Analysis of Cancer Risks in Populations Near Nuclear Facilities: Phase I

ISBN
978-0-309-25571-4

424 pages
6 x 9
PAPERBACK (2012)

Committee on the Analysis of Cancer Risks in Populations near Nuclear Facilities-Phase I; Nuclear and Radiation Studies Board; Division on Earth and Life Studies; National Research Council

ANALYSIS OF CANCER RISKS IN POPULATIONS NEAR NUCLEAR FACILITIES

Phase 1

Committee on the Analysis of Cancer Risks in Populations near Nuclear Facilities—Phase 1

Nuclear and Radiation Studies Board
Division of Earth and Life Studies

NATIONAL RESEARCH COUNCIL
OF THE NATIONAL ACADEMIES

THE NATIONAL ACADEMIES PRESS
Washington, D.C.
www.nap.edu

THE NATIONAL ACADEMIES PRESS 500 Fifth Street, NW Washington, DC 20001

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This study was supported by Contract/Grant No. NRC-04-10-152 between the National Academy of Sciences and the U.S. Nuclear Regulatory Commission. Any opinions, findings, conclusions, or recommendations expressed in this publication are those of the author(s) and do not necessarily reflect the views of the organizations or agencies that provided support for the project.

International Standard Book Number-13: 978-0-309-25571-4
International Standard Book Number-10: 0-309-25571-6

Cover images: Regional maps of northeastern (front cover) and northwestern (back cover) states and populations living in census tracts within 30 miles of nuclear power plants. See Figures 4.4a-d for more information. Maps were created by committee member Lance Waller (Emory University, Atlanta, Georgia).

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Printed in the United States of America

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Acknowledgments

The committee wishes to acknowledge and thank a number of individuals and organizations for their valuable contributions to this study:

- The U.S. Nuclear Regulatory Commission (USNRC) for its sponsorship of this study, and especially USNRC staff members Brian Sheron, Terry Brock, Vered Shaffer, Marilyn Diaz, John Tomon, Scott Burnell, Dave McIntyre, Richard Conatser, John Cassidy, and Don Stearns. Dr. Brock, the USNRC's designated liaison to the committee, served as a guide to the USNRC's regulatory programs and the agency's vast collection of technical documents.
- The presenters at the committee's information-gathering meetings, who are listed in Appendix C.
- Members of the public and nongovernmental organizations who shared their perspectives and concerns about cancer risks associated with living near nuclear facilities. Some submitted extensive written information, notably: Beyond Nuclear, the Erwin Citizens Awareness Network, Inc., the C-10 Foundation, and Anthony and Ipatia Apostolides. Their comments helped to shape the epidemiologic study designs that are recommended in this report.
- The Electric Power Research Institute (especially Phung Tran), Raid Amin, University of West Florida, and Steve Wing, University of North Carolina at Chapel Hill, for providing written advice on epidemiologic study designs.
- Willie Harris and Robert Osgood, Dresden Nuclear Power Station, Kathy Yhip and Mike Russell, San Onofre Nuclear Generating Sta-

tion, and Marie Moore and Mark Elliott, Nuclear Fuel Services, Inc (NFS), for supporting facility visits by subgroups of committee members. Ms. Moore also provided information about NFS' effluent release records.

- Rosanne Aaberg (Pacific Northwest National Laboratory [PNNL]) and David Baker (PNNL, retired) for assistance in obtaining PNNL's archived data on nuclear plant effluent releases.
- Ralph Andersen and Andrew Mauer, Nuclear Energy Institute, for providing information about operating practices and effluent release records at nuclear power plants.
- Robert Anderson, National Center of Health Statistics, Centers for Disease Control and Prevention (CDC), Christie Eheman, National Center for Chronic Disease Prevention and Health Promotion, CDC, Kevin Ward, Georgia Center for Cancer Statistics, and Scott Boggess, U.S. Census Bureau, for guidance on sources of health and population data.
- Directors and staff of state departments of public health, cancer registries, and vital statistics offices for providing information about the availability and release of health data (see Appendixes K-M).

The committee extends special thanks to the staff of the National Research Council for supporting this study. Study director Kevin Crowley and program officer Ourania Kosti organized the committee meetings and assisted the committee with report writing and review. They also provided valuable insights and perspectives on many aspects of the study. Shaunteé Whetstone managed the logistics of the meetings, report review, and publication. Erin Wingo led the public communication efforts with help from Catherine Allen-West. Jennifer Walsh led the media communication efforts. Timothy Bouley assisted with information gathering and analysis. Roger Herdman, Institute of Medicine, provided guidance on identifying surveillance systems that collect information on population characteristics over time.

This report has been reviewed in draft form by individuals chosen for their diverse perspectives and technical expertise, in accordance with procedures approved by the Report Review Committee of the National Research Council. The purpose of this independent review is to provide candid and critical comments that will assist the National Research Council in making its published report as sound as possible and will ensure that this report meets institutional standards for objectivity, evidence, and responsiveness to the study charge. The review comments and draft manuscript remain confidential to protect the integrity of the deliberative process. We thank the following individuals for their participation in the review of this report:

- John Applegate, Indiana University
- John Boice, International Epidemiology Institute
- Christie Eheman, Centers for Disease Control and Prevention
- David Hoel, Medical University of South Carolina
- David Kocher, SENES Oak Ridge, Inc.
- Martha Linet, National Cancer Institute
- Paul Locke, Johns Hopkins Bloomberg School of Public Health
- Fred Mettler, New Mexico VA Healthcare System
- Beth Mueller, Fred Hutchinson Cancer Research Center
- Donald Pierce, Oregon Health and Science University
- Daniel Strom, Pacific Northwest National Laboratory
- John Till, Risk Assessment Corporation
- Richard Vetter, Mayo Clinic (retired)

Although the reviewers listed above provided many constructive comments and suggestions, they were not asked to endorse the contents of this report, nor did they see the final draft of the report before its release. The review of this report was overseen by Lynn Goldman, George Washington University, and Jon Samet, University of Southern California. Appointed by the National Research Council, Drs. Goldman and Samet were responsible for making certain that an independent examination of this report was carried out in accordance with institutional procedures and that all review comments were considered carefully. Responsibility for the final content of this report rests entirely with the authorizing committee and the institution.

Contents

Summary

The U.S. Nuclear Regulatory Commission (USNRC) requested that the National Academy of Sciences (NAS) provide an assessment of cancer risks in populations near USNRC-licensed nuclear facilities that utilize or process uranium for the production of electricity (see Sidebar 1.1 in Chapter 1 for the complete statement of task). These facilities presently include 104 operating nuclear reactors at 65 sites in 31 states and 13 fuel-cycle facilities in operation in 10 states. The operating fuel-cycle facilities include four in situ uranium recovery facilities, one conventional uranium mill, one conversion facility, two uranium enrichment facilities, and five fuel fabrication facilities (see Sidebar 1.2 in Chapter 1 for a description of these facilities). There are additional state-licensed conventional uranium milling facilities and in situ leaching facilities.

This USNRC-requested assessment is being carried out in two consecutive phases. The focus of the Phase 1 scoping study, which is the subject of this report, is to identify scientifically sound approaches for carrying out an assessment of cancer risks associated with living near a nuclear facility. The results of this Phase 1 study will be used to inform the design of the cancer risk assessment, which will be carried out in Phase 2. This report provides the committee's judgments about the strengths and weaknesses of various study approaches; these approaches differ in their broadness of approach, anticipated statistical power, ability to assess potential confounding factors, possible biases, and required effort.

Three findings and three recommendations emerged from this study. These are presented and discussed below. Additional supporting information can be found in the report.

FINDING 1: There are several challenges for carrying out epidemiologic studies of cancer risks in populations near U.S. Nuclear Regulatory Commission-licensed nuclear facilities in the United States, including the following:

- *Uneven availability and quality of data on cancer mortality and incidence at geographic levels smaller than a county.* Cancer mortality and incidence are tracked by individual states, and the availability and quality of data varies from state to state. In general, cancer mortality data are available electronically from about 1970, but subject address at time of death is not captured until much later in some states. (In the absence of subject address at time of death, mortality data cannot be geocoded at levels of geographic interest for an epidemiologic study, such as census tracts.) Cancer incidence data of known quality are generally available from about 1995, although such data are available for earlier times in some states. These data include address at time of diagnosis and have been widely geocoded, although there are residual problems associated with post office boxes and rural delivery addresses.
- *Uneven availability and quality of data on nuclear facility effluent releases*. Effluent release data may not be available and data quality may be poor for some nuclear facilities. Effluent releases from many nuclear facilities were much higher in the past and their radionuclide compositions have changed over time. Uncertainties in dose estimates may be much higher in years when effluent releases were highest.
- *Inability to reliably capture information on population mobility, risk factors, and potential confounding factors*. There is no centralized source of information on residential histories or lifestyle characteristics of individuals who live in the United States. The U.S. Census provides decadal snapshots of some population characteristics, including population size and distribution with respect to age, race/ethnicity, gender, educational level, and income. However, data on population lifestyle risk factors, including exposure to cigarette smoking and access to healthcare, are limited to state-level health surveys and are not consistently available from state to state at the same level of resolution. Moreover, populations near nuclear facilities receive radiation doses from multiple sources that are unrelated to facility effluent releases, for example, doses from natural background radiation and medical radiation. There may be other risk factors and potential confounding factors, for example, exposures to toxic chemicals and unidentified lifestyle factors, that can influence cancer risks.

- *Low expected statistical power.* Doses resulting from monitored and reported radioactive effluent releases from nuclear facilities are expected to be low. As a consequence, epidemiologic studies of cancer risk in populations near nuclear facilities may not have adequate statistical power to detect the presumed small increases in cancer risks arising from these monitored and reported releases.

The committee paid close attention to these challenges as it assessed the scientific merit of various epidemiologic study designs.

FINDING 2: An assessment of cancer risks in populations near nuclear facilities could be carried out using several study designs. Each design has strengths and limitations for estimating cancer risks.

- *Risk-projection models* estimate cancer risks by combining population radiation dose and/or dose surrogate (e.g., distance and direction from a nuclear facility) estimates with risk coefficients derived from epidemiologic studies of other exposed populations, for example, Japanese atomic bombing survivors. Risk-projection models can be used to estimate population-based cancer risks for any facility type, population size, and time period. However, because risk estimates are based on extrapolations from other epidemiologic studies and not on actual cancer incidence and/or mortality rates in populations near nuclear facilities, risk-projection models cannot be used to assess whether any predicted excess cancer risks correspond to observed patterns of cancer incidence or mortality.
- *Ecologic studies* estimate cancer risks by comparing observed cancer incidence and/or mortality rates in populations, considered as a group rather than as individuals, as a function of average radiation doses and/or dose surrogates for those populations. This design allows for the study of multiple cancer types during past and recent times, which helps to improve statistical power and provides a comprehensive picture of cancer risks. However, ecologic studies involve a large number of comparisons among population age groups, nuclear facilities, years of operation, and cancer types. This can lead to false associations resulting from chance alone. Moreover, ecologic studies can account only for population characteristics and potential confounding factors using group averages that are available from the decennial census and from survey information that can be linked to the census data (such as the American Community Survey). Individual characteristics can diverge sharply from group averages.
- *Cohort studies* estimate cancer risks by following individuals for a

specified period of time to determine the rate or risk of cancer as a function of doses and/or dose surrogates. In a *prospective* cohort study, subjects are followed from the present to a future time; in a *retrospective* cohort study, subjects are followed from a past time to a more recent time, usually via records. Prospective cohort studies can in principle provide the least-biased estimates of associations of multiple cancer types and radiation doses and/or dose surrogates compared to studies that rely on retrospective collection of information, such as case-control studies (described below) or retrospective cohorts. However, prospective cohort studies need to follow subjects for long time periods and could therefore require decades to complete. Retrospective cohort studies are more efficient than prospective studies because the follow-up period has already occurred. However, such studies rely on linkages such as those between birth certificates and state cancer registries; logistical and administrative barriers to such linkages could limit the feasibility of this study design in some states. Moreover, in- and out-migration issues need to be considered.

- *Case-control studies* estimate cancer risks by comparing radiation dose and or dose surrogates between individuals selected because they have (cases) or do not have (controls) cancer. The individuals under study and cancer outcomes of interest must be predefined and for practical reasons would be limited to one or a few cancer types (for example, pediatric cancers). A challenge in case-control studies is to select suitable controls in a way that does not bias the study results.

In the absence of information on residential history, most studies by necessity make assumptions about relevant exposures based on information about location of residence at one time point in the lifetime of the study cases, such as place of residence at time of birth or place of residence at time of diagnosis or death, with the equivalent time for controls. This single time point of place of residence may not be the most relevant regarding exposure from the nuclear facilities.

Studies that are based on individuals, such as cohort and case-control studies, can potentially provide stronger evidence for or against an association between radiation exposure and cancer compared to an ecologic study that is based on groups of individuals (i.e., populations). However, such studies are likely to involve fewer cancer cases than an ecologic study due to the effort involved in subject selection and individual data collection. The effort involved in conducting a cohort or a case-control study could be reduced by partnering with existing multistate cancer studies that have already linked cancer and birth registration data.

Case-control studies can involve contacting subjects to collect residential history and lifestyle information through interviews and questionnaires. Such studies would need to be limited to recently diagnosed cancer cases (i.e., diagnoses made during the past 5 years) and would likely be subject to additional selection and information biases. There are added difficulties in obtaining appropriate approvals from the cancer registries before subjects could be contacted. However, such studies can also be carried out without subject contacts by using information from birth and other administrative records.

FINDING 3: Effluent release, direct exposure, and meteorology data, if available, can be used to obtain rough estimates of annual variations in dose as a function of distance and direction from nuclear facilities.

Effluent release and direct exposure data collected by facility licensees are likely to be sufficiently accurate to develop a population-level dose reconstruction that provides rough estimates in annual variations in dose as a function of distance and direction from nuclear facilities. However, such data would not be sufficient to support detailed reconstructions of doses to specific individuals living near nuclear facilities. However, it will be necessary to develop a methodology for estimating releases of carbon-14 prior to 2010 to support dose estimation (carbon-14 may be a significant contributor to dose from nuclear plant releases, especially in recent years). Moreover, facility-specific evaluations will be required to determine the quality and availability of effluent release and meteorology data as well as meteorology data for batch releases. Obtaining and digitizing effluent release and meteorology data for use in an epidemiologic study will be a large and costly effort.

Environmental monitoring data have limited usefulness for estimating absorbed doses from effluent releases around nuclear plants and fuel-cycle facilities. Almost all environmental measurements reported by facilities are either below the minimum detection limits or are not sensitive enough to allow for the development of useful dose estimates.

Computer models have been developed to estimate absorbed doses resulting from airborne and waterborne radioactive effluent releases. These models combine information on effluent release timing and magnitude, transport of the released effluents through the environment, and the exposure of individuals to radiation from these releases to estimate absorbed doses. Such models could be used to obtain rough estimates of doses to support an epidemiologic study. An existing model could be adapted for this purpose or a new model could be developed. Regardless of the approach used, it is essential that the model reflect modern practices for dose reconstruction, including approaches for estimating uncertainties.

Absorbed doses near nuclear facilities are anticipated to be low, in most cases well below variations in levels of natural background radiation in the vicinity of individual facilities. Absorbed doses are also anticipated to be below levels of radiation received by some members of the public from medical procedures and air travel. Consequently, dose estimates used in an epidemiologic study would ideally account for these other sources of radiation exposures and possibly for other risk factors such as exposure to hazardous (and potentially carcinogenic) materials released from nearby industrial facilities.

RECOMMENDATION 1: Should the U.S. Nuclear Regulatory Commission decide to proceed with an epidemiologic study of cancer risks in populations near nuclear facilities, the committee recommends that this investigation be carried out by conducting the following two studies, subject to the feasibility assessment described in Recommendation 2: (1) an ecologic study of multiple cancer types of populations living near nuclear facilities and (2) a record-linkage-based case-control study of cancers in children born near nuclear facilities.

Brief descriptions of these recommended studies are provided below. A list of strengths and weaknesses of the recommended studies and additional details on the study designs can be found in Chapter 4.

The ecologic study should assess cancer incidence and mortality in populations within approximately 50 kilometers (30 miles) of nuclear facilities for the operational histories of those facilities to the extent allowed by available data. A study zone of this size would incorporate both the most potentially exposed as well as essentially unexposed regions to be used for comparison purposes. The study should examine all relatively common cancer types by age interval and gender, including cancers that are not considered to have a radiogenic origin (presumed nonradiogenic cancers such as prostate cancer can serve as useful negative controls) and also take into account temporal changes in estimated radiation dose. A subanalysis should specifically be carried out for highly radiogenic cancers such as leukemia in children. The study should examine associations between (i) cancer and distance and direction from the nuclear facility and (ii) cancer and estimated radiation dose, both at the census-tract level. The committee recommends that absorbed doses to individual organs be estimated using the methodology outlined in Chapter 3.

The record-linkage-based case-control study should assess the association of childhood cancers (diagnosed at younger than 15 years of age) in relation to maternal residential proximity at the time of birth of the child, among those whose address at time of delivery was within a 50-kilometer radius of a nuclear facility. The study period for individual facilities should

be based on the quality and availability of cancer registration in each state. Controls born within the same 50-kilometer radius as the cases should be selected from birth records to match cases on birth year at a minimum. Absorbed doses and/or dose surrogates should be based on address of the mother's place of residence at time of delivery, as determined from birth records.

These recommended studies are complementary: The ecologic study would provide a broad investigation of both cancer incidence and mortality over the operational histories of nuclear facilities to the extent allowed by available data. The analysis will be based on place of residence at time of cancer diagnosis or at time of death from cancer. The committee's recommended approach for carrying out this study would improve on the 1990 National Cancer Institute survey[1] (these improvements are described in Chapter 4). The record-linkage-based case-control study of childhood cancers would attempt to provide a more focused assessment of the association of these cancers in relation to early life exposure to radiation during more recent operating periods of nuclear facilities. An analysis based on maternal residence at time of delivery of the child may be considered more appropriate for capturing relevant exposures.

The committee has recommended these two studies based primarily on scientific merit, feasibility, and utility for addressing public concerns about cancer risks. However, the decision about whether to carry out one or both of these studies is the responsibility of the USNRC. In making this decision, the Commission will consider a number of factors, some of which are outside the charge for this Phase 1 study such as cost and priority of addressing public concerns about cancer risks near Commission-licensed nuclear facilities versus other agency priorities. As noted in this summary and discussed in detail in Chapter 4, the statistical power of epidemiologic studies of cancer risks in populations near nuclear facilities is likely to be low based on currently reported effluent releases from those facilities. Moreover, the magnitude of the variation of other risk factors that may not be measurable such as smoking or exposure to medical radiation may surpass the expected effect from the releases of the nuclear facilities and therefore overwhelm the actual effect attributed to the releases. Nevertheless, there may be sound policy reasons for proceeding with these studies: They can help to address public concerns about cancer risks and also demonstrate the USNRC's commitment to working constructively with its stakeholders.

[1]Jablon, S., Z. Hrubec, J.D. Boice, Jr., and B.J. Stone (1990). Cancer in populations living near nuclear facilities, Volumes 1-3, NIH Publication No. 90-874; Jablon, S., Z. Hrubec, et al. (1991). Cancer in populations living near nuclear facilities. A survey of mortality nationwide and incidence in two states. *JAMA* 265(11):1403-1408.

RECOMMENDATION 2: A pilot study should be carried out to assess the feasibility of the committee-recommended dose assessment and epidemiologic studies and to estimate the required time and resources.

Additional work beyond the scope of this Phase 1 study will be required to assess the feasibility of these recommended studies and to estimate the time and resources needed to carry them out. The recommended pilot study is designed to develop this information. The pilot study should focus on the four activities described below. Additional details can be found in Chapters 3 and 4.

- Obtain effluent release and meteorology data for six nuclear plants and one fuel-cycle facility (the committee suggests Dresden, Millstone, Oyster Creek, Haddam Neck, Big Rock Point, San Onofre, and Nuclear Fuel Services; see Chapter 2) and digitize these data into a form that is usable for dose estimation. The pilot should also develop a methodology for estimating releases of carbon-14 from the six nuclear plants for all years of operations for which effluent release data are available.
- Develop a computer model (i.e., by modifying an existing model or developing a new model) to obtain estimates of absorbed doses to individual organs resulting from airborne and waterborne effluent releases, and use this model to obtain dose estimates as a function of distance (0 to 50 kilometers from the plant) and direction for each of these seven facilities. Methodologies should also be developed to account for natural background radiation and, to the extent feasible, other sources of radiation in the dose estimates, especially medical radiation. An analysis should be carried out to estimate dose uncertainties.
- Retrieve cancer incidence and mortality data at the census-tract level within 50 kilometers of these seven facilities to assess feasibility of the recommended ecologic study.
- Confer with investigators who are conducting linkages of cancer and birth registration data to identify eligible cases of pediatric cancers and matched controls to assess feasibility of the recommended record-linkage-based case-control study. Where such linkages are not already in place, link birth registration and cancer incidence data to identify eligible cases of pediatric cancers and matched controls.

RECOMMENDATION 3: The epidemiologic studies should include processes for involving and communicating with stakeholders. A plan for

stakeholder engagement should be developed prior to the initiation of data gathering and analysis for these studies.

Stakeholder engagement is an essential element of any risk assessment process that addresses important public interests (see Chapter 5). Several approaches were used in this Phase 1 study to engage with stakeholders. The Phase 2 study can build on these Phase 1 efforts to achieve effective collaboration with local people and officials and increase social trust and confidence. To this end, the Phase 2 study should develop and execute an engagement plan that includes processes to:

- Identify key stakeholders and stakeholder groups with whom engagement is essential.
- Assess stakeholder concerns, perceptions, and knowledge.
- Communicate the questions that the Phase 2 study can address and its strengths and limitations, and communicate the results from the Phase 2 study in forms that are useful to different stakeholder groups.
- Make the information used in the Phase 2 study publicly accessible to the extent possible.

It is important that the plan be developed prior to the initiation of data gathering and analysis to ensure early engagement with stakeholders in the Phase 2 study.

1

Introduction

The U.S. Nuclear Regulatory Commission (USNRC) requested that the National Academy of Sciences (NAS) provide an assessment of cancer risks in populations near USNRC-licensed nuclear facilities. This assessment is being carried out in two consecutive phases. The focus of the Phase 1 scoping study, which is the subject of this report, is to identify scientifically sound approaches for carrying out an assessment of cancer risks. The results of this Phase 1 study will be used to inform the design of the assessment, which will be carried out in Phase 2. The complete study task is shown in Sidebar 1.1.

The USNRC-licensed nuclear facilities referred to in the statement of task are nuclear power reactors and nuclear fuel-cycle facilities that utilize uranium for the production of electricity.[1] These facilities are described in Sidebar 1.2. A large number of nuclear facilities have been constructed in the United States during the past six decades. Presently licensed USNRC facilities include:

- 104 operating nuclear reactors (35 boiling water reactors and 69 pressurized water reactors) at 65 sites in 31 states (Table 1.1).
- 13 fuel-cycle facilities in operation in 10 states. The operating facilities include four in situ uranium recovery facilities, one conventional uranium mill,[2] one conversion facility, two uranium enrichment facilities, and five fuel fabrication facilities. There are

[1]These are referred to as *nuclear plants* and *fuel-cycle facilities* in this report; the more generic term *nuclear facilities* is used to refer to nuclear plants and fuel-cycle facilities collectively.

[2]Currently on standby (i.e., available for operations but not currently operating).

SIDEBAR 1.1
Statement of Task

The National Academies will provide an assessment of cancer risks in populations living near U.S. Nuclear Regulatory Commission-licensed nuclear facilities. This assessment will be carried out in two consecutive phases:

A Phase 1 scoping study will identify scientifically sound approaches for carrying out the cancer epidemiology study that has been requested by the U.S. Nuclear Regulatory Commission. It will address the following tasks:

1. Methodological approaches for assessing off-site radiation dose, including consideration of:

   - Pathways, receptors, and source terms
   - Availability, completeness, and quality of information on gaseous and liquid radioactive releases and direct radiation exposure from nuclear facilities
   - Approaches for overcoming potential methodological limitations arising from the variability in radioactive releases over time and other confounding factors
   - Approaches for characterizing and communicating uncertainties.

2. Methodological approaches for assessing cancer epidemiology, including consideration of:

   - Characteristics of the study populations (e.g., socioeconomic factors, all age groups, children only, and nuclear facility workers)
   - Geographic areas to use in the study (e.g., county, zip codes, census tracts, or annular rings around the facility at some nominal distances)
   - Cancer types and health outcomes of morbidity and mortality
   - Availability, completeness, and quality of cancer incidence and mortality data
   - Different epidemiological study designs and statistical assessment methods (e.g., ecologic or case-control study designs)
   - Approaches for overcoming potential methodological limitations arising from low statistical power, random clustering, changes in population characteristics over time, and other confounding factors
   - Approaches for characterizing and communicating uncertainties.

The results of this Phase 1 scoping study will be used to inform the design of the cancer risk assessment, which will be carried out in Phase 2.

additional state-licensed[3] conventional uranium milling facilities and in situ leaching facilities that are not shown on Table 1.2.[4]

Figures 1.1a and 1.1b show the locations of currently operating nuclear plants and USNRC-licensed fuel-cycle facilities in the United States. Applications for 24 additional nuclear reactors were under active review by the USNRC while the present study was in progress.[5]

1.1 BACKGROUND ON THE STUDY REQUEST

In the late 1980s, the National Cancer Institute (NCI) initiated an investigation of cancer risks in populations near 52 commercial nuclear power plants and 10 Department of Energy nuclear facilities (including research and nuclear weapons production facilities and one reprocessing plant) in the United States (Jablon et al., 1990). The investigation compared cancer mortality rates in "study" counties (i.e., counties that contained nuclear facilities) with rates in "control" counties (i.e., counties that were similar to the study counties in terms of population size, income, education, and other socioeconomic factors but did not contain nuclear facilities). The NCI investigation also compared cancer registration (i.e., cancer incidence) rates in study and control counties in two states: Connecticut and Iowa. No differences in cancer mortality or incidence rates were observed between study and control counties. The authors of the study concluded that "if nuclear facilities posed a risk to neighboring populations, the risk was too small to be detected by a survey such as this one" (Jablon et al., 1991).

The USNRC has been using the results of this NCI investigation as a primary resource for communicating with the public about cancer risks near the nuclear facilities that it regulates. However, this study is now over 20 years old. There have been substantial demographic shifts in populations around some of these facilities, and the facility inventory itself has changed; some facilities have shut down and new facilities have started up. Additionally, at least one facility that was not included in the NCI investigation (Nuclear Fuel Services in Tennessee) has become a focus of public interest.

