distributions used in the modeling effort. For each exposure group, weighted average lifetime risks per pCi consumed over that age interval were developed. Gross alpha was modeled as radium 224. Beta was not modeled.

Table VI.3
NTNC Population and Exposure Time Data

	Number of Systems	Total SDWIS Population	Number of Customer Cycles/Year	Worker Per Person Per Day	Percent of Worker's Daily Consumption	Worker Days Per Year	Worker Years	Percent of Customer's Daily Consumption	Customer Days Per Year	Customer Years	Total Worker Population	Total Customer Population
Water Wholesalers	266	66,018	1.00	0	n/a	n/a	n/a	25.0%	270.00	70.00	0	66,018
Mobile Home Parks	104	19,240	1.33	0.046	50.0%	250	40	100.0%	270.00	35.00	885	24,412
Nursing Homes	130	13,910	1.00	0.23	50.0%	250	40	100.0%	365.00	10.00	3,199	10,711
Churches	230	11,500	1.00	0.01	50.0%	250	40	50.0%	52.00	70.00	115	11,385
Golf and Country Clubs	116	11,716	4.50	0.11	50.0%	250	40	50.0%	52.00	70.00	1,289	46,923
Retailers (Food related)	142	45,724	2.00	0.07	50.0%	250	40	25.0%	185.00	70.00	3,201	85,047
Retailers (Non-food related)	695	120,930	4.50	0.09	50.0%	250	40	25.0%	52.00	70.00	10,884	495,208
Restaurants	418	154,660	2.00	0.07	50.0%	250	40	25.0%	185.00	70.00	10,826	287,668
Hotels/Motels	351	46,683	86.00	0.27	50.0%	250	40	100.0%	3.40	40.00	12,604	2,930,759
Prisons/Jails	67	121,940	1.33	0.1	50.0%	250	40	100.0%	270.00	3.00	12,194	145,962
Service Stations	53	12,190	7.00	0.06	50.0%	250	40	25.0%	52.00	54.00	731	80,210
Agricultural Products/Services	368	27,968	7.00	0.125	50.0%	250	40	25.0%	52.00	50.00	3,496	171,304
Daycare Centers	809	61,484	1.00	0.145	50.0%	250	10	50.0%	250.00	5.00	8,915	52,569
Schools	8,414	3,086,012	1.00	0.073	50.0%	200	40	50.0%	200.00	12.00	225,279	2,860,733
State Parks	83	106,895	26.00	0.016	50.0%	250	40	50.0%	14.00	70.00	1,710	2,734,802
Medical Facilities	367	163,631	16.40	0.022	50.0%	250	40	100.0%	6.70	10.30	3,600	2,624,510
Campgrounds/RV Parks	123	19,680	22.50	0.041	50.0%	180	40	100.0%	5.00	50.00	807	424,645
Federal Parks	20	780	26.00	0.016	50.0%	250	40	50.0%	14.00	70.00	12	19,956
Highway Rest Areas	15	6,105	50.70	0.01	50.0%	250	40	50.0%	7.20	70.00	61	306,428
Misc. Recreation Services	259	22,533	26.00	0.016	50.0%	250	40	100.0%	14.00	70.00	361	576,484
Forest Service	107	4,494	26.00	0.016	100.0%	250	40	100.0%	14.00	50.00	72	114,974
Interstate Carriers	287	35,301	93.00	0.304	50.0%	250	40	50.0%	2.00	70.00	10,732	2,284,963
Amusement Parks	159	76,462	90.00	0.18	50.0%	250	10	50.0%	1.00	70.00	13,763	5,642,896
Summer Camps	46	6,716	8.50	0.1	100.0%	180	10	100.0%	7.00	10.00	672	51,377
Airports	101	326,860	36.50	0.308	50.0%	250	40	25.0%	10.00	70.00	100,673	8,255,830
Military Bases	95	67,525	n/a	1	50.0%	250	40	n/a	n/a	n/a	67,525	0
Non-Water Utilities	497	84,490	n/a	1	50.0%	250	40	n/a	n/a	n/a	84,490	0
Office Parks	950	181,600	n/a	1	50.0%	250	40	n/a	n/a	n/a	181,600	0
Manufacturing: Food	768	285,696	n/a	1	50.0%	250	40	n/a	n/a	n/a	285,696	0
Manufacturing: Non-Food	3,356	588,792	n/a	1	50.0%	250	40	n/a	n/a	n/a	588,792	0
Landfills	78	3,432	n/a	1	100.0%	250	40	n/a	n/a	n/a	3,432	0
Fire Departments	41	4,018	n/a	1	100.0%	250	40	n/a	n/a	n/a	4,018	0
Construction	99	5,247	n/a	1	100.0%	250	40	n/a	n/a	n/a	5,247	0
Mining	119	13,447	n/a	1	100.0%	250	40	n/a	n/a	n/a	13,447	0
Migrant Labor Camps	33	2,079	n/a	1	100.0%	250	40	n/a	n/a	n/a	2,079	0
										Subtotal =	1,662,407	30,305,774
										TOTAL =		31,968,181

Table VI.4
NTNC Percent of Lifetime Exposure and Age at Exposure

	Percent of Worker Lifetime Exposure	Percent of Customer Lifetime Exposure	Worker Age Bracket	Customer Age Bracket
Water Wholesalers	0.00%	18.49%	n/a	all
Mobile Home Parks	19.57%	36.99%	20 to 64	all
Nursing Homes	19.57%	14.29%	20 to 64	65+
Churches	19.57%	7.12%	20 to 64	all
Golf and Country Clubs	19.57%	7.12%	20 to 64	all
Retailers (Food related)	19.57%	12.67%	20 to 64	all
Retailers (Non-food related)	19.57%	3.56%	20 to 64	all
Restaurants	19.57%	12.67%	20 to 64	all
Hotels/Motels	19.57%	0.53%	20 to 64	all
Prisons/Jails	19.57%	3.17%	20 to 64	20 to 64
Service Stations	19.57%	2.75%	20 to 64	16 to 70
Agricultural Products/Services	19.57%	2.54%	20 to 64	all
Daycare Centers	4.89%	2.45%	20 to 64	<5
Schools	15.66%	4.70%	20 to 64	6 to 18
State Parks	19.57%	1.92%	20 to 64	all
Medical Facilities	19.57%	0.27%	20 to 64	all
Campgrounds/RV Parks	14.09%	0.98%	20 to 64	all
Federal Parks	19.57%	1.92%	20 to 64	all
Highway Rest Areas	19.57%	0.99%	20 to 64	all
Misc. Recreation Services	19.57%	3.84%	20 to 64	all
Forest Service	39.14%	2.74%	20 to 64	all
Interstate Carriers	19.57%	0.27%	20 to 64	all
Amusement Parks	4.89%	0.14%	20 to 64	all
Summer Camps	7.05%	0.27%	20 to 64	11 to 19
Airports	19.57%	0.68%	20 to 64	all
Military Bases	19.57%	0.00%	20 to 64	n/a
Non-Water Utilities	19.57%	0.00%	20 to 64	n/a
Office Parks	19.57%	0.00%	20 to 64	n/a
Manufacturing: Food	19.57%	0.00%	20 to 64	n/a
Manufacturing: Non-Food	19.57%	0.00%	20 to 64	n/a
Landfills	39.14%	0.00%	20 to 64	n/a
Fire Departments	39.14%	0.00%	20 to 64	n/a
Construction	39.14%	0.00%	20 to 64	n/a
Mining	39.14%	0.00%	20 to 64	n/a
Migrant Labor Camps	39.14%	0.00%	all	n/a

E. Occurrence

There is no readily available information relating to the occurrence of radionuclides in NCWS. Further, there are serious limitations associated with the use of the earlier referenced National Inorganics and Radionuclide Survey (NIRS), even for CWS estimation. Nevertheless, given that NIRS data were obtained early in the implementation of SDWA, among those systems which exceed present MCLs, NIRS may still provide a reasonable estimate of the relative distribution of the MCL exceedences. By limiting the model to only those systems which are in excess of the MCL, the Agency believes it is possible to characterize the relative distribution of individual risks due to radionuclide exposure. For each of the modeled radionuclides, means and 90th percentile concentrations of the exceedences were calculated by assuming that the NIRS exceedences were lognormally distributed. For example, in the case of radium-228, the log mean and variance of all NIRS values in excess of 5 pCi/l were determined. The resultant mean and 90th percentile values were used in calculating the NTNC benefits. Table VI.5 presents the mean and 90th percentile values derived for each radionuclide.

Table VI.5
Assumed NTNC Water Concentrations
pCi/l

	Gross alpha (Ra-224)	Radium-226	Radium-228	Uranium
Mean	28	7.6	7.2	39
90th %ile	54	12	11	80

F. Estimates of Distribution of Consumer Overlap

Very little is presently known about the extent of consumer overlap among NTNC water systems. It is known that the total number of workers served by these systems is only about one percent of the Nation's workforce. On the customer side of the equation, day care centers and schools constitute the two most sensitive sectors from an exposure standpoint. In both cases, NTNC served populations represent a very small portion of the total sector population. Add to these facts that only a very small percentage of water systems would be expected to have high radionuclide levels and one would conclude that it is fairly improbable that there would be considerable consumer overlap across these groups and of these types of water systems[9].

[9]To have overlap, there would need to be significant groups of NTNC water systems concentrated in one geographic region. Such a concentration of industrial and commercial enterprise, however, would imply a fair degree of urbanization and collectivization of water supply.

On the other hand, there are some sectors representing a significant portion of the total population. Foremost among these is airports, which serve approximately seventeen percent of the major airport traffic. However, in this case, predicted consumption is less than one percent of the lifetime amount. Therefore, even if there is overlap, it will have a fairly inconsequential impact on individual lifetime risks[10]. Consequently, the Agency believes it is appropriate to model these groups separately, but to leave some buffer in the interpretation of results to reflect the uncertainty.

G. NCWS Risk Model

The risk analysis is a limited Monte-Carlo model. Ordinarily, in assessing chronic carcinogenic components, the model would simulate the distribution by randomly selecting values for water consumption, body weight, occurrence concentration, and service area type to obtain each realization of the Monte Carlo model. Because there is so much uncertainty about the actual contaminant levels, occurrence distributions were input as a constant in the analysis. In addition, body weight was irrelevant to radionuclide potency and did not require simulation. The formula for each iteration of the model is as follows:

$$WLR_i = OCC \times R_i \times Cons_i \times fpd \times dpy \times Z$$

and

$$CLR_i = OCC \times R_i \times Cons_i \times fpd \times dpy \times Z$$

where

WLR_i = worker lifetime risk (per 100,000 people)
CLR_i = customer lifetime risk (per 100,000 people)
OCC = water concentration in pCi/l
R_i = risk per pCi for the individual in the age category corresponding to the selected service area type
$Cons_i$ = daily water consumption rate
fpd = fraction of daily water consumption obtained from source
dpy = number of days per year source is used
Z = number of years of exposure to source

These formula are calculated for each service area type on each iteration. In addition, on each iteration, the model selects the service area being simulated for the composite risk using a population weighted probability distribution.

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[10]Further, airports are predominantly purchased water systems. This point is significant because, regardless of the decision on the regulation of radionuclides in NTNC systems, puchased water systems will already have realized the benefits of regulation by virtue of treatment performed by the community water system.

