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Review of the Technical Bases of 40 CFR Part 190

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Abstract

The dose limits for emissions from the nuclear fuel cycle were established by the Environmental Protection Agency in 40 CFR Part 190 in 1977. These limits were based on assumptions regarding the growth of nuclear power and the technical capabilities of decontamination systems as well as the then-current knowledge of atmospheric dispersion and the biological effects of ionizing radiation. In the more than thirty years since the adoption of the limits, much has changed with respect to the scale of nuclear energy deployment in the United States and the scientific knowledge associated with modeling health effects from radioactivity release. Sandia National Laboratories conducted a study to examine and understand the methodologies and technical bases of 40 CFR 190 and also to determine if the conclusions of the earlier work would be different today given the current projected growth of nuclear power and the advances in scientific understanding. This report documents the results of that work.

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Acronyms

AEC	Atomic Energy Commission
AFCI	Advanced Fuel Cycle Initiative
AGNS	Allied General Nuclear Services
ALARA	As Low As Reasonably Achievable
BEIR	[National Research Council Committee on the] Biological Effects of Ionizing Radiation
BRC	Below regulatory concern
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act of 1980
CEM	Cost-effectiveness metric
CFR	Code of Federal Regulations
DCF	Dose conversion factor
DF	Decontamination factor
DOE	U.S. Department of Energy
DOT	U.S. Department of Transportation
DDREF	Dose and dose rate effectiveness factor
DREF	Dose rate effectiveness factor
DR	Dose response
EDE	Effective dose equivalent
EIA	U.S. Energy Information Agency
EIS	Environmental impact statement
EPA	U.S. Environmental Protection Agency
FES	Final Environmental Statement
EPRI	Electric Power Research Institute
FGR	Federal Guidance Report
GE	General Electric
Gy	Gray; one joule of energy per kg of absorbing material
ICRP	International Commission on Radiation Protection
INRA	International Nuclear Recycling Alliance
ISCORS	U.S. Interagency Steering Committee on Radiation Standards
LCF	Latent cancer fatality
LNT	Linear No Threshold (theory)
LSS	Life Span Study
LQ	Linear quadratic
MACCS2	MELCOR Accident Consequence Code System, version 2. (MACCS2 is the NRC consequence analysis tool for performing risk assessment involving release of radioactive vapor and aerosols into the atmosphere.)
MEI	Maximum exposed individual
NESHAP	National Emission Standards for Hazardous Air Pollutants
NID	Negligible individual dose
NCRP	National Council on Radiation Protection
NRC	U.S. Nuclear Regulatory Commission
OSHA	U.S. Department of Labor, Occupational Safety and Health Administration

RBE	Relative Biological Effectiveness
TEDE	Total effective dose equivalent
VSL	Value of a Statistical Life, a measure of society's willingness to pay for the control measures and regulations designed to reduce risk and protect human life

1.0 Introduction

Heightened international concern over global climate change and increased appreciation of the relationship of energy to the environment and economic health have increased the interest in nuclear power as an important energy source for the future. The potential contributions of nuclear energy domestically and internationally are subject to considerable debate over concerns about wastes, proliferation, safety, and economics. Several initiatives have addressed these concerns, including an examination of alternatives and/or improvements to the current fuel cycle. A central element in this examination is whether or how alternative or supplemental fuel cycle processes such as separation, recycling, and transmutation could address many of those concerns.

This report examines a part of the regulatory structure in the United States, 40 CFR 190 – "Environmental Radiation Protection Standards for Nuclear Power Operations," which considerably influences nuclear fuel cycle activities. Promulgated in 1977, it was driven by conditions, perceptions, and assumptions applicable at the time. Since then, our understanding of the transport of pollutants and their effects on populations (and our ability to model them) has evolved, including the use of collective dose methodologies. Projections of the number and types of nuclear facilities have changed. Prior assumptions regarding the risk reduction effectiveness of various emission-capture technologies appear to be significantly outdated. New chemical processes and process flows are being developed. Regulatory philosophies and the societal perspectives that drive them (such as risks of global climate change and energy shortages) have also evolved. All these factors motivated the interest in re-examining the technical bases for 40 CFR 190.

While the U.S. has focused on a "once-through" fuel cycle, other nations have proceeded with separations and recycling. There is now a body of foreign experience from which there is much to learn. In addition, from the standpoint of foreign policy and nonproliferation objectives, the ability of the U.S. to influence the further growth and evolution of the nuclear fuel cycle globally is certainly affected by its actions domestically. Lastly, it is noted that the Nuclear Regulatory Commission (NRC) has received letters from vendors expressing interest in building and operating commercial recycling plants in the U.S. The NRC believes that the interests of the U.S. government, industry, and utility entities have grown sufficiently, and that circumstances have changed enough, to warrant an evaluation of the need to update regulations affecting the nuclear fuel cycle. Of particular interest is whether changes since 40 CFR 190 was promulgated might also warrant a similar evaluation by the Environmental Protection Agency (EPA).

Government, academic, and private sector efforts are in part driving the effort to evaluate the technical and economic feasibility of various alternatives and/or improvements to the current U.S. once-through fuel cycle. Study of those alternatives in turn is stimulated by a wide range of interests, as previously discussed. Regulatory considerations will have a major effect on the technical and commercial viability of various fuel-cycle options. Accordingly, it appears appropriate to begin considering and exchanging information among the federal agencies that could inform the evaluation and discussion of alternatives.

40 CFR 190 was the EPA's first regulation of the nuclear fuel cycle and was intended to protect the general public from the adverse health effects of ionizing radiation. The regulation is

important from a regulatory perspective because it sets limits on emissions of selected radionuclides and imposes decontamination system effectiveness goals or requirements that may prove expensive to achieve.

The regulation sets a dose limit for any ionizing radiation and a release limit for selected radionuclides. 40 CFR 190.10(a) sets the dose limit. It limits the annual dose equivalent to any member of the public not to exceed 25 mrem (0.25 mSv) to whole body, 75 mrem (0.75 mSv) to the thyroid, and 25 mrem (0.25 mSv) to any other organ. 40 CFR 190.10(b) sets the release limits for selected radionuclides. It limits the total quantity of radioactive materials entering the general environment per gigawatt-year of electrical energy to less than 50,000 Ci Kr-85, 5 mCi I-129, and 0.5 mCi Pu-239. These isotopic limits were set to control the build-up in the atmosphere of the selected radionuclides.

This report presents the findings of a review of the technical bases of the dose and release limits in 40 CFR 190 and assesses changes in assumptions and technical understanding over the last thirty years that may affect the underlying technical basis for the standard. The report focuses principally on the release limits in 40 CFR 190.10(b) because recycling facilities are expected to be able to meet the dose limits in 40 CFR 190.10(a) based on an evaluation of facilities currently in operation. It is unlikely that there would be an issue meeting the dose limits in the regulations. However, the release limits in 40 CFR 190 10(b) could have significant technical and cost implications, especially for sequestering Kr-85 and I-129. The two issues for Kr-85 sequestration are safety related (specifically the mode of storage) and cost. The issue associated with I-129 sequestration is a technical issue driven by the carry-over of I-129 in the process flow (about 2% of the total I-129), which is difficult to sequester (INRA, 2009).

To understand the basis for the release limits in 40 CFR 190, the background information and technical basis documents published by the EPA were reviewed and analyzed to develop an in-depth understanding of the EPA methodology and its underlying assumptions. Methodologies utilized by the EPA were then re-created and verified. Once the EPA methodologies were understood, a comparative analysis of major changes in technologies, scientific understanding, and current methodologies was conducted to determine if there were major changes in the technical basis that would justify updating the release limits. Finally, a prioritized list of areas needing further study was prepared.

This report presents the findings of the review of the technical bases for 40 CFR 190. Section 2.0 is a synopsis of the EPA technical approach used in the development of the regulation. Section 3.0 examines the technical bases and assesses the current state of knowledge and assumptions. In particular, the following topics are reviewed:

- Radionuclide isotopic source (Section 3.1);
- Transport and dose modeling (Section 3.2);
- Health effects modeling (Section 3.3); and
- Cost effectiveness (Section 3.4).

Section 4.0 presents key observations and concludes with a summary and recommendations for further work.

2.0 Synopsis of 40 CFR 190

2.1. Background

When the EPA was created in 1970, the responsibility for establishing radiation standards was transferred to the EPA from the Atomic Energy Commission (AEC). This responsibility included establishing generally applicable environmental standards to limit man-made radioactive materials in the environment and developing national radiation protection guidance for use by federal and state agencies in their radiation protection programs and regulations. Initially, there was some confusion between the EPA and the AEC regarding their roles and responsibilities for this area. After much interagency discussion, it was decided that the EPA would set the standards and the AEC (and its successor the NRC) would regulate the industry to the standards.

As one of its first efforts, the EPA undertook to establish radiation standards for all phases of the normal operation of nuclear power production. The standard was envisioned to cover all fuel-cycle facilities (enrichment, fuel fabrication, reactor operation, and reprocessing). The EPA performed an extensive study that included information gathering and environmental assessment of the impact of release of radioactive materials from fuel-cycle facilities.

The historical evidence from that period suggests that a number of considerations influenced the EPA during the standard-setting process. First, as nuclear power technology became more sophisticated and widespread, concern was growing over potential public health and environmental impacts stemming from the use of radioactive materials. One should remember that significant uncertainty existed in the technical community regarding the acceptable level of dose, and some of those uncertainties persist to today. Hence, there was a need for better information on the impacts of nuclear power as well as radiation standards to ensure its safe use.

Second, radiation standards had evolved as more experience was gained with nuclear power. It appears that the initial standards strove to set a balance between the risk to the public and the cost impacts to the nuclear power industry. However, there were some views that the standards favored industry and were not stringent enough. As the industry gained experience and as data on real emissions became available, it appeared that stricter standards could be imposed with little cost impact to the industry. There was also a tendency to tighten the standards if the impact to the industry could be accommodated.

Third, the projections for growth in nuclear power were staggering. In 1970, 17 nuclear power plants were in operation, 49 were under construction, and an additional 48 were in the planning stage. These 114 plants were to be located in 29 states and were expected to provide more than 85 GW of electric generating capacity. The AEC estimated that nuclear power would generate 150 GW by 1980 and 1000 GW by the year 2000. This would be about a thousand plants or 10 times the number actually built. Reprocessing was also assumed to be an integral part of the nuclear industry, and it was estimated that there would be one reprocessing plant (800 MT/yr) for every forty reactors. In other words, along with the 1000 reactors, there would be approximately 25 reprocessing plants by the year 2000. There would also be additional enrichment and fuel

fabrication facilities, but these were not viewed to pose the same radiological impact to people as did reactors or reprocessing plants.

While normal operating emissions from nuclear plants at that time indicated that releases were acceptable, the large growth in nuclear power, coupled with the potential emissions from reprocessing plants, led to questions about the health effects from normal operations. In evaluating the situation, there was considerable uncertainty in the technical community regarding the amount of dose that would be considered safe enough. The EPA seemed to take the position of trying to establish a standard that would protect people and not adversely limit the benefits of nuclear power.

In January 1977, the uranium fuel-cycle standard was promulgated as 40 CFR 190. It establishes both dose limits and, for certain radionuclides, release limits. The regulation of dose limits for nuclear facilities is common practice, and evidence from foreign reprocessing facilities indicates these dose limits are relatively straightforward to meet. In contrast, limits on releases appear much more constraining, and there has been little experience demonstrating the ability to meet these release limits. It appears that the release limits were largely driven by a desire to regulate reprocessing operations that have never been deployed in the United States. Accordingly, the release limits are of most interest and the focus of this report.

2.2. Overview of EPA Radiation Standard Development Process

The process used by EPA to develop the radiation standard included four basic considerations:

- Total radiation dose to populations
- Maximum dose to individuals
- Risk of health effects attributable to those doses, including future risks arising from the release of long-lived radionuclides to the environment
- Effectiveness and costs of the technology available to mitigate those risks

The initial basis of the proposed rule was the 1973 report, *Environmental Analysis of the Uranium Fuel Cycle, Part III Nuclear Fuel Reprocessing, plus errata* (EPA, 1973). The following excerpts from that report reflect the best analysis and thinking of those times.

The EPA considered "…the quantities of certain radionuclides which may be released, if industry growth projections are correct, are large. Consequently, the reprocessing step represents the point in the nuclear fuel cycle where the consequences of releases of the long-lived isotopes, such as tritium, krypton-85, iodine-129 and plutonium-239 should be carefully considered..."

The EPA believed "…the consequences of the build-up of very long-lived radionuclides (such as iodine-129 and plutonium-239) from the fuel reprocessing industry can produce cumulative and irreversible environmental levels which can be projected to cause adverse health effects on a national and worldwide scale. The consequences of the release of the shorter-lived isotopes, while possibly more severe on a short-term scale, are more easily reversed by effective control techniques…"

Based on information available to the EPA at that time, "…Control technology exists to reduce emissions of these materials. During the near-term future, all the tritium and krypton-85 produced during the fission process will be released to the environment during the reprocessing stage. Some control currently is applied for iodine and plutonium releases. Control technology to achieve confinement of krypton is essentially developed and some reduction for nearly all radioisotopes would appear feasible on a long-term basis…"

It was further believed that "…Removal of plutonium and other actinides is the most cost effective method of health risk reduction. Next in order of effectiveness are systems for removal of krypton and iodine which are about equal from a cost effectiveness standpoint. Tritium control technology is least cost effective at the present time, but future developments should be pursued to alleviate this problem…"

From a cost-impact perspective, it was noted that "…available data allows an estimate of the incremental additional cost to the cost of nuclear production of electricity from the imposition of waste control systems at fuel reprocessing facilities to be about 0.1 percent of the total present cost to the producer of electricity..."

Further, the "…evaluation of the total environmental impact of radioactive effluents requires a consideration of dose commitments beyond those delivered immediately in the site area, and, because of the long life of those materials in the biosphere, must include the exposures to national and world populations, beyond those delivered in the year of release…"

Finally, "…although the estimated health impact from the operation of a single tonne/day reprocessing facility is relatively small, extrapolation of the industry as a whole over the next 50 years of operation indicates that long-term cumulative effects may be quite large..."