The NCI investigation had several limitations: The investigation utilized county-level mortality and, when available, incidence data. The use

[3]Section 274 of the Atomic Energy Act of 1954 authorizes the USNRC to enter into agreements with state governors to discontinue the Commission's regulatory authority for byproduct materials (radioisotopes), source materials (uranium and thorium), and certain quantities of special nuclear materials. States that have assumed regulatory authority for these materials are referred to as *agreement states*.

[4]A listing of these facilities as of 2010 can be found at http://www.eia.gov/uranium/production/annual/.

[5]See http://www.nrc.gov/reactors/new-reactors/col/new-reactor-map.html.

SIDEBAR 1.2
Nuclear Fuel Cycle

The *nuclear fuel cycle* comprises a set of industrial processes for producing electricity from uranium. These processes are carried out in nuclear fuel-cycle facilities, as illustrated in Figure S.1. Facilities comprising the *front end* of the nuclear fuel cycle are involved in the extraction of uranium from the environment and its fabrication into fuel for nuclear reactors. The uranium fuel is utilized in *nuclear power reactors* to produce electricity. Modern reactors typically generate on the order of 3000 megawatts of thermal power and produce about 1000 megawatts of electrical power. Facilities comprising the *back end* of the nuclear fuel cycle are involved in managing this fuel after it has been utilized in reactors; fuel management activities can involve recycling, storage, and/or disposal. The only civilian back-end facilities currently in operation in the United States are interim storage facilities for managing used fuel, most of which are located at commercial nuclear power plants. In the United States, almost all of these fuel storage facilities are co-located with nuclear plants.

The USNRC regulates five types of front-end fuel-cycle facilities:

Mining facilities: Facilities that are used to extract uranium from the environment. Currently, uranium is extracted using either *conventional mining* or *leaching* methods. The former method involves the physical removal of uranium-bearing ores from the subsurface in underground and open-pit mines. The latter method includes *in situ leaching*, in which solutions are pumped into the subsurface to extract uranium, and *heap leaching*, in which solutions are sprayed onto piles of mined rock to extract uranium. This study is concerned only with in situ leaching facilities. (The USNRC did not ask the NAS to examine conventional mining facilities because these facilities are not regulated by that agency.)

Milling facilities: Facilities that are used to process uranium ore or leach solutions to produce uranium oxide (U_3O_8) powder, or *yellowcake*. Mills can be standalone facilities, or they can be integrated into a uranium extraction operation. The former type of facility is used for conventional mining operations, where a single mill can service several mines, whereas the latter type of facility is used for in situ leaching operations.

Conversion facilities: Facilities that are used to convert yellowcake into a solid hexafluoride form (uranium hexafluoride, UF_6). This compound sublimes to form a gas at about 56°C at standard atmospheric pressures. The gaseous form of this material is used in subsequent processing steps.

Enrichment facilities: Facilities that are used to increase the concentration of

of countywide data made it difficult to discern local effects around nuclear facilities, especially in geographically extensive counties. The investigation also focused primarily on cancer mortality, because good-quality cancer incidence data were largely unavailable at the time the study was conducted. (Incidence may be a better indicator of risk than mortality because advances in cancer treatments have lowered mortality rates for many types of cancer, including leukemia.)

uranium-235 in uranium hexafluoride. Almost all natural uranium contains about 99.3 percent uranium-238 and about 0.7 percent uranium-235 by mass. Enrichment increases the mass percentage of uranium-235, the fissile (i.e., the component of the nuclear fuel that can be induced to fission with thermal [low-energy] neutrons) component of nuclear fuel, to between about 4 and 5 percent. In the United States, uranium enrichment is currently being carried out in gaseous diffusion and centrifuge plants. New plants that use laser enrichment technologies are under construction.

Fuel fabrication facilities: Facilities that are used to convert enriched uranium hexafluoride into a uranium dioxide (UO_2) solid and fabricate it into nuclear fuel for civilian reactors.

Some of the fuel facilities being considered in this study have had or currently have dual civilian and defense missions. Prior to the USNRC assuming regulatory control, some of these facilities were previously regulated by the U.S. Department of Energy and its predecessor agencies.

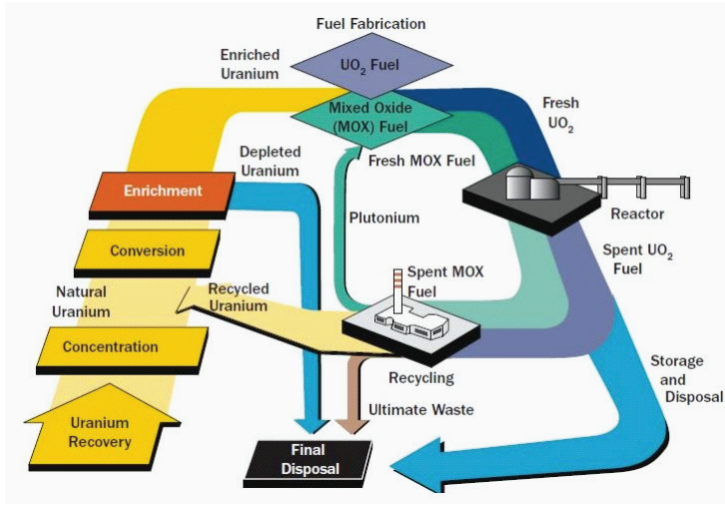

FIGURE S.1 Schematic depiction of the nuclear fuel cycle. SOURCE: USNRC.

The NCI investigation also did not attempt to estimate radiation exposures resulting from the operation of nuclear facilities. However, NCI investigators noted that such exposures are likely to be "too small to result in detectable harm" (Jablon et al., 1991, p. 1407). Absent reliable information about radiation exposures, it is difficult to provide scientifically supportable explanations for any observed associations between a nuclear facility and cancer incidence or mortality.

TABLE 1.1 Civilian Nuclear Power Plants in the United States

State	Number of Active Reactors in State	Name of Nuclear Power Plant (USNRC-abbreviated plant names)	Reactor Unit	Operating License Issue Date	Shutdown Date
Alabama	5	Browns Ferry Nuclear Plant (Browns Ferry)	1	1973	
			2	1974	
			3	1976	
		Joseph M. Farley Nuclear Plant (Farley)	1	1977	
			2	1981	
Arizona	3	Palo Verde Nuclear Generating Station (Palo Verde)	1	1985	
			2	1986	
			3	1987	
Arkansas	2	Arkansas Nuclear One (Arkansas Nuclear)	1	1974	
			2	1978	
California	4	Diablo Canyon Power Plant (Diablo Canyon)	1	1984	
			2	1985	
		San Onofre Nuclear Generating Station (San Onofre)	1	1967	1992
			2	1982	
			3	1982	
		Humboldt Bay Nuclear Power Plant (Humboldt Bay)	3	1963	1976
		Rancho Seco Nuclear Generating Station (Rancho Seco)		1974	1989
Colorado	1	Fort Saint Vrain Generating Station (Fort Saint Vrain)		1973	1989
Connecticut	2	Millstone Power Station (Millstone)	1	1970	1998
			2	1975	
			3	1986	
		Haddam Neck (Connecticut Yankee)		1968	1996
Florida	5	Crystal River Nuclear Generating Plant (Crystal River)	3	1976	
		St. Lucie Plant (St. Lucie)	1	1976	
			2	1986	
		Turkey Point Nuclear Plant (Turkey Point)	3	1972	
			4	1973	
Georgia	4	Edwin I. Hatch Nuclear Plant (Edwin I. Hatch)	1	1974	
			2	1978	
		Vogtle Electric Generating Plant (Vogtle)	1	1987	
			2	1989	

TABLE 1.1 Continued

State	Number of Active Reactors in State	Name of Nuclear Power Plant (USNRC-abbreviated plant names)	Reactor Unit	Operating License Issue Date	Shutdown Date
Illinois	11	Braidwood Station (Braidwood)	1 2	1987 1988	
		Byron Station (Byron)	1 2	1985 1987	
		Clinton Power Station (Clinton)	1	1987	
		Dresden Nuclear Power Station (Dresden)	1 2 3	1959 1969 1971	1978
		LaSalle County Station (LaSalle)	1 2	1982 1983	
		Quad Cities Nuclear Power Station (Quad Cities)	1 2	1972 1972	
		Zion Nuclear Power Station (Zion)	1 2	1973 1973	1997 1996
Iowa	1	Duane Arnold Energy Center (Duane Arnold)		1974	
Kansas	1	Wolf Creek Generating Station (Wolf Creek)	1	1985	
Louisiana	2	River Bend Station (River Bend)	1	1985	
		Waterford Steam Electric Station (Waterford)	3	1985	
Maine	0	Maine Yankee Nuclear Power Plant (Maine Yankee)		1972	1996
Maryland	2	Calvert Cliffs Nuclear Power Plant (Calvert Cliffs)	1 2	1974 1976	
Massachusetts	1	Pilgrim Nuclear Power Station (Pilgrim)		1972	
		Yankee Rowe Nuclear Power Station (Yankee-Rowe)		1961	1991

continued

TABLE 1.1 Continued

State	Number of Active Reactors in State	Name of Nuclear Power Plant (USNRC-abbreviated plant names)	Reactor Unit	Operating License Issue Date	Shutdown Date
Michigan	4	Donald C. Cook Nuclear Plant (Cook)	1 2	1974 1977	
		Palisades Nuclear Plant (Palisades)		1971	
		Fermi	1 2	1966 1985	1992
		Big Rock Point Nuclear Plant (Big Rock Point)		1962	1997
Minnesota	3	Monticello Nuclear Generating Plant (Monticello)	1	1970	
		Prairie Island Nuclear Generating Plant (Prairie Island)	1 2	1974 1974	
Mississippi	1	Grand Gulf Nuclear Station (Grand Gulf)	1	1984	
Missouri	1	Callaway Plant (Callaway)	1	1984	
Nebraska	2	Cooper Nuclear Station (Cooper)		1974	
		Fort Calhoun Station (Fort Calhoun)	1	1973	
New Hampshire	1	Seabrook Station (Seabrook)	1	1990	
New Jersey	4	Hope Creek Generating Station (Hope Creek)	1	1986	
		Oyster Creek Nuclear Generating Station (Oyster Creek)		1969	
		Salem Nuclear Generating Station (Salem)	1 2	1976 1981	

TABLE 1.1 Continued

State	Number of Active Reactors in State	Name of Nuclear Power Plant (USNRC-abbreviated plant names)	Reactor Unit	Operating License Issue Date	Shutdown Date
New York	6	James A. FitzPatrick Nuclear Power Plant (FitzPatrick)		1974	
		R. E. Ginna Nuclear Power Plant (Ginna)		1969	
		Indian Point Nuclear Generating (Indian Point)	1 2 3	1962 1973 1975	1974
		Nine Mile Point Nuclear Station (Nine Mile Point)	1 2	1969 1987	
		Shoreham Nuclear Power Station (Shoreham)		1989	1992
North Carolina	5	Brunswick Steam Electric Plant (Brunswick)	1 2	1976 1974	
		McGuire Nuclear Station (McGuire)	1 2	1981 1983	
		Shearon Harris Nuclear Power Plant (Harris)	1	1986	
Ohio	2	Davis-Besse Nuclear Power Station (Davis-Besse)	1	1977	
		Perry Nuclear Power Plant (Perry)	1	1986	
Oregon	0	Trojan Nuclear Power Plant (Trojan)	1	1976	1992
Pennsylvania	9	Beaver Valley Power Station (Beaver Valley)	1 2	1976 1987	
		Limerick Generating Station (Limerick)	1 2	1985 1989	
		Peach Bottom Atomic Power Station (Peach Bottom)	1 2 3	1967 1973 1974	1974
		Susquehanna Steam Electric Station (Susquehanna)	1 2	1982 1984	
		Three Mile Island Nuclear Station (Three Mile Island)	1 2	1974 1978	1979
		Shippingport Atomic Power Station		1957	1982
		Saxton		1962	1972

continued

TABLE 1.1 Continued

State	Number of Active Reactors in State	Name of Nuclear Power Plant (USNRC-abbreviated plant names)	Reactor Unit	Operating License Issue Date	Shutdown Date
South Carolina	7	Carolinas-Virginia Tube Reactor		1963	1967
		Oconee Nuclear Station (Oconee)	1 2 3	1973 1973 1974	
		H.B. Robinson Steam Electric Plant (Robinson)	2	1970	
		Virgil C. Summer Nuclear Station (Summer)	1	1982	
		Catawba Nuclear Station (Catawba)	1 2	1985 1986	
South Dakota	0	Pathfinder Atomic Plant (Pathfinder)		1964	1967
Tennessee	3	Sequoyah Nuclear Plant (Sequoyah)	1 2	1980 1981	
		Watts Bar Nuclear Plant (Watts Bar)	1	1996	
Texas	4	Comanche Peak Nuclear Power Plant (Comanche Peak)	1 2	1990 1993	
		South Texas Project	1 2	1988 1989	
Vermont	1	Vermont Yankee Nuclear Power Station (Vermont Yankee)		1972	
Virginia	4	North Anna Power Station (North Anna)	1 2	1978 1980	
		Surry Power Station (Surry)	1 2	1972 1973	
Washington	1	Columbia Generating Station (Columbia)		1984	
Wisconsin	3	Kewaunee Power Station (Kewaunee)		1973	
		Point Beach Nuclear Plant (Point Beach)	1 2	1970 1973	
		La Crosse Nuclear Generating Station (La Crosse)		1969	1987

TABLE 1.2 USNRC-Licensed Facilities that Are Part of the Nuclear Fuel Cycle

Site Name, Location	Licensee	Operational Status
In situ Recovery Facilities[a]		
Crow Butte, NE	Crow Butte Resources, Inc.	Active
Crownpoint, NM	Hydro Resources, Inc.	Not yet constructed
Moore Ranch, WY	Uranium One	Active
Smith Ranch and Highlands, WY	Power Resources, Inc.	Active
Willow Creek, WY	Uranium One	Active
Conventional Uranium Mill Recovery Facilities[a]		
Ambrosia Lake, NM	Rio Algom Mining, LLC	Decommissioning
Church Rock, NM	United Nuclear Corp.	Decommissioning
Homestake, NM	Homestake Mining Co.	Decommissioning
Bear Creek, WY	Bear Creek Uranium Co.	Decommissioning
Gas Hills, WY	American Nuclear Corp.	Decommissioning
Gas Hills, WY	Umetco Minerals Corp.	Decommissioning
Highlands, WY	Exxon Mobil Corp.	Decommissioning
Lucky Mc, WY	Pathfinder Mines Corp.	Decommissioning
Shirley Basin, WY	Pathfinder Mines Corp.	Decommissioning
Split Rock, WY	Western Nuclear, Inc.	Decommissioning
Sweetwater, WY	Kennecott Uranium Corp.	Stand-by
Uranium Hexafluoride Conversion Facility		
Metropolis, IL	Honeywell International, Inc.	Active
Uranium Fuel Fabrication Facilities		
Wilmington, NC	Global Nuclear Fuels-Americas, LLC	Active
Columbia, SC	Westinghouse Electric Company, LLC Columbia Fuel Fabrication Fac.	Active
Erwin, TN	Nuclear Fuel Services, Inc.	Active
Lynchburg, VA	AREVA NP, Inc. Mt. Athos Road	Inactive
Lynchburg, VA	B&W Nuclear Operations Group	Active
Richland, WA	AREVA NP , Inc.	Active
Mixed Oxide Fuel Fabrication Facility		
Aiken, SC	Shaw AREVA MOX Services, LLC	Under construction
Gaseous Diffusion Uranium Enrichment Facilities		
Paducah, KY	USEC Inc.	Active
Piketon, OH	USEC Inc.	In cold shutdown
Gas Centrifuge Uranium Enrichment Facilities		
Piketon, OH	USEC Inc.	In construction
Eunice, NM	Louisiana Energy Services	Active
Idaho Falls, ID	AREVA Enrichment Services	Under review

continued

TABLE 1.2 Continued

Laser Separation Enrichment Facility		
Wilmington, NC	GE-Hitachi	Under review
Uranium Hexafluoride Deconversion Facility		
Hobbes, NM	International Isotopes	Under review

[a]There are additional in situ recovery facilities and conventional uranium mill recovery facilities that are licensed by USNRC agreement states. See the text for details.
SOURCE: USNRC (2011).

The USNRC initially contracted with the Center for Epidemiologic Research at Oak Ridge Associated Universities (ORAU) to assess the feasibility of updating the 1990 NCI investigation. Two methodological approaches were outlined by ORAU: The first was the methodology used in the original 1990 NCI investigation, which utilized county-level data. The second involved an analysis of cancer mortality within 3, 10, 30, and 50 miles from nuclear facilities using more advanced spatial analysis techniques. The ORAU investigators concluded that both approaches were feasible (ORISE, 2009a).

ORAU also studied the feasibility of utilizing cancer incidence data collected either at the county level or by spatial analysis using census tracts or residential addresses. ORAU investigators concluded that there was sufficient cancer incidence data available in electronic form that could be used to update the NCI investigation (ORISE, 2009b).

Subsequently, the USNRC requested that the NAS undertake a de novo assessment of methodologies for carrying out an assessment of cancer risks that could go well beyond an update of the 1990 NCI study. That request resulted in the present study.

The NAS was asked to develop a design for a cancer epidemiologic study to assess potential cancer risks associated with living near USNRC-licensed nuclear facilities (see Sidebar 1.1). A decision about whether to carry out the Phase 2 epidemiologic study is the responsibility of the USNRC. In making this decision, the USNRC will consider a number of factors, some of which are outside the charge for this Phase 1 study. Factors may include scientific merit; the priority of addressing public concerns about cancer risks near USNRC-licensed nuclear facilities versus other agency priorities; and cost.

Epidemiologic studies may have a limited ability to discern associations between radiation exposure and cancer risk at low doses, even when large populations are examined. Additionally, epidemiologic studies of populations exposed to low radiation doses are likely to produce "false positive" associations (i.e., associations that occur purely by chance) if multiple

Index	Nuclear Power Plant, State	Index	Nuclear Power Plant, State
1	Browns Ferry, Alabama	34	Seabrook, New Hampshire
2	Farley, Alabama	35	Hope Creek, New Jersey
3	Palo Verde, Arizona	36	Oyster Creek, New Jersey
4	Arkansas Nuclear, Arkansas	37	Salem, New Jersey
5	Diablo Canyon, California	38	Fitzpatrick, New York
6	San Onofre, California	39	Ginna, New York
7	Millstone, Connecticut	40	Indian Point, New York
8	Crystal River, Florida	41	Nine Mile Point, New York
9	St. Lucie, Florida	42	Brunswick, North Carolina
10	Turkey Point, Florida	43	McGuire, North Carolina
11	Edwin I. Hatch, Georgia	44	Harris, North Carolina
12	Vogtle, Georgia	45	Davis-Besse, Ohio
13	Braidwood, Illinois	46	Perry, Ohio
14	Byron, Illinois	47	Beaver Valley, Pennsylvania
15	Clinton, Illinois	48	Limerick, Pennsylvania
16	Dresden, Illinois	49	Peach Bottom, Pennsylvania
17	LaSalle, Illinois	50	Susquehanna, Pennsylvania
18	Quad Cities, Illinois	51	Three Mile Island, Pennsylvania
19	Duane Arnold, Iowa	52	Oconee, South Carolina
20	Wolf Creek, Kansas	53	Robinson, South Carolina
21	River Bend, Louisiana	54	Summer, South Carolina
22	Waterford, Louisiana	55	Catawba, South Carolina
23	Calvert Cliffs, Maryland	56	Sequoyah, Tennessee
24	Pilgrim, Massachusetts	57	Watts Bar, Tennessee
25	Cook, Michigan	58	Comanche Peak, Texas
26	Palisades, Michigan	59	South Texas Project, Texas
27	Fermi, Michigan	60	Vermont Yankee, Vermont
28	Monticello, Minnesota	61	North Anna, Virginia
29	Prairie Island, Minnesota	62	Surry, Virginia
30	Grand Gulf, Mississippi	63	Columbia, Washington
31	Callaway, Missouri	64	Kewaunee, Wisconsin
32	Cooper, Nebraska	65	Point Beach, Wisconsin
33	Fort Calhoun, Nebraska		

FIGURE 1.1a Currently operating nuclear power plants in the United States.

Index	Licensee, State
1	Crow Butte Resources, Inc., Nebraska
2	Uranium One, Wyoming
3	Power Resources, Inc, Wyoming
4	Uranium One, Wyoming
5	Kennecott Uranium Corp.,[a] Wyoming
6	Honeywell International, Inc, Illinois
7	Global Nuclear Fuels-Americas, LLC, North Carolina
8	Westinghouse Electric Company, LLC Columbia Fuel Fabrication Fac., South Carolina
9	Nuclear Fuel Services, Inc., Tennessee
10	B&W Nuclear Operations Group, Virginia
11	AREVA NP, Inc., Washington
12	USEC Inc., Kentucky
13	Louisiana Energy Services, New Mexico

[a]Standby

FIGURE 1.1b Currently operating USNRC-licensed nuclear fuel-cycle facilities in the United States.

comparisons are made (e.g., for multiple cancer types) as well as "false negative" associations (i.e., associations not established because statistical power is low) because effect size is small. There is little way of knowing whether any such associations (or lack of associations) are anything more than statistical effects.

On the other hand, epidemiologic studies provide the most direct evidence for associations between suspected risk factors (e.g., radiation) and disease (e.g., cancer). Perhaps for this reason, epidemiologic studies continue to be used to assess cancer risks in populations near nuclear facilities in other countries (see Section 1.2 in this chapter and Appendix A). A well-

designed epidemiologic study can be used to formulate or test hypotheses about cancer risks in populations around nuclear facilities.

The committee received two somewhat conflicting messages from presenters at its information-gathering meetings (see Section 1.4 in this chapter) and peer reviewers for this report: (1) A Phase 2 epidemiologic study should be carried out; (2) the study will be a "political" rather than a "scientific" exercise. The committee has endeavored to recommend a technically sound approach for carrying out an epidemiologic study while at the same time clearly identifying the challenges for assessing cancer risks at low doses. The committee hopes that the USNRC will be able to use this information to help make an informed decision about whether to undertake a new epidemiologic study and what type of study to conduct.

1.2 PREVIOUS STUDIES OF CANCER RISKS

Concerns about the potential health impacts from living near nuclear facilities are not new or unique to the United States. A British television program in 1983 reported a cluster of childhood leukemia in Seascale, a village located on the coast of the Irish Sea about 3 kilometers from the nuclear fuel reprocessing facility at Sellafield. The television program reported on seven childhood leukemia cases in the village over the previous 30 years, whereas fewer than one case was expected (Urquhart et al., 1984). Given the proximity of the village to Sellafield, and the absence of other obvious causative agents, radioactive discharges from the reprocessing plant were hypothesized to be responsible for the excess leukemia.

The British government appointed an independent advisory group to investigate these claims. The group (Black, 1984) confirmed the leukemia cluster but could not link it to radioactive discharges. A governmental Committee on Medical Aspects of Radiation in the Environment (COMARE) was subsequently established in 1985 to undertake further investigations. To date, this committee has published 14 reports using data from the national registry of children's tumors (see Appendix A for literature review).

Since 1985, epidemiologic studies of cancer risks in populations near nuclear facilities have been carried out in at least 11 countries.[6] The majority of these studies investigated rates of cancer deaths or cancer occurrence in populations living in various-size geographic areas including counties and municipalities, zones of increasing distance, or zones based on models of dispersion of releases from the nuclear facilities (see Table 4.2, Chapter 4). These studies have come to different conclusions, with some suggesting a positive association between living in proximity to a nuclear facility

[6]Canada, Finland, France, Germany, Great Britain, Israel, Japan, Spain, Sweden, Switzerland, and the United States.

and cancer risk. However, studies have been unable to attribute positive associations to radioactive releases from the facilities.

A widely publicized study with a positive finding is the German *Kinderkrebs in der Umgebung von Kernkraftwerken* (KiKK) study, which was carried out by researchers from the German Childhood Cancer Registry in Mainz on behalf of the Federal Office of Radiation Protection. Study results were published in 2008 (Kaatsch et al., 2008; Spix et al., 2008). They indicated that for a child of age 0-5 years, the risk of developing leukemia doubles if that child lives in close vicinity of a nuclear plant. However, the methodology, presentation, and interpretation of results from the study have been strongly criticized by others (COMARE, 2011; Kinlen, 2011). Additional information about these studies is provided in Appendix A.

Results from two other epidemiologic studies were published during this Phase 1 study: the 14th report of COMARE, which provided further consideration of the incidence of childhood leukemia around nuclear plants in Great Britain (COMARE, 2011), and a study on the risk of childhood leukemia and all childhood cancers in the vicinity of Swiss nuclear plants (Spycher et al., 2011). Neither provided significant evidence of a positive association between distance from nuclear plants and cancer risk.

A third report from France showed that children living within 5 kilometers of nuclear plants are twice as likely to develop leukemia compared to those living 20 kilometers or farther away from the plants. However, analysis of the same population of children using a dose-based geographic zoning approach, instead of distance, did not support the findings. The authors suggest that the absence of any association with the dose-based geographic zoning approach may indicate that the observed association of distance and cancer risk may be due to some unidentified factors other than the releases from the nuclear power plants (Sermage-Faure et al., 2012). Current joint efforts from France and Germany are focusing on developing studies that would improve understanding of the positive associations between childhood leukemia and distance from nuclear power plants by improving current knowledge on the etiology of the disease.

Epidemiologic studies of cancer risks in populations near nuclear facilities have used a number of approaches to assess exposures of study populations to radiation from facility releases (see Section 4.2.1 in Chapter 4). In most cases, exposures are based on surrogate measures (e.g., distance from a facility) that are not related to quantifiable radiation doses. However, some recent studies have attempted to obtain dose estimates based on facility effluent releases. Evrard et al. (2006) grouped communes within 40 kilometers of nuclear plants in France into five categories based on estimated doses based on airborne radioactive effluent discharges (see Chapter 2) and local climate data. The Nuclear Safety Council and the Carlos III

Institute of Health (2009) estimated effective doses in populations living in municipalities at various distances from nuclear facilities in Spain.

More detailed dose reconstructions have been carried out for other applications. These include reconstruction of doses for World War II atomic bombing survivors in Japan; U.S. military personnel exposed to radiation from atmospheric nuclear-weapons testing; U.S. Department of Energy workers who were exposed to occupational radiation at nuclear weapons production and testing facilities and residents in nearby states who were exposed to radiation that was released from these facilities; and individuals who responded to the 1986 Chernobyl accident. These dose reconstruction efforts are described in a number of reports; see, for example, NCRP (2009) and NAS (1995).