VII - SUPPORTING MATERIAL FOR ECONOMICS AND IMPACTS ANALYSIS

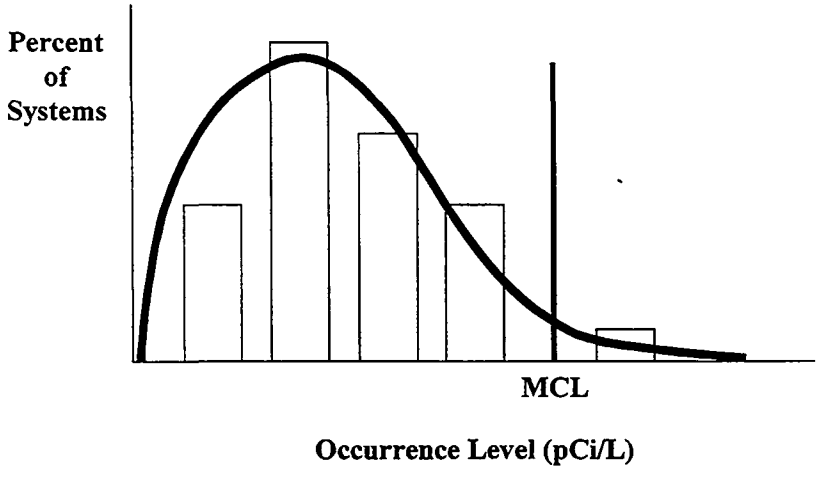
A. Strengths and Weaknesses of the Direct Proportions and Lognormal Model Approaches to Estimating Radionuclides Occurrence in Drinking Water

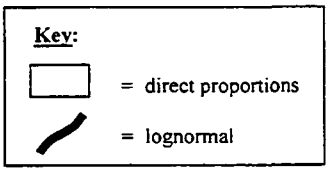
The difference between the direct proportions and lognormal model approaches to extrapolating from the NIRS data to national estimates of occurrence is illustrated by the hypothetical example shown in Figure VII-1. The lognormal distribution follows the pattern of the bars (which indicate the proportions extrapolated directly from NIRS), but spreads systems more evenly over the continuum of possible concentration levels. It also tends to place systems into the right tail of the distribution (i.e., to lead to higher estimates of occurrence for those systems most out of compliance). However, the exact relationship between the direct proportion and lognormal estimates depends on the underlying NIRS data, as is indicated by the estimates reported in the Economics and Impacts Analysis section of the NODA.

The use of both approaches allows the distinct advantages of each to be exploited and the disadvantages of each to be considered in the interpretation of the results. To summarize, the advantage of the direct proportions approach is that it does not assume that the data behaves in a certain way, so model errors are not introduced into predictions. Its disadvantage is that it introduces "zero probabilities" in some cases, which are not meaningful. The advantage of the lognormal model approach is that it allows the replacement of these artificial "zero probabilities" with an estimate of the small probabilities involved, which is more realistic. Its disadvantage is that it predicts artificial probabilities of very high radionuclides levels, since the data are "smoothed" into the tails of the distribution. For example, the highest reported uranium level in a national survey (28,239 samples representing all U.S. States except Alaska) of finished drinking water performed by Oak Ridge National Laboratory (ORNL) is 660 pCi/L (EPA 1981). However, a lognormal model based on the NIRS data for uranium would predict that almost 1% of ground water samples would have uranium levels greater than 660 pCi/L, which is clearly not the case based on the ORNL study. Since both approaches have advantages and disadvantages, we used them both for occurrence estimates, keeping in mind that the direct proportions approach may under-predict occurrence in situations where "zero impacts" are predicted and that the lognormal approach may over-predict the percentages of systems having high occurrence levels.

Figure VII.1.

HYPOTHETICAL ILLUSTRATION OF DIRECT PROPORTIONS VS. LOGNORMAL MODEL APPROACHES TO OCCURRENCE





B. Risk Coefficients

For each individual radionuclide, EPA developed a central-tendency risk coefficient that expresses the estimated probability of radiogenic cancer morbidity or mortality per unit activity intake.[11] For this analysis, we used the draft September 1999 risk coefficients developed as part of EPA's revisions to Federal Guidance Report 13 (FGR-13).[12] FGR-13 compiled the results of several models predicting the cancer risks associated with radioactivity. The cancer sites considered in these models include the esophagus, stomach, colon, liver, lung, bone, skin, breast, ovary, bladder, kidney, thyroid, red marrow (leukemia), as well as residual impacts on all remaining cancer sites combined.

The available occurrence data from NIRS do not provide information on the contribution of individual radionuclides or isotopes to the total concentrations of gross alpha or uranium. Therefore, we cannot apply the individual risk coefficients directly from FGR-13 in these cases. Our approach to estimating the risk coefficients in these cases is described here.

1. Gross Alpha Risk Coefficient

Ideally, a risk coefficient used to model the risks presented by gross alpha in drinking water would reflect the actual mix of alpha emitters commonly found in drinking water from those systems affected by each potential regulatory change. However, sufficient information on the prevalence of the individual alpha emitters is not available. Instead, we estimated the risk factors based on two prevalent alpha emitters: radium-224 and radium-226. The risk coefficients for the naturally occurring alpha emitters, including Ra-224 and Ra-226, are listed in Table VII-1, which follows.

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[11] Morbidity indicates total cancer incidence (fatal and nonfatal); mortality indicates the incidence of fatal cancers.

[12] Eckerman, Keith F., Richard W. Leggett, Christopher B. Nelson, Jerome S. Puskin, and Allan C.B. Richardson, Cancer Risk Coefficients for Environmental Exposure to Radionuclides, Federal Guidance Report No. 13 (Draft), September 1999.

Table VII-1 RISK COEFFICIENTS FOR NATURALLY-OCCURRING ALPHA EMITTERS				
Series	Radionuclide	Half Life	Mortality Risk Coefficient per pCi	Morbidity Risk Coefficient per pCi
Th-232	Th-232	1.4E10 years	6.92E-11	1.01E-10
Th-232	Th-228	1.9 years	6.73E-11	1.07E-10
Th-232	Ra-224	3.66 days	1.01E-10	1.67E-10
Th-232	Po-216	0.15 seconds	NA	NA
Th-232	Bi-212	61 minutes	5.00E-13	7.10E-13
Th-232	Po-212	3.0E-6 seconds	NA	NA
U-238	Ra-226	1,600 years	2.65E-10	3.85E-10
U-238	Po-218	3 minutes	NA	NA
U-238	Po-214	1.6E-4 seconds	NA	NA
U-238	Po-210 (inorganic)	138 days	2.74E-10	3.77E-10
U-235	Po-231	3.3E4 years	NA	NA
U-235	Th-227	18.7 days	2.67E-11	4.74E-11
U-235	Ra-223	11 days	1.48E-10	2.38E-10
U-235	Po-215	1.8E4 seconds	NA	NA
U-235	Bi-211	2.2 minutes	NA	NA

Notes:
NA indicates that the coefficient is not available.
Lifetime risk coefficients are based on Table 2.2a of the September 1999 draft of *Federal Guidance Report No. 13*; becquerels are converted to picocuries using a conversion factor of 3.70E-02 Bq/pCi.

For the preliminary analysis, we used a weighted-average of the risk coefficients for radium-224 and radium-226 to calculate the cancer risks associated with the changes to the monitoring requirements for gross alpha. Based on occurrence data from NIRS and from a recent U.S. Geological Survey draft report (currently under review for publication), we weighted the risk coefficients according to the estimated relative presence of these two radionuclides (see EPA 1999a for details). The resulting coefficients were used in assessing the affects of closing the gross alpha monitoring deficiency. For risk reductions associated with proposed revisions to the gross alpha MCL, we employed the mortality and morbidity risk coefficients for radium-224 alone, since the revised MCL would exclude radium-226 under the proposed changes.

Table VII-2 presents the resulting risk coefficients for gross alpha. The values in the last row of the table, the weighted averages of the risk coefficients for radium-224 and radium-226, were used in the analysis of risk reductions associated with closing the monitoring deficiencies.

Table VII-2

RISK COEFFICIENTS USED IN GROSS ALPHA ANALYSIS

Radionuclide	Half Life	Mortality Risk Coefficient per pCi	Morbidity Risk Coefficient per pCi
Ra-224	3.6 days	1.01E-10	1.67E-10
Ra-226	1,600 years	2.65E-10	3.85E-10
Average Weighted by Relative Prevalence of Ra-224 and Ra-226		1.14E-10	1.83E-10

Notes:
Lifetime risk coefficients are based on Table 2.2a of the September 1999 draft of *Federal Guidance Report No. 13*; becquerels are converted to picocuries using a conversion factor of 3.70E-02 Bq/pCi.
Weighted average values for gross alpha are based on the estimated relative prevalence of these two radionuclides in systems affected by closure of the monitoring loopholes.

2. Combined Radium Risk Coefficient

The approach used to develop risk coefficients for combined radium is similar to the approach used for gross alpha. To estimate cancer risk reductions from changes to the monitoring requirements for combined radium, we used a weighted average of the risk coefficients for radium-226 and radium-228; this weighted average is based upon the occurrence data for those systems legally out of compliance with the combined radium standard. For risk reductions resulting from limiting the contribution of radium-228 to the MCL, we use the risk coefficient for radium-228 alone. Table VII-3 below presents the risk coefficients employed in the preliminary analysis for combined radium.

Table VII-3 RISK COEFFICIENTS USED IN COMBINED RADIUM ANALYSIS			
Radionuclide	**Half Life**	**Mortality Risk Coefficient per pCi**	**Morbidity Risk Coefficient per pCi**
Ra-226	**1,600 years**	**2.65E-10**	**3.85E-10**
Ra-228	**5.75 years**	**7.40E-10**	**1.04E-09**
Average Weighted by Relative Prevalence of Ra-226 and Ra-228		**5.66E-10**	**8.03E-10**

Notes:
Lifetime risk coefficients are based on Table 2.2a of the September 1999 draft of *Federal Guidance Report No. 13*; becquerels are converted to picocuries using a conversion factor of 3.70E-02 Bq/pCi.
Weighted average values for combined radium are based on the estimated relative prevalence of these two radionuclides in systems affected by closure of the monitoring loopholes, based directly on the NIRS data.

3. Uranium Risk Coefficient

To determine the cancer risk coefficients for uranium, we calculated the simple average of the coefficients for uranium-234, -235, and -238, due to the lack of data on the prevalence of each isotope in those drinking water supplies potentially affected by each regulatory option. As shown in Table VII-4, the coefficients for each of these isotopes are similar, so we expect that this simplified approach will not result in significant under- or over-estimates of risk in spite of the fact that the expected prevalences of the three uranium isotopes are different than that implied by the averaging process.

Table VII-4 RISK COEFFICIENTS USED IN URANIUM ANALYSIS			
Radionuclide	**Half Life**	**Mortality Risk Coefficient per pCi**	**Morbidity Risk Coefficient per pCi**
U-238	**4.5E9 years**	**4.18E-11**	**6.40E-11**
U-235	**7.4E6 years**	**4.48E-11**	**6.96E-11**
U-234	**2.4E5 years**	**4.59E-11**	**7.07E-11**
Simple Average for Uranium (U-234, U-235, U-238)		**4.40E-11**	**6.81E-11**

Notes:
Lifetime risk coefficients are based on Table 2.2a of the September 1999 draft of *Federal Guidance Report No. 13*; becquerels are converted to picocuries using a conversion factor of 3.70E-02 Bq/pCi.
The average values for uranium are simple (unweighted) averages of the risk coefficients for the isotopes listed, based directly on the NIRS data.