Following the 1973 Environmental Analysis, the EPA published a proposed rule supported by draft and final environmental statements, supplemental analyses, and extensive public comment. First was the *Environmental Radiation Protection Requirements for Normal Operations of Activities in the Uranium Fuel Cycle: Draft Environmental Statement* (EPA, 1975). This was followed by a supplementary analysis in response to public comments: *Environmental Analysis of the Uranium Fuel Cycle, Part IV – Supplementary Analysis: 1976* (EPA, 1976a). Then *40 CFR 190, Environmental Radiation Protection Requirements for Normal Operation of Activities in the Uranium Fuel Cycle, Final Environmental Statement* (EPA, 1976b) was published. This then led to publication in the Federal Register of the final rule, Part 190, "Environmental Radiation Protection Standards for Nuclear Power Operations" (40 CFR Part 190, 1977).

Throughout this rule-making process, from the original 1973 analysis to the Final Rule, there was considerable debate on the philosophy behind the rule and how that philosophy was applied. This included debate on what were the right nuclear fuel cycle projections and source terms to use, on use of the Linear No-Threshold (LNT) methodology for estimating health effects on world populations from an accumulation of long-lived isotopes, on costs and availability of effluent control technologies, on how other countries were dealing with the issues, and many more issues. The EPA's response to those various issues and consideration of alternatives is well documented. In the end, projections of health effects appear to have been truncated to estimates

through the year 2000 (versus 2020 in the 1973 report), projections for nuclear growth (including reprocessing) were scaled back, and the availability and costs of control technologies were revised somewhat, but the basic methodologies and results from the 1973 Environmental Analysis still appear to be the technical foundation for 40 CFR 190 dose and release limits.

With respect to Kr-85, in the Supplementary Analysis (EPA, 1976a) leading up to the Final Rule, the EPA concluded, "It can be seen that the costs per man-rem for all of the systems and organs considered are rather small especially when compared to the NRC's interim value of $1,000/ whole body or thyroid man-rem applicable to light water power reactors."

The January 1977 Federal Register Notice stated, "The standards limit irreversible contamination of the local, national and global environment due to releases of radioactive Kr-85 (half-life 10.7 years), I-129 (half-life 17 million years), and alpha-emitting transuranics (half-lives 18 years to 2 million years). The total reduction in potential health impact attributable to operations through the year 2000 is estimated to be in excess of 1000 cases of cancer, leukemia, and serious genetic effects in human populations, based upon the assumed achievement of an annual nuclear production of 1000 GW(e)-yr of electrical power by that year."

It should also be noted that in several places, the EPA indicated an intention to periodically revisit the rule. In the Federal Register Notice promulgating the rule, the EPA states, "In addition, the Agency has made its intent regarding realistic implementation clear, as, for example, in the discussion of these matters in the Final Environmental Statement and will continue to do so if necessary as implementation proceeds, to assure that unnecessary conservatism does not occur."

In summary, nuclear power, reprocessing, and the fuel-cycle itself were relatively new and growing when the EPA was formed, and there was mounting public concern about radioactive releases. Key engineered barriers to releases during reactor operations (solid pellets, intact cladding, high-integrity reactor vessels, and robust containment structures) were deliberately breached or absent during reprocessing. Reprocessing was the point in the nuclear fuel cycle where releases particularly needed to be controlled. Whatever limits were placed on what was allowed to be released needed to take into account not just any immediate doses to individuals and populations, but also a recognition that long-lived isotopes could accumulate and continue to be a source of dosage well into the future. This concern about accumulation explains why the 40 CFR 190 limits on releases are considerably more restrictive than the dose limits. One can understand why the regulation developed as it did after considering first that there were significant uncertainties as to the effects at the local, regional, national, and global levels on expected releases (hence, a need for conservatism) and second that technologies to control releases seemed to be on the horizon at a cost commensurate with (hypothetically) avoided health effects. It is worth noting that the EPA clearly took cost considerations into account and did not believe that 40 CFR 190 would make reprocessing prohibitively expensive.

2.3. Summary of Basis for Dose and Release Limits

To understand the basis for the quantitative limits in 40 CFR 190, it is noted that the standard has two parts and that the limits in each part are assessed separately. The first part (Part a) sets a

dose limit on the maximally exposed individual. This limit appears to be risk based and is intended to protect people in the local area surrounding the plant. The limit is 25 mrem/yr, which is about 10% of the annual background dose in the U.S. The limit seems to be based on recommendations coming from esteemed expert panels such as the National Research Committee on the Biological Effects of Ionizing Radiation (BEIR) and National Council on Radiation Protection (NCRP). It is not entirely apparent that there is a quantitative basis for the 25 mrem/yr limit, but if an individual were exposed to 25 mrem/yr for 70 years and the LNT model were applied (based on total dose), the resulting total dose would increase the cancer risk by less than 1/10 of 1%, which could be viewed to be an acceptable risk. Actual dose measurements from nuclear plants at that time indicated that this limit could be met without expensive retrofits.

The second part of the standard sets release limits of certain radionuclides (I-129, Kr-85, Pu-239, and other actinides) that are prone to build up in the biosphere. The specific radionuclides of concern during normal operation depend on the nuclear system in question. For reactors, since most of the fission products would be retained in the fuel and clad, the radionuclides of concern were activation products and fission products released from failed fuel. For reprocessing facilities, the radionuclides of concern were radioactive noble gases and volatile products, especially those that would build up in the atmosphere because of their long half-life. There were also questions on how the release of these radionuclides during reprocessing operations and their subsequent build-up would impact populations beyond the local area. The EPA analysis indicated that the health effects from some of these radionuclides could be global.

A discussion of these release limits should begin with what these limits actually mean with respect to the operations of a reprocessing plant. The total radionuclide inventory in spent fuel is a function of the fuel burn-up. For the long-lived isotopes, the build-up is roughly proportional to the burn-up because the rate of production greatly exceeds the decay rate. Burn-up, of course, is directly proportional to electrical energy produced, and so the amount of a particular isotope in spent fuel can be calculated as a function of burn-up or electrical power produced. For example, the amount of I-129 in 150 day cooled spent fuel with a burn-up of 50,000 MWD/MT converted to Ci per electrical energy produced is 930 mCi per GWe-yr. A similar calculation can be performed for the other isotopes.

Given the 40 CFR 190 limits, the ratio of the amount produced divided by the release limit is the effective decontamination factor (DF) for the facility. Table 2.1 provides numerical values of the production rate and effective DF for the isotopes of interest.

Table 2.1. Comparison of Production Rate, Release Limit, and Effective Decontamination Factor.

Isotope	Production per GWe-yr	40 CFR 190 Release Limit	Effective Decontamination Factor
I-129	930 mCi	5 mCi	190
Kr-85	2.8×10^5 Ci	50,000 Ci	5.6
Pu-239	8.5×10^6 mCi	0.5 mCi	~2×10^7

As discussed below, the EPA methodology actually determined the DFs for each isotope in order to achieve a desired reduction in the calculated health effects. With the selected DF, the release limits were then derived by dividing the production rate by the DF. The values for these DFs were selected to achieve an optimal reduction of potential health effects in a cost-effective manner. The EPA characterized its process as cost-effective health risk minimization.

Briefly, the methodology used to determine the release limits may be summarized as (1) estimating the release from a "projected" reprocessing plant per ton of fuel reprocessed, (2) estimating world population health effects from such releases assuming the nuclear growth projections then available, (3) calculating what control technologies were capable of achieving and the cost for doing so, (4) comparing the theoretical health effects that could be averted by reducing effluents with these technologies, and then lastly (5) examining the cost per health effect avoided and determining if it is reasonable.

A summary of health-effects calculations is provided here, and details are discussed in Section 3.0 of this report. To calculate the health effects from the release of a particular isotope, one calculates the dose to an individual and multiples this by the LNT risk factor, that is:

$$\text{Health Effects} = \text{Dose} \bullet \text{LNT Risk Factor}$$

The dose is calculated generically via an equation in which the dose is proportional to the isotopic source release rate (Q), the decontamination factor (DF), an atmospheric transport factor (TF), the dose conversion factor (DCF), and population (P) affected.

$$\text{Dose} \sim Q \bullet 1/DF \bullet TF \bullet DCF \bullet P$$

The isotopic source release rate (Q) is directly related to the fuel burn-up, which is proportional to the energy produced, and the processing capability of a given facility. The decontamination factor (DF) is a function of the decontamination system, which would likely be dependent on the specific chemical form of the isotope in question. For example, a silver zeolite bed would have a specific DF for iodine, but would not be effective in sequestering Kr. The transport factor (TF) accounts for the transport of a specific isotope from the facility to a receptor at a particular location or within a region. The dose conversion factor (DCF) converts the exposure (in Ci) to dose (in rem). The population (P) represents the number of people in a specific region. For the case of the maximum exposed individual (MEI) dose, P is set to 1.

As an illustration of the health-effects methodology, it is useful to consider only the effects from the reprocessing facility. The EPA analysis considered three distinct zones for calculating health effects: (1) the local regional area (3 km to 80 km from the source of emission), (2) the U.S. population defined as within the U.S. but beyond the 80 km regional limit (modeled within the Eastern U.S.), and (3) worldwide. In each case, the DF for a given isotope was parameterized, and a transport model was developed.

For the first zone (regionally), the collective dose to the regional population, as well as the dose at the site boundary, was calculated. As seen in Section 3.2, a $\frac{\chi}{Q}$ method was used, and this is a fairly standard approach. Doses and health effects from I-129, Kr-85, and H-3 were calculated.

The second zone (Eastern U.S.) was chosen because it was assumed that reprocessing facilities would be located near reactor sites, and most of these were located in the Eastern U.S. Collective dose was used exclusively in the calculations. A set of simplifying assumptions was made for the transport calculation to make the problem mathematically tractable. Once again, doses and health effects were calculated for the isotopes of interest.

The third zone (worldwide) principally considered the collected dose to the world population from the build-up of Kr-85 and H-3 in the biosphere. For each isotope, simplifying assumptions were used in the analysis that basically led to a uniform dose to the entire worldwide population. Obviously in such an approach, the average individual dose was very small (<0.002 mrem). The collective world population dose was calculated by multiplying an exceedingly small average dose ("micro-dose") by the world population (a "mega population"). World health effects were then calculated by applying the LNT to this collective dose. This method of estimating health effects is highly uncertain for two reasons: (1) the use of a "micro-dose to a megapopulation" as an absolute number is a method that the NRC has questioned and (2) there is large uncertainty in the use of the LNT to "convert" collective dose to health effects.

The final part of the analysis consisted of a survey and analysis of the available technologies used to decontaminate specific elements. Technologies evaluated include HEPA filters, zeolite beds, cryogenic distillation, and many others. For each technology evaluated, a cost-effectiveness metric was developed in terms of Health Effects Avoided / Unit Cost. To illustrate this, consider a hypothetical case in which the health effects for a particular isotope were 1000 (assuming a DF=1) and the cost of achieving a DF of 10 for that isotope was $9 million. The cost-effectiveness metric (CEM) would be as follows:

$$\text{CEM} = \text{Health Effects Avoided / Unit Cost}$$

$$= 900 \text{ Health Effects Avoided / } \$9 \times 10^{6} = 1 \times 10^{-4} \text{ Health Effects Avoided / } \$$$

$$= \$10{,}000 \text{ per Health Effect Avoided.}$$

Using this approach, a large number of systems were evaluated and compared to each other. From this analysis, it became apparent that some systems were highly cost effective and others were not. As Figure 2.1 (EPA, 1976B: Vol. 1) indicates, there is actually a point where the cost effectiveness rapidly diminishes, and it is at this cost-effectiveness value that the EPA chose to set the limit for a DF for a particular isotope. In other words, the selected cost-effectiveness measure actually sets the maximum required DF for a particular isotope. It then appears that there was some margin added (i.e., a lower DF was selected) because the EPA acknowledged that improvements in decontamination effectiveness would be expected over time. Finally, the selected DF was used to back-calculate the permissible emissions from a facility, and these were normalized on a per unit of energy produced basis.

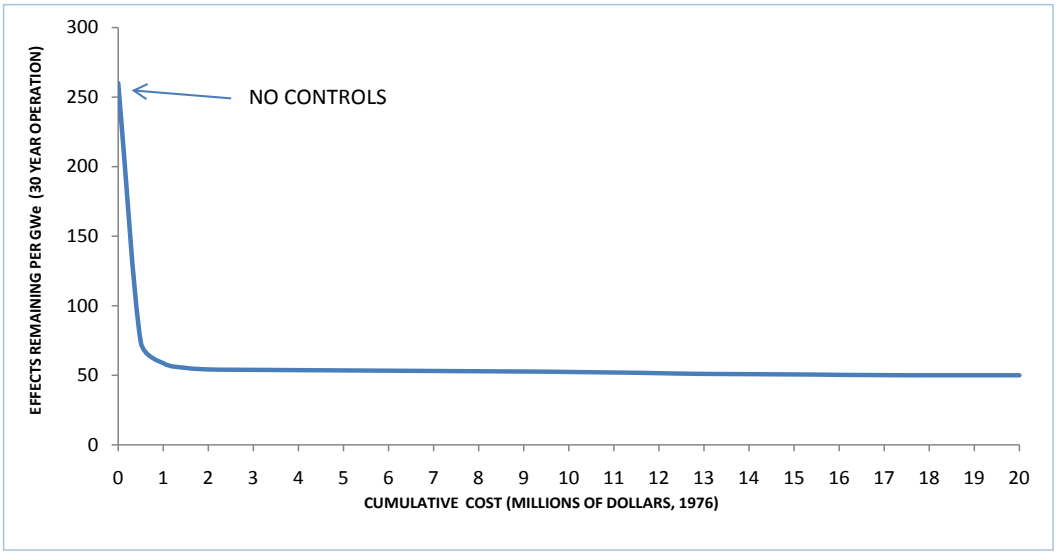

Figure 2.1. Diminishing Returns for Applying Decontamination Technologies.