1.3 STRATEGY TO ADDRESS THE STUDY CHARGE

This study was carried out by a committee of experts appointed by the NAS. The committee consists of 20 members with expertise that spans the disciplines relevant to the study task: biostatistics, contaminant fate and transport, environmental exposure monitoring, epidemiology, medicine, public health, radiation dosimetry, radiobiology, social science and risk communication, and toxicology. In selecting the committee, the NAS sought to obtain a balance between experts in the design and execution of risk assessment studies for low-dose radiation exposures and experts with relevant disciplinary expertise but no direct experience with low-dose radiation risk assessment. Biographical sketches of the committee members are provided in Appendix B.

The committee was tasked to recommend appropriate study design(s) to assess cancer risks associated with living near nuclear facilities. The selection of suitable study designs primarily involved judgments about scientific soundness, data availability and accessibility, and level of effort versus likely scientific return. The committee's judgments were also informed by information that it received from technical experts (see Appendix C) and comments from the public (see Chapter 5). The committee attempted to identify study approaches that were scientifically sound and that addressed public concerns.

The focus for this study is on cancer risks arising from exposures to radiation from nuclear plants and fuel-cycle facilities past and present in the course of their ordinary day-to-day operations. The study is not focused on risks arising from nuclear accidents (e.g., Chernobyl or, more recently, Fukushima). Nevertheless, the committee recognizes that public perceptions about the risks related to nuclear plants and fuel-cycle facilities may be shaped by these events.

One of the scientific challenges for carrying out assessments of cancer risks in populations near nuclear facilities is the lack of sufficient statistical power[7] to detect relatively small associations between cancer incidence or mortality and exposures to radiation from facility releases. This is primarily the result of the small radiation doses that are typically received by individuals living near nuclear facilities as a result of normal operations at those facilities (see Chapter 3). As a consequence, epidemiologic assessments of cancer risk require the study of very large populations to have any hope of having adequate statistical power to detect positive associations between cancer and radiation exposure. Modest improvements in the statistical power can be achieved by examining dose-response gradients, especially when the population under study is exposed to a range of doses.

Tables 1.3 and 1.4 show the populations living within 5 and 30 miles of currently operating nuclear facilities in the United States as determined in the 2010 census.[8] As can be seen in this table, there was a wide variation in the numbers of persons living near nuclear facilities in 2010:

- Approximately 1 million people lived within 5 miles of operating nuclear plants in 2010; over 45 million people lived within 30 miles.
- Approximately 116,000 people lived within 5 miles of USNRC-licensed operating fuel-cycle facilities in 2010; over 2 million people lived within 30 miles.
- Approximately 210 people lived within 5 miles of a USNRC-licensed operating in situ recovery or conventional uranium mill recovery facility in 2010; about 11,000 lived within 30 miles.[9]

The committee decided to focus most of its efforts in this Phase 1 study on nuclear plants because of their large associated populations. The committee decided not to consider mining and milling facilities in this Phase 1 study because of their low associated populations. The committee recognizes that people who live near these mining and milling facilities may be just as concerned about cancer risks as people who live near nuclear plants. However, epidemiologic studies of cancer risk would have no statistical

[7]That is, the ability of a statistical test to detect a predetermined difference in risk (e.g., a doubling in cancer mortality associated with radiation exposure) if it exists. In this context, statistical power depends on the risk in the control population, the smallest increase in risk the investigator wants to be reasonably sure of finding (if it is present), and the acceptable probabilities of a false positive result (if there is no increase) and a false negative result (if there is an increase of at least the size to be sought).

[8]The 2010 census data are used here simply to illustrate population differences for various facilities. The 2010 data do not reflect the population distribution around sites in prior years.

[9]Note: These are median estimates for individual in situ recovery or conventional uranium mill recovery facilities, not total populations for all facilities.

TABLE 1.3 Populations in the 5- and 30-Mile (Approximately 8- and 50-Kilometer) Zones around Currently Operating Nuclear Power Plants Based on the 2010 U.S. Census Data

Index	State	Name	5 Mile	30 Mile
1	Alabama	Browns Ferry Nuclear Plant	6,098	530,011
2		Joseph M Farley Nuclear Plant	2,534	186,768
3	Arizona	Palo Verde Nuclear Generating Station	1,117	273,806
4	Arkansas	Arkansas Nuclear One	14,177	137,107
5	California	Diablo Canyon Power Plant	1,648	338,602
6		San Onofre Nuclear Generating Station	23,525	2,410,113
7	Connecticut	Millstone Power Station	53,321	667,492
8	Florida	Crystal River Nuclear Generating Plant	6,142	271,625
9		St. Lucie Plant	34,017	584,465
10		Turkey Point	7,963	1,838,689
11	Georgia	Edwin I. Hatch Nuclear Plant	2,063	135,568
12		Vogtle Electric Generating Plant	1,941	398,181
13	Illinois	Braidwood Station	16,834	971,587
14		Byron Station	12,339	600,581
15		Clinton Power Station	1,643	419,698
16		Dresden Nuclear Power Station	22,872	1,815,892
17		LaSalle County Station	3,211	345,966
18		Quad Cities Nuclear Power Station	6,252	451,281
19	Iowa	Duane Arnold Arnold Energy Center	12,180	351,236
20	Kansas	Wolf Creek Generating Station	1,690	75,810
21	Louisiana	River Bend Station	5,647	536,645
22		Waterford Steam Electric Station	13,774	1,119,079
23	Maryland	Calvert Cliffs Nuclear Power Plant	18,438	443,962
24	Massachusetts	Pilgrim Nuclear Power Station	23,108	1,245,016
25	Michigan	Donald C. Cook Nuclear Plant	16,977	563,815
26		Palisades Nuclear Plant	7,693	288,716
27		Fermi	18,035	2,230,762
28	Minnesota	Monticello Nuclear Generating Plant	21,107	964,863
29		Prairie Island Nuclear Generating Plant	6,650	789,039
30	Mississippi	Grand Gulf Nuclear Station	1,657	87,677
31	Missouri	Callaway Plant	1,620	225,301
32	Nebraska	Cooper Nuclear Station	892	54,338
33		Fort Calhoun Station	9,305	829,567
34	New Hampshire	Seabrook Station	47,004	1,667,009

continued

TABLE 1.3 Continued

35	New Jersey	Hope Creek Generating Station	5,681	1,512,768
36		Oyster Creek Nuclear Generating Station	44,156	1,010,661
37		Salem Nuclear Generating Station	5,434	1,490,771
38	New York	James A. Fitzpatrick Nuclear Power Plant	10,838	615,046
39		R.E. Ginna Nuclear Power Plant	14,788	894,227
40		Indian Point Nuclear Generating	88,189	5,695,758
41		Nine Mile Point	6,729	307,622
42	North Carolina	Brunswick Steam Electric Plant	13,398	315,360
43		McGuire Nuclear Station	51,561	2,014,369
44		Shearon Harris Nuclear Power Plant	29,445	1,567,691
45	Ohio	Davis-Besse Nuclear Power Plant	3,390	733,031
46		Perry Nuclear Power Plant	24,164	810,777
47	Pennsylvania	Beaver Valley Power Station	16,181	1,656,510
48		Limerick Generating Station	97,649	4,453,399
49		Peach Bottom Atomic Power Station	11,326	1,787,122
50		Susquehanna Steam Electric Station	15,462	664,767
51		Three Mile Island Nuclear Station	48,714	1,520,777
52	South Carolina	Oconee Nuclear Station	15,616	634,339
53		H.B. Robinson Steam Electric Plant	11,927	292,920
54		Virgil C. Summer Nuclear Station	2,940	663,629
55		Catawba Nuclear Station	50,337	1,768,246
56	Tennessee	Sequoyah Nuclear Plant	29,485	714,473
57		Watts Bar Nuclear Plant	5,152	362,142
58	Texas	Comanche Peak Nuclear Power Plant	6,842	285,159
59		South Texas Project	1,691	66,066
60	Vermont	Vermont Yankee Nuclear Power Station	12,737	345,863
61	Virginia	North Anna Power Station	6,903	507,945
62		Surry Power Station	13,081	984,927
63	Washington	Columbia Generating Station	407	282,505
64		Kewaunee Power Station	2,974	324,911
65	Wisconsin	Point Beach Nuclear Plant	3,297	304,151
		Total:	934,488	45,020,247

NOTE: Plants in close geographic proximity may have overlapping populations, so persons living near those plants could be included (i.e., counted) in more than one plant population. The population total shown at the bottom of the table corrects for multiple counting (i.e., each person living near a plant is only counted once). As a consequence, the sum of the populations for the individual plants does not equal the population total at the bottom of the table.

TABLE 1.4 Populations in the 5- and 30-Mile (Approximately 8- and 50-Kilometer) Zones around Currently Operating USNRC-Licensed Facilities that Are Part of the Nuclear Fuel Cycle Based on the 2010 U.S. Census Data

Index	State	Licensee	Type	5 mile	30 mile
1	Nebraska	Crow Butte Resources, Inc	Mining	196	10,796
2	Wyoming	Uranium One	Mining	237	5,986
3		Power Resources, Inc	Mining	72	14,378
4		Uranium One	Mining	123	5,340
5	Wyoming	Kennecott Uranium Corp.[a]	Milling	21	1,438
6	Illinois	Honeywell International, Inc	Conversion	11,334	184,442
7	North Carolina	Global Nuclear Fuels-Americas, LLC	Fuel Fabrication	35,854	349,780
8	South Carolina	Westinghouse Electric Company, LLC Columbia Fuel Fabrication Fac.	Fuel Fabrication	14,512	796,391
9	Tennessee	Nuclear Fuel Services, Inc.	Fuel Fabrication	12,765	432,825
10	Virginia	B&W Nuclear Operations Group	Fuel Fabrication	21,810	280,396
11	Washington	AREVA NP , Inc.	Fuel Fabrication	33,253	276,038
12	Kentucky	USEC Inc.	Enrichment	7,370	190,772
13	New Mexico	Louisiana Energy Services	Enrichment	934	48,631
			Total:	116,282	2,308,747

NOTE: Facilities in close geographic proximity may have overlapping populations, so persons living near those facilities could be included (i.e., counted) in more than one facility population. The population total shown at the bottom of the table corrects for multiple counting (i.e., each person living near a facility is only counted once). As a consequence, the sum of the populations for the individual facilities does not equal the population total at the bottom of the table.

[a]Standby

power to detect associations between radiation and cancer because of these small populations.

With respect to the other types of fuel-cycle facilities, the committee focused most of its efforts on one facility, Nuclear Fuel Services in Erwin, Tennessee, primarily because of the public interest in cancer risks resulting from radioactive releases from that facility. The methodology proposed by the committee for assessing cancer risk at this facility is applicable to other fuel-cycle facilities as well.

1.4 INFORMATION GATHERING AND REPORT ORGANIZATION

The committee held five information-gathering meetings to receive briefings from subject-matter experts, including experts in the fields of epidemiology, dosimetry, and social science; representatives of the USNRC and the nuclear industry; representatives of cancer registries; and interested members of the public. Small groups of committee members visited the Dresden Nuclear Power Station (Illinois) in April 2011, the San Onofre Nuclear Generating Station (California) in July 2011, and the Nuclear Fuel Services facility (Tennessee) in October 2011 to learn about the design and operation of these facilities' radioactive effluent release and environmental monitoring programs. A list of committee meeting briefings is provided in Appendix C.

The committee's information-gathering sessions were webcast in an effort to enhance public awareness and participation in the study. Copies of these webcasts are available at http://www.nationalacademies.org/cancerriskstudy.

The committee received a large number of oral and written comments from nongovernmental organizations and other members of the public. These were helpful for informing the committee about public concerns related to the study and for uncovering data sources and documents that were useful to the committee.

This report is organized into five chapters that address the statement of task (Sidebar 1.1) in its entirety:

- Chapter 1 (this chapter) provides background on the study.
- Chapter 2 describes the effluent releases from nuclear facilities.
- Chapter 3 describes methods to estimate radiation exposure and dose from radioactive effluent releases and other sources.
- Chapter 4 describes epidemiologic study designs that could be used to investigate whether populations near nuclear facilities are at an increased risk of developing cancer.
- Chapter 5 describes the public engagement process used in this Phase 1 study and suggests how it can be extended for Phase 2.

Definitions of terms and acronyms are provided in Appendixes N and O, respectively.

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2

Effluent Releases from Nuclear Power Plants and Fuel-Cycle Facilities

This chapter addresses the following charge in the statement of task for this study (see Sidebar 1.1 in Chapter 1):

- Availability, completeness, and quality of information on gaseous and liquid radioactive releases and direct radiation exposure from nuclear facilities required to estimate doses for an epidemiologic study.

There are two potential sources of data on radiation releases from nuclear facilities that could be used to estimate doses for an epidemiologic study:

(1) Measurements of radioactivity contained in airborne[1] and liquid effluents that are released from nuclear facilities.
(2) Measurements of radiation in the environment around nuclear facilities.

This chapter describes these effluent release and environmental monitoring data and assesses their suitability for dose estimation. The primary focus is on effluent release data; as will be shown in this chapter, these data are more useful than currently available environmental monitoring data for estimating radiation doses for an epidemiologic study.

[1]The committee uses the term *airborne* to refer to gaseous and particulate releases to air and *liquid* or *waterborne* to refer to releases to water.

The effluent release and meteorological data collected by plant licensees and reported to the U.S. Nuclear Regulatory Commission (USNRC) are intended to demonstrate compliance with applicable USNRC regulations. These data were not intended to be used for dose reconstruction to support an epidemiologic study. The suitability of this information to support an epidemiologic study depends on the intended use of the dose reconstruction. For example, it might be necessary to obtain hourly or daily data on effluent releases and meteorological conditions at each facility to reconstruct doses to specific individuals living near those facilities. One the other hand, data that are averaged over longer time periods (weeks and months) might be sufficient to obtain rough estimates of annual doses to populations as a function of distance and direction from those facilities. Dose reconstruction is discussed in Chapter 3.

2.1 EFFLUENT RELEASES FROM NUCLEAR PLANTS

The operation of nuclear plants produces large quantities of radioactive materials (Appendix D). Quantities of radioactive materials are most readily expressed in terms of *activity*, defined as the rate of radioactive decay of that material. Activity is usually expressed in units of becquerels (abbreviated Bq; 1 Bq = 1 decay per second) or curies (abbreviated Ci; 1 Ci = 3.7×10^{10} [37 billion] decays per second).[2] An operating nuclear reactor can contain on the order of 10^{14} Ci of activity excluding very-short-lived radionuclides (NCRP, 1987). Most of this activity is the result of fission of the reactor fuel (see Appendix D).

A small fraction[3] of this activity is typically emitted to the environment each year as a result of normal plant operations. Radioactive effluents are released in airborne and liquid form. They originate from several sources within a nuclear plant:

- Fission of residual uranium contained on the exterior of the fuel rods, referred to as *tramp uranium*.
- Leaks from failed fuel rods.
- Diffusion of radioactive gases through intact fuel rods.
- Activation of materials in reactor cooling water.

[2]These units are used interchangeably in this chapter, depending on the source of data. International organizations generally use becquerels. Nuclear facility licensees and the regulator generally use curies.

[3]As will be shown elsewhere in this chapter (see Figures 2.1 through 2.4), operating nuclear plants currently release a few curies to a few hundred curies of activity per year to the environment. However, some plants emitted several hundred thousand curies of activity per year to the environment in the past.

- Erosion and entrainment of activated materials from pipes, valves, and pumps in the cooling system.

Effluent releases from nuclear plants are permitted under regulations promulgated by the USNRC, but they must be controlled, monitored, and reported to regulatory authorities. Appendix F describes USNRC requirements for reporting effluent releases from nuclear plants, and Appendix G describes the Radiological Effluents Technical Specifications (RETS) guidance for monitoring and reporting such releases.

Nuclear plant licensees are required to report emissions of radionuclides to the environment to the USNRC on an annual basis. Because nuclear power plants are industrial sites, plant licensees also are subject to environmental reporting requirements mandated by other federal and state regulatory agencies. These include industrial waste discharges (Clean Water Act), air emissions (Clean Air Act), chemical inventory reporting (Emergency Planning Community Right-to-Know Act), hazardous waste disposal (Resource Conservation and Recovery Act), storage tank management, and spill prevention (Oil Pollution Act).

Tables 2.1 and 2.2 provide lists of the radionuclides that are typically reported in effluent releases from nuclear plants. The characteristics and quantities of typical releases are described in the following sections. The radioactive isotope carbon-14, which is not shown in the tables, is mainly produced by neutron activation of oxygen-17 in the coolant of reactors of all types. The production of carbon-14 is estimated to be about 5 Ci per gigawatt (thermal)-year (GW_{th}-y) in boiling-water reactors (BWRs) and 4

TABLE 2.1 Common Radionuclides in Reported Airborne Effluent Releases from Nuclear Plants

Category	Commonly Reported Radionuclides
Fission and activation gases	Krypton (85, 85m, 87, 88) Xenon (131, 131m, 133, 133m, 135, 135m, 138) Argon (41)
Iodines/halogens	Iodine (131, 132, 133, 134, 135) Bromine (82)
Particulates	Cobalt (58, 60) Cesium (134, 137) Chromium (51) Manganese (54) Niobium (95)
Tritium	Hydrogen (3)

SOURCE: USNRC (2007), Table 2.1.

TABLE 2.2 Common Radionuclides in Reported Liquid Effluent Releases from Nuclear Plants

Category	Commonly Reported Radionuclides
Mixed Fission and Activation Products	Iron (55) Cobalt (58, 60) Cesium (134, 137) Chromium (51) Manganese (54) Zirconium (95) Niobium (95) Iodine (131, 133, 135)
Tritium	Hydrogen (3)
Dissolved and Entrained Noble Gases	Krypton (85, 85m, 87, 88) Xenon (131, 133, 133m, 135, 135m)

SOURCE: USNRC (2007), Table 2.2.

Ci per GW_{th}-y in pressurized-water reactors (PWRs) (EPRI, 2010). Most of the activity produced is released into the atmosphere. Effluent releases of carbon-14 have not been required to be reported to the USNRC in the past. However, starting in 2010, plant licenses are required to estimate and report releases of this radionuclide to the USNRC. It has been estimated by some that the atmospheric releases of carbon-14 result in a relatively large contribution to population dose (Kahn et al., 1985; NEA, 2003). Additional discussion of the carbon-14 contribution to dose is provided in Chapter 3.

2.1.1 Airborne Effluent Releases

Figures 2.1 through 2.4 provide graphical illustrations of selected airborne effluent releases reported to the USNRC for operating plants in the United States in 2008. The figures show noble gas releases (Figure 2.1), iodine-131 releases (Figure 2.2), particulate releases (Figure 2.3), and tritium releases (Figure 2.4) from BWRs and PWRs.

The following observations emerge from an inspection of these figures:

- At present, nuclear plants typically release between a few curies and several hundred curies per year in airborne effluents.
- Most of the activity released in airborne effluents is from fission/activation gases and tritium. The median activities of these releases are (currently) approximately the same for BWRs and PWRs, in spite of the fact that tritium production in PWRs is higher than in

(A)

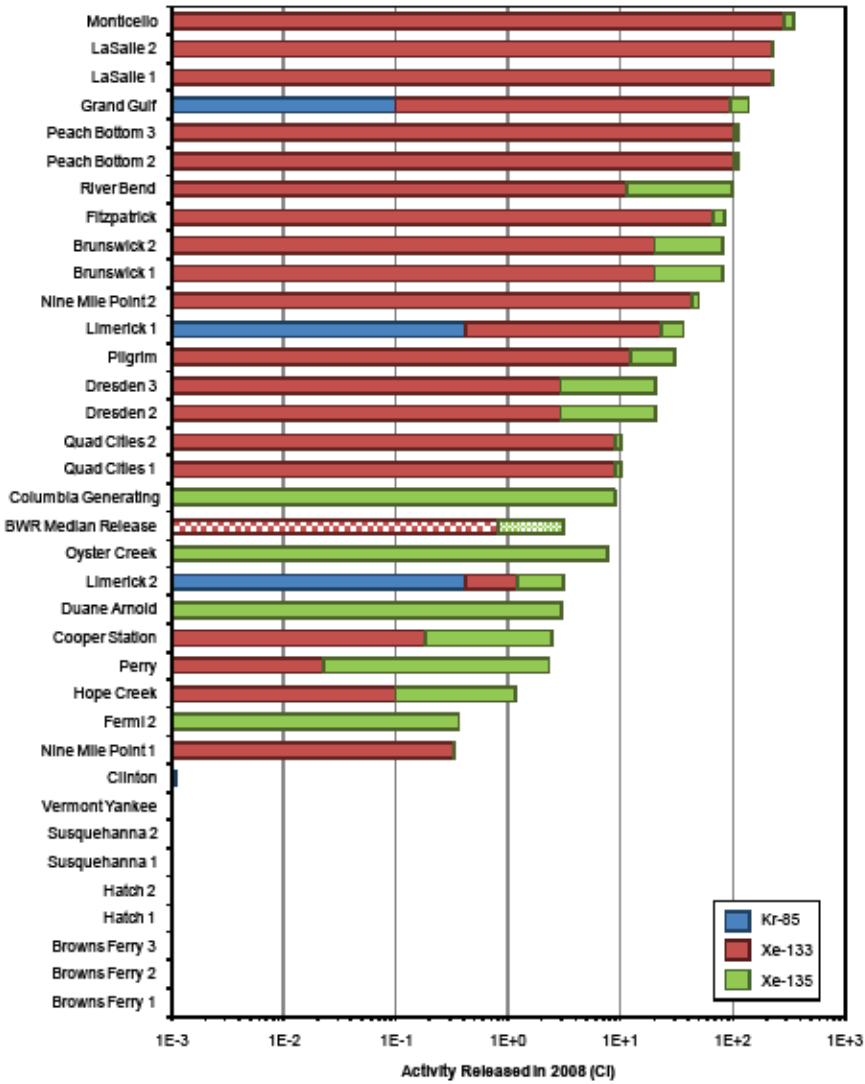

FIGURE 2.1 Noble gas releases from (A) BWRs and (B) PWRs in 2008. SOURCE: Daugherty and Conatser (2008).

(B)

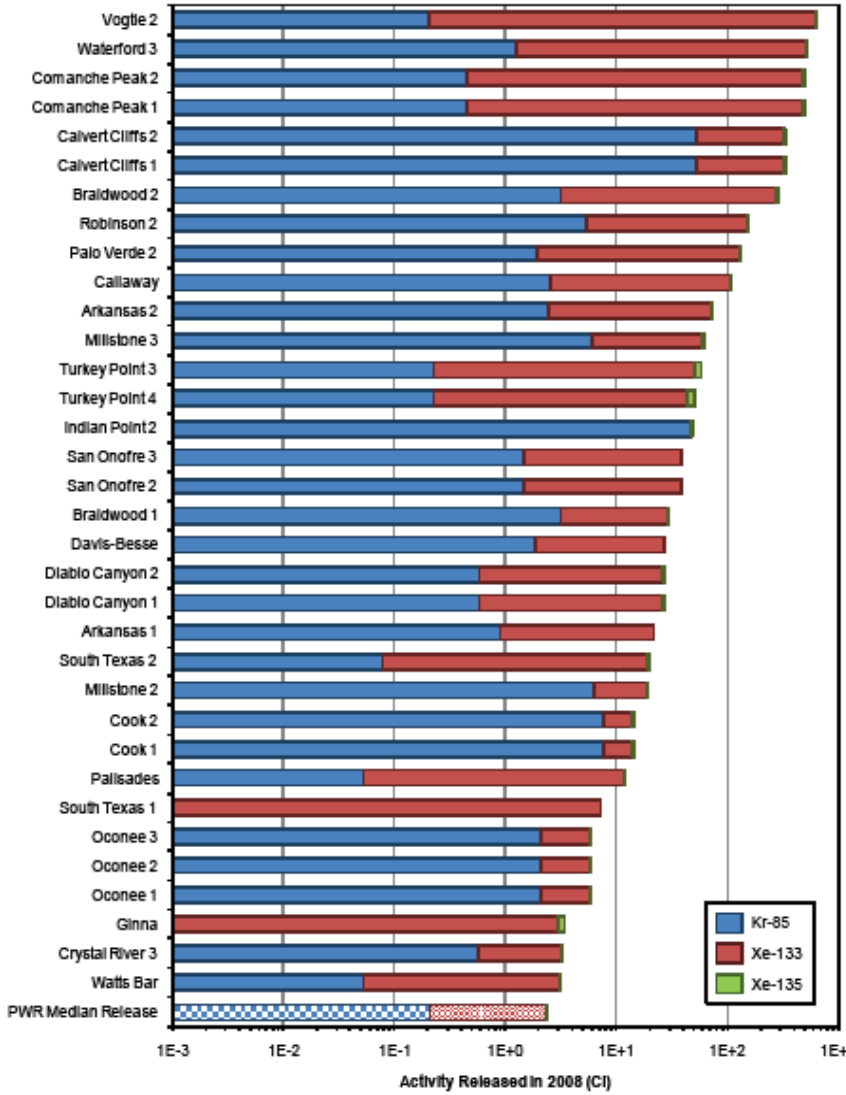

FIGURE 2.1 Continued

(B, continued)