C. Risk Factors

Next, we used the unit risk coefficients discussed above to determine lifetime and annual factors that indicate the cancer risks faced by individuals who ingest tap water. We converted the risk coefficients from FGR-13 into individual risk factors (expressed in terms of activity concentration -- per pCi/L of water) by assuming that, on average, an individual consumes 1.1 liters of water per person per day. This is the estimated mean value for direct and indirect community tap water ingestion for the total U.S. populations (all ages) based on a recent EPA study. "High-end" risk factors based on water ingestion rates of 2.2 L/person/day (90th percentile value) were also calculated to risks to individuals who consume much larger quantities of water, on average. Other assumptions are standard: water ingestion is assumed to occur over the individual average life expectancy of 70 years. All assumptions are discussed in detail in EPA 1999a.

Using these assumptions, we calculated risk factors for both morbidity and mortality in terms of statistical cases per person per pCi/L, which are presented in Table VII-5.

Table VII-5 AVERAGE INDIVIDUAL RISK FACTORS, AVERAGE WATER CONSUMPTION (1.1 L/person/day) (per pCi/L)				
Regulatory Option	**Morbidity**		**Mortality**	
	lifetime ingestion	**annual ingestion**	**lifetime ingestion**	**annual ingestion**
Gross Alpha: changes in monitoring requirements (weighted average of Ra-224 and Ra-226)	5.24E-06	7.48E-08	3.26E-06	4.65E-08
Gross Alpha: changes in MCL (Ra-224 only)	4.77E-06	6.81E-08	2.90E-06	4.15E-08
Combined Radium: changes in monitoring requirements (weighted average of Ra-226 and Ra-228)	2.30E-05	3.28E-07	1.63E-05	2.32E-07
Combined Radium: changes in MCL (Ra-228 only)	2.98E-05	4.26E-07	2.12E-05	3.03E-07
Uranium: establish MCL (simple average of U-234, U-235, and U-238)	1.95E-06	2.79E-08	1.26E-06	1.81E-08
AVERAGE INDIVIDUAL RISK FACTORS, 90th PERCENTILE WATER CONSUMPTION (per pCi/L)				
Gross Alpha: changes in monitoring requirements (weighted average of Ra-224 and Ra-226)	1.03E-05	1.47E-07	6.39E-06	9.13E-08
Gross Alpha: changes in MCL (Ra-224 only)	9.37E-06	1.34E-07	5.70E-06	8.15E-08
Combined Radium: changes in monitoring requirements (weighted average of Ra-226 and Ra-228)	4.51E-05	6.44E-07	3.19E-05	4.56E-07
Combined Radium: changes in MCL (Ra-228 only)	5.85E-05	8.35E-07	4.16E-05	5.95E-07
Uranium: establish MCL (simple average of U-234, U-235, and U-238)	3.83E-06	5.47E-08	2.48E-06	3.55E-08

D. Estimation of Monetized Benefits

1. Methodologies Used for the Analysis

The practice of benefits valuation is based on economic theories which attempt to measure the societal value of a particular improvement by estimating the "satisfaction" or "utility" that individuals derive from it. Individuals reveal these values through their "willingness to pay" for the effects of these improvements. Willingness to pay is the maximum amount of money an individual would voluntarily exchange to obtain an improvement (e.g., a reduction in health risks), given his or her available financial resources and desired spending on other goods and services.

Note that willingness to pay is not the same as price or cost. Price is determined by the interactions of buyers and sellers in the marketplace, while cost is a function of the materials, processes, and labor which are necessary to create a good or service. An individual's willingness to pay for a particular good or service may either greater or less than the market price, depending upon individual preferences.

Because willingness to pay for health risk reductions is difficult to directly observe in the marketplace, economists most commonly use three types of studies to estimate the value of reduced fatal and nonfatal risks: wage-risk studies; cost of illness studies; and contingent valuation studies. EPA regulatory analyses often transfer estimates from existing studies to value the benefits of alternative policies, as discussed later in this section. Each of these types of studies is discussed.

- Wage-risk studies are often used to value changes in fatal risks; i.e., premature mortality. These studies examine the additional compensation workers demand for taking riskier jobs, typically focusing on small changes in the risk of accidental workplace fatalities. Researchers use statistical methods to separate the changes in compensation that are associated with changes in risks from the changes in compensation that are associated with other job characteristics. The wage-risk method has several advantages; for example, the data and methods it uses are well-established, and it directly measures changes in the risk of premature mortality. This method is widely used to value reductions in fatal risks, and the available studies have been subject to extensive peer review. However, these studies generally address risks from work place accidents that differ in significant ways from the cancer and other risks associated with environmental regulations.

- Cost of illness studies are often used to value changes in nonfatal (morbidity) risks, but are not a measure of willingness to pay. These studies examine the actual direct (e.g., medical expenses) and indirect (e.g., lost work or leisure time) costs incurred by affected individuals. In general, the logic for using cost of illness studies to value benefits is as follows: if illness imposes the cost of medical expenditures and foregone earnings, then a regulation leading to a reduction in illness yields benefits equal at minimum to the costs saved. The cost of illness method is well-developed, widely applied, and easily explained. It also addresses direct and indirect costs that are relatively easy to measure and has been used to provide estimates for large numbers of illnesses. In most cases, however, cost of illness studies may significantly underestimate individuals' willingness to pay for decreased health risks because they do not address factors such as pain and suffering.[13] In addition, environmental regulations generally reduce future risks, while the cost of illness method considers effects that have already occurred -- and hence does not address risk aversion. Nonetheless, because of their widespread availability and ease

[13] Cost of illness estimates may also occasionally overstate willingness to pay, particularly if the availability of insurance leads people to agree to treatments that they would not fully finance themselves.

of use, cost of illness estimates are often used to value the nonfatal effects of environmental regulations.

- Contingent valuation studies use surveys to elicit statements of willingness to pay and are often applied to value both fatal and nonfatal health effects. For example, researchers might ask individuals what they would be willing to pay for a specified reduction in the risk of developing stomach cancer from long-term exposure to contaminants in drinking water. The researchers can define the scenario to address factors that may influence total willingness to pay, such as the pain and suffering associated with an illness, thereby providing a more complete estimate of willingness to pay. Such surveys must be carefully designed and administered, however, if they are to provide reliable and precise estimates, because the individuals surveyed are usually not required to make actual payments and may have difficulty understanding the scenario presented. Contingent valuation surveys have been completed for a relatively small subset of the health effects associated with environmental regulations.

"Benefits transfers" from the above types of studies are often used in EPA regulatory analyses. Rather than conducting resource-intensive new primary research on the value of reducing specific risks to human health, analysts often use data from existing wage-risk, cost of illness, contingent valuation, or other studies as surrogates for the needed values. Because the available studies usually do not address the specific effects of concern (e.g., cancer risks from radionuclides in drinking water), EPA's approach requires the application of "benefit transfer" techniques. Benefit transfer refers to the use of valuation information from one or more existing studies to assess similar, but not identical, effects associated with a regulation or policy. To conduct a benefit transfer, analysts first must evaluate the quality and applicability (e.g., the similarity of the health effects and populations experiencing the effects) of the available studies, then apply the results of selected studies (with any necessary adjustments) to the policy of concern.

Using these generally accepted economic valuation techniques in conjunction with the estimates of fatal and nonfatal cancer risks reductions associated with each regulatory option (described in the previous section), we projected valuation estimates. To estimate the monetary value of reduced fatal risks (i.e., risks of premature death from cancer) predicted under different regulatory options, we apply the value of a statistical life (VSL) approach. VSL does not refer to the value of an identifiable life, but instead to the value of small reductions in mortality risks in a population. A "statistical life" is thus the sum of small individual risk reductions across an entire exposed population.

EPA has identified twenty-six VSL studies (that use the wage-risk or contingent valuation method) which have been peer reviewed and recommended for use in EPA policy analyses.[14] The best estimates from these studies range from $0.7 million to $16.3 million and

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[14] U.S. Environmental Protection Agency, *The Benefits and Costs of the Clean Air Act, 1970 to 1990*, October 1997, Appendix I; and U.S. Environmental Protection Agency, *Guidelines for Preparing Economic*

approximate a Wiebull distribution with a mean of $5.8 million (in 1997 dollars). To value the changes in fatal risks associated with the radionuclides regulation, we apply a "best" estimate of $5.8 million and low and high end estimates of $1.4 million and $11.2 million, reflecting the uncertainty in these estimates. The low and high end estimates represent the tenth and ninetieth percentile of the distribution of VSL estimates, respectively.

To estimate the monetary valu of reduced nonfatal cancers under different regulatory options, ideally we would be able to predict the types of cancers averted by the regulations. However, exposure to radionuclides can result in a range of cancers. The type of cancer depends largely on where the radionuclides localize in the body as a result of one's metabolism. While some radionuclides are associated with specific cancer types (such as leukemia or colon, stomach, thyroid, bone, and liver cancer), many are not, and radiation risk models generally consider fourteen cancer sites (discussed in the earlier section on risk coefficients).

Given the difficulties inherent in predicting the types of cancers averted by the radionuclides regulations, we review the cost of illness estimates available for a range of nonfatal cancers that may be most likely to result from exposure to radionuclides via tap water ingestion.[15] EPA has developed cost of illness estimates for selected cancers, as reported in Table VII-6.[16] Note that these estimates are preliminary and are now undergoing review.

Analysis (Review Draft), June 1999, Chapter 7.

[15] U.S. Environmental Protection Agency, *Cost of Illness Handbook (draft),* prepared by Abt Associates, September 30, 1998, Chapters II.1, II.2, and II.9.

[16] Because liver cancer has a high mortality rate (approximately 97 percent after 21 years), we do not consider it in the valuation of nonfatal risks; fatal risks are assessed using the VSL method discussed earlier in this section. We also do not include the estimates for kidney cancer, because the cost of illness analysis does not separate the costs for survivors from the costs for non-survivors.

Table VII-6 LIFETIME AVOIDED MEDICAL COSTS FOR SURVIVORS (1997 dollars)				
Type of Cancer	**Date Data Collected**	**Number of Cases Studied**	**Estimated Survival Rate**	**Mean Value per Nonfatal Case**
Colorectal cancer	1974-1981	19,673 Medicare patients	53 percent (after 10 years)	$106,900 (for typical individual diagnosed at age 70)
Stomach cancer	1974-1981	3,228 Medicare patients	< 20 percent (after 5 years)	$88,100 (for typical individual diagnosed at age 70)
Bone cancer	N/A; theoretical approach		64 percent (after 5 years, includes bone and joint cancers)	$89,400 - $110,000

Notes:
Exhibit reports present value (at the time of onset) of the lifetime costs of the illness (using a 7 percent discount rate). Values were inflated to 1997 dollars based on the consumer price index for the costs of medical commodities and services.
Source: U.S. Environmental Protection Agency, *Cost of Illness Handbook (draft)*, September 1998.

For colorectal and stomach cancers, these estimates of direct medical costs are derived from a study conducted by Baker et al., which uses data from a sample of Medicare records for 1974 - 1981.[17] These data include the total charges for inpatient hospital stays, skilled nursing facility stays, home health agency charges, physician services, and other outpatient and medical services. These costs were inflated using the medical care components of the consumer price index. EPA combined these data with estimates of survival rates and treatment time periods to determine the average costs of initial treatment and maintenance care for patients who do not die of the disease. Information on mean age at diagnosis and survival rates were generally derived from a database maintained by the National Cancer Institute, that covers the years 1973 - 1993.