2.4. Key Assumptions in EPA Methodology

As illustrated in the synopsis of the EPA methodology, the technical basis for 40 CFR 190 was the knowledge base of the early 1970s and a number of then reasonable assumptions. Under today's circumstances and given all that has changed and been learned since the 1970s, it might be appropriate to re-examine the technical basis and the derived limits found in 40 CFR 190. The specific assumptions in particular seem to have changed considerably:

1. The basis for estimates of cost effectiveness of controls in terms of costs avoided per health effect as compared to other industries or parts of the nuclear fuel cycle, as well as assumptions regarding emission-capture technologies, has changed.
2. Estimates of the amount of radionuclides that could be expected to be released and accumulate over the next century have changed markedly. Specifically, the projected maximum source term is much less, and rather than tens of reprocessing plants by 2020, it is unlikely that there would be more than a few such facilities in the next century.
3. Conventional wisdom on the use of collective dose methodologies for estimating worldwide health effects from a given emission or accumulation of radionuclides has changed.
4. More information is now available to assess fully the dose and risks to workers resulting from the capture and sequestration of the isotopes of concern against the dose and risk to world populations if these were released.
5. There are now data and experience from the use of international standards that could affect the EPA analysis.

The implications of these assumptions are discussed more fully in Section 3.0 of this report.

3.0 Technical Bases of EPA Methodology

An outline of the methodology used by the EPA for developing the release limits in 40 CFR Part 190 is shown in Figure 3.1. There were two main objectives in setting the release limits: (1) a complete assessment of the potential **impact on public health** be made and (2) the **cost effectiveness** of measures available to reduce or eliminate radioactive effluents to the environment be carefully considered (EPA, 1976b: Vol. I). Both objectives were predicated on an estimate of projected growth of commercial nuclear power, which was developed by the AEC in the 1970s.

The EPA goal in setting the standard appears to have been to achieve a balance between avoiding unnecessary health risks to the public and imposing unreasonable costs on the industry. This required an in-depth study of potential health effects from the emissions as well as a cost-effectiveness assessment of decontamination technologies.

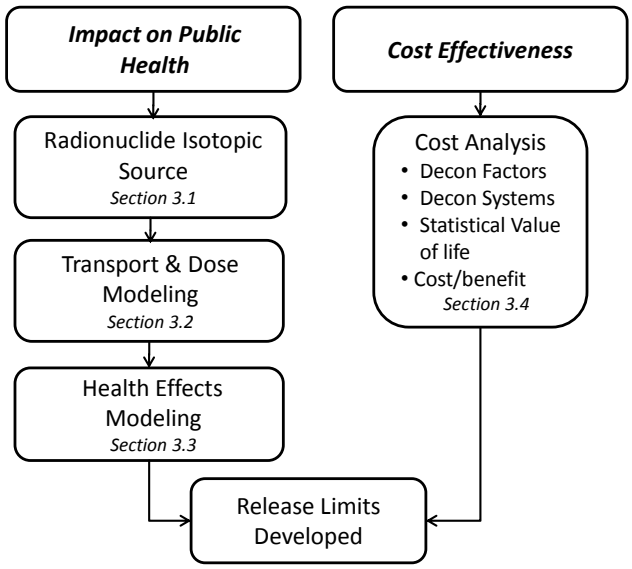

Figure 3.1. Outline of EPA Methodology.

In general, radiological health protection is achieved by preventing, as much as possible, the emission of ionizing radiation and the dispersal of radioactive materials into any environment to which people have access. It should be kept in mind that there are many natural sources of ionizing radiation such as uranium and its radioactive progeny in building materials, radio-potassium in bananas and potassium salts, and radon gas. Further, exposure to medical radioisotopes and diagnostic medical and dental x-rays contributes to the dose an individual receives. Therefore, an adequate level of protection is expected to be achieved when the incremental dose to an individual is a small fraction of the natural background dose.

The evaluation of the technical bases in this report examined the key components of the EPA analyses, which included the following:

Radionuclide Isotopic Source – The source is the amount of an isotope available for release to the environment in a given period of time. This is the input to the consequence modeling and is dependent on the magnitude of nuclear power generation. For the long-lived isotopes, the source would be proportional to the fuel burn-up that is being reprocessed.

Transport and Dose Modeling – This aspect deals with how specific isotopes are released from the facility and transported through the environment to deliver a radiation dose to individuals. Physical processes considered here include decontamination within the facility, dispersion in the atmosphere, dilution in the biosphere, and the biological effects of the specific radiation from the isotope.

Health Effects Modeling – Modeling deals with the conversion of dose into potential health effects. It brings in the findings of expert panels, such as the National Research Committee on the Biological Effects of Ionizing Radiation (BEIR) and NCRP, and the concepts of LNT and collective dose.

Cost Effectiveness – This aspect deals with the cost effectiveness for decontamination systems. Normally, the effectiveness of a system (or a set of systems) in sequestering radionuclides increases as more is spent until a point of diminishing returns is reached. Beyond this point, the increased cost of increased decontamination is very limited and is deemed to be not cost effective. Cost and decontamination potential are evaluated to determine the optimal cost effectiveness.

In the following sections, these key elements of the EPA methodology are reviewed. This review includes attempts to replicate the EPA results and assesses the robustness of the underlying assumptions given today's knowledge and all that has changed and been learned since the 1970s.

3.1 Radionuclide Isotopic Source

3.1.1 Historical Perspective

The decade 1970 to 1980 saw rapid expansion of commercial nuclear power. Twenty-two commercial nuclear plants were in full operation in 1971, and 41 plants were ordered by utilities in 1973 (EIA, 1998). By 1979, 72 licensed reactors produced 12% of commercially generated electricity in the U.S. Interest in extending the commercial nuclear fuel cycle by "breeding" fissile plutonium grew from the successful breeding and isolation of plutonium for weapons. To extract the plutonium, spent fuel needed to be reprocessed, and this generated commercial interest in reprocessing. However, during reprocessing, many fission products normally contained in the spent fuel are liberated and potentially available for release to the environment. With the large projected growth in nuclear power and the widespread adoption of reprocessing, the potential radioactive source term was very large. There was significant concern that even if a small fraction of the fission products was routinely released, then significant adverse health effects would occur. This concern was a major driver for the 40 CFR 190 radiation standard.

In 1970, Allied General Nuclear Services (AGNS) began constructing a large commercial reprocessing plant at Barnwell, SC. Both General Electric (GE) and Exxon considered commercial nuclear fuel reprocessing in the early 1970s (Andrews, 2008). In 1975, the NRC delayed licensing proceedings for the AGNS plant until the environmental impact of the reprocessing facility could be assessed adequately. The proposed environmental impact statement (EIS) was never completed and time ran out for the AGNS facility. In 1977, President Carter announced that commercial recycling and reprocessing of plutonium would be "deferred indefinitely," and the NRC terminated licensing proceedings and the EIS (Andrews, 2008).

40 CFR 190 originated during the period of commercial nuclear power expansion and expanding interest in reprocessing, and it received little attention until 2001 when the U.S. interest in

reprocessing was revived. The regulation is now important because it sets release limits (emission standards) for the nuclear fuel cycle, including any reprocessing that includes uranium.

3.1.2 Assumption: Growth in Nuclear Power

The actual growth of nuclear power in the U.S. has been considerably less than was projected by the EPA in 1977. The significant difference in growth would reduce the amount of I-129 and Kr-85 in the atmosphere when compared to that predicted by the EPA in the 1970s. Figure 3.2 compares the actual growth of commercial nuclear power in GWYe over time with that projected by the EPA. The difference today is over an order of magnitude.

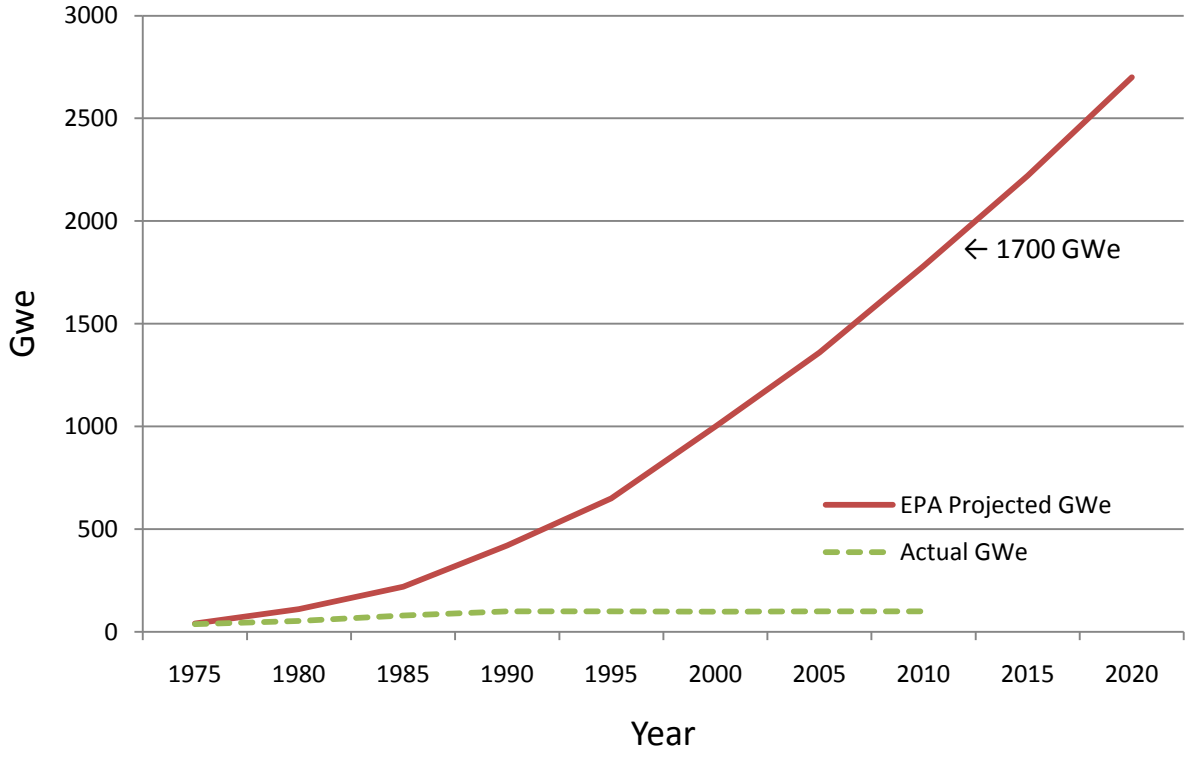

Figure 3.2. Comparison of EPA Nuclear Energy Growth Projections to Actual Nuclear Energy Growth.

Sources: EPA projections from EPA, 1973: Table A.1. Actual nuclear energy growth obtained from EIA, 2009.

The growth of commercial nuclear power projected by the EPA was used as a basis for subsequent calculations and the eventual development of the release limit regulations. This projected growth also assumed that reprocessing activities would release some fraction of the available I-129, Kr-85, Pu, H-3, and C-14 to the biosphere.

As shown in Figure 3.2, the build-up of Kr-85 and I-129 in the environment, the EPA assumed a significant nuclear energy growth rate (i.e., that nuclear power generation in the U.S. would reach 2700 GWe/year by the year 2020; see EPA, 1976b: Figure 8). The solid lines in Figures 3.3 and 3.4 show the projected environmental burden of Kr-85 and I-129, respectively, in the atmosphere. In generating these numbers, the EPA recognized that only about 5% of these gases would be released by operating nuclear power plants (Benedict et al., 1981) and assumed that the

remaining 95% would be released during reprocessing (EPA, 1976b). The solid lines thus represent total projected releases of Kr-85 and I-129 into the environment. The total release is based on the release from reprocessed spent fuel from operating power plants and is proportional to the projected growth of nuclear power generation.

The dashed lines in the figures show the calculated release of each radionuclide in the atmosphere based on actual nuclear power production during the period of time shown, and taking into account startup, shutdown, and outages during which no power is generated. Like the EPA projection, the analysis that produced the dashed lines assumed that all fuel would eventually be reprocessed, and that all Kr-85 would eventually be released. The difference between the solid and dashed curves is the difference between the EPA projected nuclear power production and actual nuclear power production. There may be some reprocessing during the next 30 to 50 years, but the U.S. is unlikely to reprocess all spent fuel for another century, if ever. The EPA projected release of Kr-85 to the atmosphere was greatly overestimated by a combination of an optimistic estimate of reprocessing and an unrealized estimate of nuclear power production. If the dashed curve is extended, the Kr-85 release will still remain well below that projected by EPA.

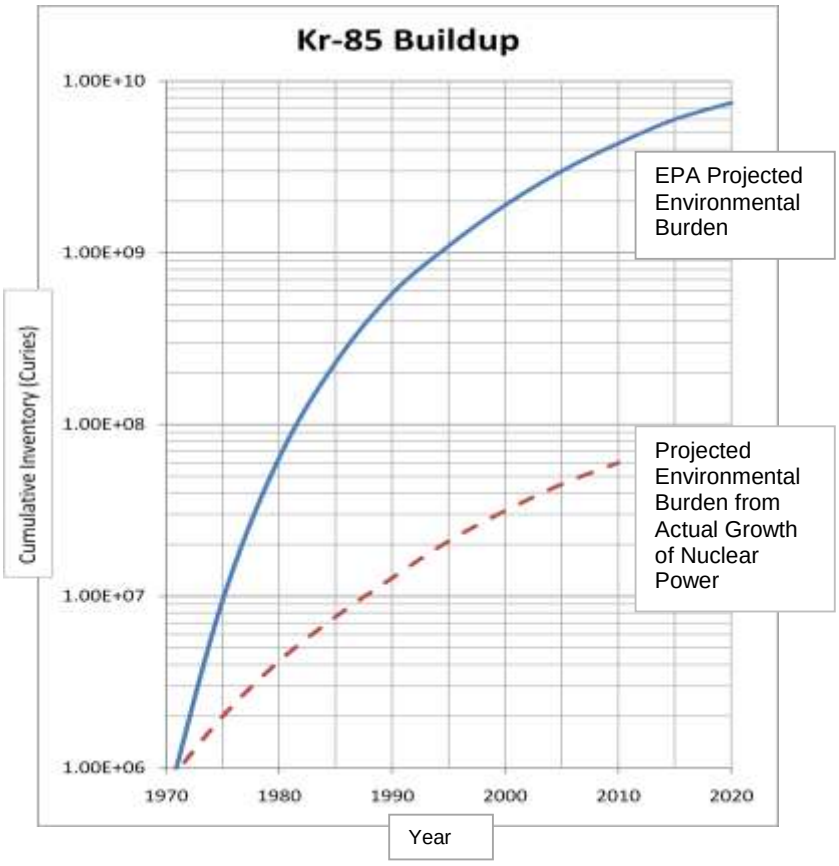

Figure 3.3. Projected Environmental Burden of Kr-85 from the U.S. Nuclear Power Industry.