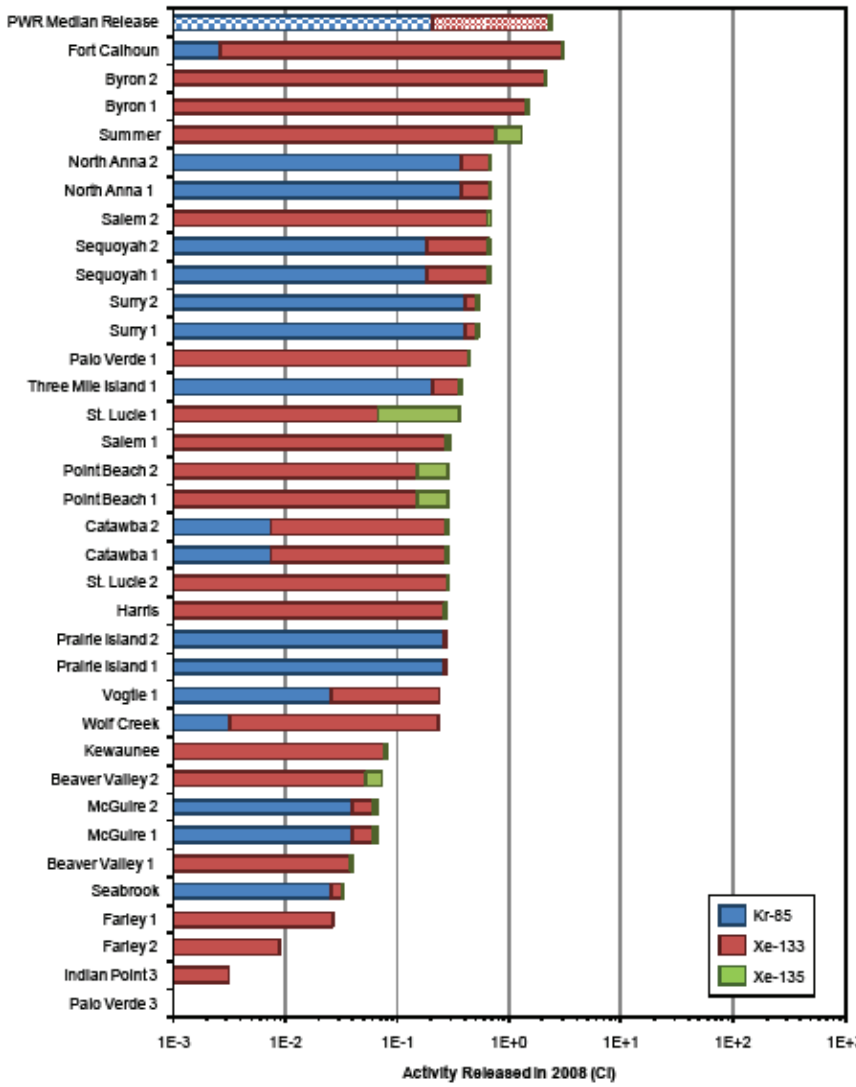

FIGURE 2.1 Continued

(A)

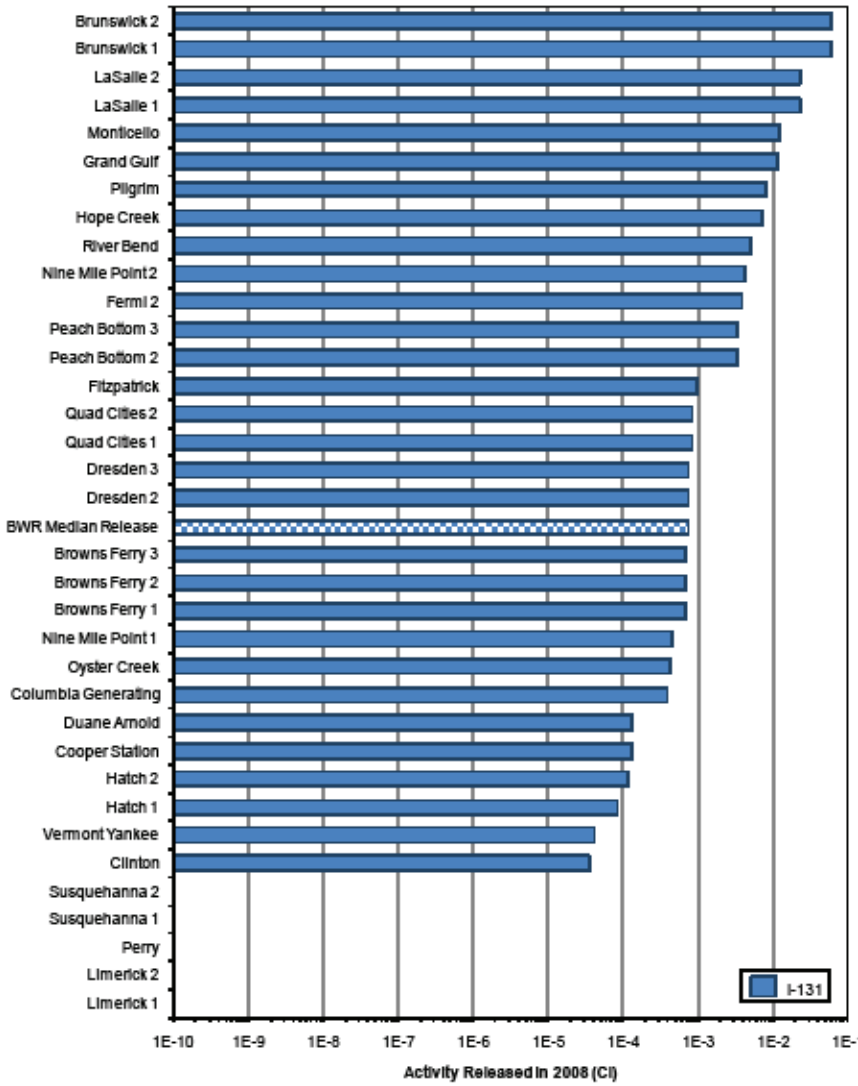

FIGURE 2.2 Iodine-131 releases from (A) BWRs and (B) PWRs in 2008. SOURCE: Daugherty and Conatser (2008).

(B)

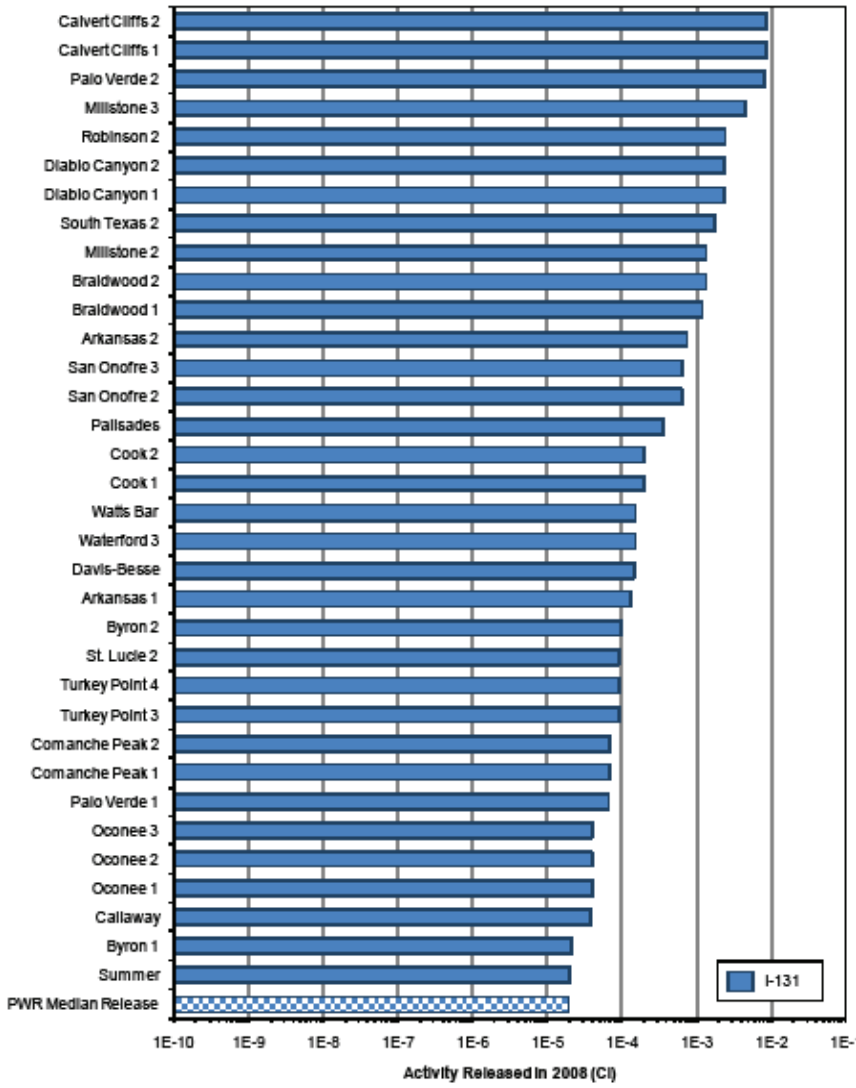

FIGURE 2.2 Continued

(B, continued)

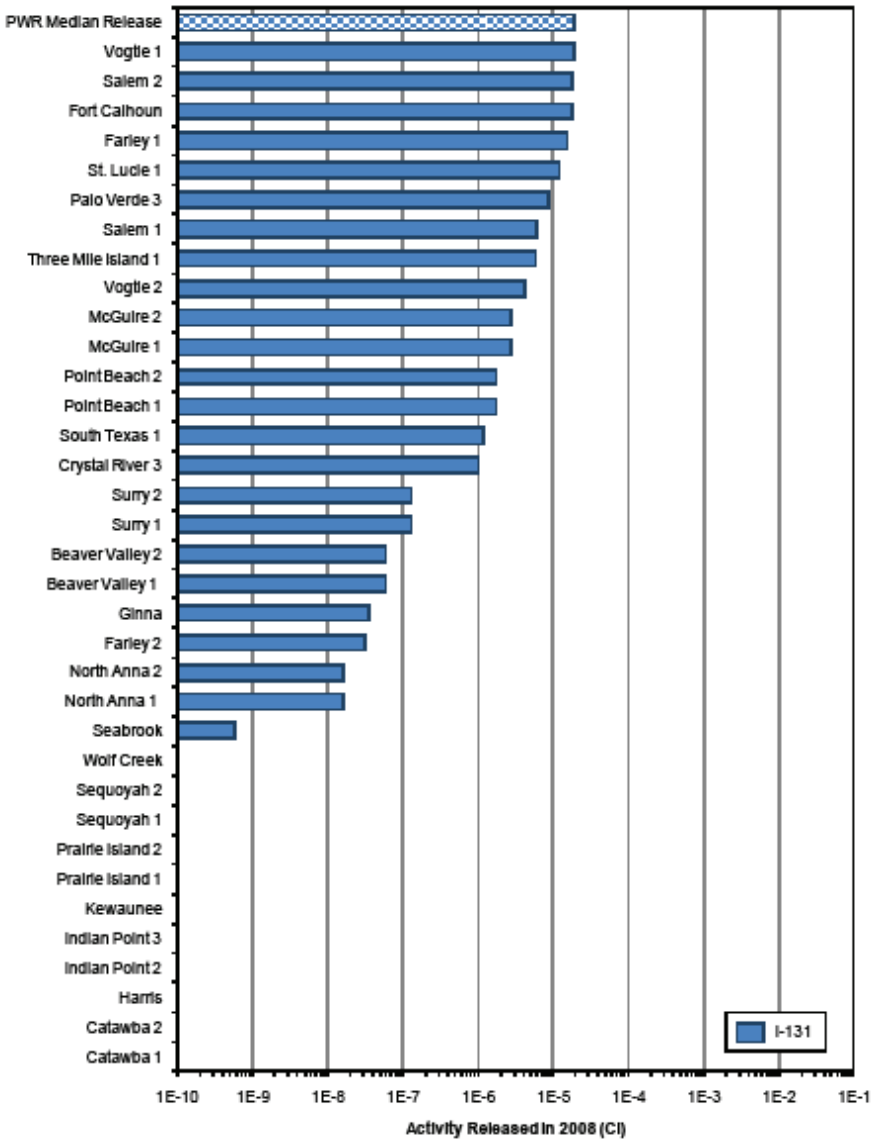

FIGURE 2.2 Continued

(A)

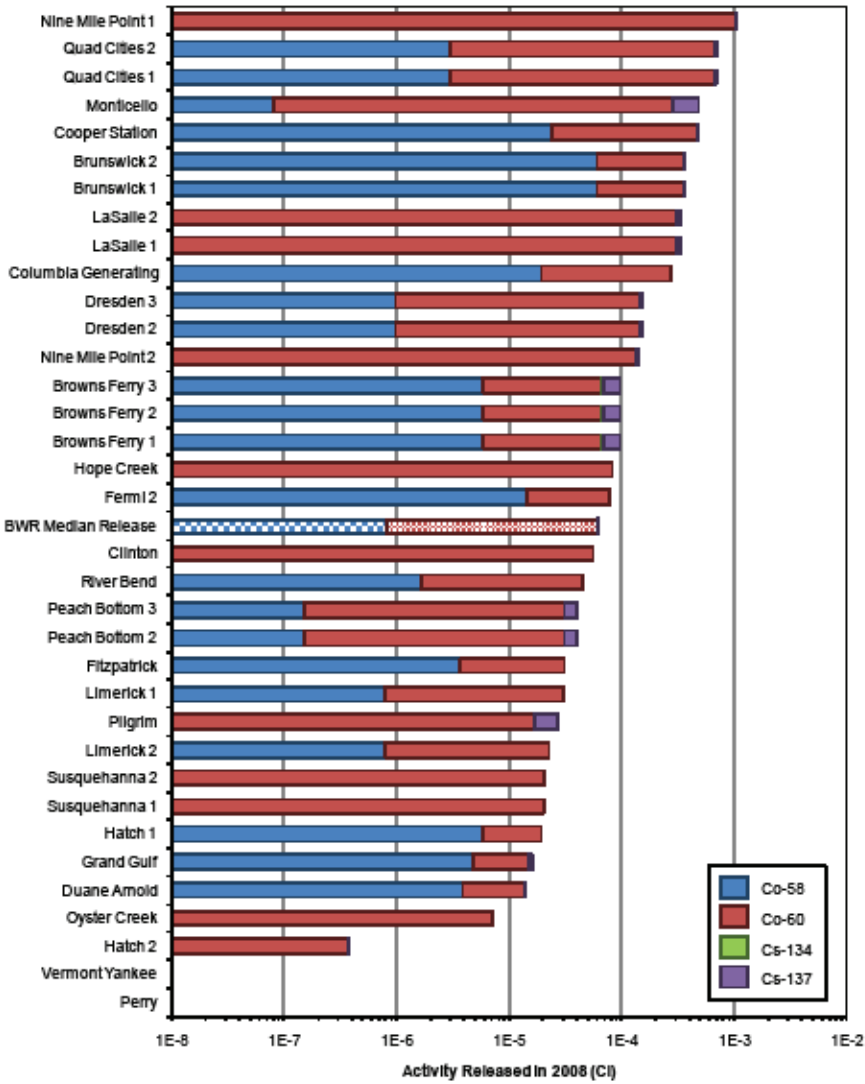

FIGURE 2.3 Particulate releases from (A) BWRs and (B) PWRs in 2008. SOURCE: Daugherty and Conatser (2008).

(B)

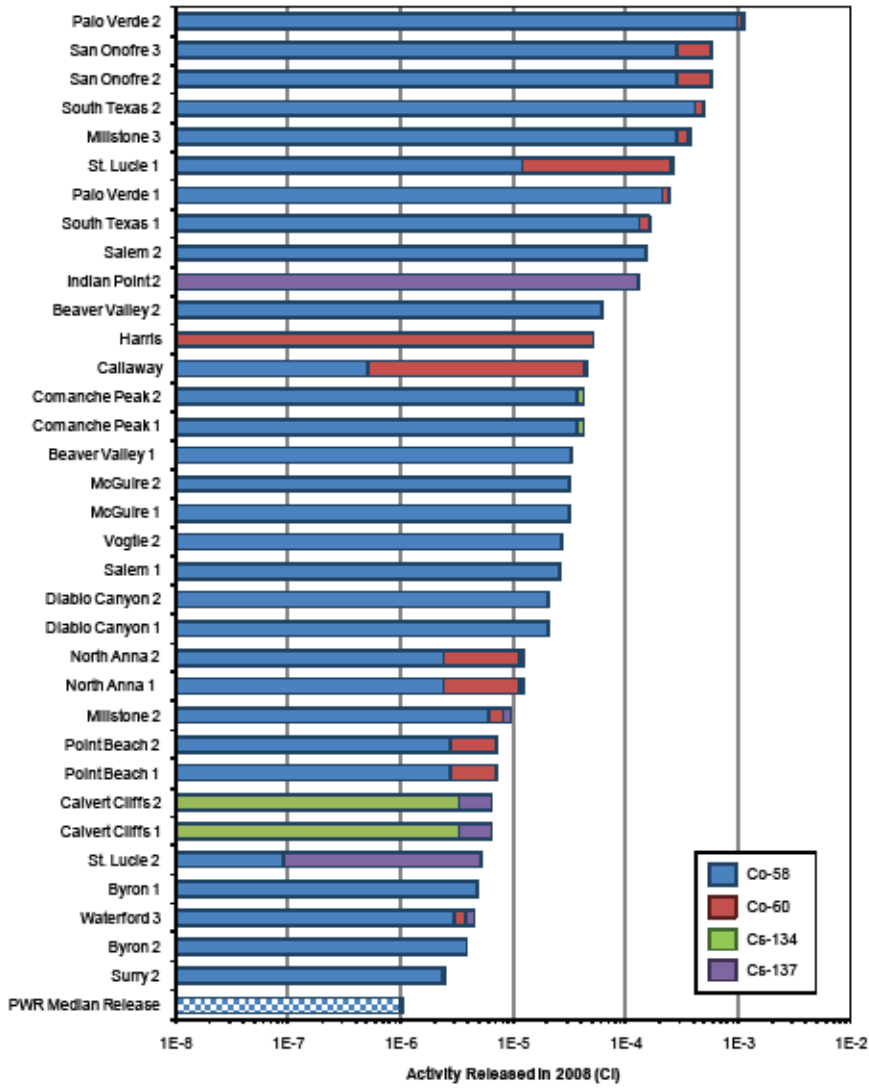

FIGURE 2.3 Continued

(B, Continued)

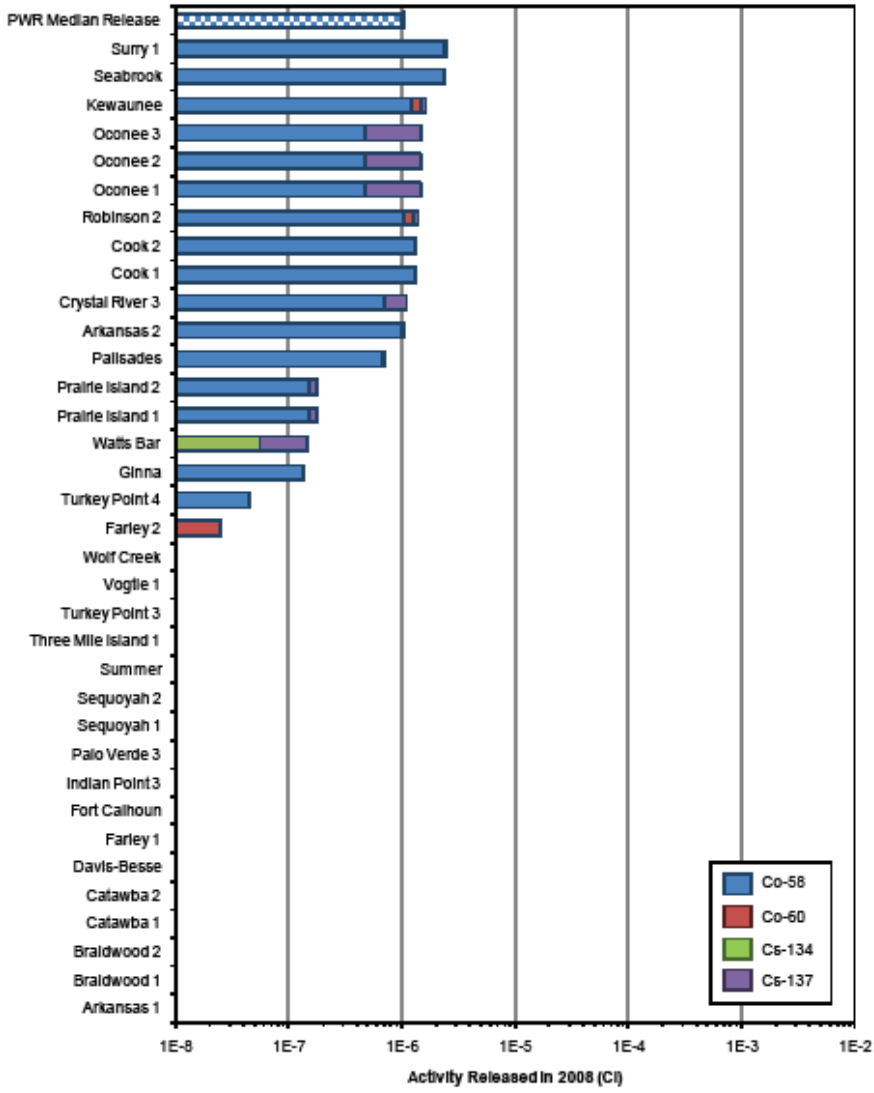

FIGURE 2.3 Continued

(A)

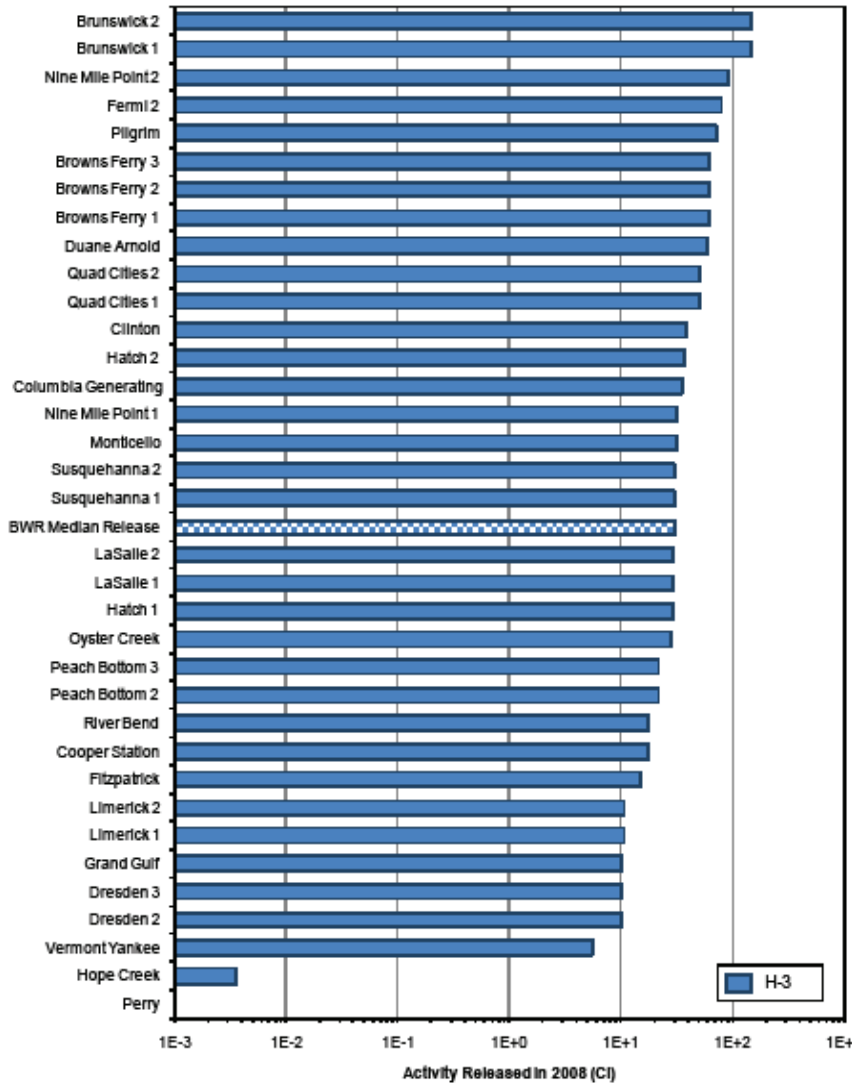

FIGURE 2.4 Tritium (H-3) releases from (A) BWRs and (B) PWRs in 2008. SOURCE: Daugherty and Conatser (2008).

(B)

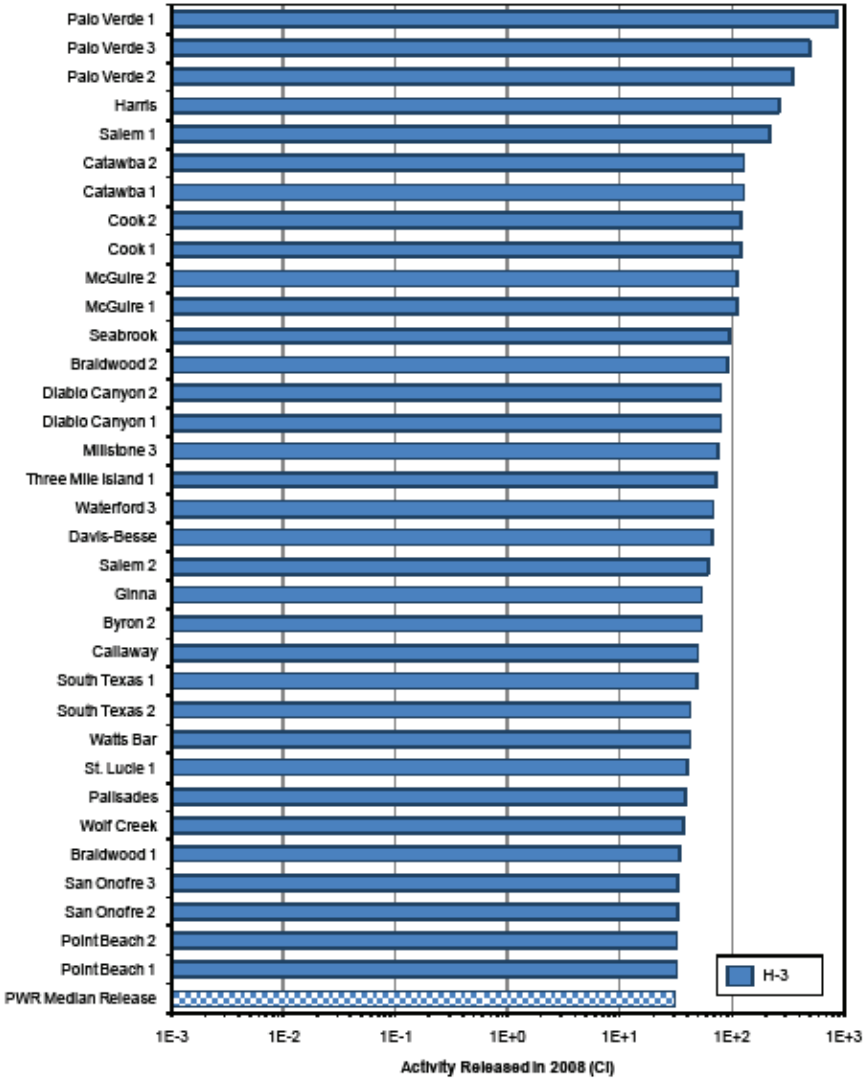

FIGURE 2.4 Continued

(B, continued)

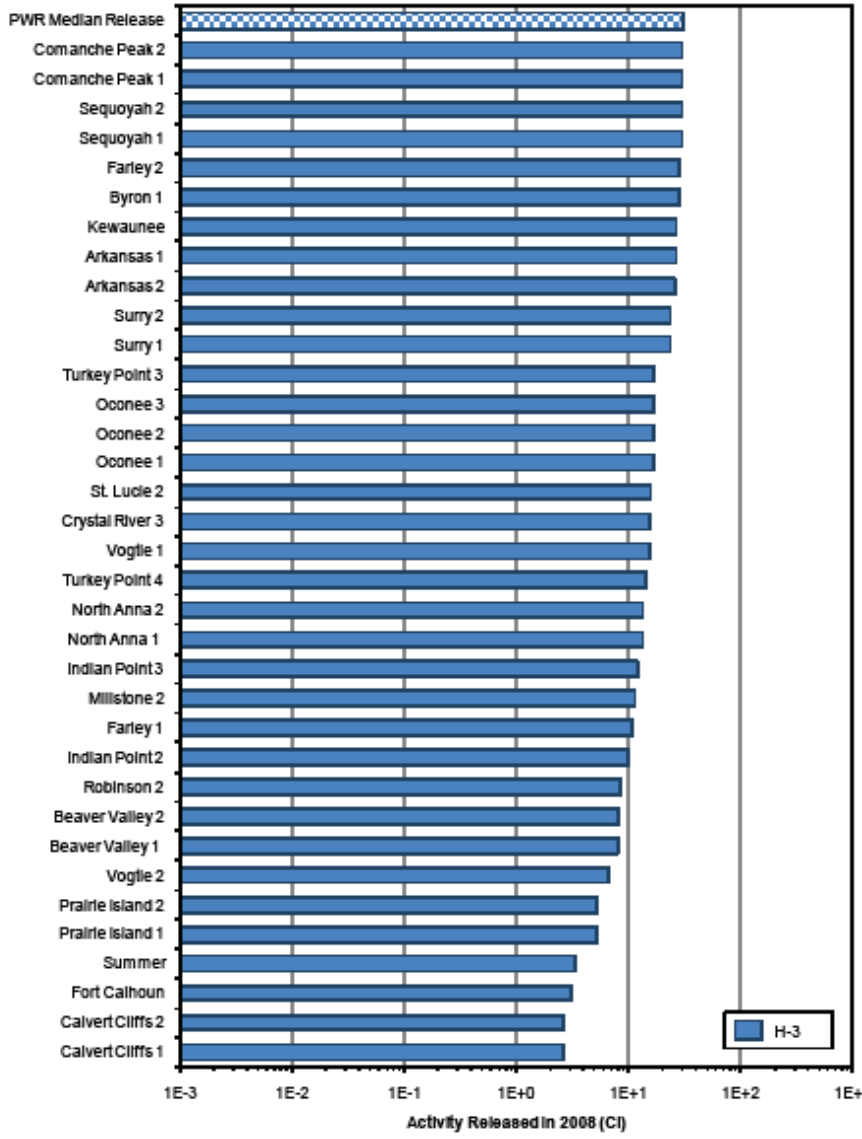

FIGURE 2.4 Continued

BWRs.[4] However, as will be discussed later in this chapter, BWRs generally released greater quantities of radionuclides than PWRs prior to about 1980.