For bone cancer (which is not addressed by Baker et al.), EPA used a theoretical approach that combine average values for initial and maintenance care from the Baker study with estimates of the time period over which maintenance care is needed. The range reported in the exhibit above reflects two different assumptions regarding the duration of maintenance care; a 10 year duration vs. a duration based on average life expectancy at the age of diagnosis.

This study also provides estimates of time lost due to illness for colorectal and stomach cancer. For individuals diagnosed at age 70, the average lifetime lost hours are 2,266 for colorectal cancer and 2,942 for stomach cancer. These estimates are based on a study conducted

[17] Baker, Mary S. et al., "Site Specific Treatment Costs for Cancer: An Analysis of the Medicare Continuous History Sample File," *Cancer Care and Cost. DRGs and Beyond.* Richard M. Scheffler and Neil C. Andrews, Editors, Ann Arbor, MI: Health Administration Press Perspectives, 1989.

by Hartunian et al., which calculated lost work time for the first year post-diagnoses.[18] EPA then adjusted these estimates to reflect lifetime lost hours including lost leisure time.[19] For the typical stomach and colorectal cancer survivor, all of the lost hours are assumed to occur in the first year post-diagnoses. Because the cancers most often linked to the radionuclides of concern are usually diagnosed late in life, this lost time is most likely to be leisure time during retirement. Determining the appropriate value for such lost time is difficult, and is hence not included in the valuation estimates.

For the preliminary analysis, we use the approximate mid-point and high and low estimates from Table VII-6 to estimate the avoided medical costs attributable to reducing nonfatal cancer risks, as summarized below.

Best Estimate: *Value of nonfatal risk reductions (medical costs only) = Statistical cases averted * $0.099 million*

Low End Estimate: *Value of nonfatal risk reductions (medical costs only) = Statistical cases averted * $0.088 million*

High End Estimate: *Value of nonfatal risk reductions (medical costs only) = Statistical cases averted * $0.110 million*

These cost of illness estimates are likely to understate total willingness to pay for avoiding these cancers. They exclude certain types of avoided costs (e.g., lost work or leisure time). In addition, the cost of illness approach does not address other factors that influence willingness to pay, such as risk aversion and the desire to avoid pain and suffering.

While little information is available on individuals' willingness to pay to avoid cancer risks, studies of other diseases suggest that cost of illness values may significantly understate total willingness to pay. Table VII-7 summarizes the studies that compare cost of illness estimates to estimates of total willingness to pay. These studies estimate total willingness to pay based on contingent valuation or averting behavior studies. They vary in terms of the types of expenditures addressed in the cost of illness studies; some exclude lost earnings and some exclude costs borne by others (e.g., through insurance).

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[18] Hartunian, N.S., C.N. Smart, and M.S. Thompson, *The Incidence of Economic Costs of Major Health Impairments*. Lexington, MA: Lexington Books, 1981.

[19] U.S. Environmental Protection Agency. *Cost of Illness Handbook (draft)*. September 1998, Chapters II.1, II.2, and II.9

Table VII-7 COMPARISON OF COST OF ILLNESS AND WILLINGNESS TO PAY ESTIMATES		
Health Effect	**Study**	**Ratio of Willingness to Pay to Cost of Illness Estimates**
Several minor health effects (cough, congestion, headache, etc.)	Berger et al., 1987	3.1 - 78.9
Angina episodes	Chestnut et al., 1988, 1996	2.9 - 8.0
Asthma	Rowe and Chestnut, 1985	3.2 - 9.8
Unspecified effects of ozone	Dickie and Gerking, 1991	1.9 - 4.2
Childhood exposure to lead	Agee and Crocker, 1996	2.1 - 20.0
Chronic bronchitis	U.S. EPA, 1997	3.4 - 6.3
For more information on these studies as well as full citations, see: U.S. Environmental Protection Agency, *Handbook for Noncancer Health Effects Valuation (Draft)*, September 30, 1998.		

The ratios reported in Table VII-7 cover a broad range, suggesting that the relationship between cost of illness and willingness to pay values varies greatly depending on the health effect of concern and the study methodology. Therefore we do not apply these ratios when considering the extent to which cost of illness estimates may understate the value of nonfatal cancer risks averted by the radionuclides rule. However, as discussed in the limitations section of this chapter, these ratios indicate that the use of cost of illness estimates may substantially understate the value of related benefits.

2. Limitations in the Estimation of Monetized Benefits

a. Latency Periods

The benefits calculated in this work are assumed to begin to accrue on the effective date of the rule and are based on a calculation referred to as the ``value of a statistical life" (VSL). For benefits-costs analyses supporting drinking water regulations, some stakeholders have argued that the Agency should consider an assumed time lag or latency period in these calculations. Latency refers to the difference between the time of initial exposure to environmental carcinogens and the onset of any resulting cancer. Use of such an approach might reduce significantly the present value estimate.

The BEIR VI model and U.S. vital statistics, on which the estimate of cancers avoided is based, imply a range of latency periods between exposure to radionuclides and increased probability of cancer death. For some cancer types, there is a significant spread in the credible latency period, ranging from a few years to decades. In addition, there is uncertainty which type of cancer will be induced from a given exposure to a given radionuclide. For these reasons, it is intrinsically difficult to model latency periods. Furthermore, the relationship between cancer latency, benefits

discounting, and other factors is sufficiently complicated to warrant caution in application. EPA is currently working with the Science Advisory Board to come up with an acceptable approach to this problem.

b. Other Factors Affecting the Estimation of Benefits

In fact, cancer latency is only one of a number of adjustments or factors that are related to an evaluation of potential benefits associated with these options, how those benefits are calculated, and when those economic benefits occur. Other factors which may influence the estimate of economic benefits associated with avoided cancer fatalities include (1) A possible ``cancer premium" (i.e., the additional value or sum that people may be willing to pay to avoid the experiences of dread, pain and suffering, and diminished quality of life associated with cancer-related illness and ultimate fatality); (2) the willingness of people to pay more over time to avoid mortality risk as their income rises; (3) a possible premium for accepting involuntary risks as opposed to voluntary assumed risks; (4) the greater or lesser risk aversion of the general population compared to the workers in the wage-risk valuation studies; (5) ``altruism" or the willingness of people to pay more to reduce risk in other sectors of the population; and (6) a consideration of health status and life years remaining at the time of premature mortality. Most of these factors may significantly increase the present value estimate. EPA therefore believes that latency adjustments should be considered simultaneously with accounting for these other factors. It is unclear how long it will take to resolved these issues.

In summary, there is currently neither a clear consensus among economists about how to simultaneously analyze each of these adjustments, nor is there adequate empirical data to support definitive quantitative estimates for these potentially significant adjustment factors. As a result, the primary estimates of economic benefits presented in the analysis of these options rely on the unadjusted $5.9 million estimate. However, EPA solicits comment on whether and how to conduct these potential adjustments to economic benefits estimates together with any rationale or supporting data commenters wish to offer. As mentioned, EPA is currently working the Science Advisory Board (SAB) to conduct a review of these benefits transfer issues associated with economic valuation of adjustments in mortality risks. In its analysis of the final rule, EPA will attempt to develop and present an analysis and estimate of the latency structure and associated benefits transfer issues outlined previously consistent with the recommendations of the SAB and subject to resolution of any technical limitations of the data and models.

E. Estimation of Costs of Compliance

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1. Unit Compliance Costs

Estimated ranges of small and large system total water production costs ($/kgal) for the treatment technologies modeled are summarized in Table VII-8, "Central-Tendency Model Production Costs for Removal of Radionuclides from Community Water Systems", which include estimated disposal costs. A discussion of the assumptions about water treatment residuals disposal can be found in Section V and Appendix V in today's notice. A comparison of the modeled costs to total production costs from case studies, shown in the Table VII-9 ("Case Studies: Production

Costs for Removal of Radionuclides from Community Water Systems"), suggests that EPA's unit cost estimates are reasonable. Figure VII-2 graphically displays the production costs for these case studies. Note that these unit costs are "central-tendency" estimates, which means that some "real-world costs" will be higher, others lower, but that, on average, the central-tendency estimates should be accurate. Also note that the case study sample sizes are too small to compare the case study averages to our central-tendency costs, but that the case study values bracket the central-tendency values, as expected. Figure VIII-3 graphically displays production costs for case studies of systems using blending, purchased water, and new wells. For more details about comparisons of modeled costs to costs documented in case studies, including breakdowns of capital and operations & maintenance costs, see EPA 1999b.

Table VII-8

CENTRAL-TENDENCY MODEL PRODUCTION COSTS FOR REMOVAL OF RADIONUCLIDES FROM COMMUNITY WATER SYSTEMS
(per thousand gallons treated)

	30 Percent Removal Efficiency		**80 Percent Removal Efficiency**		**Point-of-Use Devices (systems serving 25-500 persons)**
Treatment to Remove Gross Alpha, Radium, and Uranium from Ground Water					
	Small Systems	**Large Systems**	**Small Systems**	**Large Systems**	
Ion Exchange/ Softening	$0.37 - $1.78	$0.21 - $0.28	$0.78 - $2.69	$0.54 - $0.66	$2.26 - $2.63
Greensand Filtration	$0.43 - $2.92	NA	NA	NA	NA
Treatment to Remove Uranium from Surface Water					
Anion Exchange	$0.30 - $1.47	$0.24	$0.68 - $2.22	$0.66	$2.26 - $2.63
Enhanced Coagulation/Filtration	$0.28 - $5.24	$0.18 - $0.25	$0.28 - $5.24	$0.18 - $0.25	NA

Notes:
NA means "not applicable."
Source:
EPA 1999a.

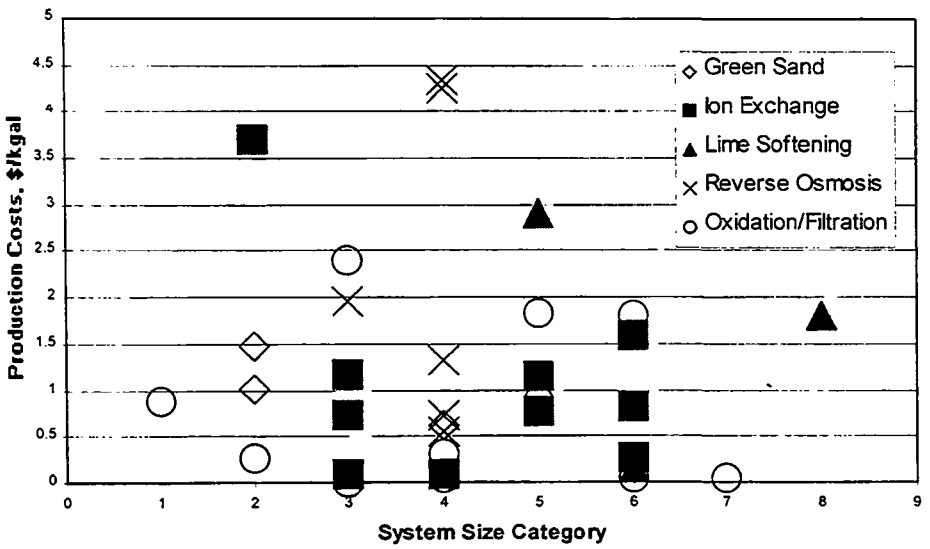
Table VII-9

CASE STUDIES: PRODUCTION COSTS FOR REMOVAL OF RADIONUCLIDES FROM COMMUNITY WATER SYSTEMS
(Dollars per thousand gallons treated)

Treatment to Remove Radium from Ground Water [1]				
	Small Systems[2]		Large Systems[3]	
	Range	Average (No. of Studies)	Range	Average (No. of Studies)
Cation Exchange	$0.08 - $3.69	$1.10 (7)	$0.27 - $1.58	$0.89 (3)
Lime Softening	$2.91	NA (1)	$0.15 - $1.80	$0.97 (2)
Reverse Osmosis	$0.54 - $4.34	$2.19 (6)	NA	NA
Greensand Filtration	$0.63 - $1.47	$1.03 (3)	NA	NA
Other Oxidation/Filtration[4]	$0.01 $2.40	$0.82 (9)	$0.04 - $1.83	$0.63 (5)

Notes:
1. Data source is EPA 1998a, "Actual Costs of Compliance with the Safe Drinking Water Act Standard for Radium-226 and Radium-228", otherwise stated otherwise.
2. Small systems are defined as those serving 10,000 persons or fewer.
3. Large systems are defined as those serving greater than 10,000 persons.
4. Data source is EPA 1998b.