Source: Figure and solid lines redrawn from EPA, 1976b: Figure 8, p. 83. Dashed line calculated from EIA, 1998: Tables 9.1 and 9.2. Plant closures and periodic outages were included in the calculation.

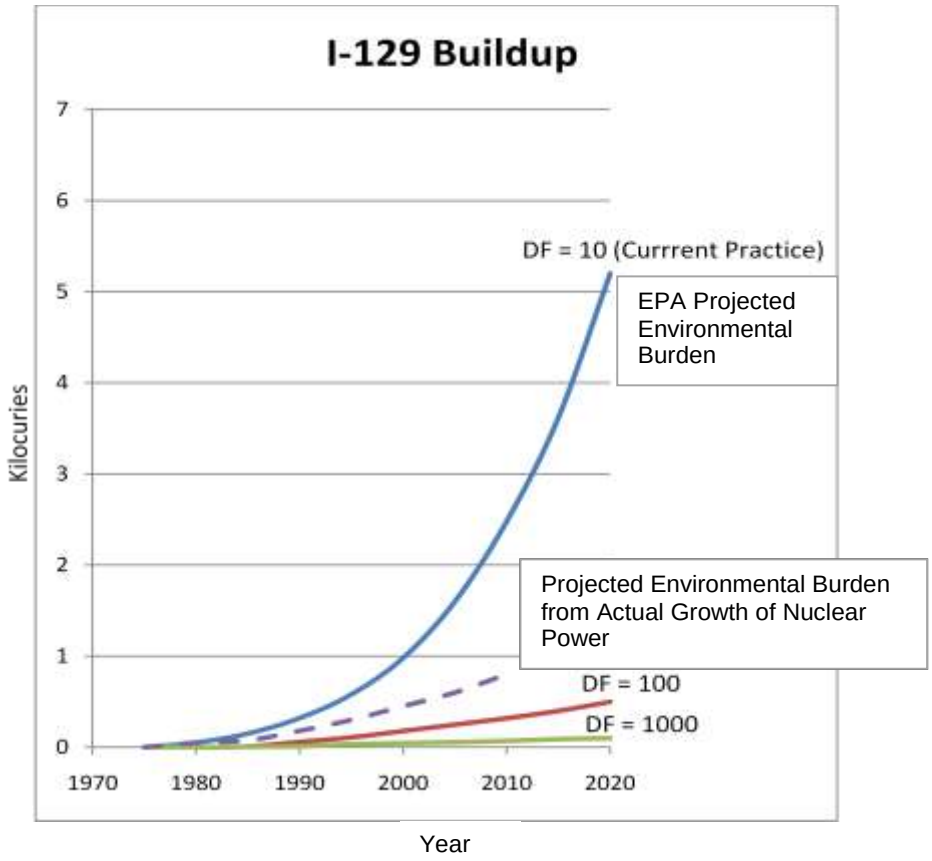

Figure 3.4. Global Environmental Burden of I-129.

Source: Figure and solid lines redrawn from EPA, 1976b: Figure 9, p. 84. Dashed line calculated from EIA, 1998: Tables 9.1 and 9.2. Plant closures and periodic outages were included in the calculation.

Since the EPA projected growth of I-129 build-up is significantly more than the actual growth, and since reprocessing activities have not been undertaken in the U.S., the projected environmental burden used to set the release limits was significantly overestimated. In fact, the 1977 estimates for nuclear power growth are unlikely to occur over the next 100 years, assuming that commercial nuclear energy power growth continues at the current rate. It seems clear that the use of more realistic projections on the magnitude of nuclear power would significantly affect the acceptable release.

3.1.3 Assumption: Cooling Time

The EPA estimates of Kr-85 and I-129 releases were based on an assumption of 150 days cooling time for the fuel. While extending the cooling time of the fuel has no measurable effect on the release of I-129 (half-life ~16 million years), it does affect the release of Kr-85 (half-life ~11 years) as shown in Table 3.1. The table assumes a 50 GWD fuel burn-up, and it shows how the DF required to meet the release limit in 40 CFR 190.10(b) changes as the cooling time increases.

Table 3.1. Effect of Cooling Time Assumptions.

Isotope	Fuel Cooling Time	mCi produced per GWY(e)	40 CFR 190.10(b) Release Limit	DF
Kr-85	150 days	2.8×10^8	5.00×10^7	5.6
Kr-85	10 years	1.5×10^8	5.00×10^7	3
Kr-85	27 years	5.00×10^7	5.00×10^7	1
I-129	27 years	9.3×10^2	5.00×10^7	190

A short cooling time and a high burn-up have proved unrealistic as the U.S. nuclear industry has developed. The industry is going to high burn-up (45 GWD/MTU rather than 50 GWD/MTU) (EPRI, 2003), but it almost never transports fuel with less than five years of cooling time and usually allows fuel to cool for ten years before transporting it. By cooling the fuel for about 30 years prior to reprocessing, it should be possible either to meet the Kr-85 release limit in 40 CFR 190.10(b) without any controls or at least to reduce the decontamination requirement. By using a 150-day cooling time, the EPA DF for the Kr-85 source is higher by a factor of almost 6 when compared to a 27 year cooled fuel. Of course, the same is not true for I-129 because of the long half-life. Since Kr-85 sequestration is identified by industry as being very costly (see Section 3.4, Table 3.9, of this report), the use of ~30 year cooling would seem to be a simple solution consistent with current wet and dry storage at nuclear plants.

3.2 Transport and Dose Modeling

The EPA performed transport and dose analysis for three distinct distance scales: (1) regional population dose modeling, including the Maximally Exposed Individual (MEI) defined as being from 3 km to 80 km from the source of emission; (2) U.S. population dose modeling defined as within the U.S., but beyond the 80 km regional limit; and (3) worldwide population dose modeling, defined as the remainder of the world. In addition to a review of transport and dose modeling and results, the Dose Conversion Factors (DCFs) used by the EPA are reviewed as well and discussed in the following sections.

3.2.1 Regional Population Dose Modeling

The general equation used to determine regional doses is the standard equation for dilution at a specified distance. The equation is a simple form of the equation for atmospheric transport and dispersion and is given below.

$$D_{ij} = Q_j \cdot (\frac{\chi}{Q})_{3km} \cdot F_{ij} \cdot C \cdot P_R$$

Here,

D_{ij} = dose (rem/yr or person-rem/yr) to organ i from exposure to radionuclide j.
Q_j = release rate (Ci/s) of radionuclide j.
χ = concentration of radionuclide (μCi·s)

$(\frac{\chi}{Q})_{3km}$ = meteorological dispersion factor at 3 km from the source of emission. The value used by the EPA is $5 \cdot 10^{-8}$ μCi·s/ Ci·cm^3, which is equivalent to units of s/m^3.

F_{ij} = pathway-dependent DCF for organ i and radionuclide j [(rem/yr)/(μCi/cm^3)].

C = regional dilution and population correction factor (dimensionless).

P_R = regional population. This value is set to unity to calculate the individual dose (rem) at the 3 km location and set to $1.5 \cdot 10^6$ people for 1980 and is assumed to double over a 40 year period.

This equation relates dose to an individual or group as the product of the release rate of radionuclide *j*, Q_j, a dilution factor that expresses the amount of atmospheric mixing that occurs as a plume moves from its initial release point to a downwind location at three kilometers from the source, $(\frac{\chi}{Q})_{3km}$, a DCF that relates exposure to airborne radionuclide to dose to organ *i*, F_{ij}, a regional dilution factor that accounts for additional dilution over a larger region, *C*, and the population used in the calculation, P_R. The value of *C* is simply the ratio of the average value of $(\frac{\chi}{Q})$ for the area within 80 km of the plant to the value of $(\frac{\chi}{Q})_{3km}$. This value can be obtained by performing dispersion analyses for the appropriate distances.

For calculating dose to an individual at the site boundary (MEI Dose), both *C* and P_R are set to unity, and the result is in units of rem/yr (i.e., committed dose to an individual per year of plant operation). For calculating regional population dose (within a radius of 80 km), *C* is chosen to be 0.028, and P_R is chosen to be the average population over the 40-year lifetime of the plant, that is, (1.5 x 1.5) x10^6 = 2.25 x 10^6. For the regional population dose, the product, $C\ (\frac{\chi}{Q})_{3km}$, represents the average dilution within 80 km of the site (i.e., the average dilution of released material that the population living within an 80 km radius of the plant would encounter). The initial factor of 1.5 for the population is the average population correction factor over the 40-year lifetime of the plant, which is based on a doubling of the population over 40 years. The result in this case is an average population dose corresponding to a year of plant operation in units of person-rem/yr.

The meteorological dispersion factor is about an order of magnitude smaller than would typically be used for short-term dispersion problems. This order of magnitude difference is not explained in the EPA report, but probably reflects changes in wind direction over year-long periods such that an individual at a specific location would receive a dose only for the fraction of time that the plume is traveling in his/her direction. Because a plume typically occupies a pie-shaped sector that is about 1/10 to 1/20 of the entire compass, the order-of-magnitude difference appears to reasonably account for a long-term average.

The population within an 80-km radius was chosen to reflect the average population surrounding existing U.S. nuclear power plants at the time of the EPA study. This assumes that a recycling plant would be built in an area with a similar population density as the existing fleet of nuclear power plants. This assumption seems reasonable.

In summary, the regional dose model uses techniques that are comparable to those in use today. This was confirmed by running a set of calculations with the MACCS2 code and comparing it with the EPA results. Furthermore, the underlying assumptions and parameter selections seem to be reasonable. Therefore, the results from this part of the EPA analysis would not likely be changed using today's methods. One exception is ingestion dose from I-129. EPA's assumptions for the ingestion pathway of this radionuclide are shown to be conservative in Subsection 3.2.5.

3.2.2 U.S. Population Dose Modeling

Doses to the U.S. population beyond the 80-km region described above are treated separately for four groups of radionuclides (H-3, Kr-85, I-129, and I-131) and the actinides. The difference in treatment stems from differences in the dose pathways for these groups. Evaluation of the calculation methods is presented in the following.

Tritium (H-3)

H-3 is assumed to become oxidized and to join the hydrological cycle in the form of tritiated water. This is a very reasonable assumption given that hydrogen is very reactive. It follows that the primary pathway for exposure to H-3 is ingestion. The following equation is used to treat the ingestion of H-3:

$$D_{ij} = C_j \cdot f \cdot F_{ij} \cdot 0.80 \cdot P_u$$

Here

C_j = water concentration of H-3 (denoted by j) determined by diluting the annual release by the average annual rainfall over the eastern half of the U.S. The equation for determining C_j is $C_j = \dfrac{Q_j \cdot [3.15 \cdot 10^7 s/yr] \cdot [10^6 \mu Ci/Ci]}{[3.7 \cdot 10^6 km^2] \cdot [100 cm] \cdot [10^5 cm/km]^2}$. Here, $3.7 \cdot 10^6$ km^2 is half the land area of the U.S., and 100 cm is the average annual rainfall over the eastern U.S. The units on the concentration are μCi/cm^3/yr, where the rate is per year of plant operation.

f = factor that accounts for the fraction of H-3 that does not rain out over the U.S. and the fraction that is further diluted in groundwater. A value of 0.5 is specified in the EPA study.

F_{ij} = pathway-dependent DCF for organ i and radionuclide j [(rem/yr)/(μCi/cm^3)].

P_u = population of the U.S. The factor of 0.80 in the equation is the fraction of population in the eastern half of the U.S. The U.S. population is assumed to grow linearly from $2 \cdot 10^8$ in 1970 to $4 \cdot 10^8$ in 2030. Only half the population is used when calculating genetic effects.

The average concentration of tritium in drinking water is calculated by assuming that 50% of the annual release of tritium is diluted into the rainwater of the eastern half of the continental U.S. This rainwater is ultimately consumed as drinking water.

The suggested values of the factors in the above equation are conservative because no delay is included to account for the time that it takes for the rainwater falling onto the ground to enter reservoirs and other drinking water systems. If delay times were treated, some of the H-3 would have decayed, and estimated doses would be less.

Krypton (Kr-85)

The EPA assumes that released Kr-85 would initially pass over the eastern half of the U.S. before moving across the Atlantic Ocean and ultimately being dispersed on a worldwide basis. The assumptions made by the EPA (1973) report for the U.S. dose come from empirical data from the National Oceanic and Atmospheric Administration. The equation used in the EPA report is as follows:

$$D_{ij} = [2.5 \cdot 10^{-16}\, person \cdot Ci / cm^3 / Ci] \cdot [10^6\, \mu Ci / Ci] \cdot N \cdot A_j / d_j \cdot F_{ij} \cdot 1.5 = [3.8 \cdot 10^{-10}] \cdot N \cdot A_j / d_j \cdot F_{ij}$$

Here

N	=	number of tonnes processed per year (1500 tonnes/yr is assumed).
A_j	=	activity of radionuclide *j* contained in 1 tonne of spent fuel (Ci).
d_j	=	system DF (dimensionless). DFs of 1 for H-3 and Kr-85 and of 1000 for iodine are assumed.