- The activities of iodine and particulates in releases are typically several orders of magnitude lower than activities from fission/activation gases and tritium. Additionally, median activities of iodine and particulates are about one to two orders of magnitude lower in PWRs than in BWRs.
- Both BWRs and PWRs exhibit significant variability in releases of all airborne effluent categories: about six orders of magnitude of variability in noble gas releases; over seven orders of magnitude of variability in iodine releases; over four orders of magnitude of variability in particulate releases; and (with one exception) about three orders of magnitude of variability in tritium releases. In general, the variability differences are greater among PWRs than BWRs.
- The variability in airborne effluent releases that are exhibited in these figures is the result of several factors, including differences in the plant designs and operations; designs and operations of radioactive waste management and effluent control systems; plant equipment performance; and analytical methods used to monitor effluent releases. A detailed discussion of these differences is beyond the scope of this report; additional information is available in NCRP (1987) and in Marley (1979).

Airborne effluent releases from nuclear plants also display significant variability across time. To illustrate, Figure 2.5 provides comparative examples of annual releases of noble gases from operating PWR and BWR nuclear plants for two different years separated by two decades. In general, noble gas releases have decreased over time, even though plant capacity factors have increased and some plants have received power uprates.[5] This decrease is likely due to several factors, including improved fuel cladding performance and improved design and operation of effluent control and waste treatment systems.

The intraplant variability of releases as a function of time can also be high, as illustrated in Figure 2.6, which compares atmospheric releases of

[4]Although tritium is produced in both reactors as a result of ternary fission and activation of deuterium that is naturally present in cooling water, PWRs also produce tritium from neutron capture in boron that is added to the cooling water to control reactivity, i.e., through the reaction $^{10}B(n, 2\alpha)T$ (see Appendix D).

[5]A nuclear reactor is licensed by the USNRC to operate up to a specified maximum power. Plant licensees can request approval from the USNRC to increase (or *uprate*) the maximum power at which the reactor can operate. A reactor's power is typically increased by changing the enrichment or other design elements of the reactor fuel.

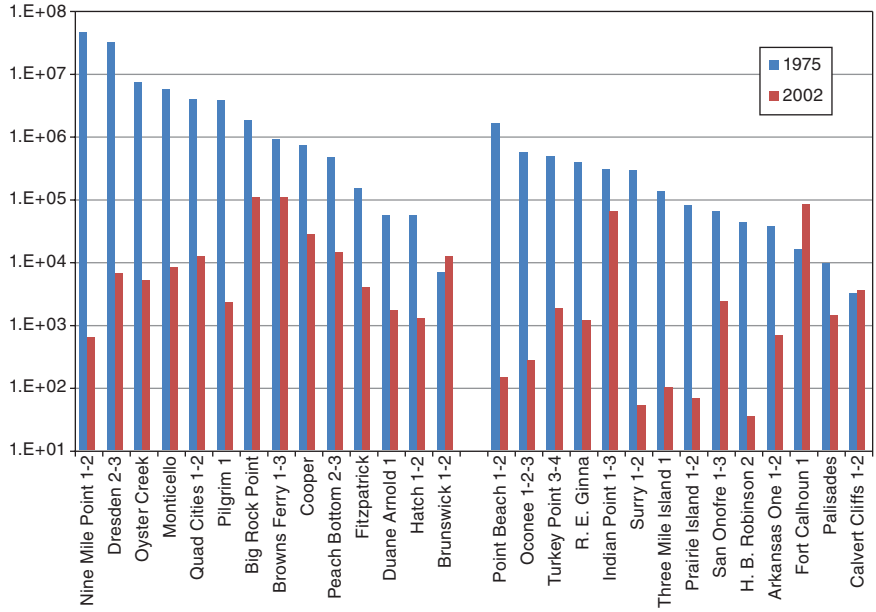

FIGURE 2.5 Comparison of atmospheric releases of noble gases for selected BWRs (left) and PWRs (right) in the United States. The units on the vertical scale are in gigabecquerels (GBq = 0.03 Ci). SOURCE: Data from the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR).

noble gases, iodine-131, and radioactive particulates from the Dresden plant (located near Chicago, Illinois) from 1975 to 2002. Noble gases constituted the largest source of releases from the Dresden plant during this time period, which again is typical for effluent releases from nuclear plants. Notice also that the total quantities of releases decreased from the mid 1970s to the mid 1990s, likely the result of improvements in effluent controls and plant operations. The increase in emissions starting in the mid 1990s was likely due in part to an increase in power output (Figure 2.7). Improved operating practices resulted in higher plant utilization levels as well as higher allowed power levels.

A further illustration of intraplant variability of effluent releases from nuclear plants is shown in Table 2.3. This table shows releases from four plants (two BWRs and two PWRs) for two time periods (1980 and 2008-2010). Note particularly the much higher noble gas releases in 1980 compared to 2008-2010, which reflects higher releases of short-lived nuclides such as krypton-87 (76-minute half-life) and krypton-88 (2.8-hour half-life) from BWRs. In 2008-2010, effluent releases were primarily xenon-133 (5.2-day half-life).

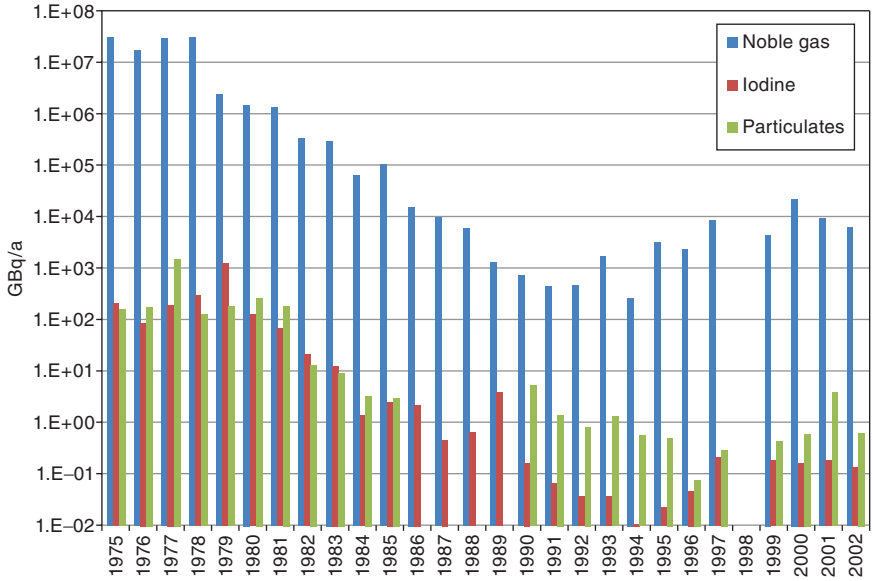

FIGURE 2.6 Comparison of annual atmospheric releases of noble gases (blue bars), iodine-131 (red bars), and radioactive particulates (green bars) for the Dresden plant from 1975 to 2002. The units on the vertical scale are in GBq (=0.03 Ci). SOURCE: Data from UNSCEAR.

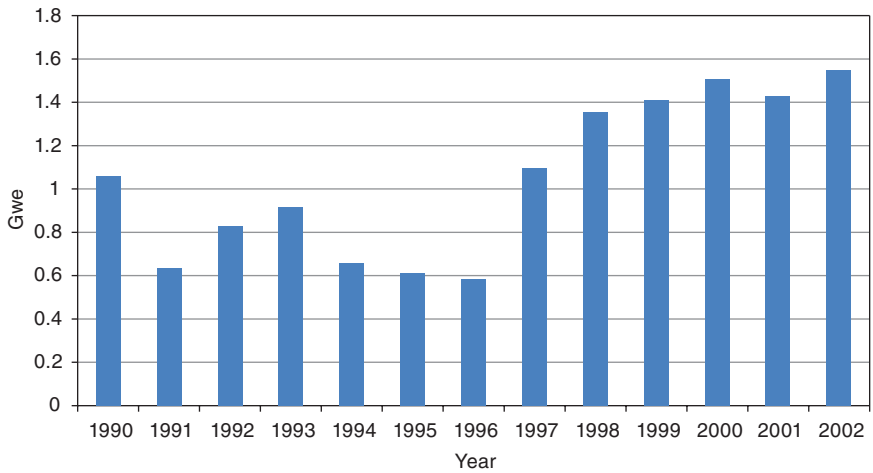

FIGURE 2.7 Variation with time of the electricity generated by the Dresden plant. SOURCE: UNSCEAR (2008).

TABLE 2.3 Comparison of Airborne Radioactive Effluent Releases (in curies) from Four Nuclear Plants, 1980 and 2008-2010

	Millstone Point 2 & 3 (PWR)		Dresden 2 & 3 (BWR)		Oyster Creek (BWR)		North Anna (PWR)	
Radionuclide	1980	2010	1980	2009	1980	2008	1980	2010
Helium-3	850	12	1180	2.3	9.4	28	56	4.8
Cobolt-60		0	0.64	3.20×10^{-4}	8.90×10^{-3}	7.20×10^{-6}	5.30×10^{-6}	0
Krypton-85	9.7	2.1		0		0	2.4	5.5
Krypton-85M	2.5	0	0.093	2.1	1260	0	0.65	0
Krypton-87	NR	0	0.042	1.3	4260	0	0.018	0
Krypton-88	7.6	0	0.0017	1.7	4060	0	0.24	0
Iodine-131	6.30×10^{-3}	7.20×10^{-5}	3.5	1.20×10^{-2}	0.95	4.10×10^{-5}	1.20×10^{-2}	8.20×10^{-4}
Iodine-133	3.30×10^{-3}	3.30×10^{-4}	11	3.50×10^{-3}	3.2	1.30×10^{-3}	1.40×10^{-3}	0.00
Xenon-133	1280	0.025	10800	3.6	861	0	3390	27
Xenon-135	63	0.022	10800	8.7	6980	68	84	0.095
Xenon-138	NR	6.60E-05	15000	28	10300	0	nm	0
Cesium-137	7.60×10^{-3}	0	8.00×10^{-2}	2.00×10^{-6}	4.50×10^{-3}	0	3.30×10^{-4}	0

SOURCE: BNL (1983) for 1980 data; Annual Radioactive Effluent Release Reports for 2008-2010 data.

In fact, releases of shorter-lived radionuclides (i.e., iodine-133, xenon-135) from nuclear plants have been reduced in recent years compared to earlier years. This is a result of increased holdup times[6] to reduce effluents and doses to meet ALARA[7] goals. This reduction in releases also accounts for much of the dramatic decrease in population doses[8] from airborne effluent releases: For example, xenon-133 emits only weak gamma rays, whereas the krypton isotopes and some of the other xenon isotopes emit relatively high-energy gamma radiation. The relatively lower activities of airborne effluents from PWRs compared to BWRs is also partly due to the fact that most of the PWR releases are batch releases; releasing effluents in batches allows more time for decay of short-lived radionuclides.

2.1.2 Liquid Effluent Releases

Liquid radioactive effluents that are released in surface waters (rivers, estuaries, and oceans) are monitored. In addition, uncontrolled leaks of liquid radioactive effluents have resulted in contamination of groundwater. Groundwater contamination is discussed in Section 2.1.4.3.

Figures 2.8 through 2.11 provide graphical illustrations of selected liquid effluent releases for nuclear plants in the United States. The figures show the variation with time of liquid effluent releases from the Dresden plant (BWR) (Figure 2.8); a comparison of liquid effluent releases from a number of other BWRs and PWRs in 1975 and 2002 (Figures 2.9 and 2.10), and the variation with time of tritium releases in liquid effluents for selected BWRs and PWRs (Figure 2.11). The following observations emerge from an inspection of these figures:

- Currently, nuclear plants typically release between a few curies and one thousand curies per year of tritium in liquid effluents; releases

[6]That is, effluents were stored in the plant for longer times before being released to the environment. Such storage is especially effective for reducing concentrations of short-lived radionuclides through radioactive decay.

[7]ALARA stands for As Low As (is) Reasonably Achievable. ALARA is defined in Title 10, Part 20.1003 of the Code of Federal Regulations (CFR) to mean "making every reasonable effort to maintain exposures to radiation as far below the dose limits in this part as is practical consistent with the purpose for which the licensed activity is undertaken, taking into account the state of technology, the economics of improvements in relation to state of technology, the economics of improvements in relation to benefits to the public health and safety, and other societal and socioeconomic considerations, and in relation to utilization of nuclear energy and licensed materials in the public interest."

[8]The distribution of dose versus distance from a nuclear plant depends on the half-lives of the radionuclides in released effluents as well as the energy of their emitted radiations. The longer the half-life, the longer the radionuclide persists in the environment and the more people who are potentially exposed.

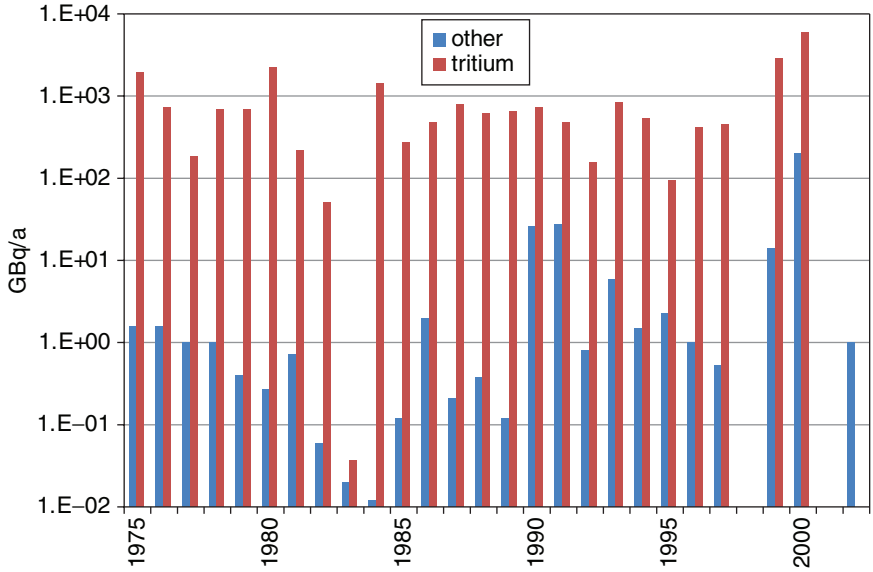

FIGURE 2.8 Variation of annual liquid radioactive effluent releases from the Dresden plant between 1975 and 2003.

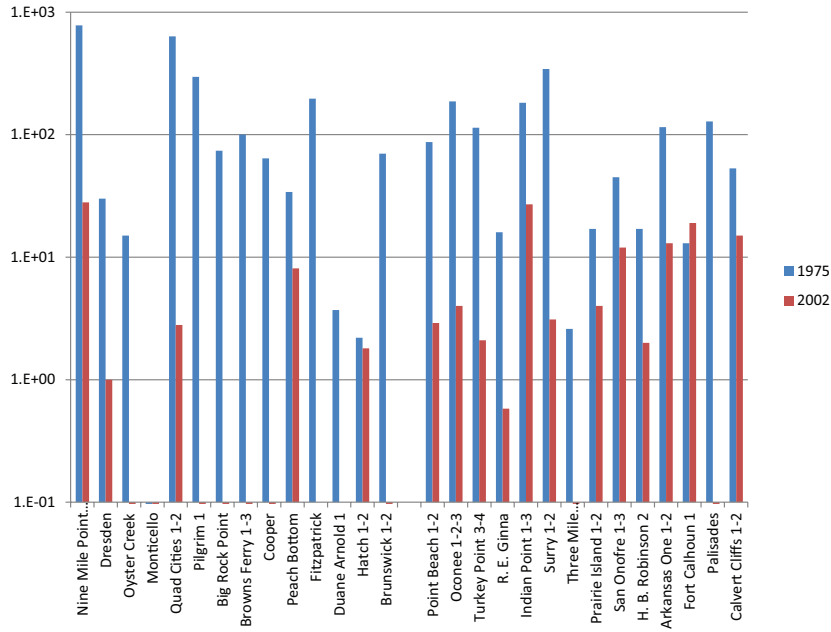

FIGURE 2.9 Comparison of liquid radioactive effluent releases, excluding tritium, for selected BWRs (left) and PWRs (right) in 1975 and 2002. SOURCE: Data from UNSCEAR.

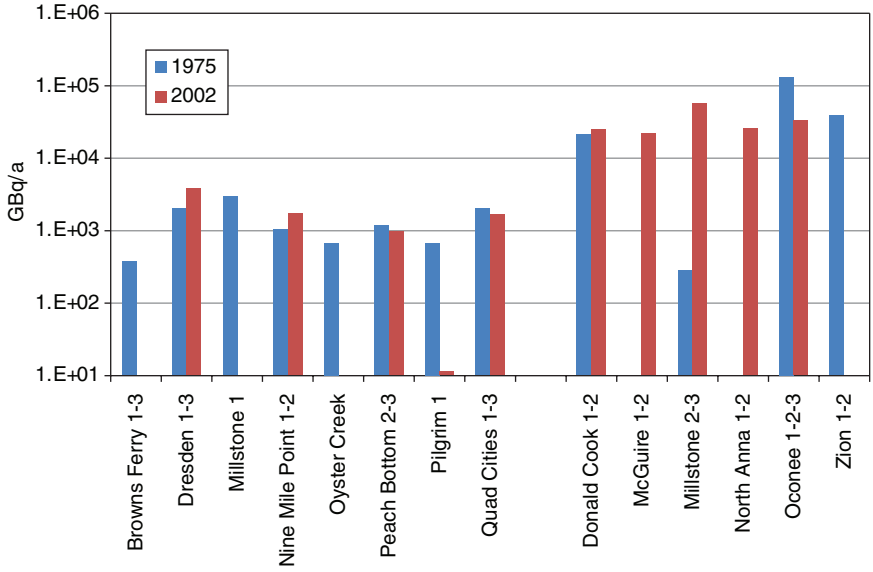

FIGURE 2.10 Tritium released in liquid effluents for selected nuclear plants (left, BWRs; right, PWRs) in 1975 and 2002. NOTE: The North Anna and McGuire plants were not operational in 1975. SOURCE: Data from UNSCEAR.

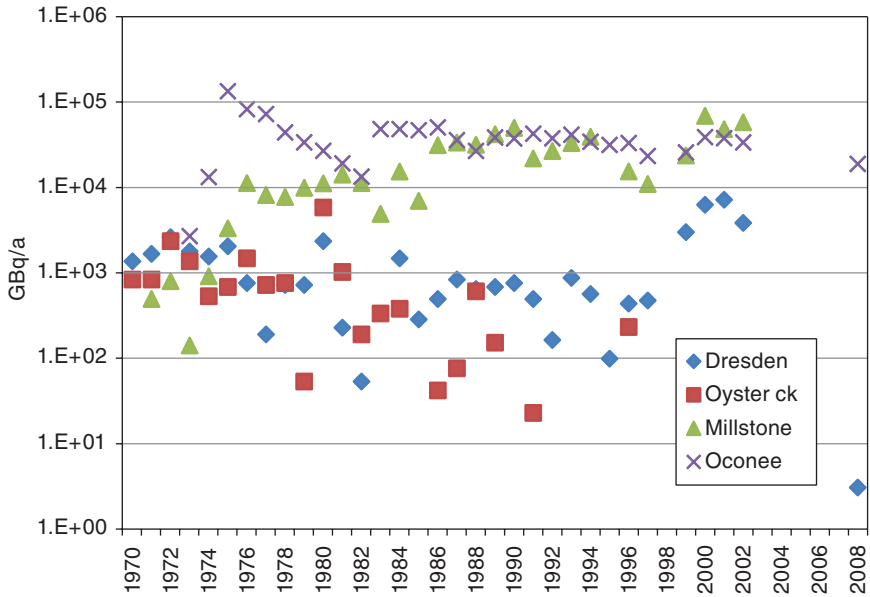

FIGURE 2.11 Variation of annual tritium releases in liquid effluents from selected nuclear plants. SOURCE: Data from UNSCEAR.

of mixed fission and activation products are much smaller (in the range from 0.001 to 0.01 curies per year).

- Tritium activity in liquid effluents is much greater for PWRs (about 500 curies per year) than for BWRs (about 30 curies per year). Tritium releases have changed little through time.
- Releases of mixed fission and activation products are greater for BWRs than for PWRs and show a decreasing trend with time.

Table 2.4 compares levels of selected radionuclides in liquid effluent releases in 1980 and 2008-2010 for the two PWR and two BWR plants shown in Table 2.3. For the PWRs (Millstone and North Anna), tritium levels were higher in 2010, whereas the other liquid effluents were much lower in 2008-2010 for both types of plants.

2.1.3 Availability of Information on Effluent Releases

Information on releases of airborne and liquid radioactive effluents from nuclear plants to the environment is available in reports that are submitted by plant licensees to the USNRC. These reports are available in pdf format for all operating nuclear plants in the United States beginning in 2005 (http://www.nrc.gov/reactors/operating/ops-experience/tritium/plant-info.html). Electronic summaries of the data in these reports for the period 1998-2007 are also available in the Effluent Database for Nuclear Power Plants, which is available on the USNRC website (www.reirs.com/effluent/).

Several summaries of total airborne and liquid radioactive effluent releases (and sometimes total tritium and iodine-131 releases) have been published over the years. The United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) has published summaries of data from nuclear plants worldwide that list total releases of noble gases, particulates, and iodine from 1975 up to 2004 (UNSCEAR, 1982, 1988, 1993, 2000, 2008). The U.S. Environmental Protection Agency (USEPA) published a report (Phillips, 1978) summarizing more detailed isotopic releases for the period 1973-1976. Additional information on airborne emissions is provided in Harris and Miller (2008), Hull (1973), and Marley (1979).

These summary data are useful for understanding the magnitudes of and trends in effluent releases, but they are not sufficiently detailed for use in reconstructing doses to persons living near nuclear facilities to support an epidemiologic study. For this purpose, more detailed information on radionuclide releases are required, including release quantities of specific radionuclides; method of release (i.e., continuous or batch); points of release (i.e., locations of stacks, vents, and liquid discharge points); time of release; and local meteorological conditions at the time of release. These

TABLE 2.4 Comparison of Liquid Radioactive Effluent Releases (in curies) from Four Nuclear Plants, 1980 and 2008-2010

	Millstone 2 & 3 (PWR)		Dresden 2 & 3 (BWR)		Oyster Creek (BWR)		North Anna (PWR)	
Radionuclide	1980	2010	1980	2009	1980	2008	1980	2010
Helium-3	268	427	62	6.8	154	<MDL	403	1270
Chromium-51	0.14	1.40×10^{-5}	1.60×10^{-2}	1.00×10^{-4}	9.60×10^{-3}	<MDL	3.90×10^{-3}	
Manganese-54	0.12	1.20×10^{-4}	4.00×10^{-2}	6.60×10^{-4}	5.70×10^{-2}	<MDL	6.80×10^{-3}	3.60×10^{-5}
Cobalt-58	0.69	1.24×10^{-3}	8.90×10^{-3}	2.00×10^{-5}	6.08×10^{-3}	<MDL	5.50×10^{-1}	3.50×10^{-3}
Cobalt-60	0.91	3.29×10^{-3}	2.00×10^{-1}	1.10×10^{-3}	3.20×10^{-1}	<MDL	2.20×10^{-1}	3.10×10^{-3}
Zirconium-95	0.022	1.34×10^{-4}	1.50×10^{-2}	0.00		<MDL	5.50×10^{-4}	

NOTE: MDL, minimum detection limit.
SOURCE: BNL (1983) for 1980 data; Annual Radioactive Effluent Release Reports for 2009-2010 data.

detailed data are available in reports provided to the USNRC by nuclear plant licensees.