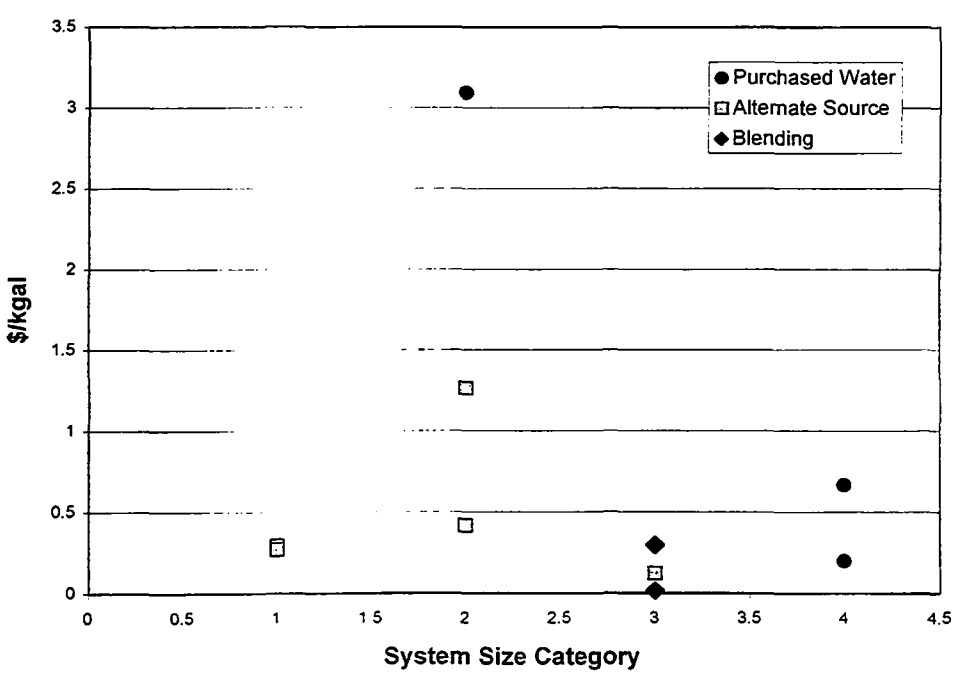
Figure VII-2. Treatment Case Studies, Total Production Costs in Dollars Per Thousand Gallons



2. Characteristics of Community Water Systems

Tables VII-10, VII-11, and VII-12 provide the detailed data on the water system characteristics used in the economic analysis. These tables are based on data from the "Drinking Water Baseline Handbook" (EPA 1999c), which is a compilation of EPA's best estimates of drinking water system characteristics necessary for modeling. In particular, the numbers and populations of community water systems are based on validated 1997 data from EPA's Safe Drinking Water Information System (SDWIS/FED; information can be obtained on-line at "http://www.epa.gov/OGWDW/sdwisfed/sdwis.htm"). The SDWIS database classifies water systems according to type of source water (surface versus ground), ownership (public versus private), and whether or not they purchase water from another system. In our analyses, we included community water systems classified as "purchased water systems", but excluded systems classified as "other". The inclusion of purchased water systems is a conservative assumption, since some double-counting would be expected to result (the water seller and buyer would both be in violation, but, in most cases, only the seller would have to take a compliance action). We count them because the "population served" data for all types include only "retail populations", with the result that the customers served by a purchased water system would be mistakenly classified as a "non-retail" population and would be excluded from the regulatory impact analysis. By including purchased water systems, we potentially over-estimate compliance costs by over-counting the numbers of systems that have to take a compliance action; in reality, while the system purchasing water may

Figure VII-3. Non-Treatment Production Costs, Case Studies



have a water rate increase due to the compliance action taken by the system selling water, the per system and per customer costs for both systems would tend to be smaller than the costs that would be incurred if both systems had to independently take compliance actions. At the present time, EPA does not have the necessary information to take this into consideration, so the conservative simplifying assumption that purchased water systems would behave as non-purchased water systems was used. By including purchased water systems in this way, we avoid the larger error of ignoring the benefits that would accrue to the relatively large impacted residential populations currently labeled as "non-retail populations" because of their SDWIS classification status as "purchased water systems". Regarding the systems classified as "other", based on recent work done to validate the data found in SDWIS, these systems were determined to be most likely inactive. The numbers of systems classified as "other" comprise a small number of the total number of systems, so any resulting errors due their exclusion would be expected to be inconsequential. We also excluded systems reported in SDWIS as regularly serving fewer than 25 persons since they do not meet the definition of public water systems. Community water systems serving more than one million persons are assessed separately in this analysis, and hence are also excluded from this part of the analysis.

Table VII-10

COMMUNITY WATER SYSTEMS: NUMBER OF SYSTEMS
(1997 SDWIS data)

	Population Categories								
System Type	**25-100**	**101-500**	**501-1,000**	**1,001-3,300**	**3,301-10,000**	**10,001-50,000**	**50,001-100,000**	**100,001-1,000,000**	**Total**
Ground Water	13,848	14,654	4,645	5,674	2,472	1,279	139	70	42,781
Public	1,202	4,104	2,574	3,792	1,916	997	113	52	14,750
Private	12,361	9,776	1,705	1,531	459	243	24	14	26,113
Purchased-public	114	427	265	272	84	36	1	4	1,203
Purchased-private	171	347	101	79	13	3	1	0	715
Surface Water	942	1,967	1,167	2,435	1,821	1,528	268	247	10,375
Public	151	385	331	928	882	810	146	161	3,794
Private	307	389	111	211	107	113	33	39	1,310
Purchased-public	185	687	511	1,015	720	560	86	40	3,804
Purchased-private	299	506	214	281	112	45	3	7	1,467
TOTAL	**14,790**	**16,621**	**5,812**	**8,109**	**4,293**	**2,769**	**403**	**317**	**53,156**

Table VII-11

COMMUNITY WATER SYSTEMS: AVERAGE NUMBER OF PERSONS SERVED
(1997 SDWIS data)

	Population Categories							
System Type	**25-100**	**101-500**	**501-1,000**	**1,001-3,300**	**3,301-10,000**	**10,001-50,000**	**50,001-100,000**	**100,001-1,000,000**
Ground Water	61	249	737	1,858	5,739	21,168	67,661	225,473
Public	65	290	745	1,885	5,758	20,875	67,543	213,794
Private	60	230	721	1,805	5,669	22,562	67,670	297,449
Purchased-public	71	282	741	1,809	5,609	20,076	96,000	125,381
Purchased-private	64	271	772	1,775	6,317	18,654	52,500	0
Surface Water	62	283	751	1,982	5,964	22,656	68,441	247,380
Public	55	300	769	2,064	6,012	23,080	69,224	257,483
Private	59	259	770	1,975	6,182	25,438	67,985	277,442
Purchased-public	71	297	745	1,935	5,928	21,712	67,082	193,330
Purchased-private	64	270	722	1,879	5,603	19,776	74,293	156,384

Table VII-12

Numbers of Wells and Entry Points for Typical Systems[1]

	System Size Category							
	25-100	101-500	501- 1,000	1,001-3,301	3,301-10,000	10,001-50,000	50,001-100,000	100,001 - 1,000,000
Average Number of Wells per System (95% Confidence Interval)	1.5 (1.3 - 1.7)	2.0 (1.8 - 2.2)	2.3 (2.1 - 2.5)	3.1 (2.8 - 3.4)	4.6 (3.5 - 5.7)	9.8 (8.0 - 11.6)	16.1 (13.9 - 18.3)	49.9 (48.5 - 51.3)
Average Number of Entry Points per System (95% Confidence Interval)	1.2 (1.1 - 1.3)	1.3 (1.2 - 1.5)	1.5 (1.4 - 1.7)	1.8 (1.6 - 2.0)	2.3 (2.0 - 2.6)	3.9 (3.3 - 4.6)	6.4 (4.6 - 8.6)	9.2 (6.1 - 12.6)

1 Source USEPA 1999c

3. Predicted Compliance Actions (Decision Trees)

Tables VII-13 and VII-14 present the decision trees used in the cost analysis. Decision trees are arrays of estimated probabilities that typical systems will choose particular compliance actions. These decision trees were developed for community water systems as functions of system size, removal rate, and treatment technology or alternate source. Table VII-13 presents the decision tree for removal of gross alpha, combined radium, and uranium in ground water systems; Tables VII-14 present the decision tree for removal of uranium from surface water systems. More details on the decision trees can be found in the background documentation (EPA 1999a).

Table VII-13
DECISION TREE FOR COMBINED RADIUM, GROSS ALPHA, AND URANIUM[1] (Ground Water Systems)

Decision Tree for Systems Requiring 80% up to Max Removal

Technology	Water Softening[2]/Iron Removal	Greens and Filtration	Point-of-Use Reverse Osmosis	Point-of-Use Cation Exchange	Regionalization/Blending/Other	Alternative Source	Totals
Population Size Category							
1 (25-100)	56%	0%	5%	5%	17%	17%	100%
2 (101-500)	56%	0%	5%	5%	17%	17%	100%
3 (501-1,000)	66%	0%	0%	0%	17%	17%	100%
4 (1,001-3,300)	66%	0%	0%	0%	17%	17%	100%
5 (3,301-10,000)	66%	0%	0%	0%	17%	17%	100%
6 (10,001-50,000)	66%	0%	0%	0%	17%	17%	100%
7 (50,001-100,000)	66%	0%	0%	0%	17%	17%	100%
8 (100,001- 1 million)	66%	0%	0%	0%	17%	17%	100%

Decision Tree for Systems Requiring 50% Removal

Technology	Water Softening/Iron Removal	Greens and Filtration	Point-of-Use Reverse Osmosis	Point-of-Use Cation Exchange	Regionalization/Blending/Other	Alternative Source	Totals
Population Size Category							
1 (25-100)	46% (56%)	10% (0%)	5%	5%	17%	17%	100%
2 (101-500)	46% (56%)	10% (0%)	5%	5%	17%	17%	100%
3 (501-1,000)	46% (56%)	10% (0%)	0%	0%	17%	17%	100%
4 (1,001-3,300)	46% (56%)	10% (0%)	0%	0%	17%	17%	100%
5 (3,301-10,000)	66%	0%	0%	0%	17%	17%	100%
6 (10,001-50,000)	66%	0%	0%	0%	17%	17%	100%
7 (50,001-100,000)	66%	0%	0%	0%	17%	17%	100%
8 (100,001- 1 million)	66%	0%	0%	0%	17%	17%	100%

Decision Tree for Systems Requiring 30% Removal

Technology	Water Softening/Iron Removal	Greens and Filtration	Point-of-Use Reverse Osmosis	Point-of-Use Cation Exchange	Regionalization/Blending/Other	Alternative Source	Totals
Population Size Category							
1 (25-100)	46% (56%)	10% (0%)	5%	5%	17%	17%	100%
2 (101-500)	46% (56%)	10% (0%)	5%	5%	17%	17%	100%
3 (501-1,000)	36% (66%)	20% (0%)	0%	0%	17%	17%	100%
4 (1,001-3,300)	36% (66%)	20% (0%)	0%	0%	17%	17%	100%
5 (3,301-10,000)	66%	0%	0%	0%	17%	17%	100%
6 (10,001-50,000)	66%	0%	0%	0%	17%	17%	100%
7 (50,001-100,000)	66%	0%	0%	0%	17%	17%	100%
8 (100,001- 1 million)	66%	0%	0%	0%	17%	17%	100%

Notes: 1) When the uranium decision tree differs, its value is shown in parentheses 2) Water Softening refers to ion exchange softening, lime softening, membrane softening, etc.