This equation is comparable to the one above under the category of Regional Population Dose Modeling (Section 3.2.1). The population dose is proportional to the activity released over one year of plant operation.

The factor used in this equation for average exposure to the population in the eastern half of the U.S. is somewhat difficult to judge. To evaluate this factor, Sandia performed an analysis with MACCS2 for a hypothetical plant located coincident with the Sequoyah nuclear power plant in Tennessee and obtained a reasonably consistent result. The factor of 1.5 in the above equation accounts for population growth over the lifetime of the plant. This equation is judged as providing a reasonable estimate of population dose to the eastern half of the U.S.

Iodine (I-129, I-131)

Most of the iodine dose is caused by deposition onto soil and subsequent ingestion through the food pathway. The EPA dose estimates for ingestion of iodine are evaluated below, but only for the MEI at the site boundary and for the region.

It should be noted that the EPA analyses for the iodine isotopes were based on an assumed DF of 1000. This DF may be hard to obtain in practice and may not be necessary to provide adequate protection to the public (refer to Section 3.3).

Actinides (Pu-239, etc.)

Essentially all of the actinide doses are from inhalation. It is further assumed by the EPA that the inhalation doses are primarily from resuspension of actinides that have deposited onto the ground. Anspaugh (1990) has demonstrated that resuspension of material deposited on the ground is minimal by about 150 days after deposition. These doses and resulting health effects are predicted to be very small and so are not evaluated here. However, it should be noted that

the EPA assumed a DF of 10^9 for the actinides. Because the actinides have very low volatilities, releases would be almost exclusively as aerosols rather than as vapors. Such a large DF requires a highly efficient filtration system to capture any released actinides.

3.2.3 Worldwide Population Dose Modeling

In the EPA analysis, Kr-85 and H-3 are considered when estimating worldwide doses and health effects. These two isotopes are considered separately in the following discussion.

Krypton (Kr-85)

Kr-85 is very inert and would persist in the atmosphere until it decays to rubidium-85, a stable isotope. The primary assumption for calculating the worldwide dose from release of Kr-85 is that, after its initial pass over the U.S., it becomes uniformly dispersed throughout the atmosphere. This assumption leads to the following equation for calculating worldwide population dose from exposure to one year's release of Kr-85:

$$D_{ij} = [10^6 \mu Ci/Ci] \cdot [\frac{0.00129 g/cm^3}{5.14 \cdot 10^{21} g}] \cdot N \cdot A_j / d_j \cdot F_{ij} \cdot \int_t^{\infty} P_w(t') \cdot e^{-\lambda t'} dt'$$

$$= [1.9 \cdot 10^{-8} \mu Ci/cm^3/Ci] \cdot N \cdot A_j / d_j \cdot F_{ij} \cdot e^{0.019t}$$

Here

$P_w(t)$ = worldwide population at time t. $P_w(t)$ is $3.56 \cdot 10^9$ for 1970 and is assumed to increase at an exponential rate of 0.019/yr.

λ = exponential decay rate of Kr-85, which is 0.0645/yr.

This equation is for the committed population dose resulting from a single year of plant operation corresponding to year *t*. The final term in the equation is a population correction factor that escalates population from 1970 to the year of interest. Thus, *t* is years from 1970. The EPA assumes that the plant begins operation in 1980, so t = 10 yr.

Tritium (H-3)

The estimation of worldwide dose from release of H-3 is based on the assumption that the H-3 is diluted into the circulating waters of the Northern Hemisphere. The volume of these waters is given as one-half of $2.7 \cdot 10^{19}$ liters.

The entire quantity of water at or near the surface of the earth is $1.46 \cdot 10^{21}$ kg, which is approximately the same number of liters. Of this water, 1.6% is below ground in aquifers, and 0.001% is suspended in the atmosphere. Of the remaining water, 97% is salt water, 2.4% is in the form of ice, and the remaining 0.6% is above-ground freshwater in the form of streams, rivers, ponds, and lakes (AGU, 1995).

The fraction of the water characterized in the EPA (1973) report as circulating water is about 1.8% of the total water on the earth. The assumption in the report is apparently that the released H-3 does not become diluted into most of the salt water, but primarily into above-ground freshwater and some of the groundwater. If the assumption were that the H-3 was diluted into all

surface water, freshwater and salt water, the resulting doses and health effects would be reduced by almost two orders of magnitude. The worldwide H-3 doses estimated in the EPA report should be viewed as conservative, possibly by one or more orders of magnitude.

3.2.4 Verification of EPA Results

Using the EPA equations and assumptions, an attempt was made to reproduce the dose estimates given by the EPA for all regions (regional, U.S., and worldwide). The values obtained by this analysis are provided in Table 3.2. These agree with Table 2 from the 1973 EPA report to two significant figures. Table 2 from the EPA report is included for comparison purposes and appears as Table 3.3 in this report.

Note that, as previously discussed in Section 3.2.3, the DFs assumed for krypton and tritium in Table 3.2 and 3.3 are 1 (no decontamination); the DF assumed for iodine is 1000 (only 0.1% of the iodine from reprocessing is released into the environment). The estimated iodine doses would be higher if the DF were lower than 1000. For example, if the iodine DF were 100 instead of 1000, regional, individual, adult thyroid doses would increase from 12 μrem/yr to 120 μrem/yr.

Table 3.2. Doses from 1 Year of Plant Operation from a 5 Tonne/Day Nuclear Fuel Reprocessing Plant, as Estimated by Sandia.

		Average Individual and Population Doses from One-Year Release for 1980 Startup						
Radionuclide	Organ	3 km mrem/yr	Regional μrem/yr	Regional person-rem/yr	U.S. μrem/yr	U.S. person-rem/yr	World μrem/yr	World person-rem/yr
Kr-85	Effective	0.37	10	24	2	560	2	7,900
	Lung	0.75	21	47	5	1,100	4	16,000
	Skin	12	350	790	79	19,000	60	260,000
	Gonads	0.44	12	14	3	330	2	4,600
H-3	Effective	3.2	89	200	16	3,900	0.2	1,000
	Gonads	3.2	44	100	17	2,000	0.2	500
I-129	Thyroid Infant	1.4	40					
	Thyroid Adult	0.4	12					

Table 3.3. Table 2 from EPA 1973. Doses from 1 Year of Plant Operation from a 5 Tonne/Day Nuclear Fuel Reprocessing Plant, as Estimated by the EPA.

Average Individual and Population Doses from One-Year Release for 1980 Startup					
Radionuclide	Organ	3 km mrem/yr	Regional person-rem/yr	U.S. person-rem/yr	World person-rem/yr
Kr-85	Effective	0.38	24	520	8,100
	Lung	0.75	47	1,000	16,000
	Skin	13	790	17,000	270,000
	Gonads	0.50	14	300	4,700
H-3	Effective	3.2	200	3,700	1,100
	Gonads	3.2	100	1,800	570
I-129	Thyroid Infant	1.4	2.3	2	
	Thyroid Adult	0.4	27	85	

The EPA also considered doses resulting from the release of I-131. According to the EPA (EPA, 1973), the I-131 doses dominate over the ones from release of I-129 by about an order of magnitude for infants and about a factor of 2 for adults. This estimate appears to be based on a fuel cooling time of 150 days. It should be noted that I-131 has a half-life of only eight days. In a reactor accident at a nuclear power plant, where there may be less than a day for I-131 to decay before it is released into the atmosphere, this isotope is extremely important. For reprocessing, it is unlikely that any measurable quantity of I-131 would remain in the fuel, especially if cooling time is one year or more. Here, we consider that levels of I-131 are negligible and do not need to be considered for estimating health effects to the public. This would be true even if the iodine DF were 1 instead of 1000.

3.2.5 Dose Conversion Factors

This section discusses DCFs because current values are more accurate than 1973 values. DCFs are the factors used to convert radioactivity (curies) to a dose sustained by an individual receptor (rem). The DCFs were studied to see if the changes make any difference in the effect of a release. Table 3.3 compares the DCFs used in the EPA (1973: Appendix C) report, with more current values that are consistent with ICRP-72 (ICRP, 1999) and Federal Guidance Report (FGR)-13 (EPA, 1999). The DCFs used in the EPA report are taken from ICRP-8. The DCFs for immersion can be compared directly; comparison of the DCFs for inhalation of H-3 requires that a breathing rate and retention factor be assumed to convert from time integrated air concentration to quantity retained by inhalation; comparison of the DCFs for ingestion of H-3 requires that a drinking rate be assumed to convert from water concentration to quantity ingested. Comparisons of the overall DCFs for ingestion of I-129 are more difficult to evaluate because they involve the food pathway leading to concentration of iodine in milk. Ingestion DCFs for I-129 are not included in Table 3. 4 but are discussed below.

Table 3.4. Comparison of DCFs from the EPA Report (EPA, 1973, taken from ICRP, 1965) with Current DCFs (taken from ICRP, 1999, or EPA, 1999). All DCFs are in units of rem·cm^3/(yr·µCi).

Dose Conversion Factors for Immersion			
Isotope	**Organ**	**ICRP-8**	**ICRP-72**
Kr-85	Effective	1.5×10^4	2.8×10^4
	Gonads F	1.5×10^4	9.8×10^3
	Gonads M	2.0×10^4	1.4×10^4
	Lung	3.0×10^4	1.3×10^4
	Skin	5.0×10^5	1.5×10^6
Dose Conversion Factors for Inhalation			
H-3	Effective	1.7×10^6	5.7×10^5
Dose Conversion Factors for Ingestion			
H-3	Effective	1.0×10^2	5.5×10^1

There is no consistent trend between the older and newer DCFs; some have increased, and some have decreased over the years. The most important values in the table are the effective DCFs for Kr-85 and H-3, since these are the ones used to estimate health effects. The value for Kr-85 has increased by almost a factor of 2; on the other hand, the DCFs for H-3 have decreased by a factor of 2 to 3, depending on the dose pathway. This indicates that current dose estimates for release of Kr-85 would be higher than those in the EPA (1973) report, and those for H-3 would be lower than those in the report if the same assumptions were made for dilutions and populations.

DCFs for ingestion provided in ICRP documents are generally in units of dose per unit of ingested activity (e.g., Sv/Bq). The values given for I-129 in the EPA (1973, App. C) report are in units of dose per year of plant operation per unit concentration in air, (rem/yr) / (µCi/cm^3 air). The values in the EPA report for I-129 ingestion dose to the thyroid for infants and adults are 1.5×10^{13} and 4.6×10^{12} (rem/yr) / (µCi/cm^3 air), respectively. To obtain these values, EPA makes the following assumptions:

- Infants consume 0.76 l/dy of cow's milk; adults consume milk at the rate of 0.5 l/dy. From current agricultural statistics, the EPA value for adults overestimates milk consumption by about a factor of 2.
- The mean deposition velocity of aerosols containing I-129 is 0.5 cm/s. This assumption is reasonable.
- A key assumption is the fraction of I-129 to nonradioactive iodine (primarily I-127) that is present in the environment. This ratio influences the relative concentration of I-129 in the diet. This assumption is important because, once a steady state is achieved, the ratio of iodine isotopes in the thyroid is essentially the same as those in the diet. The amount of

environmental iodine varies from place to place within the U.S. The EPA report states that their assumption for environmental iodine overestimates the DCF by a factor of about 2.5 on the average. However, they do not acknowledge the fact that most of the U.S. population consumes iodized salt, which would further dilute the amount of I-129 in the diet, and thus in the thyroid.

- The EPA report assumes that dairy is produced locally (i.e., the cows producing the milk are raised in the same general area as the individual consuming the dairy products). This assumption is in opposition to the fact that most dairy in the U.S. is not produced locally. Furthermore, it does not account for the fact that plant siting or remedial action could ensure that milk was not produced in areas with significantly elevated levels of I-129.

Thus, taken together, the assumptions used by the EPA for doses from I-129 are very conservative. An independent calculation using the MACCS2 code indicates that more realistic estimates of I-129 doses are about two orders below those in the EPA report. This implies that the I-129 doses shown in Tables 3.2 and 3.3 could be achieved if the DF were only 10 instead of 1000, as assumed in the EPA report.

3.3 Health Effects Modeling

3.3.1 EPA Estimate of Health Effects from Kr-85, I-129, and H-3

Health effects are estimated by the EPA by multiplying the individual or population doses (in rem or person-rem) calculated in Section 3.2.1 through Section 3.2.3 by a risk factor that expresses the risk to an individual of a cancer incidence or cancer fatality per unit dose received. The risk factors for estimating health effects corresponding to population doses are also provided by the EPA. It should be noted that the EPA use of the term "health effects" is not synonymous with estimated latent cancer fatalities (LCFs). The overall number of health effects is shown in the category "Whole-body" by the EPA (1973) report. A footnote to these results states that mortality is 50%. In other words, half the health effects are estimated to be treatable and do not result in a fatality.

Table 3.5 presents the health effects derived by applying the risk factors used in the EPA (1973) report to the doses in Table 3.3 and accounting for 40 years of plant operation. The results in this table agree very closely with Table 3 of the EPA (1973) report, which is also shown below as Table 3.6. The caution noted in Subsection 3.2.3, that doses and health effects associated with H-3 release might have been overestimated in the EPA report, applies to the health effects estimated for H-3 shown in Table 3.5. This same comment also applies to the health effects from exposure to I-129 because those doses are very conservatively estimated, as discussed above in Subsection 3.2.5.

Table 3.5. Health Effects from 40 Years of Plant Operation from a 5 Tonne/Day Nuclear Fuel Reprocessing Plant (estimated by Sandia).

Radionuclide	Organ	Estimated Health Effects from 40 Years of Operation				
		3 km	Regional	U.S.	World	Total
Kr-85	Effective	6.0×10^{-6}	0.38	9.0	130	140
	Lung	1.5×10^{-6}	0.094	2.2	30	32
	Skin	1.5×10^{-6}	0.095	2.3	30	32
	Gonads	5.2×10^{-6}	0.17	4.0	55	59
H-3	Effective	5.2×10^{-5}	3.2	62	16	82
	Gonads	3.9×10^{-5}	1.2	24	6	31
I-129	Thyroid Infant	8.6×10^{-6}				
	Thyroid Adult	8.7×10^{-8}				

Table 3.6. Health Effects from 40 Years of Plant Operation from a 5 Tonne/Day Nuclear Fuel Reprocessing Plant (reproduced directly from Table 3 of EPA, 1973).