The committee undertook a detailed investigation for a sample of plants to determine whether these reports are available for all years of plant operations. The committee selected the following six plants for this investigation: Dresden (Illinois), Millstone (Connecticut), Oyster Creek (New Jersey), Haddam Neck (Connecticut), Big Rock Point (Michigan), and San Onofre (California). These plants were selected because they provide a broad representation of the nuclear plant designs and operating histories:

- Dresden, Big Rock Point, and Oyster Creek are BWRs; Haddam Neck, Millstone, and San Onofre are PWRs.
- Plant sizes range from 240 MWt (Big Rock Point) to 6876 MWt (San Onofre 2 and 3).
- Reactors at these plant sites began operations from the late 1950s (Dresden) to the early 1980s (San Onofre). Two plants (Big Rock Point and Haddam Neck) are no longer operating.

The committee first assessed the availability of these semiannual reports through the USNRC's public records system (ADAMS).[9] The committee, with the assistance of USNRC staff, searched ADAMS and visited the Public Reading Room at USNRC headquarters in Rockville, Maryland, to examine microfiche records. The committee was not able to locate many of the reports for these plants, especially prior to 1975, and some of the reports on microfiche were not legible.

The committee then asked the Nuclear Energy Institute (NEI)[10] to contact plant licensees to determine whether they have maintained records of effluent releases and associated meteorological data. Licensees at these six plants were asked the following questions:

1. Does the plant licensee maintain records of its effluent monitoring reports that are submitted to the USNRC?
2. If so, how far back in time are these records maintained?
3. If meteorology data are not included in the effluent monitoring reports, are records of those data also maintained? If so, for how far back in time?

[9]The Agencywide Documents Access and Management System (ADAMS) is the USNRC's official recordkeeping system (see http://www.nrc.gov/reading-rm/adams.html). Two collections of documents are available through this system: The Publicly Available Records System contains full-text documents released since November 1, 1999. The Public Legacy Library contains more than 2 million bibliographic citations for earlier documents. These earlier documents are stored on microfiche.

[10]NEI is the policy arm of the nuclear and technology industry. See nei.org.

4. In what format(s) are effluent monitoring and meteorology records maintained (i.e., digital or paper)?
5. Would these effluent monitoring and meteorology records be made available to support a USNRC-requested epidemiologic study if requested?

The committee obtained the following information from NEI (Ralph Andersen, NEI, verbal communication to K.D. Crowley, February 23, 2012):

- Licensees of operating nuclear plants maintain records of effluent monitoring reports that are submitted to the USNRC. Licensees are not required by the USNRC to maintain these records after their licenses are terminated, but most licensees maintain these records anyway to meet insurance company requirements.
- Prior to the mid 1980s, effluent release and meteorology data are available only in hard copy (paper or microfiche) format. Between the mid 1980s and mid 1990s, these data are available in mixed (i.e., digital or hard copy) format. Some data may be available in digital format from the mid 1990s forward.
- Some plant licensees may be able to provide digital data if requested. Licensees would probably defer to the USNRC for hard copy records because of the significant expense of retrieving these records from their archives.

Information on effluent releases may be available from other sources as well. In the late 1970s the USNRC contracted with Brookhaven National Laboratory to enter the semiannual effluent data from each nuclear plant into an electronic database.[11] This effort continued until 1990 and was then replaced by the Effluent Database for Nuclear Power Plants which was described previously. Annual reports summarizing these data are available electronically. However, these annual reports do not provide specific information about effluent release points or associated meteorological data required to estimate atmospheric dispersion. They also do not distinguish batch releases from continuous releases.

[11]These data are available in paper form for the following years: 1974 (NUREG-0077, June 1976); 1975 (NUREG-0218, March 1977); 1976 (NUREG-0367, March 1978); 1977 (NUREG-0521, January 1979); 1978 (NUREG/CR-1497 [BNL-NUREG-51192], March 1981); 1979 (NUREG/CR-2227 [BNL-NUREG-51416], November 1981); 1980-1994 (NUREG-CR-2907, vol.1-14).

More detailed data on effluent releases were provided to Pacific Northwest Laboratory (now Pacific Northwest National Laboratory[12]) researchers who were contracted by the USNRC to develop annual estimates of population exposures around nuclear plants.[13] The committee was able to locate the electronic media containing these data covering the years 1978 and 1981-1989 along with some of the corresponding meteorological data used to calculate atmospheric dispersion.[14] (Often the meteorological data were averaged over several years.)

The committee judges that these PNL data will be of marginal utility for dose estimation to support an epidemiologic study. The data do not distinguish between batch and continuous releases and reflect only two release heights, "elevated" and "mixed" (i.e., a combination of elevated and ground level). It is also not clear whether the files contain all of the radionuclides that were reported by plant licensees. These data may be helpful if the effluent releases for some particular site cannot be located, but otherwise there appears to be little data in these files beyond what is contained in the reports cited in footnote 12.

Detailed data on effluent releases will need to be obtained from the plant licensee's effluent release reports to the USNRC. It may be necessary to contact plant licensees to obtain these reports if they cannot be located in the USNRC library. Additionally, data relating to dispersion of effluents in surface waters and to the use that is made of the environment may have to be requested from plant licensees. Obtaining and digitizing these data will be a large and costly job.

2.1.4 Data Quality and Suitability for Estimating Radiation Doses

The committee assessed the quality of the effluent release data and its suitability for use in dose estimation for an epidemiologic study. These assessments are described below.

[12]PNL was renamed Pacific Northwest National Laboratory (PNNL) in 1995. This laboratory is located in Richland, Washington, adjacent to the Hanford Site.

[13]PNL issued a series of reports entitled Dose Commitments Due to Radioactive Releases from Nuclear Power Plant Sites that covered nuclear plant operations from 1977-1992. The first four reports in the series were issued as PNL-2439 (1977), NUREG/CR-1125/PNL-2940 (1979), NUREG/CR-1498/PNL-3324 (1980), and NUREG/CR-2201/PNL-4039 (1982). The remaining reports were issued from 1982 to 1996 as NUREG/CR-2850 (PNL-4221), vols. 1-14.

[14]The data were stored on 5.25-inch and 3.5-inch floppy disks. The committee was able to obtain these disks from a PNNL storage facility and transfer almost all of the data to a CD. The data are available in the Public Access File for this study.

2.1.4.1 Airborne Effluent Releases

As noted in Section 2.1.3, estimating doses to individuals living near nuclear plants from airborne effluent releases in a thorough manner requires detailed information on release quantities of specific radionuclides, method of release (i.e., continuous or batch), points of release (i.e., locations of air and liquid discharge points), time of release, and local meteorological conditions at the time of release (see Section 2.4). In its review of available data, the committee noted that the format of reported data, the specific radionuclides monitored, and the completeness of the data varied significantly from plant to plant, particularly during their early years of operation (i.e., prior to the mid 1980s). As discussed previously, the population doses from airborne releases in early years of plant operations were from short-lived radionuclides in the effluents. The estimated release rates for short-lived radionuclides are very sensitive to the assumed stack flow rate and probable holdup times.

The quality of the reported data was likely much poorer in the early years of operation prior to implementation over time of improved quality-assurance (QA) procedures. There are some unpublished data suggesting that plant licensees may have sometimes overestimated stack flow rates and thus actual effluent activities of shorter-lived radionuclides. There are also documented instances of facilities discovering errors in flow rates (and thus the magnitude of releases), sometimes years after the fact.[15]

The committee evaluated the quality and availability of airborne effluent release data for a few selected plants and years (see Section 2.1.3). However, a plant-by-plant evaluation will be required to assess data availability and sufficiency for use in a Phase 2 epidemiologic study. The committee judges that if such data are available, they are likely to be sufficiently accurate to develop credible dosimetry estimates that will adequately reflect variations in annual dose from plant to plant as a function of distance and direction from plant boundaries.

The releases of some nuclides may be very uncertain or not available, particularly for earlier years of operation. Also, as previously noted, atmospheric releases of carbon-14 have not been reported until 2010, although their contribution to the collective dose may be substantial (Dominion, 2010a; Kahn et al., 1985). However, because it can be assumed that carbon-14 activity released is approximately proportional to the thermal en-

[15]For example, from ML09057085 (2009): "The Dresden Nuclear Power Station (DNPS) Units 2/3 Chimney flow indication was found to be inaccurate in 2008, due to fouling of its flow elements. Further investigation showed that this issue began in April 2004, which resulted in non-conservative reporting of station effluents and calculated offsite doses for this period. This affected the data reported in the Annual Radioactive Effluent Release Reports for the calendar years 2004, 2005, 2006, and 2007 … and the Annual Radiological Environmental Operating Reports for the calendar years 2004, 2005, and 2006."

ergy generated by the plants, the annual doses resulting from carbon-14 releases can be crudely estimated. It is likely that simplifying assumptions will have to be made to reconstruct complete sets of airborne releases during the entire periods of operation of the nuclear plants considered in any Phase 2 epidemiologic study.

2.1.4.2 Liquid Effluent Releases

Estimating doses from liquid releases in surface waters requires detailed information on the specific radionuclides released; the total amount of activity of each radionuclide released; the time of release; the hydrology at the time of release; and the use that humans make of the water. In its review of available data, the committee noted that, as was the case for airborne effluent releases, the availability and completeness of the data varied significantly from plant to plant, particularly during the early years of operation. Also, the quality of the reported data was likely much poorer in the early years of operation prior to implementation of improved QA procedures.

The committee evaluated the quality and availability of liquid effluent release data for a few selected plants and years (see the discussion in Section 2.1.3). However, a plant-by-plant evaluation will be required to assess data availability and sufficiency for use in a Phase 2 epidemiologic study. The committee judges that if release data are available, they are likely to be sufficiently accurate to develop credible dose estimates. The most important uncertainties in terms of data sufficiency involve liquid effluent releases, particularly the determination of the dispersion of liquid effluents in receiving waters, the evaluation of the contamination of sediments, and the use of the contaminated water for human purposes (e.g., drinking water, consumption of aquatic foodstuffs, and consumption of irrigated terrestrial foodstuffs).

2.1.4.3 Uncontrolled Liquid Releases

Although there are no specific regulatory requirements for licensees to conduct routine onsite environmental surveys and monitoring for potential abnormal spills and leaks of radioactive liquids, regulations do require that licensees keep records of information important to the safe and effective decommissioning of their plants. Because the decommissioning of a nuclear plant requires licensees to clean up radioactive spills and leaks at the site, facility records include information on known spills or other unusual occurrences involving the spread of contamination that might require action as part of any decommissioning activities. These records can be limited to instances where significant contamination remains after procedures to remediate an uncontrolled liquid release, or when there is reasonable likelihood that contamination may have spread to inaccessible areas.

Table 2.5 provides a summary of known uncontrolled/inadvertent releases of radioactive liquids at nuclear plants over the period 1986 to 2006 (USNRC, 2006). These releases include leaks from spent fuel pool or condensate storage tank structures and/or associated equipment. They also include routine liquid releases initially prepared and monitored in accordance with regulatory guidance, but which were discharged to an unanalyzed environmental pathway as a result of degraded radioactive waste equipment or piping.

TABLE 2.5 Summary of Inadvertent Releases of Radioactive Liquid Effluents at Nuclear Plants

Nuclear Power Plant	Date of Release Discovery	Source of Release	Radionuclides Detected
Braidwood	March 2005	Vacuum breaker valves on the circulating water blowdown line	Tritium
Byron	February 2006	Vacuum breaker valves on the circulating water blowdown line	Tritium
Callaway	June 2006	Vacuum breaker valves on the circulating water blowdown line	Tritium, cobalt-58, cobalt-60, cesium-134, cesium-137
Dresden	August 2004, January 2006	Non-safety related HPCI suction and return line	Tritium
Hatch	December 1986	Fuel transfer canal due to operator action	Tritium
Indian Point	August 2005-Unit 1 leakage predates August 2005	Unit 1 and Unit 2 spent fuel pools	Tritium nickel-63, cesium-137, strontium-90, and cobalt-60
Oyster Creek	September 1996	Condensate transfer system due to operator action	Tritium
Palo Verde	March 2006	Rain condensing onto property after a gaseous release	Tritium
Perry	March 2006	Feedwater venturi	Tritium
Point Beach	1999	Retention pond	Tritium, cesium-137
Seabrook	June 1999	Spent fuel pool	Tritium
Salem	September 2002	Spent fuel pool	Tritium
Three Mile Island	May 2006	Condensate storage tank	Tritium
Watts Bar	August 2002	Effluent release pipe and SFP transfer tube sleeve	Tritium and mixed fission products

SOURCE: USNRC (2006).

Many of the uncontrolled liquid release events documented in Table 2.5 have resulted in groundwater contamination at plant sites. Liquid leakage that enters the subsurface can frequently go undetected because groundwater monitoring within a licensee's site is only required if the groundwater is used for drinking or irrigation purposes. In the offsite environment, groundwater monitoring is required only if groundwater sources are likely to be impacted by the operation of the nuclear plant. Consequently, there are no regulatory requirements for the regular monitoring of groundwater for the purpose of detecting inadvertent radioactive contamination and its fate and transport either on- or offsite.

As a result of lack of historical groundwater monitoring data, estimation of public dose impacts arising from uncontrolled liquid releases at many sites has required licensees to retroactively undertake the following activities:

1. Install new groundwater and/or surface water monitoring networks to evaluate current and potential movement of the released liquid(s).
2. Conduct additional radionuclide analyses to define the actual source-term radionuclides and their quantities.
3. Perform supplemental bounding dose calculations to back-calculate potential public health impacts associated with releases.

The USNRC's Liquid Radioactive Release Lessons Learned Task Force (USNRC, 2006) examined available data on uncontrolled release events, including additional monitoring data gathered by licensees after releases were identified. The Task Force did not find any instances where the available data indicated that the near-term health of the public was impacted by uncontrolled liquid releases to the environment (USNRC, 2006, p. 13):

> Based on currently available data for sites with detailed evaluations or monitoring, the inadvertent releases of radioactive liquids to surface and/or to ground-water pathways had a negligible impact on public radiation doses. For many of the identified sites, the lack of a public dose impact resulted from the radioactive contamination remaining within owner controlled areas. For the few events which resulted in detectable radionuclide concentrations in the surface and/or ground-water samples collected outside of the owner controlled area, Dose impacts on members of the public still were determined to be negligible. However, several of the reviewed abnormal release event scenarios did, or potentially could, impact groundwater sources relative to established EPA drinking water standards.

It is beyond the scope of the present study to evaluate the results of this USNRC report. However, if this finding is correct, there is no obvious sci-

entific advantage[16] to including these data as part of any Phase 2 dosimetry study.

A complete understanding of the dose impacts to the public arising from uncontrolled liquid release events would require detailed knowledge of the liquid source terms at the time of release as well as the distribution of released radionuclide concentrations in the environment through time; the latter would require a comprehensive spatial and temporal understanding of the environmental parameters influencing the fate and transport of the released liquid(s). There is considerable uncertainty associated with source terms, subsurface environmental conditions, and subsurface fate and transport behavior at most nuclear plant sites where uncontrolled liquid releases have occurred. The same is true at industrial sites where hazardous chemicals have inadvertently been released to groundwater.

Indeed, it is notoriously difficult to recreate distributions of released subsurface contaminants over time and, hence, difficult to estimate the risks such contaminants have posed, or continue to pose, to public health. The quality and completeness of available data on uncontrolled liquid releases at nuclear plants differs from site to site but, in all cases, uncertainty exists in how these liquids have migrated over time and, thus, the exposure pathways and possible historic doses associated with these releases.

As a result of groundwater contamination associated with uncontrolled liquid releases, the nuclear industry took action in 2006 to implement a voluntary Groundwater Protection Initiative (GPI) (Yhip et al., 2010). In January 2010, the NEI also issued guidelines for the management of buried pipe integrity (NEI, 2010); these guidelines are intended to provide proactive assessment and management of buried piping systems at plants to reduce possibilities of future inadvertent radioactive liquid releases. Both steps have potential to provide future data that might better inform dose impacts to the public living in the vicinity of a nuclear power plant, depending on the quantity and quality of the data being gathered.

2.2 EFFLUENT RELEASES FROM FUEL-CYCLE FACILITIES

Unlike nuclear plants, it is difficult to make general statements about airborne effluent releases from front-end nuclear fuel-cycle facilities, beyond the fact that the majority of releases involve uranium and uranium progeny with lesser amounts of other radionuclides (see Appendix E). Four examples of recent effluent release data for front-end nuclear fuel-cycle facilities are shown in Tables 2.6 through 2.9.

[16]The committee notes that there may be other advantages to taking account of these data in dose estimates, including addressing public concerns.

TABLE 2.6 Stack Effluent Release Rates for the Second Quarter of 2011 for the White Mesa Mill in Utah

	Effluent Release Rates at Release Point (μCi/s)			
Radionuclide	North YC Dryer, Run 1	North YC Dryer, Run 2	Yellowcake Baghouse	Grizzly Baghouse
Natural U	9.21×10^{-3}	7.56×10^{-3}	8.22×10^{-3}	5.96×10^{-6}
Thorium-230	9.25×10^{-7}	1.04×10^{-6}	1.89×10^{-5}	6.67×10^{-8}
Radium-226	2.20×10^{-8}	2.35×10^{-8}	1.88×10^{-7}	Not required
Lead-210	3.94×10^{-6}	3.90×10^{-6}	7.97×10^{-7}	Not required

NOTE: Radon is not measured at this site. Instead, the radiation dose from radon is estimated through calculation.
SOURCE: Denison Mines (2011).

TABLE 2.7 Airborne and Liquid Effluent Releases from the Honeywell Conversion Facility during the Period January 1, 2010, to June 30, 2010

	Reported Releases (curies)	
Radionuclide	Airborne Effluents	Liquid Effluents
Natural U	4.28×10^{-2} (measured)	9.12×10^{-1} (measured)
Radium-226	1.20×10^{-5} (calculated)	3.08×10^{-3} (measured)
Thorium-230	1.22×10^{-4} (calculated)	1.60×10^{-3} (measured)

SOURCE: Honeywell (2010).

- Table 2.6 shows airborne effluent releases from the White Mesa Mill near Blanding, Utah, for the second quarter of 2011. The releases include natural uranium, thorium-230, radium-226, and lead-210.
- Table 2.7 shows airborne effluent releases from the Honeywell Conversion Facility in Metropolis, Illinois, for the first half of calendar year 2010. The releases include natural uranium and two progeny, radium-226 and thorium-230.
- Table 2.8 shows airborne effluent releases for the Paducah Gaseous Diffusion Plant for calendar year 2006. Released effluents include the three naturally occurring isotopes of uranium (uranium-234, 235, and 238), uranium decay progeny (thorium-230), and one fission product (technetium-99) and two actinide isotopes (neptunium-237 and plutonium-239).[17]

[17]Recycled uranium (i.e., uranium obtained from reprocessing spent nuclear fuel) was enriched at the Paducah Gaseous Diffusion Plant between 1953 and 1975. This plant is still reporting releases of fission product and actinide effluents from this recycled uranium, albeit in very small quantities.

TABLE 2.8 Airborne Effluent Releases for the Paducah Gaseous Diffusion Plant for Calendar Year 2006

	Reported Release from Location (curies)							
Radionuclide	C-400 Grouped Sources	C-400 Cylinder Drying Station	C-709/ C-710 Drying Hoods	C-310 Stack	Seal Exhaust/ Wet Air Group	C-409 Dissolver	C-360	Total
Uranium-234	1.39×10^{-4}	1.27×10^{-5}	5.91×10^{-4}	2.04×10^{-3}	1.05×10^{-2}	1.03×10^{-7}	1.31×10^{-4}	1.34×10^{-2}
Uranium-235	4.84×10^{-6}	4.42×10^{-7}	2.05×10^{-5}	7.09×10^{-5}	3.64×10^{-4}	3.59×10^{-9}	4.56×10^{-6}	4.65×10^{-4}
Uranium-238	2.86×10^{-5}	9.94×10^{-6}	5.49×10^{-5}	2.12×10^{-4}	3.65×10^{-3}	1.88×10^{-8}	1.36×10^{-5}	3.97×10^{-3}
Technetium-99	9.40×10^{-3}	1.56×10^{-8}	0.00	3.65×10^{-4}	3.96×10^{-5}	1.07×10^{-7}	0.00	9.80×10^{-3}
Thorium-230	3.95×10^{-7}	8.73×10^{-9}	0.00	3.48×10^{-6}	0.00	1.25×10^{-9}	0.00	3.66×10^{-6}
Neptunium-237	1.76×10^{-5}	3.01×10^{-7}	1.31×10^{-7}	2.93×10^{-5}	1.22×10^{-5}	2.59×10^{-9}	1.66×10^{-5}	7.61×10^{-5}
Plutonium-239	4.65×10^{-8}	5.99×10^{-9}	0.00	1.47×10^{-6}	0.00	8.5×10^{-10}	0.00	1.52×10^{-6}

SOURCE: USEC (2008).

TABLE 2.9 Airborne Effluent Releases for the Nuclear Fuel Services Facility in Erwin, Tennessee for the Period July 1, 2010, to December 31, 2010

	Quantity Released for Radionuclide (curies)				
Release Point	Technetium-99	Thorium-228	Thorium-230	Thorium-232	Uranium-234
Main Stack 416		1.17×10^{-7}	1.17×10^{-7}	1.17×10^{-7}	2.76×10^{-5}
Stack 185	2.67×10^{-7}				1.00×10^{-7}
Stack 234					
Stack 327	1.83×10^{-5}				1.45×10^{-5}
Stack 421	8.07×10^{-7}				1.01×10^{-6}
Stack 424	2.14×10^{-7}				3.81×10^{-8}
Stack 501		4.22×10^{-9}	5.42×10^{-9}	3.62×10^{-9}	1.12×10^{-8}
Stack 502		0.00	0.00	0.00	0.00
Stack 503		2.66×10^{-10}	3.42×10^{-10}	2.28×10^{-10}	7.03×10^{-10}
Stack 573	1.92×10^{-7}				4.32×10^{-8}
Stack 600	2.93×10^{-5}				4.18×10^{-5}
Stack 615	1.07×10^{-7}				3.46×10^{-8}
Stack 646	2.28×10^{-7}				4.23×10^{-8}
Stack 649	1.48×10^{-7}				2.59×10^{-8}
Stack 701	1.85×10^{-6}				1.43×10^{-6}
Stack 702	6.21×10^{-7}				2.52×10^{-7}
Stack 703		1.08×10^{-7}	6.22×10^{-8}	8.83×10^{-8}	6.76×10^{-7}
Stack 704		6.53×10^{-9}	3.76×10^{-9}	5.34×10^{-9}	4.09×10^{-8}
Stack 773		4.47×10^{-8}	5.75×10^{-8}	3.83×10^{-8}	1.18×10^{-7}
Stack 774		2.74×10^{-7}	2.16×10^{-7}	2.38×10^{-7}	5.93×10^{-6}

SOURCE: NFS (2011).

- Table 2.9 shows airborne effluent data for the Nuclear Fuel Services facility in Erwin, Tennessee, for the last half of calendar year 2010. Released effluents include the three naturally occurring isotopes of uranium; natural thorium (thorium-232); uranium and thorium progeny (thorium-228, 230); one fission product (technetium-99); and several actinide isotopes (plutonium-238, 239, 240, and 241 and americium-241).[18]

A key take-away message from an examination of Tables 2.6 through 2.9 is that reported effluent releases from fuel-cycle facilities in recent years

[18]The presence of plutonium-238 in the effluents would not be expected to result from commercial nuclear fuel production. This isotope is produced by irradiating uranium-238 with deuterons and is produced for use in thermoelectric generators. The fission products and actinide effluents are likely from the processing of recycled uranium.

Uranium-235	Uranium-238	Plutonium-238	Plutonium-239	Plutonium-240	Plutonium-241	Americium-241
1.06×10^{-6}	2.93×10^{-7}					
3.10×10^{-9}					8.26×10^{-9}	
		1.15×10^{-8}	9.60×10^{-8}	3.39×10^{-8}	8.55×10^{-7}	5.66×10^{-8}
4.50×10^{-7}					5.65×10^{-7}	
3.13×10^{-8}					2.50×10^{-8}	
1.18×10^{-9}					6.63×10^{-9}	
1.96×10^{-9}	3.92×10^{-9}				4.80×10^{-7}	
0.00	0.00				4.44×10^{-7}	
1.24×10^{-10}	2.47×10^{-10}				3.90×10^{-8}	
1.34×10^{-9}					5.93×10^{-9}	
1.29×10^{-6}					9.05×10^{-7}	
1.07×10^{-9}					3.32×10^{-9}	
1.31×10^{-9}					7.06×10^{-9}	
8.00×10^{-10}					4.56×10^{-9}	
4.43×10^{-8}					5.72×10^{-8}	
7.80×10^{-9}					1.92×10^{-8}	
6.98×10^{-8}	8.51×10^{-8}				2.48×10^{-6}	
4.22×10^{-9}	5.14×10^{-9}				2.36×10^{-7}	
2.08×10^{-8}	4.15×10^{-8}				2.09×10^{-6}	
2.52×10^{-7}	2.95×10^{-7}				6.01×10^{-6}	

are substantially smaller than reported releases from nuclear plants, typically only fractions of curies for each radionuclide.[19] However, it is quite likely that releases were significantly higher in the early years of operation of these facilities similar to what was found for nuclear plants.

The reported releases shown in the table are for normal operations only; they do not include unplanned releases. As for any operating industrial facility, significant unplanned releases from fuel-cycle facilities (as well from nuclear plants) could have large impacts on doses to populations. Moreover, the toxicological risks of uranium releases (in addition to the radiation risks) also need to be taken into account in any epidemiologic study.

[19]Release quantities do not tell the whole story about relative risks. Intake of alpha emitters through inhalation or ingestion can result in substantially higher doses per unit activity released than external exposure to gamma emitters.

2.2.1 Availability of Information on Effluent Releases

With one exception, fuel-cycle facility licensees are required to report their effluent releases to the USNRC (or to agreement-state regulators[20]) on a semiannual basis. The exception is for licensees of gaseous diffusion plants (e.g., the Paducah Gaseous Diffusion Plant; see Table 1.2 in Chapter 1). Prior to 2008, gaseous diffusion plant licensees were required to report their effluent releases on a quarterly basis. From 2008 onward, licensees are only required to report their effluent releases when they renew their facility operating licenses. However, annual reporting of effluent releases to the USEPA is required to meet the 40 CFR 61[21] requirements. In cases where unplanned releases have occurred, such releases would need to be taken into account when making dose estimates for an epidemiologic study.