Table VII-14

DECISION TREE FOR URANIUM (Surface Water Systems)

Decision Tree for Systems, All Removal Requirements

Technology	Water Softening[1]/Anion Exchange	Enhanced Coagulation/ Filtration	Point-of-Use Reverse Osmosis	Point-of-Use IX or AA[2]	Regionalization/ Blending/Other	Alternative Source	Totals
Size Category							
1 (25-100)	51%	5%	5%	5%	17%	17%	100%
2 (101-500)	51%	5%	5%	5%	17%	17%	100%
3 (501-1,000)	16%	50%	0%	0%	17%	17%	100%
4 (1,001-3,300)	10%	56%	0%	0%	17%	17%	100%
5 (3,301-10,000)	10%	56%	0%	0%	17%	17%	100%
6 (10,001-50,000)	0%	66%	0%	0%	17%	17%	100%
7 (50,001-100,000)	0%	66%	0%	0%	17%	17%	100%
8 (100,001- 1 million)	0%	66%	0%	0%	17%	17%	100%

Notes: 1) Water Softening refers to ion exchange softening, lime softening, membrane softening, etc. 2) IX = ion exhange; AA = activated alumina

4. Compliance Actions: Treatment Installation Versus Other Actions

Recent work (ASDWA/EPA 1999) recently reviewed the actions that water systems have taken to come into compliance with the MCLs for combined radium, nitrate and nitrite, and atrazine. These comprehensive analyses indicate that most water systems choose compliance options other than treatment. The most common of these options include modifications and/or additions to the present treatment system, blending with less contaminated water (i.e., water below the MCL), adding new wells for blending or replacement of contaminated wells, purchasing water from other water systems, and discontinuing the use of contaminated wells when they are not necessary to meet water demand. Table VII-15 presents the frequency of these alternative compliance actions, based on a preliminary analysis of the analytic results.

Table VII-15

ACTUAL COMPLIANCE ACTIONS FROM RETROSPECTIVE STUDIES OF GROUND WATER AND NON-MICROBIAL SURFACE WATER VIOLATIONS

Contaminant	Number of Systems	Compliance Action and Frequency of Use (percent of systems)					
		Installed Treatment	Modified Existing Operations	Blended	Added New Well(s)	Purchased Water	Discontinued Use of Contaminated Well
Nitrate/ nitrite/ atrazine[1]	208	23.9%	18.2%	13.6%	27.8%	13.6%	2.8%
Radium[2]	76	27.6%	none	3.9%	10.5%	51.3%	6.6%

1. Results for the States of OH, SD, FL, MO, CT, CA, IL, WI, MN, NY, MD, OR, PA, and IN.
2. Results are for the State of Illinois, submitted by U. S. Environmental Protection Agency, Region 5.

These results suggest that the decision trees used in this analysis may overstate the extent to which systems will choose to install treatment to comply with the regulatory options for radionuclides. Since compliance costs for non-treatment options are expected to lower, on average, than compliance costs for treatment options, EPA believes that the current decision tree would tend to inflate the estimate of compliance costs, other factors being equal.

5. Summary of Compliance Costs

Table VII-16 shows the breakdown of compliance costs into total capital, annualized capital, monitoring, and operations and maintenance annual costs (O&M) using the system impact estimates based on the "direct proportions approach". Table VII-17 shows the same information for system impact estimates using the "lognormal model approach".

Table VII-16

BREAKDOWN OF TREATMENT AND MONITORING COSTS: DIRECT PROPORTIONS APPROACH
(community water systems)

Regulatory Option		Total Capital Costs	Annualized Capital Costs	Annual O&M Costs	Annual Monitoring Costs	Total Annual Costs
Gross alpha loophole (MCL = 15 pCi/L)		$ 4,720,000	$ 440,000	$ 880,000	$ 1,160,000	$ 2,480,000
Combined radium loophole (MCL = 5 pCi/L)		$ 96,220,000	$ 9,040,000	$ 12,380,000	$ 170,000	$ 21,590,000
Gross alpha MCL = 10 pCi/L, net of radium-226		$ 282,440,000	$ 26,550,000	$ 36,170,000	---	$ 62,720,000
Radium-228 limited at 3 pCi/L, within combined radium MCL of 5 pCi/L		$ 184,480,000	$ 17,340,000	$ 23,320,000	---	$ 40,660,000
Uranium MCL = 20 pCi/L (20 μg/L)	Ground Water	$ 117,510,000	$ 11,050,000	$ 15,310,000	$2,890,000	$ 29,240,000
	Surface Water	$ 110,000	$ 10,000	$ 20,000	$ 2,330,000	$ 2,370,000
Uranium MCL = 40 pCi/L (40 μg/L)	Ground Water	$ 6,840,000	$ 640,000	$ 1,140,000	$2,580,000	$ 4,370,000
	Surface Water	---	---	---	$ 2,310,000	$ 2,310,000
Uranium MCL = 80 pCi/L (80 μg/L)	Ground Water	$ 800,000	$ 75,000	$ 150,000	$ 2,450,000	$ 2,670,000
	Surface Water	---	---	---	$ 2,310,000	$ 2,310,000

Notes:
1. Detail may not add to total due to rounding; estimates are not adjusted for double-counting of systems out of compliance with more than one option..

Table VII-17

BREAKDOWN OF TREATMENT AND MONITORING COSTS: LOGNORMAL DISTRIBUTION APPROACH
(community water systems)

Regulatory Option		Total Capital Costs	Total Annualized Capital Costs	Annual Operations & Maintenance Costs	Annual Monitoring Costs	Total Annual Costs
Eliminate gross alpha loophole (MCL = 15 pCi/L)		$ 145,990,000	$ 13,720,000	$ 19,610,000	$ 1,180,000	$ 34,510,000
Eliminate combined radium loophole (MCL = 5 pCi/L)		$ 166,150,000	$ 15,620,000	$ 23,020,000	$ 170,000	$ 38,810,000
Gross alpha MCL = 10 pCi/L, net of radium-226		$ 314,890,000	$ 29,600,000	$ 42,030,000	—	$ 71,630,000
Radium-228 limited to 3 pCi/L within combined radium MCL of 5 pCi/L		$ 165,580,000	$ 15,560,000	$ 22,300,000	—	$ 37,860,000
Uranium MCL = 20 pCi/L (20 μg/L)	Ground Water	$ 608,780,000	$ 57,230,000	$ 87,710,000	$ 3,020,000	$ 147,950,000
	Surface Water	$ 16,250,000	$ 1,530,000	$ 5,080,000	$ 2,460,000	$ 9,080,000
Uranium MCL = 40 pCi/L (40 μg/L)	Ground Water	$ 254,670,000	$ 23,940,000	$ 36,630,000	$ 2,680,000	$ 63,260,000
	Surface Water	$ 5,850,000	$ 550,000	$ 1,830,000	$ 2,370,000	$ 4,740,000
Uranium MCL = 80 pCi/L (80 μg/L)	Ground Water	$ 101,460,000	$ 9,540,000	$ 14,740,000	$ 2,530,000	$ 26,800,000
	Surface Water	$ 1,950,000	$ 180,000	$ 610,000	$ 2,330,000	$ 3,120,000

Notes:
1. Detail may not add to total due to rounding.
Source:
Based on data provided by William Labiosa, EPA/OGWDW November 22 and 23, 1999.

Tables VII-18 through VII- 24 present the detailed estimated numbers of impacted systems and cost results for each system size category. Each table indicates the total national annual operations and maintenance costs, annualized capital expenditures, annual monitoring costs, and total annual costs by system size category. The number of systems affected nationally by each regulatory option is also reported. In addition, the tables presenting the results for the uranium MCLs contain separate estimates for surface water and ground water systems. Each table presents the results obtained through both the direct proportion and lognormal distribution approaches.

Table VII-18 contains the estimates for closing the gross alpha monitoring deficiency; Table VII-19 presents the estimates for closing the combined radium deficiency; Tables VII-20 and VII-21 contain the estimates for changing the MCL for gross alpha to 10 pCi/L and established an MCL of 3 pCi/L for Ra-228, respectively; and Tables VII-22 through VII-22 present the estimates for establishing uranium MCLs of 20, 40, and 80 ug/L (pCi/L[20]), respectively. Only Tables VII-19, VII-22, VII-23, and VII-24 refer to options being considered for finalization. The others are not being considered for reasons discussed in the NODA.

[20] As, previously noted, the analysis assumed an activity-to-mass ratio of 1:1 for uranium.

Table VII-18

NATIONAL COSTS DUE TO CLOSING MONITORING LOOPHOLE FOR GROSS ALPHA
(ground water systems only)

System Size Class	Directly Proportional					Lognormally Distributed				
	Number of Affected Systems	Annual Capital Costs	Annual O&M Costs	Annual Monitoring Costs	Total Annual Costs	Number of Affected Systems	Annual Capital Costs	Annual O&M Costs	Annual Monitoring Costs	Total Annual Costs
25-100	102	$ 115,817	$ 235,136	$ 299,850	$ 650,803	84	$ 96,593	$ 191,916	$ 295,072	$ 583,581
101-500	108	$ 327,806	$ 647,056	$ 342,500	$ 1,317,363	89	$ 277,667	$ 538,345	$ 337,044	$ 1,153,056
501-1,000	0	$ 0	$ 0	$ 117,165	$ 117,165	25	$ 404,049	$ 425,307	$ 125,374	$ 954,730
1,001-3,300	0	$ 0	$ 0	$ 171,699	$ 171,699	31	$ 1,080,659	$ 1,149,991	$ 183,729	$ 2,414,380
3,301-10,000	0	$ 0	$ 0	$ 101,934	$ 101,934	13	$ 1,559,203	$ 1,689,217	$ 109,075	$ 3,357,496
10,001-50,000	0	$ 0	$0	$ 90,538	$ 90,538	7	$ 3,986,067	$ 6,129,901	$ 96,881	$ 10,212,849
50,001-100,000	0	$ 0	$ 0	$ 22,217	$ 22,217	1	$ 2,336,806	$ 3,855,565	$ 23,774	$ 6,216,145
100,001-1,000,000	0	$ 0	$ 0	$ 11,229	$11,229	0*	$ 3,981,627	$ 5,627,407	$ 12,016	$ 9,621,050
TOTAL	211	$ 443,624	$ 882,192	$ 1,157,132	$2,482,948	250	$ 13,722,671	$ 19,607,649	$ 1,182,966	$ 34,513,286

* Model predicts an expected value of less than 0.5 systems affected nationally.