Radionuclide	Organ	Estimated Health Effects from 40 Years of Operation				
		3 km	Regional	U.S.	World	Total
Kr-85	Effective	6.0×10^{-6}	0.38	8.3	130	140
	Lung	1.5×10^{-6}	0.095	2.1	32	34
	Skin	1.5×10^{-6}	0.095	2.0	32	34
	Gonads	6.0×10^{-6}	0.17	3.6	57	61
H-3	Effective	5.2×10^{-5}	3.3	59	18	80
	Gonads	5.2×10^{-5}	1.2	22	6.9	30
I-129	Thyroid Infant	7.4×10^{-7}	0.012	0.024		0.04
	Thyroid Adult		0.02	0.12		0.14

The health effects under the category "Effective" represent the estimated health effects from the total radiation dose to all organs of the body. Specific types of cancers corresponding to the organs listed as "Lung," "Skin," and other organs not listed in the above table should approximately add up to the total health effects listed as "Effective." Although genetic effects from doses to the gonads were considered to be important at the time of the EPA study, they are now considered much less important than somatic effects for reasons discussed in the remainder of Section 3.

The risk of health effects to a person at 3 km from the site from exposure to I-129 cannot be directly compared with the value given in the EPA (1973) report. The values in Table 3.5 are for two specific age categories, infants (less than 1 year of age) and adults (above age 20). The value in the EPA report is a population-weighted average, including infants, persons aged 1 through 20, and adults.

The EPA assumed a DF for iodine of 1000. Because of this assumption, the health effects associated with the release of I-129 are roughly 2 orders of magnitude lower than those for H-3 for the region and the U.S., and the contribution to the total is negligible compared with either Kr-85 or H-3. Thus, the impact on predicted health effects would be small even if the DF for

iodine were allowed to be 10 or 100 instead of 1000. Furthermore, as noted above, the I-129 doses leading to these estimates of health effects are conservative by about two orders of magnitude. As also noted previously, health effects associated with I-131 can be made completely negligible by allowing sufficient cooling time for the spent fuel. Because the half-life for this isotope is only eight days, this requirement is easily fulfilled.

Keeping in mind that only half of the total health effects are fatalities, the EPA predictions are for a total of 70 fatalities per year of plant operation from Kr-85 and 41 fatalities per year of plant operation from H-3. The predicted fatalities from all other radionuclides combined are less than 1 per year of plant operation. However, it should be further noted that most of the induced health effects are assumed to be from doses that are in the range of μrem/yr. As discussed in Section 3.3.3, the health effects induced by such small doses cannot currently be quantified with confidence. The health effect estimates provided above are based on the LNT dose-response model, which is highly uncertain at low dose rates.

3.3.2 Historical Evolution of the Dose-Response Model

Table 3.7 briefly traces the assessment of exposure to ionizing radiation and the development of lifetime dose-response effects models of ionizing radiation from 1965 to the present. Pre-1980 data on health effects of ionizing radiation were developed primarily by studying the survivors of the Hiroshima and Nagasaki bombings, since this was the first sizable exposed population identified. The survivors have been studied throughout their lives, a study generally referred to as the Life Span Study, or LSS (NAS, 2006). As more became known about the chronic effects of ionizing radiation, effects on workers who handled radioactive materials (e.g., the Radium Dial Painters' Study; NCRP, 1987) and on individuals who had been treated therapeutically with ionizing radiation (e.g., treatment for ankylosing spondylitis; Hall, 1994) were included in the data. The National Academy of Sciences / National Research Council Committee on the Biological Effects of Ionizing Radiation (BEIR Committee) began its analysis of these data in 1972 (NAS, 1972).

As Table 3.7 shows, estimates of U.S. background radiation increased markedly from 100 mrem/year in 1965 to an average of 365 mrem/year at present. This increase is due in part to improved measurement capability and in part to the recognition of background sources that are not easily measured, like radon. In spite of this increase by a factor of about 3.5, the dose-response model—the number of chronic health effects per unit of ionizing radiation—has remained remarkably constant at 4×10^{-4} to 6×10^{-4} health effects per rem of exposure relative risk. The LNT theory of radiation damage was first formulated in BEIR I[1] (NAS, 1972) and forms the basis for the risk estimates, explaining why there has been so little variation in those estimates in 35 years. The LNT is discussed in greater detail in Section 3.3.3.

[1] This document is the first report issued by the BEIR. The authors refer to it as BEIR I rather than BEIR to avoid confusion.

Table 3.7. Estimates of Background Radiation and Postulated Dose Response Models.

DOCUMENT	DATE	"NATURAL" U.S. BACKGROUND	HEALTH EFFECTS PER UNIT DOSE – LIFETIME EXPOSURE	COMMENTS
ICRP 8	1965	80 to 100 mrem/year		General discussion of health effects and the uncertainty surrounding them.
BEIR I	1972	180 mrem/year natural (102 mrem/year) +medical +fallout+nuclear power	5.68×10^{-4}/ rem/year: relative;1.15×10^{-4} absolute	The LNT theory is used, but emphasis is put on uncertainty and the lack of any low-dose data at all. Iodine concern is with I-131.
Fuel Cycle EIS	1973	200 mrem/year natural+medical +fallout+nuclear power	0.39/rem/year over 40 years to U.S. population PROJECTED	These are from a 5 metric ton/day reprocessing plant.
UNSCEAR	1977		4.03×10^{-4}/ rem/year: relative; 1.58×10^{-4} absolute	Linear/quadratic; cited in BEIR III.
BEIR III	1980	210 mrem/year natural+medical +fallout+nuclear power	1.69×10^{-4}/ rem/year: relative; 0.67×10^{-4} absolute	Linear/quadratic model, not extrapolated to zero. Iodine concern is with I-131.
BEIR V	1990	360 mrem/year natural+medical +fallout+nuclear power	5.6×10^{-4}/ rem/year relative	Average males and females. LNT extrapolated to zero. Low-dose only.
ISCORS Technical Report 1	2002		6×10^{-4}/rem/year relative	From EPA documents.
BEIR VII	2006	365 mrem/year natural+medical +fallout+nuclear power	6.1×10^{-4}/ rem/year relative	Average males and females. LNT extrapolated to zero. Low-dose only. Includes DDREF.

The risk factors used in the EPA (1973) report are taken from BEIR I (NAS, 1972); values currently used for estimation of health effects for the NRC are taken from BEIR V (NAS, 1990); more recent values are contained in BEIR VII (NAS, 2006). Table 3.8 compares the risk factors used in the EPA (1973) report, taken from BEIR I, with the ones currently used for NRC analyses, taken from BEIR V.

Table 3.8. Comparison of Latent Cancer Risk Factors (Health Effects/Person-rem) Used in the EPA (1973) Report with Values Taken from BEIR V.

	Latent Cancer Risk Factors (Health Effects per Person-Rem)		Ratio
Organ Dose	BEIR I	BEIR V	BEIR V/BEIR I
Effective	2.0×10^{-4}	5.8×10^{-4}	2.88
Lung	5.0×10^{-5}	9.4×10^{-5}	1.88
Skin	3.0×10^{-6}	2.0×10^{-9}	6.7×10^{-4}
Gonads	3.0×10^{-4}		
Thyroid Infant	1.5×10^{-4}		
Thyroid Adult	5.0×10^{-6}	6.5×10^{-6}	1.30

Most of the risk factors in Table 3.8 are for cancer fatalities. The risk factor used in the EPA (1973) report for the "Effective" dose was 4.0×10^{-4} with the statement that half the cancers result in mortality. So that the values could be directly compared, a factor of 0.5 was applied to the value for the risk factor displayed in the table. The statement given for the lung and gonad doses is that mortality is very high; therefore, the values from the EPA report are used without modification in Table 3.8. The comment for skin is that the mortality is low or zero. This is now better quantified, and the risk factor for skin has been reduced by about 3 orders of magnitude.

The risk factor for the gonads in the EPA report is for genetic effects, not latent cancers. The genetic effects that were anticipated have not been observed because they could not be distinguished from the very large number of random inherited genetic defects that occurs in human populations (NAS, 1990: Chapter 1). Chapter 2 of BEIR V states that random inherited genetic defects are so frequent in the human population that detecting a genetic defect caused by a low level of ionizing radiation is daunting, if it is even possible.

The most important comparison in Table 3.8 is the one for the "Effective" dose because this value is used to estimate the total number of latent cancers and latent cancer fatalities in the EPA (1973) report. As can be seen, this risk factor has increased by nearly a factor of 3 between BEIR I and BEIR V. The ISCORS (2002) recommended conversion from individual and collective dose to health risk ("risk factor") is used by EPA to calculate health risk. Thus, based on the current DCFs and cancer risk factors, the predicted LCFs would increase by about a factor of 5.4 for Kr-85 and would remain about the same for H-3 if all other assumptions were the same as those in the EPA report. The concept of collective dose as used by EPA is discussed further in Section 3.3.4.

3.3.3 LNT Theory of Radiation Damage and Risk

The above estimates of health effects depend on the assumption of an LNT dose response model. Using that model, predicted health effects are proportional to dose. For the dose to an individual, the "predicted health effect" is the probability that the exposed individual will sustain that particular effect.

The most important feature of BEIR I (NAS, 1972) was the postulate of a linear relationship between low doses of ionizing radiation and cancer, the extrapolation from the region of observed effects back to zero, and the possible sublinear relationship ("S-curve") in the very low dose region. The linear relationship introduced in BEIR I is generally referred to as the LNT hypothesis. BEIR I explains the linearity as the result of very low doses that are essentially single, independent radiation tracks through the receptor tissue. The dose-response model coefficients in Table 3.7 are derived from high-dose data (primarily Ra-226 exposure). Figure 3.5 shows the 1972 postulated dose-response relationships in BIER I that provided the bases for evaluation of the risks from low-level ionizing radiation. No scales were provided for the axes of Figure 3.5; it was apparently intended to show a general, approximate relationship between doses and health effects (responses).

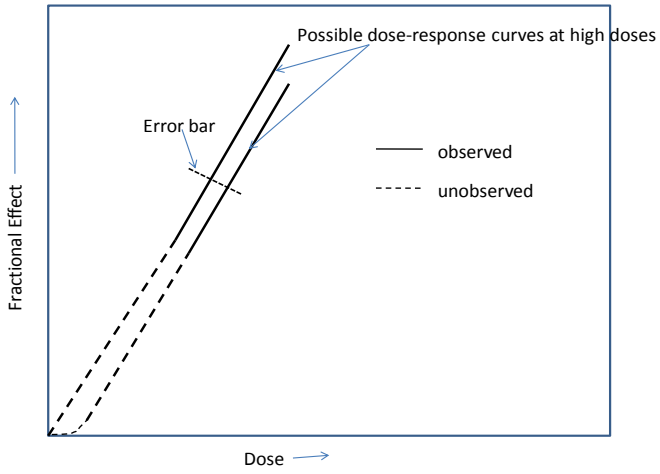

Figure 3.5. Bases of Evaluation of Risks of Low-level Radiation: Hypothetical Dose-effect Curves (reproduced from NAS, 1972: Figure 1).

BEIR III (NAS, 1980) considered both absolute risk, the risk of cancer per unit of ionizing radiation, and relative risk, the fraction of cancers above the expected number of cancers for the receptor's age and sex. BEIR III also projected cancers using both the linear model of BEIR I and a linear-quadratic model: linear at low doses, but cancer risk depending on the square of the dose at higher doses. A considerably larger data set was analyzed for BEIR III than for the 1972 study, and the resulting risks, both absolute and relative, are about 60% of the absolute risks reported in BEIR I and about 30% of the relative risks in BEIR III (NAS, 1980: Table V-25). Although the calculated risks are smaller, the relationship of health effects to radiation dose (the "dose-response" relationship) is assumed linear at low doses and assumes that there is no threshold below which there is an effect.

BEIR III (NAS, 1980) refined the LNT hypothesis with the policy of relative biological effectiveness (RBE), which assigned effectiveness factors based on the biological damage done by a particle of ionizing radiation and was generally related to linear energy transfer from an ionizing particle to biological tissue. For example, a gamma has an assigned RBE equal to 1, and a neutron, considered from 5 five to 20 times as effective in producing damage, has an RBE

of 5 to 20. Alpha particles are considered to have an RBE equal to 20. The effect of the RBE on the linear dose-response relationship is to make the slope of the line steeper.

BEIR V (NAS, 1990) greatly refined and expanded the database of studies on which risk estimates are based and included naturally emitted radon in the estimate of background dose, thereby raising the average U.S. background to 360 mrem/year (3.6 mSv/yr). Moreover, in addition to the RBE concept (called quality factor Q in BEIR V), the dose rate effectiveness factor (DREF) was introduced. The DREF is based on the observation that the effect of a radiation dose depends on the rate at which that dose is delivered: the slower the rate, the less the effect. At low doses in particular, since relatively little tissue is damaged, a low dose rate allows for repair of sublethal damage and other mitigating events. The DREF is a factor by which the slope of the linear dose-response is decreased. BEIR V (NAS, 1990: p. 22 et seq.) suggests a factor of 2 to 2.5, indicating that the LCF/rem ratio is half or less than that predicted by the LNT.

The effective dose equivalent (EDE) is also introduced in BEIR V (NAS, 1990: p. 17 et seq.). The EDE is defined as the product of the absorbed dose D, the RBE or quality factor Q, and the risk-weighted sum of the dose equivalents to individually irradiated tissues of the receptor organism. The sum of EDEs from all pathways (inhalation, external radiation, ingestion) is the total effective dose equivalent, TEDE. Pathways are shown in Figure 3.6.