To the committee's knowledge, data on radioactive effluent releases from individual fuel-cycle facilities have not been compiled into summary form. Consequently, it will be necessary to obtain this information for each facility, either through ADAMS or from plant licensees directly, for use in an epidemiologic study. Given the range of facility types, the fact that some facilities were operating as far back as the 1950s as part of the U.S. weapons program with oversight from the Atomic Energy Commission and its successor agencies (presently the U.S. Department of Energy), and the fact that reporting requirements have varied over the years, the availability of effluent release data prior to the mid 1970s (when the USNRC assumed regulatory responsibility for many of these plants) is unclear.

The committee contacted the licensee for the Nuclear Fuel Services (NFS) facility in Erwin, Tennessee, to determine whether records of effluent releases could be made available. The NFS plant was selected because it has a long operating history (it initiated operations in 1957) and has nearby residents who are concerned about effluent releases from the plant. The committee obtained the following information from NFS (Marie Moore, NFS, verbal communication to K.D. Crowley, February 15, 2012):

- NFS maintains a computerized list of its vital records that were submitted to the USNRC. Almost all of these records are in hard copy (paper or microfiche), and their retrieval would be difficult

[20]Under the USNRC's agreement-state program, states can assume authority to license and regulate certain activities within their borders, including the production and utilization of byproduct materials (radioisotopes), source materials (uranium and thorium), and certain quantities of special nuclear materials. Under the agreement-state program, for example, Utah has assumed the authority to license and regulate the White Mesa Mill in Blanding, Utah.

[21]National Emission Standards for Hazardous Air Pollutants.

and labor intensive. NFS began scanning vital records into an electronic format in 2010.

- A meteorological station was installed at NFS in the mid 1980s, but detailed meteorological data that support environmental monitoring report submittals to the USNRC are only available from 1999 to present.

2.2.2 Data Quality and Suitability for Estimating Radiation Doses

The committee judges that if release data are available, they are likely to be adequate for estimating doses for a Phase 2 epidemiologic study (see Chapter 3). The licensee reports provide effluent data for individual radionuclides for both air emissions and liquid effluents at each point of release. The committee was not able to assess the availability and quality of data for early years of plant operations when releases were highest. However, as was the case for nuclear power plants, the quality of effluent release data in recent years is likely much better than for the early years of operation due to more stringent QA requirements as well as stricter requirements to ensure releases and doses meet regulatory requirements.

2.3 ENVIRONMENTAL MONITORING

Nuclear plants and fuel-cycle facilities are required to have Radiological Environmental Monitoring Programs (REMPs) to monitor radioactivity in the environment around their sites. This program is described in Appendix H. In principle, the data gathered by a licensee's REMP could be used to validate doses estimated from effluent releases and/or provide independent estimates of radiation exposure at the monitoring sites. The potential usefulness of environmental monitoring data for this purpose is discussed in this section.

It is important to note that REMPs at nuclear facilities are not intended to provide a comprehensive assessment of radionuclide distributions and concentrations in the environment surrounding the facilities. Instead, their purpose is to demonstrate that facility operations are in compliance with regulations. Monitoring therefore focuses on sampling of environmental media that might serve as pathways for radiation exposure to humans, based on effluent release pathways and the local site characteristics. The media of interest are air, water, and foodstuffs. Pathways for exposure are internal and external radiation.

The following sections provide examples of environmental monitoring data for nuclear plants. Similar kinds of data are generated for monitoring around fuel-cycle facilities but are not presented in this chapter for the sake of brevity.

2.3.1 Atmospheric Monitoring

For environmental pathways associated with airborne releases, monitoring usually involves air sampling and TLD[22] measurements at various locations in the vicinity of the plant, in addition to the monitoring of foodstuffs (see Section 2.3.3), to determine if radioactive effluent releases are detectable in the environment. Typically, air sampling measurements are made at a minimum of five stations: three stations near the plant boundary in the direction of prevailing winds (i.e., downwind); one in the vicinity of a nearby community likely to have the greatest chance of radiation exposure; and one at a control location 15 to 30 km distant in the opposite direction of prevailing winds (i.e., upwind).

Several types of analyses are carried out on the air samples: Radioiodine is measured weekly, and gross beta activity of particulates (captured on filters) is also measured weekly. Analyses to identify alpha- and beta-emitting radionuclides are made quarterly on composite samples. Typically, radionuclide concentrations measured in air samples at downwind stations are comparable with those at the control station. That is, normal operations of a plant do not result in measurable radionuclide concentrations in air, even though the measurement techniques are quite sensitive and can identify occurrences of releases at distance (e.g., Figure 2.12).

Measurements of direct radiation exposure using TLDs are discussed in detail in Section 2.3.4. These measurements are generally not sensitive enough to detect increases above background levels except at locations close to plant boundaries.

Examples of environmental monitoring data collected at the North Anna (located in Virginia) and Dresden plants are shown in Tables 2.10 through 2.13. The data in these tables further illustrate that, for the 1970s as well as in recent years, environmental monitoring programs did not detect radioactive materials above control (or background) levels at these plants.

2.3.2 Water Monitoring

For environmental pathways associated with liquid effluent releases, monitoring usually involves sampling of surface water, groundwater, and drinking water in locations near the plant, as well as shoreline sediments from existing or potential recreational facilities (see Appendix G). Surface and groundwater samples are analyzed for gamma-emitting isotopes and tritium; drinking water samples are analyzed for gross beta, gamma-

[22]Thermoluminescent dosimeters (TLDs) contain inorganic crystalline materials, typically calcium fluoride (CaF_2) and lithium fluoride (LiF), that record exposure to ionizing radiation.

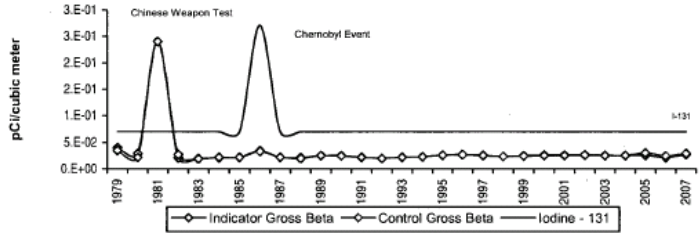

FIGURE 2.12 Measurements of gross beta and iodine-131 activity in air samples at the Fermi plant (located in Michigan) from 1979 to 2007. The measurements are sensitive enough to detect air emissions from Chinese nuclear weapons testing in the early 1980s and the Chernobyl accident in 1986. SOURCE: Detroit Edison (2007).

TABLE 2.10 Results of Environmental Monitoring at the North Anna Plant for 2009

	Indicator Locations, Mean Range (10^{-3} pCi/m^3)	Control Locations, Mean Range (10^{-3} pC/m^3)
Air particulates, gross beta	5.80-32.9	7.65-36.3
Air iodine (I-131)	<LLD	<LLD
Air particulates, beryllium-7[a]	101-267	139-171
Air particulates, cesium-134	1.83-3.34	<LLD
Air particulates, cesium-137	<LLD	<LLD
Air particulates, strontium-89	<LLD	<LLD
Air particulates, strontium-90	<LLD	<LLD

NOTE: LLD, lower limit of detection.
[a]Beryllium-7 is naturally present in the environment.
SOURCE: Dominion (2010b).

TABLE 2.11 Results of Environmental Monitoring at the North Anna Plant carried out by the Virginia Department of Health for 2009

	Indicator Location	Control Location
Air particulates, gross beta, 10^{-3} pCi/m^3	20-40	20-30
Air iodine (iodine-131), pCi/m^3	<0.05-<0.12	<0.10-<0.26

SOURCE: Virginia Department of Health (2009).

TABLE 2.12 Results of Environmental Monitoring at the Dresden Plant for 2009

	Indicator Locations, Mean Range (10^{-3} pCi/m^3)	Control, Mean Range (10^{-3} pCi/m^3)
Air particulates, gross beta	7-43	8-42
Air iodine (iodine-131)	<10-<70	<15-<69
Air particulates, cesium-137	<2-<4	<2-<4
Air particulates, cesium-134	<2-<4	<2-<4

SOURCE: Exelon (2010).

TABLE 2.13 Results of Environmental Monitoring at the Dresden Plant for 1975

	Indicator Locations, Mean Range	Control, Mean Range
Air particulates, gross beta, 10^{-3} pCi/m^3	5-6	5-7
Air iodine (iodine-131), pCi/m^3	<0.03	<0.03

SOURCE: Commonwealth Edison (1976).

emitting isotopes, tritium, and in some cases iodine-131; and sediments are analyzed for gamma-emitting isotopes.

The committee examined water monitoring data from the environmental monitoring reports for Dresden (BWR) and Millstone (PWR) plants. Reports were selected from a recent monitoring period, namely 2009, and an earlier monitoring period, namely 1975. The committee observed that the spatial distribution of monitoring stations for surface water, groundwater, well water, and sediments at these plants were not sufficient to provide a spatial map of environmental radioactivity resulting from liquid effluent releases. This is not surprising given that the goal of the REMP is to obtain measurements to demonstrate regulatory compliance, not to obtain measurements for making radiation dose estimates.

The most frequently detected radiological contaminant in water samples is tritium; see, for example, the measurements around the North Anna plant in Figure 2.13. However, reported tritium concentrations were below USEPA drinking water standards.[23] Cesium-137 was reported in sediment samples at both control and indicator measurement stations around the plant and is thus likely present in the environment due to fallout from above-ground nuclear weapons testing.

Many of the radiological concentration measurements collected under REMP yield values below detection levels. Table 2.14 presents environmental monitoring data for the Dresden plant from the plant licensees' 2009 REMP report (Exelon, 2010). All sampling locations are located within 3 km of the site. Radionuclide concentrations were below detection limits in the vast majority of cases. Tritium was detected in surface and groundwater samples but at levels well below those established by USEPA for drinking water. Monthly composites of surface water samples revealed gross beta concentrations that are similar at indicator and control locations.

[23]The USEPA has established an annual-average maximum contaminant level for tritium in drinking water of 20,000 picocuries per liter (740 becquerels per liter) based on an annual dose equivalent to the whole body of 4 mrem, assuming consumption of 2 liters per day of drinking water.

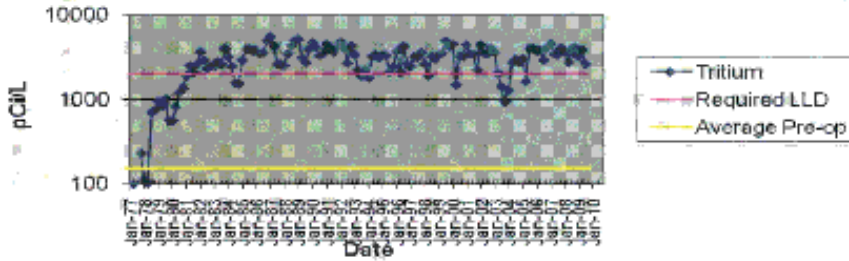

FIGURE 2.13 Variations in tritium concentrations at a surface water monitoring station in the vicinity of the North Anna plant from 1977 to 2010. SOURCE: Dominion (2010b).

Cesium-137 was detected in sediment samples and is likely due to fallout from above-ground nuclear weapons testing.

Dresden has experienced a number of leaks over its 40-year-plus operating history from underground lines and spills from above-ground systems containing radioactive water. These leaks and spills have created areas of subsurface contamination within the plant's protected area.[24] Starting in 2006, Dresden embarked on a Radiological Groundwater Monitoring Program to understand the extent and threat posed by this contamination. The program includes 39 groundwater monitoring wells within the protected area, 26 wells outside the protected area, and 6 surface water sampling points at five different canals and one cooling pond within the controlled area. These 71 locations are sampled twice per year. Short-term monitoring of select areas of shallow groundwater near historic leak points is also conducted using "sentinel" wells.

Appendix F in the 2009 Annual Radiological Environmental Operating Report includes the results of measurements of tritium and 14 radionuclides (beryllium-7, potassium-40, manganese-54, cobalt-58, iron-59, cobalt-60, zinc-65, niobium-95, zirconium-95, iodine-131, cesium-134, cesium-137, barium-140, and lanthanum-140) for the two annual sampling rounds. In 2009, only potassium-40 (in 6 out of 65 samples) and tritium (in 22 of the 39 wells inside the protected area and in 5 of the 26 wells outside the protected area) were found to be above the detectable limits. These data are probably sufficient to create spatial patterns of radiological concentra-

[24]Nuclear plants are demarcated into zones for security purposes. The controlled area of a nuclear plant includes the land on which the plant is built and any surrounding area that is controlled by the plant licensee. Public access to some parts of the controlled area may be allowed by the licensee. The protected area of the plant is a smaller parcel of land within the controlled area that has physical controls (fences, gates, and guards) to prevent public access without licensee permission.

TABLE 2.14 Environmental Monitoring Data for the Dresden Plant for 2009

Media	Frequency	Total Number of Samples	Analysis Type	Required LLD (pCi/L)	Indicator Location, Mean Range (pCi/L)	Control Location, Mean Range (pCi/L)
Surface Water (3 stations)	Monthly	36	Gross beta	4	4.8-10.9 (12/12)	4.0-10.4 (22/24)
	Monthly	36	Gamma			
			Manganese-54	15	< LLD	< LLD
			Cobalt-58	15	< LLD	< LLD
			Iron-59	30	< LLD	< LLD
			Cobalt-60	15	< LLD	< LLD
			Zinc-65	30	< LLD	< LLD
			Niobium-95	15	< LLD	< LLD
			Zirconium-95	30	< LLD	< LLD
			Iodine-131	15	< LLD	< LLD
			Cesium-134	15	< LLD	< LLD
			Cesium-137	18	< LLD	< LLD
			Barium-140	60	< LLD	< LLD
			Lanthanum-140	15	< LLD	< LLD
	Quarterly	12	Tritium (H-3)	2000	181-537 (2/4)	465-621 (3/8)
Ground/ well water (2 stations)	Quarterly or more	16	Tritium (H-3)	2000	250-725 (12/16)	N/A
	Quarterly or more	16	Gamma			
			Manganese-54	15	< LLD	N/A
			Cobalt-58	15	< LLD	N/A
			Iron-59	30	< LLD	N/A
			Cobalt-60	15	< LLD	N/A
			Zinc-65	30	< LLD	N/A
			Niobium-95	15	< LLD	N/A
			Zirconium-95	30	< LLD	N/A
			Iodine-131	15	< LLD	N/A
			Cesium-134	15	< LLD	N/A
			Cesium-137	18	< LLD	N/A
			Barium-140	60	< LLD	N/A
			Lanthanum-140	15	< LLD	N/A

TABLE 2.14 Continued

Media	Frequency	Total Number of Samples	Analysis Type	Required LLD (pCi/L)	Indicator Location, Mean Range (pCi/L)	Control Location, Mean Range (pCi/L)
Aquatic sediment (1 station)	Semi-annually	2	Gamma		pCi/kg dry	
			Manganese-54	N/A	< LLD	N/A
			Cobalt-58	N/A	< LLD	N/A
			Iron-59	N/A	< LLD	N/A
			Cobalt-60	N/A	< LLD	N/A
			Zinc-65	N/A	< LLD	N/A
			Niobium-95	N/A	< LLD	N/A
			Zirconium-95	N/A	< LLD	N/A
			Iodine-131	N/A	< LLD	N/A
			Cesium-134	150	< LLD	N/A
			Cesium-137	180	87 (1/2)	N/A
			Barium-140	N/A	< LLD	N/A
			Lanthanum-140	N/A	< LLD	N/A
Dredge spoils (2 stations)	When river was dredged	6	Gamma		pCi/kg dry	
			Manganese-54	N/A	< LLD	N/A
			Cobalt-58	N/A	< LLD	N/A
			Iron-59	N/A	< LLD	N/A
			Cobalt-60	N/A	< LLD	N/A
			Zinc-65	N/A	< LLD	N/A
			Niobium-95	N/A	< LLD	N/A
			Zirconium-95	N/A	< LLD	N/A
			Iodine-131	N/A	< LLD	N/A
			Cesium-134	150	< LLD	N/A
			Cesiums-137	180	95-142 (4/6)	N/A
			Barium-140	N/A		N/A
			Lanthanum-140	N/A		N/A

NOTE: LLD, lower limit of detection.
SOURCE: Exelon (2010, Table A-1).

tions for tritium. However, reported offsite concentrations of tritium are very low (208 to 322 pCi/L, just above minimum levels of detection). It would thus appear that most groundwater contamination currently remains onsite, limiting the value of these data for use in estimating doses for an epidemiologic study. Nonetheless, this monitoring program is important for understanding potential future risks.

Table 2.15 presents results from the environmental monitoring program at the Millstone plant for 2009. As can be seen in the table, radioisotope concentrations were below detection limits in the vast majority of instances. Tritium was detected in seawater samples at one location (location 32), which is in the vicinity of the plant's discharge point and probably has not undergone significant aquatic mixing that would dilute radiological concentrations. However, levels of tritium were well below USEPA drinking water standards. Detectable levels of naturally occurring potassium-40 were also reported in seawater, well water, and bottom sediment samples. Cesium-137 was detected in sediment samples and is likely due to fallout from above-ground nuclear weapons testing. Thorium-228 was also detectable in a number of sediment samples.

The Connecticut Department of Environmental Protection (DEP) performs independent checks on certain of Millstone's environmental measurements. A DEP comprehensive review of historical Millstone environmental monitoring data in 2006 (DEP, 2006) concluded that "the collective sampling in and around Millstone Power Station show expected levels of residual fallout from weapons testing and the Chernobyl event and are unrelated to the operation of the Millstone Power Station."

At Millstone, a cross-comparison between the liquid effluent monitoring program and the REMP program can be made by comparing tritium monitoring results at location 32-I, which is in the vicinity of the plant's effluent discharge location. Figure 2.14 shows a 5-year cross-comparison provided by the licensee. The cross-comparison indicates good agreement between the measurements from the effluent monitoring and environmental monitoring programs, providing a level of confidence in the data reported by both programs.

2.3.3 Foodstuff Monitoring

Nuclear plant licensees are required to monitor for radioactivity in foodstuffs that are grown in the vicinity of their plants. This includes monitoring for radioactivity in milk, fish and invertebrates, food products (e.g., corn and other grains), and broad-leaf vegetables. The following sampling and analysis activities are required:

- Milk: Samples from milking animals at three locations within 5 km having the highest dose potential and one sample from milking

TABLE 2.15 Environmental Monitoring Data for the Millstone Plant for 2009

Environmental Media	Radionuclide	Indicator Mean	Control Mean
Well water (pCi/L)	Barium-140	LLD	
	Beryllium-7	LLD	
	Cobalt-58	LLD	
	Cobalt-60	LLD	
	Chromium-51	LLD	
	Cesium-134	LLD	
	Cesium-137	LLD	
	Iron-59	LLD	
	Tritium	LLD	
	Iodine-131	LLD	
	Potassium-40	79	49
	Lanthanum-140	LLD	
	Manganese-54	LLD	
	Niobium-95	LLD	
	Rutheium-103	LLD	
	Ruthenium-106	LLD	
	Antimony-125	LLD	
	Strontium-89	LLD	
	Strontium-90	LLD	
	Thorium-228	LLD	
	Zinc-65	LLD	
	Zirconium-95	LLD	
Seawater (pCi/L)	Barium-140	LLD	
	Beryllium-7	LLD	
	Cobalt-58	LLD	
	Cobalt-60	LLD	
	Chromium-51	LLD	
	Cesium-134	LLD	
	Cesium-137	LLD	
	Iron-59	LLD	
	Tritium	848	
	Iodine-131	LLD	
	Potassium-40	286	279
	Lanthanum-140	LLD	
	Manganese-54	LLD	
	Niobium-95	LLD	
	Ruthenium-103	LLD	
	Ruthenium-106	LLD	
	Antimony-125	LLD	
	Thorium-228	LLD	
	Zinc-65	LLD	
	Zirconium-95	LLD	

continued

TABLE 2.15 Continued

Environmental Media	Radionuclide	Indicator Mean	Control Mean
Bottom sediment (pCi/g dry)	Silver-110m	LLD	
	Beryllium-7	LLD	
	Cobalt-58	LLD	
	Cobalt-60	LLD	
	Chromium-51	LLD	
	Cesium-134	LLD	
	Cesium-137	0.153	
	Iron-59	LLD	
	Iodine-131	LLD	
	Potassium-40	18.1	14.2
	Manganese-54	LLD	
	Niobium-95	LLD	
	Ruthenium-103	LLD	
	Ruthenium-106	LLD	
	Antimony-125	LLD	
	Thorium-228	2.71	
	Zinc-65	LLD	
	Zirconium-95	LLD	

NOTE: LLD, lower limit of detection.
SOURCE: Dominion Nuclear Connecticut, Inc. (2010, Section 3.1).

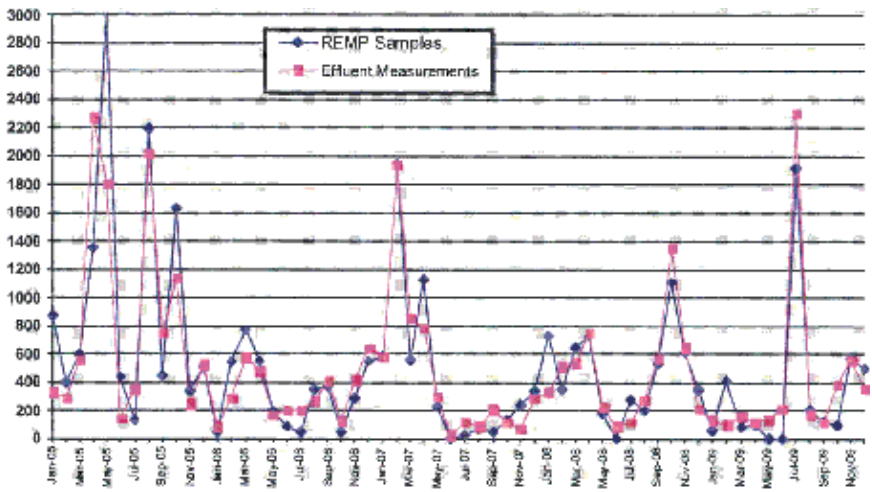

FIGURE 2.14 Five-year comparison between liquid effluent monitoring data and environmental monitoring data for tritium at location 32-I at the Millstone Plant.
SOURCE: Dominion Nuclear Connecticut, Inc. (2010).

animals at a control location. The samples must be analyzed for gamma isotopes and iodine-131.

- Fish and invertebrates: Samples of each commercially and recreationally important species in the vicinity of plant discharge areas as well as samples in areas outside the influence of plant discharges. The edible portions of samples must be analyzed for gamma isotopes.
- Food products: One sample of each principal class of food products from areas irrigated with water into which liquid effluents have been discharged. The edible portions must be analyzed for gamma isotopes.
- Broad-leaf vegetables: If milk sampling is not performed,[25] three different kinds of broad-leaf vegetables must be sampled and analyzed for gamma isotopes and iodine-131. Additionally, samples of broad-leaf vegetables grown 15-30 km distant from the plant in the least prevalent wind direction must also be analyzed for gamma isotopes and iodine-131.

Some nuclear plants have arranged with local landowners to sample from their properties. In some cases, licensees have established gardens on plant sites to obtain the necessary samples.

Environmental measurements of foodstuffs around nuclear plants generally show no activity above control levels. In fact, most measurements are below detection limits.

2.3.4 Direct Radiation Monitoring

Direct radiation exposure primarily occurs as a result of external irradiation from radioactive materials released into the atmosphere (mainly noble gases), deposited on the ground (mainly iodine and particulates), or contained in surface water and sediments (lakes or streams). Direct exposure can also occur as a result of exposure to external irradiation from radioactive waste and spent fuel stored onsite and from induced radioactivity in BWR turbines. Exposure to direct radiation from onsite sources would only be a concern for plant workers and persons living close to the plant boundary.

The USNRC requires licensees to monitor direct radiation in the environment. Licensees are required to use specific characteristics at each site to develop a surveillance program that meets regulatory requirements. The USNRC provides generic guidance to licensees on sampling and measurement types, numbers, and frequencies (USNRC, 1977, 1978). Each facility

[25]Not all nuclear plants are located in proximity to dairy farms.

develops its own site-specific sampling plan subject to approval by the USNRC (e.g., Exelon, 2011).

TLD measurements are generally made at several dozen locations in rings around the plant boundary. The inner ring is generally located close to the plant boundary, whereas the outer ring is generally located at a distance of about 5-10 km from the boundary. Additional dosimeters are placed at one or more distant control locations and at other locations of special interest, such as more highly populated areas or in prevailing downwind areas. Figure 2.15 shows the arrangement of environmental monitoring stations around the Millstone plant. Plants may supplement or substitute the passive detectors at some locations with active detectors such as continuous monitors (e.g., high-pressure ionization chambers [HPICs] or scintillation detectors). The passive detectors generally are measured (and replaced) quarterly, whereas the active detectors, if used, provide real-time data.

In addition to radiation monitors, continuous air sampling is also carried out as described in Section 2.3.1. The air sampling data can be used to estimate (or bound) the deposition density of iodine and particulates, and resultant external exposure rate, for comparison with model calculations based on measured particulate and iodine release rates.

The purpose of direct exposure monitoring is to demonstrate that the

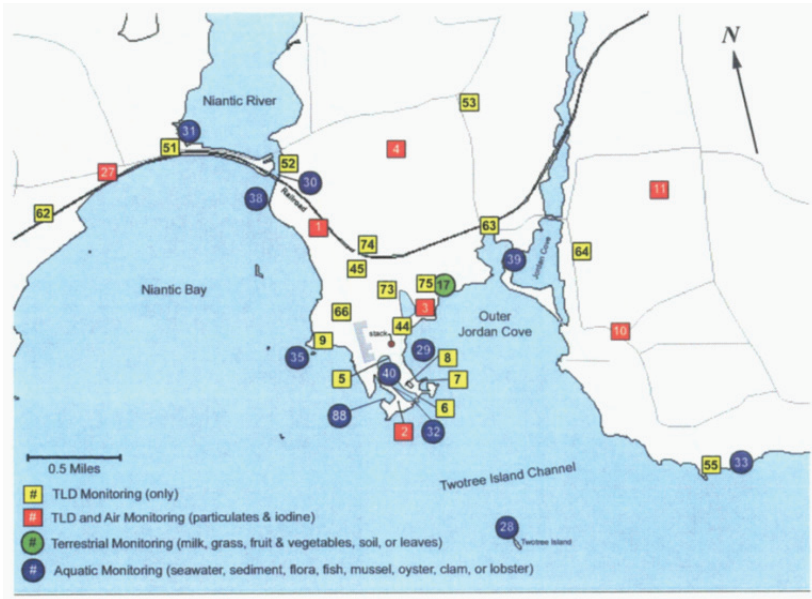

FIGURE 2.15 Environmental monitoring sites around Millstone Point Nuclear Power Station located in Connecticut. SOURCE: Dominion Nuclear Connecticut, Inc. (2010).

integrated radiation exposure at any location outside the facility boundary does not exceed levels that might have resulted in a dose to any individual greater than the operational limits set by regulations. The ability to demonstrate this depends on whether the quarterly integrated passive detector measurements are accurate and precise enough to allow one to distinguish increases in integrated exposures from the facility from the temporal and spatial variations in natural background exposures at the site (see below).