Notes:

1) Results are not adjusted for double-counting of systems out-of-compliance for both the combined radium and gross alpha loopholes.

2) Detail may not add to total due to rounding.

Table VII-19

NATIONAL COSTS DUE TO CLOSING MONITORING LOOPHOLE FOR COMBINED RADIUM
(ground water systems only)

System Size Class	Directly Proportional					Lognormally Distributed				
	Number of Affected Systems	Annual Capital Costs	Annual O&M Costs	Annual Monitoring Costs	Total Annual Costs	Number of Affected Systems	Annual Capital Costs	Annual O&M Costs	Annual Monitoring Costs	Total Annual Costs
25-100	103	$ 119,132	$ 232,254	$ 42,892	$ 394,278	118	$ 136,874	$ 269,135	$ 42,892	$ 448,900
101-500	109	$ 346,890	$ 664,221	$ 48,993	$ 1,060,104	125	$ 478,305	$ 817,129	$ 48,993	$ 1,344,426
501-1,000	20	$ 279,049	$ 283,106	$ 16,945	$ 579,100	24	$ 450,407	$ 489,490	$ 16,945	$ 956,842
1,001-3,300	24	$ 744,502	$ 755,544	$ 24,832	$ 1,524,878	30	$ 1,207,169	$ 1,338,526	$ 24,832	$ 2,570,527
3,301-10,000	11	$ 1,107,357	$ 1,116,554	$ 14,742	$ 2,238,653	13	$ 1,692,693	$ 1,953,577	$ 14,742	$ 3,661,011
10,001-50,000	5	$ 2,310,303	$ 3,249,553	$ 13,094	$ 5,572,951	7	$ 4,297,618	$ 6,931,454	$ 13,094	$ 11,242,165
50,001-100,000	1	$ 1,625,459	$ 2,516,964	$ 3,213	$ 4,145,636	1	$ 2,585,268	$ 4,498,729	$ 3,213	$ 7,087,211
100,001-1,000,000	0*	$ 2,511,998	$ 3,558,135	$ 1,624	$ 6,071,757	0*	$ 4,770,023	$ 6,726,667	$ 1,624	$ 11,498,313
TOTAL	272	$ 9,044,692	$ 12,376,331	$ 166,333	$ 21,587,356	317	$ 15,618,356	$ 23,024,706	$ 166,333	$ 38,809,396

* Model predicts an expected value of less than 0.5 systems affected nationally.

Notes:

1) *Results are not adjusted for double-counting of systems out-of-compliance for both the combined radium and gross alpha loopholes.*

2) Detail may not add to total due to rounding.

Table VII-20

NATIONAL COSTS DUE TO CHANGING THE GROSS ALPHA MCL TO 10 pCi/L
(ground water systems only)

System Size Class	Directly Proportional					Lognormally Distributed				
	Number of Affected Systems	Annual Capital Costs	Annual O&M Costs	Annual Monitoring Costs	Total Annual Costs	Number of Affected Systems	Annual Capital Costs	Annual O&M Costs	Annual Monitoring Costs	Total Annual Costs
25-100	205	$ 214,527	$ 460,811	$ 0	$ 675,338	161	$ 184,181	$ 366,908	$ 0	$ 551,089
101-500	217	$ 579,097	$ 1,243,043	$ 0	$ 1,822,139	170	$ 529,000	$ 1,028,114	$ 0	$ 1,557,114
501-1,000	60	$ 837,796	$ 849,979	$ 0	$ 1,687,775	56	$ 900,246	$ 947,348	$ 0	$ 1,847,594
1,001-3,300	73	$ 2,235,242	$ 2,268,394	$ 0	$ 4,503,636	68	$ 2,408,113	$ 2,561,929	$ 0	$ 4,970,042
3,301-10,000	32	$ 3,324,653	$ 3,352,263	$ 0	$ 6,676,916	30	$ 3,474,534	$ 3,764,390	$ 0	$ 7,238,924
10,001-50,000	16	$ 6,936,296	$ 9,756,235	$ 0	$ 16,692,531	15	$ 8,018,156	$ 12,216,672	$ 0	$ 20,234,828
50,001-100,000	2	$ 4,880,167	$ 7,556,758	$ 0	$ 12,436,925	2	$ 5,207,852	$ 8,594,268	$ 0	$ 13,802,120
100,001-1,000,000	1	$ 7,541,850	$ 10,682,700	$ 0	$ 18,224,550	1	$ 8,877,316	$ 12,546,866	$ 0	$ 21,424,181
TOTAL	606	$ 26,549,628	$ 36,170,183	$ 0	$ 62,719,810	502	$ 29,599,399	$ 42,026,495	$ 0	$ 71,625,893

Notes:
1) Results are based on full compliance with existing MCLs, after closure of the monitoring loopholes.
2) Results are not adjusted for double-counting of systems out-of-compliance with both the revised gross alpha and combined radium MCLs.
3) Detail may not add to total due to rounding.

Table VII-21

NATIONAL COSTS DUE TO ESTABLISHING AN RA-228 MCL OF 3 pCi/L
(ground water systems only)

System Size Class	Directly Proportional					Lognormally Distributed				
	Number of Affected Systems	Annual Capital Costs	Annual O&M Costs	Annual Monitoring Costs	Total Annual Costs	Number of Affected Systems	Annual Capital Costs	Annual O&M Costs	Annual Monitoring Costs	Total Annual Costs
25-100	41	$ 47,475	$ 94,705	$ 0	$ 142,180	61	$ 68,135	$ 138,785	$ 0	$ 206,920
101-500	43	$ 136,394	$ 262,389	$ 0	$ 398,783	65	$ 192,164	$ 384,233	$ 0	$ 576,397
501-1,000	40	$ 558,004	$ 566,117	$ 0	$ 1,124,121	28	$ 465,872	$ 494,913	$ 0	$ 960,785
1,001-3,300	49	$ 1,488,937	$ 1,511,032	$ 0	$ 2,999,969	34	$ 1,246,438	$ 1,342,505	$ 0	$ 2,588,942
3,301-10,000	21	$ 2,214,668	$ 2,233,126	$ 0	$ 4,447,794	15	$ 1,784,865	$ 1,963,659	$ 0	$ 3,748,524
10,001-50,000	11	$ 4,620,627	$ 6,499,124	$ 0	$ 11,119,750	8	$ 4,468,397	$ 6,925,236	$ 0	$ 11,393,633
50,001-100,000	1	$ 3,250,912	$ 5,033,911	$ 0	$ 8,284,823	1	$ 2,685,497	$ 4,481,476	$ 0	$ 7,166,973
100,001-1,000,000	1	$ 5,023,993	$ 7,116,265	$ 0	$ 12,140,258	0*	$ 4,653,286	$ 6,568,791	$ 0	$ 11,222,077
TOTAL	207	$ 17,341,009	$ 23,316,669	$ 0	$ 40,657,678	212	$ 15,564,653	$ 22,299,598	$ 0	$ 37,864,251

* Model predicts an expected value of less than 0.5 systems affected nationally.

Notes:

1) Results are based on full compliance with existing MCLs, after closure of the monitoring loopholes.

2) Results are not adjusted for double-counting of systems out-of-compliance with both the revised gross alpha and combined radium MCLs.

3) Detail may not add to total due to rounding.

Table VII-22

NATIONAL COSTS DUE TO ESTABLISHING URANIUM MCL AT 20 pCi/L

System Size Class	Directly Proportional					Lognormally Distributed				
	Number of Affected Systems	Annual Capital Costs	Annual O&M Costs	Annual Monitoring Costs	Total Annual Costs	Number of Affected Systems	Annual Capital Costs	Annual O&M Costs	Annual Monitoring Costs	Total Annual Costs
Ground Water Systems										
25-100	369	$ 511,761	$ 787,528	$ 793,578	$ 2,092,866	324	$ 457,562	$ 696,933	$ 770,307	$ 1,924,803
101-500	391	$ 1,588.823	$ 2.535,141	$ 906,457	$ 5,030,420	342	$ 1,441,744	$ 2,265,563	$ 879,877	$ 4,587,183
501-1,000	20	$ 372,326	$ 407,045	$ 269,974	$ 1,049,345	83	$ 1,948,004	$ 2,555,766	$ 310,710	$ 4,814,480
1,001-3,300	24	$ 960,535	$ 1,056,488	$ 395,632	$ 2,412,655	101	$ 5,175,360	$ 6,901,236	$ 455,328	$ 12,531,924
3,301-10,000	11	$ 1,107,558	$ 1,116,245	$ 234,877	$ 2,458,679	44	$ 6,294,165	$ 8,329,306	$ 270,317	$ 14,893,788
10,001-50,000	5	$ 2,311,986	$ 3,250,932	$ 208,618	$ 5,771,537	23	$ 12,927,409	$ 21,552,612	$ 240,097	$ 34,720,118
50,001-100,000	1	$ 1,624,413	$ 2,513,659	$ 51,194	$ 4,189,266	2	$ 9,785,965	$ 18,195,916	$ 58,918	$ 28,040,799
100,001-1,000,000	0*	$ 2,568,428	$ 3,638,107	$ 25,874	$ 6,232,409	1	$ 19,195,466	$ 27,214,162	$ 29,779	$ 46,439,407
TOTAL	821	$ 11,045,830	$ 15,305,144	$2,886,203	$ 29,237,178	921	$ 57,225,675	$ 87,711,493	$ 3,015,334	$ 147,952,502
Surface Water Systems										
25-100	1	$ 1,522	$ 3,426	$ 587,313	$ 592,261	6	$ 6,839	$ 14,676	$ 621,781	$ 643,296
101-500	3	$ 9,025	$ 18,524	$ 670,853	$ 698,402	12	$ 42,040	$ 81,976	$ 710,224	$ 834,240
501-1,000	0	$ 0	$ 0	$ 244,649	$ 244,649	5	$ 23,998	$ 61,513	$ 257,555	$ 343,066
1,001-3,300	0	$ 0	$ 0	$ 358,519	$ 358,519	10	$ 70,210	$ 204,155	$ 377,432	$ 651,797
3,301-10,000	0	$ 0	$ 0	$ 212,844	$ 212,844	8	$ 167,300	$ 418,145	$ 224,072	$ 809,517
10,001-50,000	0	$ 0	$ 0	$ 189,048	$ 189,048	7	$ 272,317	$ 931,440	$ 199,021	$ 1,402,778
50,001-100,000	0	$ 0	$ 0	$ 46,392	$ 46,392	1	$ 198,281	$ 767,609	$ 48,839	$ 1,014,729
100,001-1,000,000	0	$ 0	$ 0	$ 23,447	$ 23,447	1	$ 746,476	$ 2,604,819	$ 24,684	$ 3,375,979
TOTAL	4	$ 10,547	$ 21,950	$ 2,333,064	$ 2,365,561	50	$ 1,527,461	$ 5,084,334	$ 2,463,608	$ 9,075,403

* Model predicts an expected value of less than 0.5 systems affected nationally.

Notes:

1) Detail may not add to total due to rounding.