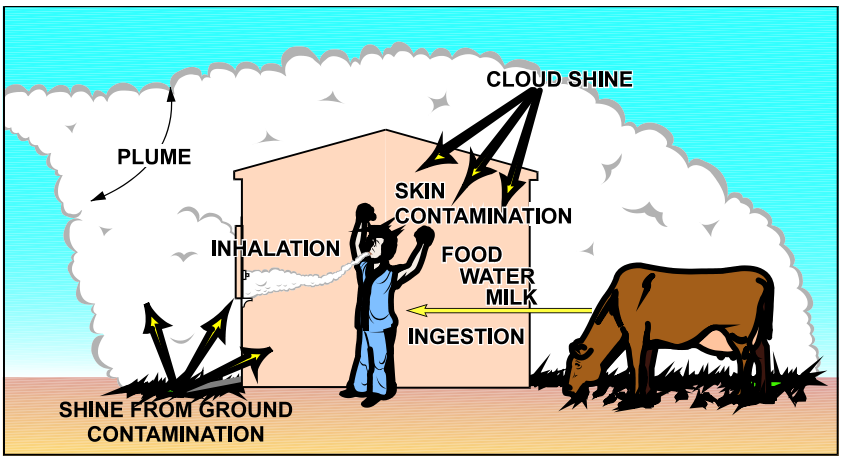

Figure 3.6. Radionuclide Pathways.

BEIR V also provided a valuable summary of the scientific phenomena underlying the risk estimates. The LCF-to-dose ratios of 5×10^{-4} LCF /rem (5×10^{-2} LCF/Sv) for a member of the public and 4×10^{-4} LCF/rem (4×10^{-2} LCF/Sv) for a worker are presented in BEIR V (1990: Table 4-2). This risk-to-dose ratio, 5×10^{-4} LCF/rem, is a risk-weighted mean of the dose to selected organs and tissues. It is derived from exposure to 0.1 rem/year (100 mrem/year or 1 mSv/yr) for a 70-year lifetime; 100 mrem/year is considered an acceptable dose internationally (e.g., ICRP, 1990).

An annual dose of 100 mrem/year and consequent risk of 5×10^{-4} LCF/rem are consistent with the 25 mrem/year whole body dose standard cited in 10 CFR 190. Moreover, the lifetime (70 year) risk of about 3.5×10^{-4} is considered an acceptable excess risk. In addition to providing the standard in 40 CFR 190, 25 mrem/year whole body dose and 75 mrem/year thyroid dose

have been established as the cleanup level for radioactive contamination under the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA) (Luftig, 1997).

In 2002, the (U.S.) Interagency Steering Committee on Radiation Standards (ISCORS) recommended a risk-to-dose ratio of 6×10^{-4} LCF/rem (6×10^{-2} LCF/Sv), increasing the risk per unit of radiation by 1.25. The rationale given is that the additional conservatism encompasses such factors as the difference between the doses delivered by external and internal radiation, any site specificity of the DREF (now called the dose and dose rate effectiveness factor, or DDREF), and other differences in dose models (ISCORS, 2002: p. 2).

BEIR VII (NAS, 2006) provides data in addition to those in BEIR V on the epidemiological and clinical data that serve as a basis for the understanding of the relationship between ionizing radiation and health effects. More important, BEIR VII addresses the growing controversy surrounding the LNT model of radiation damage. Figure 3.7 is the BEIR VII analog of Figure 3.5. The points in Figure 3.7 represent observations of health effects, usually cancers. The lines are fitted through the points and error bars, and they represent a linear fit and two linear-quadratic (L-Q) fits with different curvatures of the quadratic section. The vertical scale, "excess relative risk," represents cancer incidence beyond what is expected without the radiation dose.

Figure 3.7 illustrates the uncertainty that surrounds any estimate of risk from very small doses of ionizing radiation. Only one observation is in the "low dose" range, the range from zero to about 20 rem (0.2 Sv), and that observation appears to be at about 5 rem (0.05 Sv). The individual effective doses calculated for Table 3.1, 0.37 mrem 3 km from the source and 2 µrem (0.002 mrem) U.S. average, are so small, and such a small fraction of a dose with an observable effect, that any health effect from them is unlikely to occur. As discussed further in Section 3.3.4, estimating health effects in a population by multiplying these tiny doses by the number of people exposed yields a result that is not independent of dose distribution.

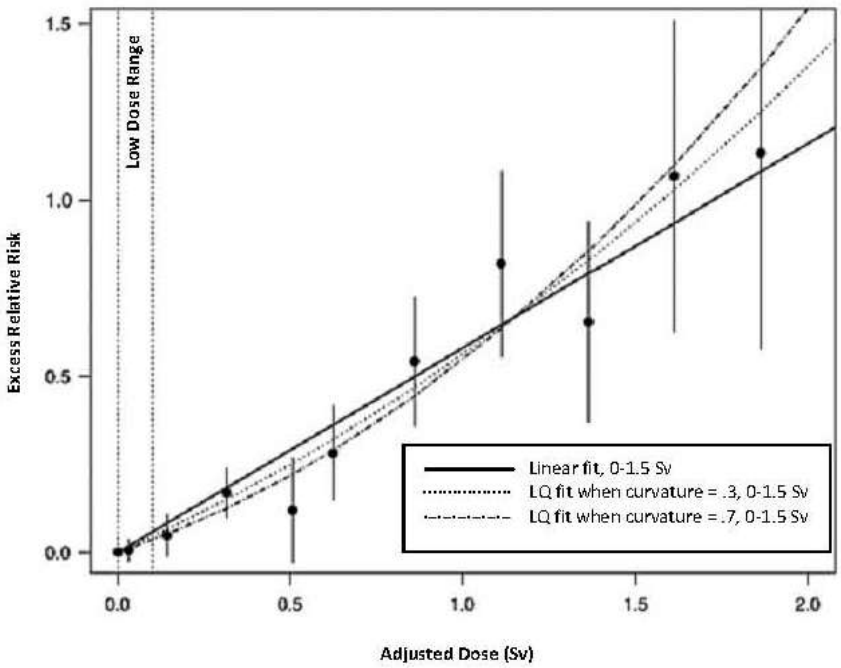

Figure 3.7. Linear and Linear-quadratic Dose-response Curves (reproduced from BEIR VII, NAS, 2006: Figure 10-2).

BEIR VII characterizes the LNT as "a computationally convenient starting point" (NAS, 2006: p. 7) and focuses on the large uncertainty in risk estimates, especially in the low-dose region where only one data point exists. Figure 3.7, adapted from BEIR VII, illustrates the statistics and uncertainty of cancer incidence from exposure to ionizing radiation. BEIR VII points out that the incidence of cancer in the U.S. population from undetermined sources is about 20%, or about 20 people out of 100. If those 100 people are exposed collectively to 10 person-rem (0.1 Sv), Figure 3.7 suggests that one additional cancer might result from the exposure. Any cancer resulting from a smaller dose delivered at a lower dose rate would be indistinguishable from expected cancers.

BEIR VII concludes that since data, albeit uncertain, are consistent with the LNT, and since no threshold has been identified, the LNT should continue to be the basis for estimating health risks of exposure to ionizing radiation. BEIR VII does not distinguish between application to individual and collective doses. The application of the LNT to cancer prediction and particularly to such predictions from collective dose estimates has been challenged recently, most notably by the Academie des Sciences and Academie Nationale de Medicine of France (Tubiana et al., 2005, 2006). These authors point out that the variety of cellular phenomena at very low doses (repair, cell death) make health effects nearly impossible to predict. Guerin (2006) noted the following:

> In the very low dose range [below 1 rad] there is a much more sensitive interplay of biological processes and phenomena than at medium (20 rad) or high doses (>100 rad)…at very low doses many different biological processes are activated or modulated, whereas at higher doses main-stream processes like cell cycle arrest, DNA repair or apoptosis…fully determine the cellular radiation response.

3.3.4 Collective Dose

The concept of collective dose, the total radiation dose to a population, was initially based on the principle that radiation exposures should be kept at the lowest practical level and was understood as applying to occupational exposures. This concept is now stated as "as low as reasonably achievable," or ALARA, and is a principle governing occupational exposure. Implementation of ALARA assumes the LNT—that some risk is associated with any radiation exposure. The risk to a population was assumed to be independent of the distribution of the dose. For example, the "risk incurred [by] 10 persons each receiving…1 rem would be identical to the risk incurred [by]…one person receiving 9.8 rem and the other nine receiving 0.1 rem each" (Auxier and Dickson, 1981).

Clearly problems exist in applying the concept of collective dose to a very small average dose distributed over a large population ("micro-doses to mega populations"). Auxier and Dickson (1981) noted that "uncertainty in measurement or calculation of dose to large populations may render the model unusable." Independence of distribution cannot be applied realistically to any large population. For example, the risk incurred by 1000 persons each receiving an average dose of 1 rem (0.01 Sv) is quite different from the risk to one person receiving 980 rem (9.8 Sv) and the other 999 receiving 20 mrem (0.2 mSv) each. In this example, the one individual suffers acute exposure, whose effects are quite different and far more severe than chronic exposure.

Tubiana et al. (2005, 2006) state that the product of a small dose and a large population—so large that distribution independence is violated—should not be multiplied by an LCF/rem factor and the result used as a prediction of cancer or other health effect because the uncertainties are too great for such a prediction to have any validity. The NRC has also been advised that collective doses should not be used to calculate health effects (Ryan, 2008; NRC, 2008a,b). Collective doses are useful in comparing different situations or radiation exposure, but not as absolute numbers.

Interpretation of collective dose is also problematic. When a risk factor like the one proposed by ISCORS (2002) is applied to a collective (population) dose (e.g., 6×10^{-4} per person-rem), the "predicted health effect" should be interpreted properly. The health effect—cancer—happens to one individual in the population. For example, in 100 people exposed to 10 rem for a total exposure of 10 person-rem, the "predicted health effect" is the probability of 0.006 (0.6%) that one individual in the 100 exposed will sustain that particular health effect. Such a risk factor also implies that 99.4% of the 100 exposed individuals will not sustain the health effect.

The cancer incidence rate in the U.S. population is about 20% (NAS, 2006). The 2009 population of the U.S. was 305.5 million (U.S. Census Bureau, 2009: www.census.gov), and 20% of these, 61 million, are statistically projected to sustain cancer from an unspecified cause. Table 3.5 cited nine potential cancers in the U.S. from the EPA projected expansion of nuclear power and reprocessing, or a cancer rate of $9/3.055 \times 10^{8}$ (3×10^{-6}%). The potential cancer risk would be increased from 20% to 20.000003% as a result of Kr-85 reprocessing emissions.

In sum, multiplying an average radiation dose by the number of people putatively exposed to it, and using that result (person-rem or person-Sv) to estimate health effects or cancers by

multiplying by 6 x 10^{-4} LCF/person-rem (6 x 10^{-2} LCF/person-Sv) or some other risk factor could give a misleading result. A better option is to concentrate on radiation dose rather than health effects and on doses to maximally exposed individual receptors rather than collective doses to populations.

The ICRP and the Health Physics Society have questioned whether population doses should be used to estimate health effects when members of the population receive significantly different doses. For example, ICRP suggests that population doses should not include doses to individuals below 10 mrem/yr. Using this recommendation, 10 mrem/yr effectively becomes a dose threshold below which no health effects are attributed. Table 3.2 shows that all the estimated doses are below this value even to individuals only 3 km from the plant. If a DF of 1000 is assumed for iodine, most of the doses are on the order of μrem/yr. Anticipated emissions from a recycling plant may not result in any observed or observable health effects.

BEIR VII (NAS, 2006: Chapter 9) discusses the weaknesses of epidemiologic studies of large or moderately sized populations (ecological studies). The primary weakness that BEIR VII ascribes to this type of study is that the unit analyzed is not the individual; individual sensitivities, life exposures to radiation, and individual risk factors differ widely and cannot be assumed to be averaged over a population. Another major weakness identified in BEIR VII is that distance from a radiation source or geographic location is used as a surrogate for dose (or exposure magnitude) and that everyone within a particular distance or at a particular location has the same exposure. Only when the exposure has been significant, as in the case of heavy fallout from the Chernobyl accident, are estimates of collective dose relevant. The collective dose concept should therefore be applied cautiously.

3.4 Cost Effectiveness

Section 3.0 of this report identified two main objectives of the EPA in setting the release limits found in 40 CFR 190 (EPA, 1976b: Vol. I). The first objective was that a complete assessment of the potential impact on public health be made, and the second was that the cost effectiveness of measures available to reduce radioactive effluents to the environment be carefully considered. The EPA went on to state that it would be irresponsible to set standards that impose unnecessary health risks on the public (unnecessary in the sense that exposures permitted by the standards can be avoided at a small or reasonable cost to the industry) and that it would be equally irresponsible to set standards that impose unreasonable costs on the industry (unreasonable in the sense that control costs imposed by the standards provide little or no benefit to the public). In this section, we examine the second of those objectives, cost effectiveness.

As discussed in Section 2.4, the EPA considered the basis for estimates of cost effectiveness of controls in terms of costs avoided per health effect as compared to other industries or parts of the nuclear fuel cycle, as well as assumptions regarding emission-capture technologies. Figure 3.8 shows the health effects remaining as a function of the cumulative cost (in 1976 dollars) of emission-capture technologies employed. The graph in the original EPA report (EPA, 1976b: Vol. I) includes a plot of emission-capture technologies using a larger number of technologies than is shown in Figure 3.8. The graph in the EPA report included emission-capture technologies available in 1976 as well as those that were anticipated to become available in the

future. Figure 3.8 includes technologies addressed by industry (INRA, 2009). Technologies identified in Figure 3.8 are shown where they appear on the graph in the EPA report.

Figure 3.8 illustrates that additional expenditures are not expected to significantly improve the reduction of health effects once technologies currently being considered are employed for I-129 (scrubber and zeolite capture) and cryogenic distillation for Kr-85. These technologies, which are currently being considered for use, are located at points on the graph beyond which additional expenditures do not yield significant reduction in effects remaining per GWe. From a cost-effectiveness perspective, technologies that lie at or near these points on the graph represent the most cost-effective expenditures for reduction of remaining health effects. Expenditures beyond these points on the graph are additional expenditures with no (or very little) reduction in remaining health effects.

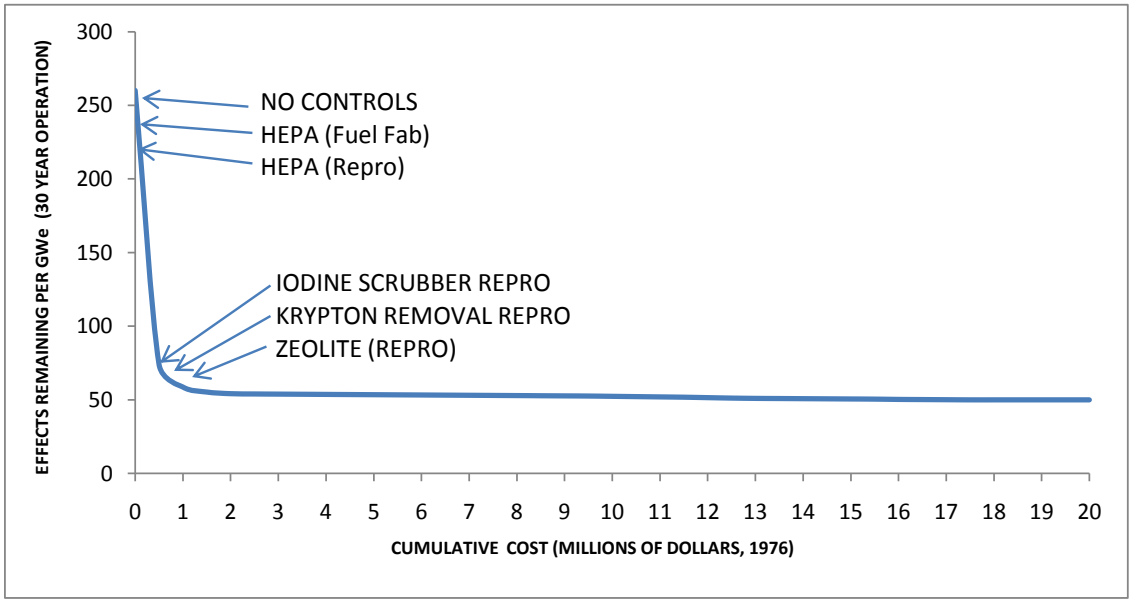

Figure 3.8. Cost Effectiveness of Decontamination Technologies.

Source: Modified from EPA (1976: Vol. 1, Figure 3.

3.4.1 Inflation Adjusted Cost Estimates for Decontamination Systems

This section examines costs associated with decontamination systems needed for compliance with 40 CFR 190. A comparison of cost estimates made by the EPA versus estimates made by industry is presented below.

Even when adjusted for inflation, industry estimates for the cost of decontamination systems needed for compliance are at least an order of magnitude higher than those developed by the EPA. The reasons for the difference are not readily apparent. The industry estimates may overestimate the cost of decontamination systems, the EPA estimates may underestimate the costs, or there may be other factors not yet known causing the difference in the estimates.

Further screening is warranted to determine the reasons for the differences in the decontamination system cost projections. It can be noted, however, that a number of the decontamination system technologies identified by the EPA were new or not yet fully developed in the 1970s, and the estimates from industry are based upon existing facilities. Consequently, the industry estimates may be closer to the current costs of decontamination systems.

The EPA cost estimates were obtained in 1975/1976 dollars for a 1500 MTHM/yr reprocessing facility (EPA, 1976b: Vol. I). To compare these estimates with recent industry estimates for a reprocessing facility, all 1975/1976 dollars were adjusted to 2008 dollars using the annual average U.S. inflation rate for each year from www.inflationdata.com (InflationData.Com, 2010).

Industry cost estimates were obtained for an 800-MTHM/yr facility (INRA, 2009) reported in 2008 dollars (Table 3.9).

Table 3.9. Financial Cost of Compliance (2008 Dollars).

EPA 1500 MTHM/yr				**Industry 800 MTHM/yr**
Isotope	Sequestration Technology	Annual Operating Cost Estimate	Capital Cost Estimate	Capital Cost Estimate
I-129	Silver zeolite beds and voloxidation	$1.8 M	$16 M	$300 M
Kr-85	Cryogenic distillation	$6.1 M	$120 M	$1.0 B

3.4.2 Value of Statistical Life Used in Support of 40 CFR 190

Cost-benefit analyses used in support of environmental regulations use a measure of society's willingness to pay for the control measures designed to reduce risk and protect human life. Often referred to as "value of a statistical life" (VSL), these values are controversial, often misunderstood, and very dependent on the method used to derive them.

The Final Environmental Statement (FES) for 40 CFR 190 noted that the breakpoint between efficient and inefficient radioactive effluent control options was about $500,000 per averted health effect. It is then stated, "Most current estimates of the acceptable limiting rate of investment for the future loss of life appear to fall at or below an upper limit of one-quarter to one-half million dollars" (EPA, 1976b: Vol. I., p. 51). The consistency between the control cost breakpoint and the FES authors' interpretation of socially accepted costs of life-saving measures (called VSL) provided additional justification for an acceptable upper value of $500,000/averted cancer for required control measures. Any discussion of possible revisions to 40 CFR 190 should take into account changes in the methods used to calculate VSL and the controversy surrounding the measure.

Even if adjusted for inflation, $500,000 in 1976 dollars adjusts to ~$2,000,000 in 2008 dollars. This value differs from VSLs used in support of regulations promulgated in the last few years (about $6.3 M–$7 M). In general, the VSLs used in policy analysis increased in the late 1970s and early 1980s because of the use of alternative methods to calculate the value of averted premature deaths. Based on his survey, Viscussi (1992) concluded that "most of the reasonable estimates of the value of life are clustered in the $3 M to $7 M range," with a mean value for the VSL of $4.8 M (in 1990 dollars). This study has been used as the basis for present-day EPA regulatory guidance. The most recent guidance issued by the EPA (2010) for economic analyses updated this VSL for inflation and recommended a Weibull distribution with a mean of $7.4 M (2006 dollars).

4.0 Key Observations

A number of key observations may be made from the material discussed in this report. Selected key observations identified are grouped according to the subsections in Section 3.0.

Radionuclide Isotopic Source

The actual growth of nuclear power in the U.S. has been considerably less than was projected by the EPA in 1977. The significant difference in growth would reduce the amount of I-129 and Kr-85 in the atmosphere when compared to that predicted by the EPA in the 1970s. Additionally, reprocessing, which is the basis of the EPA projection of Kr-85 and I-129 release, has not occurred in the U.S. The projected nuclear power growth is more than an order of magnitude higher than the actual nuclear power growth. Therefore, the build-up in the atmosphere of Kr-85 and I-129 as a consequence of nuclear power growth and reprocessing as envisioned by the EPA was greatly overestimated. Updated projections based on actual nuclear power growth and projected increases over the next few decades would significantly reduce the projected build-up of radionuclide isotopic source and likely affect the release limits, assuming release limits are to be based on the accumulated source term.

Fuel burn up and cooling time after release from the reactor assumptions made by the EPA resulted in a higher projection of Kr-85 releases than has actually been seen. Cooling spent fuel prior to reprocessing for about 30 years to allow for radioactive decay could alleviate the issue of Kr-85 release by actually reducing the Kr-85 in the reprocessing to a level below the current regulations. Assuming that the regulations did not change, this 30 year cooling by itself would eliminate the need to sequester Kr-85 and still be well within release limits. Since the U.S. nuclear industry is moving to higher burn up, they will also be moving to longer cooling time, resulting in much lower Kr-85 emissions.

Transport and Dose Modeling

The regional and U.S. dose and health-effect models used by the EPA are comparable to models in use today and are appropriate for a treatment of future risk. Furthermore, most of the underlying assumptions and parameter selections made by EPA seem to be reasonable. Therefore, the results from this part of the EPA analysis would not likely be changed using today's methods. One possible exception is ingestion dose from I-129.

EPA's assumptions for doses through the ingestion pathway of I-129 are highly conservative. An independent calculation indicates that these does are overestimated by possibly two orders of magnitude.

As a result of assuming a very short fuel cooling time, EPA also considered doses from I-131. However, this isotope has a short half-life, about eight days, and would easily decay to near zero levels by choosing an appropriate cooling time for the fuel.

EPA's treatment of worldwide doses from tritium assumes that it becomes diluted in only 1.8% of the water at or near the surface of the earth. Given that the half-life of tritium is more than 12 years, the residence time of tritium provides sufficient time for more dilution than was assumed

by the EPA. Thus, the EPA's estimate for worldwide doses from tritium exposure is likely conservative.

There is no consistent trend between the older and newer DCFs; some have increased, and some have decreased over the years. In particular, the effective dose coefficient for Kr-85 has increased by almost a factor of 2 while the one for H-3 has decreased by a factor of 2 to 3, depending on the dose pathway. This indicates that current dose estimates for release of Kr-85 would be higher than those in the EPA (1973) report, and those for H-3 would be lower than those in the report if the same assumptions were made for dilutions and populations.

Health Effects Modeling

While there has been no major change in assessing the physiological effects of ionizing radiation, biological effects of ionizing radiation on cells have been studied extensively since 1976. BIER VII (NAS, 2006) provides data in addition to BIER V on the epidemiological and clinical data that serve as a basis for the understanding of the relationship between ionizing radiation and health effects. The genetic effects that were anticipated have not been observed because they could not be distinguished from the very large number of random inherited genetic defects that occurs in human populations (NAS, 1990: Chapter 1). Therefore, somatic effects are currently considered more important.

BEIR VII concludes that since data, albeit uncertain, are consistent with the LNT, and since no threshold has been identified, the LNT should continue to be the basis for estimating health risks of exposure to ionizing radiation. However, BEIR VII does not distinguish between application to individual and collective doses. The application of the LNT to cancer prediction and particularly to such predictions from collective dose estimates has been challenged recently, most notably by the Academie des Sciences and Academie Nationale de Medicine of France (Tubiana et al., 2005, 2006). The uncertainty (error bar) in the relationship between dose and risk of health effects is very large and, unlike what one would expect from uncertainty, does not decrease for larger doses. A prediction of excessive risk should be applied with great caution in crafting any regulation, since it results in the imposition of excessive costs and can lead to poor public choices. BEIR VII also concludes that any regulation that invokes cancers should include a comparison to the general incidence of cancer of unidentified origin, about 20%. In comparing the risk from nuclear power to that from no nuclear power, the comparison is not a risk of 3×10^{-6} % as compared to zero risk, but a risk of 20.000003% as compared to 20%.

The following factors, which now have widespread use, were not utilized by the EPA.

- The "dose rate effectiveness factor (DREF)" appears to have been introduced because the prediction of cancer was seen as too conservative (i.e., too large). Although arguments are made (e.g., "…*the slower the [dose] rate, the less the effect*…"), the DREF is essentially a factor that is an attempt to reduce excessive conservatism.
- The total effective dose equivalent (TEDE) is a regulatory convenience that is, in fact, misleading since it adds external dose to committed internal dose, whereas external dose has different effects from internal committed dose.

The most problematic element of the EPA analysis was the use of the collective dose concept. In specific cases, the application of this concept is appropriate. However, applying this concept to a

very small average dose distributed over a large population ("micro-doses to mega populations") violates its underlying assumptions makes its use problematic. Auxier and Dickson (1981) noted that "uncertainty in measurement or calculation of dose to large populations may render the model unusable." In sum, multiplying an average radiation dose by the number of people putatively exposed to it, and using that result (person-rem or person-Sv) to estimate health effects or cancers by multiplying by 6×10^{-4} LCF/person-rem (6×10^{-2} LCF/person-Sv) or some other risk factor could give a misleading result. A better option is to concentrate on radiation dose rather than health effects and on doses to maximally exposed individual receptors rather than collective doses to populations.

Cost Effectiveness

The cost basis for the EPA estimates of emission-capture and decontamination technologies seems overly optimistic when compared to estimates from industry based on current operating facilities. Even when adjusted for inflation, industry estimates for the cost of compliance are at least an order of magnitude higher than those developed by the EPA. The reasons for the difference are not readily apparent. The industry estimates may overestimate the cost of decontamination systems, the EPA estimates may underestimate the costs, or there may be other factors not yet known causing the difference in the estimates. Further screening is warranted to determine the reasons for the differences in the decontamination system cost projections.

Estimates for the value of statistical life (VSL) were referenced in the EPA analysis. It was difficult to determine how important VSL was in the overall methodology, but today's estimates for VSL would be significantly higher than those used in 1977.

Path Forward

In summary, Sandia was able to reproduce results in technical basis documents that supported the development of 40 CFR Part 190. By doing so, Sandia was able to assess the technical bases by which the regulations were developed.

From this assessment a few areas were found to have important effects on the EPA analyses and these should be evaluated in more detail in future studies. These areas are summarized as follows:

- Realistic growth curves for nuclear power should be incorporated into any re-analysis for a more realistic set of regulations. Current projections for the growth of nuclear power still fall far short of the projections used in the EPA study. The projections were a major factor in risk-benefit analysis, and these should be re-baselined to reflect reality.
- The use of the collective dose methodology is problematic, especially for making worldwide projections of health effects. It would be preferable that dose and release standards should be based on the risk to a maximum exposed individual. Any re-evaluation of the risks from normal operation of the nuclear fuel cycle should take this approach.

- The decontamination system cost estimates used by EPA seem to be low compared to recent industry data. The cost estimates for decontamination systems should be re-baselined by industry to reflect currently available technologies.
- The calculated dose from I-129 appears to be inconsistent with current practice and should be evaluated. The I-129 dose is a key determinant in release regulation and should have further scrutiny.
- If doses from tritium release are re-evaluated, then the assumptions regarding tritium dilution should have further evaluation.
- The assumed cooling time period to reprocessing has a major effect on doses from Kr-85 and I-131. Realistic cooling times, probably within a typical range, would be worthwhile to pursue.

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