Table VII-23

NATIONAL COSTS DUE TO ESTABLISHING URANIUM MCL AT 40 pCi/L

System Size Class	Directly Proportional					Lognormally Distributed				
	Number of Affected Systems	Annual Capital Costs	Annual O&M Costs	Annual Monitoring Costs	Total Annual Costs	Number of Affected Systems	Annual Capital Costs	Annual O&M Costs	Annual Monitoring Costs	Total Annual Costs
Ground Water Systems										
25-100	144	$ 169,806	$ 284,584	$ 678,582	$ 1,132,971	146	$203,541	$ 312,671	$ 679,981	$ 1,196,193
101-500	152	$ 472,833	$ 859,822	$ 775,104	$ 2,107,759	155	$ 634,899	$ 1,009,588	$ 776,703	$ 2,421,190
501-1,000	0	$ 0	$ 0	$ 256,894	$ 256,894	35	$ 803,819	$ 1,022,947	$ 279,375	$ 2,106,140
1,001-3,300	0	$ 0	$ 0	$ 376,463	$ 376,463	42	$ 2,129,518	$ 2,759,639	$ 409,408	$ 5,298,565
3,301-10,000	0	$ 0	$ 0	$ 223,497	$ 223,497	19	$ 2,582,371	$ 3,328,392	$ 243,056	$ 6,153,819
10,001-50,000	0	$ 0	$ 0	$ 198,511	$ 198,511	10	$ 5,995,989	$ 10,091,830	$ 215,883	$ 16,303,702
50,001-100,000	0	$ 0	$ 0	$ 48,713	$ 48,713	1	$ 4,003,003	$ 7,341,777	$ 52,977	$ 11,397,757
100,001-1,000,000	0	$ 0	$ 0	$ 24,621	$ 24,621	1	$ 7,586,253	$ 10,767,174	$ 26,775	$ 18,380,202
TOTAL	296	$ 642,639	$ 1,144,406	$ 2,582,384	$ 4,369,429	408	$ 23,939,393	$ 36,634,017	$ 2,684,158	$ 63,257,568
Surface Water Systems										
25-100	0	$ 0	$ 0	$ 576,529	$ 576,529	2	$ 2,638	$ 5,732	$ 594,109	$ 602,480
101-500	0	$ 0	$ 0	$ 658,535	$ 658,535	5	$ 16,056	$ 31,741	$ 678,616	$ 726,413
501-1,000	0	$ 0	$ 0	$ 244,484	$ 244,484	2	$ 8,489	$ 22,023	$ 249,094	$ 279,606
1,001-3,300	0	$ 0	$ 0	$ 358,277	$ 358,277	4	$ 24,909	$ 73,383	$ 365,034	$ 463,325
3,301-10,000	0	$ 0	$ 0	$ 212,700	$ 212,700	3	$ 59,256	$ 150,221	$ 216,712	$ 426,189
10,001-50,000	0	$ 0	$ 0	$ 188,921	$ 188,921	2	$ 98,922	$ 338,246	$ 192,484	$ 629,651
50,001-100,000	0	$ 0	$ 0	$ 46,360	$ 46,360	0*	$ 71,001	$ 274,223	$ 47,235	$ 392,459
100,001-1,000,000	0	$ 0	$ 0	$ 23,431	$ 23,431	0*	$ 268,808	$ 930,555	$ 23,873	$ 1,223,236
TOTAL	0	$ 0	$ 0	$ 2,309,238	$ 2,309,238	19	$ 550,078	$ 1,826,124	$ 2,367,156	$ 4,743,358

* Model predicts an expected value of less than 0.5 systems affected nationally.

Notes:

1) Detail may not add to total due to rounding.

Table VII-24

NATIONAL COSTS DUE TO ESTABLISHING URANIUM MCL AT 80 pCi/L

System Size Class	Directly Proportional					Lognormally Distributed				
	Number of Affected Systems	Annual Capital Costs	Annual O&M Costs	Annual Monitoring Costs	Total Annual Costs	Number of Affected Systems	Annual Capital Costs	Annual O&M Costs	Annual Monitoring Costs	Total Annual Costs
Ground Water Systems										
25-100	21	$ 21,458	$ 38,563	$ 615,566	$ 675,587	60	$ 82,691	$ 128,132	$ 635,903	$ 846,726
101-500	22	$ 53,832	$ 110,907	$ 703,125	$ 867,863	64	$ 255,116	$ 410,792	$ 726,355	$ 1,392,262
501-1,000	0	$ 0	$ 0	$ 256,690	$ 256,690	13	$ 300,169	$ 372,414	$ 265,214	$ 937,796
1,001-3,300	0	$ 0	$ 0	$ 376,165	$ 376,165	16	$ 794,139	$ 1,004,150	$ 388,656	$ 2,186,945
3,301-10,000	0	$ 0	$ 0	$ 223,320	$ 223,320	7	$ 959,737	$ 1,208,058	$ 230,736	$ 2,398,530
10,001-50,000	0	$ 0	$ 0	$ 198,353	$ 198,353	4	$ 2,877,849	$ 4,985,354	$ 204,940	$ 8,068,142
50,001-100,000	0	$ 0	$ 0	$ 48,675	$ 48,675	0*	$ 1,481,967	$ 2,683,114	$ 50,291	$ 4,215,372
100.001-1.000,000	0	$ 0	$ 0	$ 24,601	$ 24,601	0*	$ 2,785,321	$ 3,944,950	$ 25,418	$ 6,755,689
TOTAL	42	$ 75,290	$ 149,470	$ 2,446,495	$ 2,671,254	165	$ 9,536,987	$ 14,736,962	$ 2,527,514	$ 26,801,462
Surface Water Systems										
25-100	0	$ 0	$ 0	$ 576,289	$ 576,289	1	$ 885	$ 1,994	$ 582,564	$ 585,443
101-500	0	$ 0	$ 0	$ 658,261	$ 658,261	2	$ 5,617	$ 11,224	$ 665,429	$ 682,269
501-1,000	0	$ 0	$ 0	$ 244,391	$ 244,391	1	$ 2,749	$ 7,195	$ 245,877	$ 255,821
1,001-3,300	0	$ 0	$ 0	$ 358,141	$ 358,141	1	$ 8,141	$ 24,182	$ 360,319	$ 392,642
3,301-10,000	0	$ 0	$ 0	$ 212,620	$ 212,620	1	$ 19,185	$ 49,095	$ 213,913	$ 282,193
10,001-50,000	0	$ 0	$ 0	$ 188,849	$ 188,849	1	$ 33,203	$ 113,335	$ 189,998	$ 336,535
50,001-100,000	0	$ 0	$ 0	$ 46,343	$ 46,343	0*	$ 23,049	$ 89,160	$ 46,624	$ 158,834
100,001-1,000,000	0	$ 0	$ 0	$ 23,423	$ 23,423	0*	$ 90,610	$ 315,702	$ 23,565	$ 429,877
TOTAL	0	$ 0	$ 0	$ 2,308,316	$ 2,308,316	6	$ 183,439	$ 611,888	$ 2,328,288	$ 3,123,615

* Model predicts an expected value of less than 0.5 systems affected nationally.

Notes:

1) Detail may not add to total due to rounding.

F. Characteristics of Non-Transient Non-Community Water System

Table VII-18 presents the numbers of systems, populations served, and estimated typical design flows used in the analyses to describe NTNCWSs by Service Area type at the national-level. "Average daily populations" were derived as described in the "Geometries and Characteristics of Public Water Systems" report. "Number of People Served" represents the number of distinct people served by each water system on an annual basis. These values were derived from the SDWIS populations through consideration of customer cycles (# cycles per yr) and worker to customer ratios (worker/pop/day), as described in Table 3.8 in Appendix III. Number of People Served is used in the Benefits analysis to determine impacted population, while average daily populations affect the sizing of water treatment systems and, hence, are used in the cost analysis. Table VII-19 presents the numbers of NTNCWSs served by water system size category.

Table VII-18. Characteristics of NTNCWSs

Service Area Type	# of Systems	Avg. Daily Population	Number of People Served	Typical Design Flow/System (MGD)
Daycare Centers	809	76	76	0.01
Highway Rest Areas	15	407	20433	0.01
Hotels/Models	351	133	8386	0.02
Interstate Carriers	287	123	7999	0.003
Medical Facilities	367	393	7161	0.12
Mobile Home Parks	104	185	243	0.03
Restaurants	418	370	714	0.00
Schools	8414	358	367	0.03
Service Stations	53	230	1527	0.01
Summer Camps	46	146	1132	0.02
Water Wholesalers	266	173	248	0.16
Agricultural Products/Services	368	76	475	0.02
Airparks	101	60	82738	0.003
Construction	99	53	53	0.001
Churches	230	50	50	0.01
Campgrounds/RV Parks	123	160	3459	0.02
Fire Departments	41	98	98	0.02
Federal Parks	20	39	998	0.01
Forest Service	107	42	1075	0.001
Golf and Country Clubs	116	101	416	0.01
Landfills	78	44	44	0.01
Mining	119	113	113	0.01
Amusement Parks	159	418	35576	0.02
Military Bases	95	395	711	0.07
Migrant Labor Camps	33	63	63	0.01
Misc. Recreation Services	259	87	2227	0.002
Nursing Homes	130	107	107	0.04
Office Parks	950	136	191	0.01
Prisons	67	1820	2361	0.53
Retailers (Non-food related)	695	174	728	0.004
Retailers (Food related)	142	322	621	0.01
State Parks	83	165	32970	0.005
Non-Water Utilities	497	170	170	0.01
Manufacturing: Food	768	372	372	0.05
Manufacturing: Non-Food	3356	168	175	0.02
Sum:	19,766	5,805,758	31,970,000	
Weighted Average:		261		0.028

Table VII-19

NON-TRANSIENT NON-COMMUNITY WATER SYSTEMS: NUMBER OF SYSTEMS
(1997 SDWIS data)

	Population Categories								
System Type	25-100	101-500	501-1,000	1,001-3,300	3,301-10,000	10,001-50,000	50,001-100,000	100,001-1,000,000	Total
Ground Water	9,169	6,873	1,912	675	59	11	0	0	18,699
Public	1,704	3,109	1,145	327	21	5	0	0	6,311
Private	7,432	3,731	752	342	33	2	0	0	12,292
Purchased-public	11	16	8	6	5	3	0	0	49
Purchased-private	22	17	7	0	0	1	0	0	47
Surface Water	209	223	77	67	16	4	1	1	598
Public	48	34	11	17	1	0	0	0	111
Private	78	119	47	33	8	0	0	0	285
Purchased-public	14	27	7	6	3	3	1	1	62
Purchased-private	69	43	12	11	4	1	0	0	140
TOTAL	9,378	7,096	1,989	742	75	15	1	1	19,297

G. References

Association of State Drinking Water Administrators and USEPA (ASDWA/EPA). Results from a survey of States regarding compliance actions taken in response to NPDWRS for nitrate and atrazine". October 1999. (ASDWA/EPA 1999)

USEPA. "Uranium in U.S. Surface Water, Ground, and Domestic Waters, Volume 1". Prepared by Oak Ridge National Laboratory. EPA 570/9-81-001. pp. 217 - 222. April 1981.

USEPA. "Preliminary Analysis of Risk Reductions and Costs: Revised National Primary Drinking Water Standards for Radionuclides. Prepared by Industrial Economics, Inc. for the Office of Ground Water and Drinking Water. Draft dated November 1999. (EPA 1999a)

USEPA. "Technologies and Costs for the Removal of Radionuclides from Potable Water Sources". Prepared by International Consultants, Inc. Draft dated April 1999. (EPA 1999b)

USEPA. "Drinking Water Baseline Handbook: First Edition". Draft dated February 24, 1999. (EPA 1999c)