The quality of environmental measurements using TLDs has improved steadily over the years (Klemic et al., 1999). Nevertheless, measured exposures are uncertain due to fading and calibration error (energy response).[26] Contemporary intercomparisons of TLD and other dosimeters used for monitoring environmental radiation levels have demonstrated that over 80 percent of the dosimeters tested were able to predict a field reference value within 30 percent (one standard deviation), only about 60 percent were able to reproduce a laboratory calibrated dosimeter value within 10 percent (one standard deviations) (NCRP, 2007).

Earlier intercomparisons suggested even greater uncertainty. In an intercomparison exercise conducted in 1974, the predicted exposure by 50 sets of passive dosimeters exposed to an integrated exposure of 16 millirad (mrad) varied around the actual exposure value by 25 percent (one standard deviation) (Beck, 1975). A study by USEPA at the Haddam Neck Station in 1974 determined that the TLD data reported by the facility predicted background levels inconsistent with USEPA's independent measurements (Kahn et al., 1985).

A careful TLD measurement program should be capable of identifying increases over background levels that might approach the design objectives for power reactors of 15 mrem to any organ.[27] However, such programs are generally not capable of verifying the small predicted increases in exposures due to routine effluent releases from nuclear plants. For example, TLD data reported for the Dresden plant during the 2009 July-September quarter (Exelon, 2010) varied from 20-28 mR over 16 locations in the inner ring around the plant. Two sets of dosimeters (two CaF_2, two LiF) were exposed at each location. At two locations the quarterly exposures differed by as much as 5 mR (22 vs. 27 and 22 vs. 26).

A location far from the facility in a sector toward which the wind blows infrequently is often used as a control site to demonstrate that no significant increases occurred at any of the measurement locations closer

[26]Note that for detecting increases in exposure due to facility releases, it is measurement precision that is most important; the accuracy of the integral exposure at a particular location is generally biased due to shielding of the TLDs as a result of their placement on walls of buildings or on telephone poles.

[27]Operating limits are established to control the amounts of radioactive materials released from nuclear plants. The USNRC requires these limits to be established in accordance with the design objectives in 10 CFR 50, Appendix I.

to the plant due to effluent releases. However, this assumes that ambient temporal variations in natural background at the control location were the same as at the other measurement locations, which is not necessarily a valid assumption. Annual exposures can vary temporally by as much as 10 mR per year due to variations in soil moisture, and they can vary spatially, even at locations only a few hundred meters apart, due to variations in soil composition (Beck and Miller, 1982), consistent with the spatial variation in the Dresden plant TLD data (see Section 3.5 in Chapter 3).

Lang et al. (1987) studied TLD data collected at the Hatch plant (located in Georgia) over a 4-year period. They concluded that it would be very difficult to detect increases in 3-month exposures below 10 percent of average background levels from TLD data because of measurement error and spatial and temporal variations in natural background radiation levels.

The maximum (i.e., MEI) annual external radiation exposure from airborne effluent releases from nuclear plants is currently estimated as << 1 mR per year (USNRC, 2009). Although airborne effluent releases from some nuclear plants in the 1970s and 1980s were up to 1000 times higher than current releases (UNSCEAR, 1982,1988, 1993, 2000, 2008; see also Section 2.1.1 in this chapter), estimated maximum quarterly integrated exposures for most plants were still likely less than 1-2 mR (see Chapter 3). Even if changes on the order of a few mR per quarter could be detected, they could not be unambiguously attributed to effluent releases from nuclear plants because of variations in natural background. Consequently, the passive monitoring systems around nuclear plants cannot be used to quantify increases in exposure resulting from routine effluent releases and therefore cannot be used to validate estimated population doses.

Real-time monitors, when used, can provide quantitative information on actual increases in exposure rates at a plant due to airborne effluent releases and can be used to validate estimates based on measured release rates. Several sites do monitor external radiation levels using HPIC detectors. For example, the state of Illinois maintains an array of HPIC detectors around the Dresden plant. An example of HPIC measurements made at various distances from a nuclear plants site in the northeastern United States is shown in Figure 2.16 (Beck et al., 1972).

As discussed later in this chapter, fluctuations in exposure rates above background can be integrated to estimate exposure for comparison with the estimated levels calculated from the reported plant effluent releases. This provides an independent verification of the reported effluent release levels.

2.3.5 Monitoring Deposited Radionuclides

Continuous air sampling measurements generally have lower limits of detection that are below the levels of airborne particulates and iodine that actually occur as a result of plant releases during normal operations. Con-

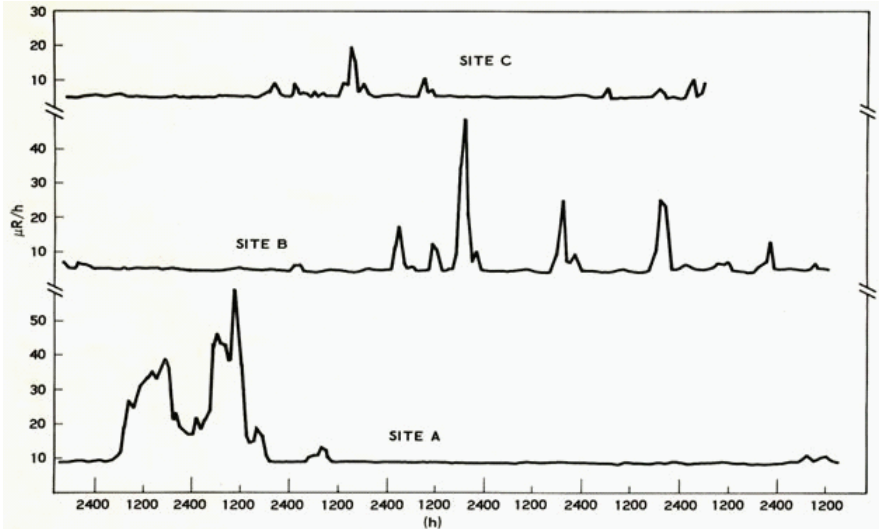

FIGURE 2.16 Mean hourly exposures over a 1-week period at three sites near at the Millstone plant. Site A is located inside the fence line; Site B is located approximately 2 km from the stack; and Site C is located several kilometers away from the stack. SOURCE: Beck et al. (1972).

sequently, such measurements are generally not useful for validating specific calculations of air activities, and possible ground contamination, based on measured release rates.[28] Plant licensees collect and analyze soil samples at a few locations around their facilities at least annually. But even after years of plant operation, the total increase in soil activity is either too low to detect or too low to distinguish from background levels. Soil and air sampling data can, however, be used to provide an upper bound on dose estimates. Because predicted levels of exposure rates from deposited radionuclides released by a plant are only small fractions of the estimated exposures from noble gas releases, these potential direct radiation exposures cannot generally be detected by the plant's passive monitoring systems either.

Monitoring programs based on arrays of passive detectors are adequate (as intended) for demonstrating compliance with operational limits on maximum exposure to any individual (i.e., the MEI), but they are not useful for confirming direct exposure at any specific location based on measured release rates, nor are they useful for estimating population doses for an epidemiologic study. Air sample data collected by plant licensees are not

[28]However, air monitors are useful for detecting and quantifying activity in air that might result from an accident or abnormal release that could result in potential doses approaching or exceeding regulatory limits.

sensitive enough to estimate deposition of radionuclides from the plant, nor are analyses of soil or vegetation samples.

2.3.6 Independent Validation Studies of Environmental Monitoring Programs

A number of independent entities conduct studies on radioactive effluent releases, environmental radioactivity, and maximum dose estimates to independently corroborate data collected by plant licensees. In the early years of nuclear plant operations, USEPA and Atomic Energy Commission research organizations conducted numerous independent studies in the environment around plants, measuring external radiation levels and radionuclide concentrations in plants, animals, and water (e.g., Beck et al., 1972; Blanchard et al., 1976; Carter et al., 1981; Kahn et al., 1970, 1971,1974; Gogolak, 1973; Gogolak and Miller, 1974a,b; Voilleque et al., 1981; Weiss et al., 1974).

In almost all instances, these studies did not detect radionuclides attributable to nuclear plants in environmental samples, even when plants were emitting much greater amounts of activity than at present. Independent estimates of MEI doses from noble gases and iodine-131 in milk were also generally of the same order as those reported by plant operators, generally confirming that radioactive effluents from the plants were not being significantly underestimated. Some of the studies also provided direct confirmation of reported release and atmospheric diffusion calculations.

Some states also conduct independent monitoring around nuclear plants.[29] For example, the state of Texas conducts environmental monitoring activities within the 10-mile emergency planning zones of its two nuclear plants (Comanche Peak and South Texas). The state deploys solid-state detectors to measure direct radiation and air monitors to measure gaseous effluents, particulates, and radioiodine. The state also samples liquids, vegetation, sediments, and fish and invertebrates for radioactivity.

The state of Illinois conducts independent monitoring near its six operating nuclear plants (Braidwood, Byron, Clinton, Dresden, LaSalle, and Quad Cities) as well as some shut down facilities. The state maintains a network of 415 environmental dosimeters to measure and document ambient gamma radiation levels within 10-mile (~16 km) radii of these plants. The state also collects samples of water, sediment, fish, milk, and vegetables from 132 locations (see iema.illinois.gov). A committee subgroup observed

[29]The USNRC provided funding to states to carry out environmental monitoring around nuclear plants from 1979 to 1997. Support was discontinued because state programs were seen to duplicate licensee REMPs. Several states (e.g., Illinois, New Jersey, Pennsylvania, Texas, and Washington) have continued to conduct environmental monitoring with their own funding.

real-time data being collected by the state around the Dresden plant using an HPIC detector.

Some states have their own onsite inspectors at nuclear plants in addition to the USNRC's resident inspectors. For example, the Pennsylvania Bureau of Radiation Protection assigns a nuclear engineer to each of the state's five nuclear plants (Beaver Valley, Limerick, Peach Bottom, Susquehanna, and Three Mile Island) to review operating procedures, conduct inspections, and maintain an awareness of environmental monitoring programs run by plan licensees.[30] The Bureau also monitors environmental dosimeters at 30 locations. New Jersey also has its own REMP.[31]

Environmental monitoring around one nuclear plant is also being carried out by a private entity. The C-10 Foundation[32] is monitoring airborne radioactivity and wind speeds and directions in Massachusetts and New Hampshire communities that are located within the 10-mile emergency planning zone for the Seabrook plant. The monitoring data are available in near real time.

In addition to the various validation studies specific to nuclear plants described above, there have been a number of more recent studies validating atmospheric transport models similar to those used at USNRC-licensed facilities (Brown, 1991; Napier et al., 1994; Rood et al., 1999; Thiessen et al., 2005). There have also been a number of other recent studies that describe the validation of models used for estimating doses resulting from releases of various radionuclides to the environment that are similar to the models used for estimating doses from USNRC-licensed facilities (BIOMOVS, 1991; IAEA, 2003; Till et al., 2000) (see Chapter 3 for a discussion of dose assessment).

2.3.7 Utility of Environmental Monitoring Data for Estimating Radiation Doses

As described in Sections 2.3.1 to 2.3.3, nuclear plant licensees are required to measure radioactivity in the environment surrounding their facilities, including in the air, water, and foodstuffs. Almost all environmental measurements reported by plant licensees, even in early years of plant operations when radioactive effluent releases were much higher than

[30]See http://www.nei.org/resourcesandstats/publicationsandmedia/insight/insight-web-extra/revealing-the-green-side-of-nuclear-energy-power-plants-closely-monitored-to-protect-the-environment/.

[31]See http://www.nj.gov/dep/rpp/bne/index.htm.

[32]This not-for-profit foundation was established in 1991, when the Seabrook plant began operations. The foundation's environmental monitoring activities are carried out under contract with the Massachusetts Department of Public Health.

at present, are either below minimum detection limits (MDLs) or are not sensitive enough for use in dose estimation. Consequently, monitoring data can play only a minimal role in the calculation of doses received by populations residing in the vicinity of nuclear facilities.

Environmental concentrations of radionuclides released from nuclear plants and the resulting absorbed doses must instead be calculated from estimated effluent releases, as described in Chapter 3. The committee judges, however, that the measured environmental concentrations, even if they are usually below MDL, are useful for assessing upper bounds of dose in the vicinity of nuclear plants. In addition, the usually rare measurements above the MDL can be used to assess the validity of the reported effluent releases or the method of calculation of environmental concentrations.

2.4 AVAILABILITY OF METEOROLOGICAL DATA

Estimates of doses from airborne emissions require detailed information on both radioactive effluent releases and the local meteorology at the time those releases occurred. All nuclear plants are required to conduct meteorological monitoring (see Appendix F) for use in estimating offsite doses from airborne effluents. For continuous releases, facilities generally use average annual values for wind speed and direction as a function of atmospheric stability and release height to estimate offsite doses. However, to estimate doses for sporadic batch releases, data are required for the actual times of release because local meteorology can vary significantly over short time intervals.

As discussed previously in this chapter, airborne releases of primary importance from nuclear plants are noble gases, tritium, and carbon-14. One needs to know the direction and strength of the wind and the state of the atmosphere to estimate transport of these releases. Transport of noble gases is unaffected by rain. However, this would not be the case for facilities that release radioactive particulates, which would include many fuel cycle facilities.

The committee could not determine the extent to which detailed meteorology data are readily available for all plants and years of operation. Some plant licensees report annual meteorological data in their REMP reports. More detailed meteorology data may need to be recovered directly from facility licensees or from nearby meteorological stations. If detailed meteorology data are not available for plants with significant batch releases or highly time-variable continuous releases, then estimated doses may be significantly more uncertain than those for plants with relatively time-invariable continuous releases. However, batch releases are generally significant only for PWRs. However, as shown earlier in this chapter, airborne releases for PWRs tend to be lower than for BWRs.

2.5 FINDINGS AND RECOMMENDATIONS

This chapter provides the committee's assessment of the availability, completeness, and quality of information on airborne and liquid radioactive effluent releases and direct radiation exposure from nuclear facilities to support an epidemiologic study. Based on its assessment, the committee finds that:

1. Effluent release and direct exposure data collected by facility licensees, when available, are likely to be sufficiently accurate to develop a population-level dose reconstruction that provides rough estimates in annual variations in dose as a function of distance and direction from nuclear facilities (see Sections 2.1.3 and 2.2.2). However, even when available, such data would not be sufficient to support detailed reconstructions of doses to specific individuals living near nuclear facilities, which would require very precise information on the whereabouts and dietary habits of the individuals under consideration. Facility-specific evaluations will be required to determine the availability and quality of the effluent release and direct exposure data. These data are likely to be of better quality for later years of facility operations relative to earlier years because of improved QA procedures (see Sections 2.1.4 and 2.2.3).
2. Carbon-14 releases from nuclear plants may make a significant contribution to population dose, especially in recent years. However, plant licensees have not been required to estimate or report carbon-14 releases until 2010. It will be necessary to develop a methodology for estimating releases of carbon-14 prior to 2010 to support dose estimation for an epidemiologic study.
3. Meteorology data collected by nuclear plants and fuel-cycle facilities are probably adequate to support estimates of radiation doses for continuous effluent releases. However, the committee was unable to determine the extent to which detailed meteorology data are readily available for all facilities and years of operation. Facility-specific evaluations will be required to determine the availability and quality of meteorology data to support dose estimation for an epidemiologic study (see Section 2.4).
4. Environmental monitoring data have limited usefulness for estimating doses from effluent releases around nuclear plants and fuel-cycle facilities. Almost all environmental measurements reported by facilities are either below the MDLs or are not sensitive enough to allow for the development of adequate dose estimates. Data from environmental monitoring that are above MDLs can, however, be used to validate reported effluent releases or the methods of dose calculation (see Sections 3.3 and 3.6 in Chapter 3).

5. Obtaining and digitizing effluent release and meteorology data for use in an epidemiologic study will be a large and costly effort. Existing digitized data for nuclear plants are of marginal usefulness (see Section 2.1.3), and to the committee's knowledge such data do not exist in electronic form for fuel-cycle facilities. It may be necessary to contact individual licensees to obtain these data, in addition to information on surface water dispersion of effluents, and information on the use that is made of the environment around facilities. Data may not be available for all facilities and all years of operation.

In light of these findings (especially Findings 1, 2, and 5), the committee recommends that a pilot study be undertaken to demonstrate the feasibility of obtaining sufficient data on effluent releases, dispersion of the released activities in the atmosphere and surface waters, and the use that is made of the environment around facilities for use in dose estimation to support an epidemiologic study. This pilot study should:

- Obtain effluent release, direct exposure, and meteorology data for the six nuclear plants and one fuel-cycle facility discussed in Section 2.1.3 for their entire periods of operation; the committee suggests Dresden (Illinois), Millstone (Connecticut), Oyster Creek (New Jersey), Haddam Neck (Connecticut), Big Rock Point (Michigan), San Onofre (California), and Nuclear Fuel Services (Tennessee) for the reasons described in Section 2.1.3. If data from these facilities are not available, then other facilities having similar characteristics should be selected.
- Digitize these data into a form that is usable for dose estimation (see Chapter 3).
- Develop interpolation algorithms for estimating effluent releases for sites and/or years when detailed effluent release data are not available.
- Develop a methodology for estimating releases of carbon-14 from the six nuclear plants for all years of plant operations.

The results of this pilot study should be used to inform decisions about any Phase 2 epidemiologic study effort.

Finally, the USNRC did not ask the National Academy of Sciences to review effluent release monitoring and reporting requirements as part of this study. Nevertheless, the committee notes that it would be useful for the USNRC to review these requirements to determine if they can be adjusted to improve the usefulness of effluent release, meteorological, and environmental monitoring data for future dose reconstructions. Making such data

freely available to the public in summary form (as the USNRC is doing now with its Effluent Database for Nuclear Plants; see Section 2.1.3) could be an important step for informing the public about these releases.

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3

Radiation Dose Assessment

This chapter addresses the first charge in the statement of task for this study (see Sidebar 1.1 in Chapter 1) on methodological approaches for assessing offsite radiation doses to populations near nuclear plants and fuel-cycle facilities in the United States. It is specifically intended to address the following issues:

- Pathways, receptors, and source terms.
- Approaches for overcoming methodological limitations arising from the variability in radioactive releases over time as well as other confounding factors.
- Approaches for characterizing and communicating uncertainties.

Information on the availability, completeness, and quality of radioactive effluent releases from nuclear facilities, which is also part of this first charge, was addressed in Chapter 2.

3.1 BACKGROUND ON DOSE ASSESSMENT AND DOSE RECONSTRUCTION

When ionizing radiation interacts with the human body it transfers part or all of its energy to the molecules and cells of body tissues. The response of these tissues to the deposition of energy in terms of physical, chemical, and biological changes is dependent on the amount of energy deposited per unit mass of tissue, or *absorbed dose* (see Table 3.1). The quantity absorbed dose (D) is defined as the mean energy imparted by ionizing radiation per

TABLE 3.1 Selected Quantities and Units for Radiation Exposure and Dose

Quantity	Old Unit	SI Unit or Its Special Name	Relationship Between Units	Field of Application	Reference
Exposure	R	C kg^{-1}	1 R = 2.58 10^{-4} C kg^{-1}	Monitoring	NCRP (2007)
Absorbed dose	rad	Gy	1 rad = 0.01 Gy	Research	ICRP (2007b)
Dose equivalent[a]	rem	Sv	1 rem = 0.01 Sv	Radiation Protection	ICRP (1977)
Equivalent dose[a]	rem	Sv	1 rem = 0.01 Sv	Radiation protection	ICRP (1991)
Effective dose equivalent[b]	rem	Sv	1 rem = 0.01 Sv	Radiation protection	ICRP (1991)
Effective dose[b]	rem	Sv	1 rem = 0.01 Sv	Radiation protection	ICRP (1991)
Committed effective dose equivalent[c] (CEDE)	rem	Sv	1 rem = 0.01 Sv	Radiation protection	ICRP (1991)
Collective dose equivalent	person-rem	person-Sv	1 person-rem = 0.01 person-Sv	Radiation protection	ICRP (1991)

[a]Dose equivalent and equivalent dose are conceptually similar. However, dose equivalent makes use of quality factors (QFs), which were replaced with radiation-weighting factors (w_R) for the calculation of equivalent doses.

[b]Effective dose equivalent and effective dose are conceptually similar. Effective dose equivalent is the weighted sum of the dose equivalents over all organs and tissues of the body, using tissue-weighting factors (w_T), whereas effective dose is the weighted sum of the equivalent doses over all organs and tissues of the body. An additional difference is that different w_T values are used in the calculation of effective dose equivalent and effective dose.

[c]Committed effective dose equivalent is the time integral of the effective dose equivalent from the time of the activity intake until the age of 70 y.

unit mass at a point of interest. The unit of absorbed dose is J/kg, and its special name is the gray (Gy) (ICRU, 2011). Although defined as a point quantity, absorbed dose usually represents an average over some finite volume or mass, such as the mass of the thyroid or the volume of red bone marrow distributed in the entire body. When the absorbed dose has approximately the same value for all organs and tissues of the body, as is the

case for direct radiation[1] from energetic gamma rays or internal irradiation from inhalation or ingestion of cesium-137, it is common to use the term *whole-body absorbed dose*.

The quantity referred to as *dose equivalent* (H_T) is also used in some dose calculations, for example, for calculating doses to the *maximally exposed individual*, or MEI[2] (USNRC, 1977a) around nuclear facilities (see Table 3.1). Dose equivalent is defined as the absorbed dose modified by a quality factor (QF) that represents the relative biological effectiveness of a radiation type:

$$H_T = D \times QF \quad (1)$$

In the U.S. Nuclear Regulatory Commission's (USNRC's) fundamental regulatory radiation protection guidance (10 CFR Part 20, Standards for Protection Against Radiation), QF takes on values of unity (1) for X rays, gamma rays, and beta radiation; 20 for alpha particles, fission fragments, and heavy particles of unknown charge; and 10 for high-energy protons and neutrons of unknown energy.

More recent radiation protection guidance from the International Commission on Radiological Protection (ICRP) defines other dose quantities. These include *equivalent dose* and *effective dose* (ICRP, 1991; see Table 3.1).

As radiation protection guidance has evolved over the years, the application of various dose quantities has become more clearly prescribed. For example, as stated in ICRP Publication 103 (2007b):

> The main and primary uses of effective dose in radiological protection for both occupational workers and the general public are:
>
> - prospective dose assessment for planning and optimization of protection; and
> - retrospective dose assessment for demonstrating compliance with dose limits, or for comparing with dose constraints or reference levels.

Thus, effective dose and equivalent dose have been used for regulatory

[1]As noted in Chapter 2, direct radiation exposure refers to external whole-body radiation exposure from ionizing radiation emitted by radionuclides in the air, soil, sediments, or water bodies as well as radiation from sources within the site boundary. The latter includes radioactive wastes buried or stored onsite as well as N-16 produced in the turbines of boiling water reactors.

[2]MEI is a regulatory construct for assessing compliance with radiation protection standards. It refers to a hypothetical individual who is postulated to receive the maximum possible radiation dose from a facility because of his or her location relative to the facility as well as lifestyle habits.

purposes worldwide, and the latter is used in the current USNRC dose compliance formalism. In essence, the calculation of effective dose for external exposure, as well as dose coefficients for internal exposure, are based on absorbed dose, weighting factors, and reference values for the human body and its organs and tissues. In general, effective and equivalent doses do not provide individual-specific doses, but rather doses for a *reference person*[3] (such as an MEI) under a given exposure situation.

Effective and equivalent doses, as well as *collective dose*[4] (see Table 3.1), were not designed for research purposes. Consequently, the use of these quantities should be avoided in epidemiologic studies because they mask many uncertainties that are embedded in their formalism, for example, uncertainties in radiation and tissue weighting. It is prudent to use the more fundamental dose quantity, D, for dose assessments used in epidemiologic studies. For such studies, absorbed dose is usually estimated for specific organs on an annual basis, expressed as rad/yr.

In the context of this discussion, the term *dose assessment* refers to the estimation of absorbed doses received by individuals as a result of exposure to ionizing radiation. Absorbed doses from direct radiation exposure[5] can be estimated using equipment that measures exposures in air in real time, for example by using radiation-sensitive materials such as thermoluminescent detectors (TLDs). Alternatively, doses can be estimated retrospectively by reconstructing an individual's past exposure to ionizing radiation. Absorbed dose from internal exposure (i.e., inhalation, ingestion, or absorption of radionuclides) can be estimated from measurements of radionuclide concentrations in air, soil, and food. Both exposure and dose can be estimated using models that relate releases of radioactivity to the environment (e.g., facility effluents) to exposure rates in air and to radionuclide concentrations in air, water, and food. Dose reconstruction is the primary concern of this chapter.

Reconstructing an individual's absorbed dose from releases of radioactive effluents from a nuclear plant or fuel-cycle facility requires knowledge of several factors, including:

[3]The most recent ICRP guidance (ICRP 101) uses the term "representative person" instead of "reference person" (ICRP, 2007a). However, the USNRC continues to use the older terminology.

[4]Collective dose is the sum of individual doses received by a specified population over a specified period of time. Collective dose is sometimes referred to as the *population dose*. ICRP (2007b) notes that collective dose is a useful concept for radiological protection but is not appropriate for use in epidemiologic studies or risk projections.

[5]As shown in Table 3.1, radiation exposures are expressed in terms of Roentgen (R). In the 1970s, it was common practice to convert exposure measurements in R to absorbed doses in air in rad using the conversion factor 1 R = 0.875 rad.

- Amount of radioactive material released from a facility, or *source term*;
- Transport of this radioactivity through the environment; and
- Uptake of (or exposure to) this radioactivity by the individual.

There are many pathways by which individuals can be exposed to radiation, be it from naturally occurring or manmade sources. As illustrated in Figure 3.1, individuals can be exposed to:

- *External radiation* from radionuclides that emit penetrating radiation (i.e., high-energy radiation such as gamma radiation that penetrates the human body). This radiation can be received directly

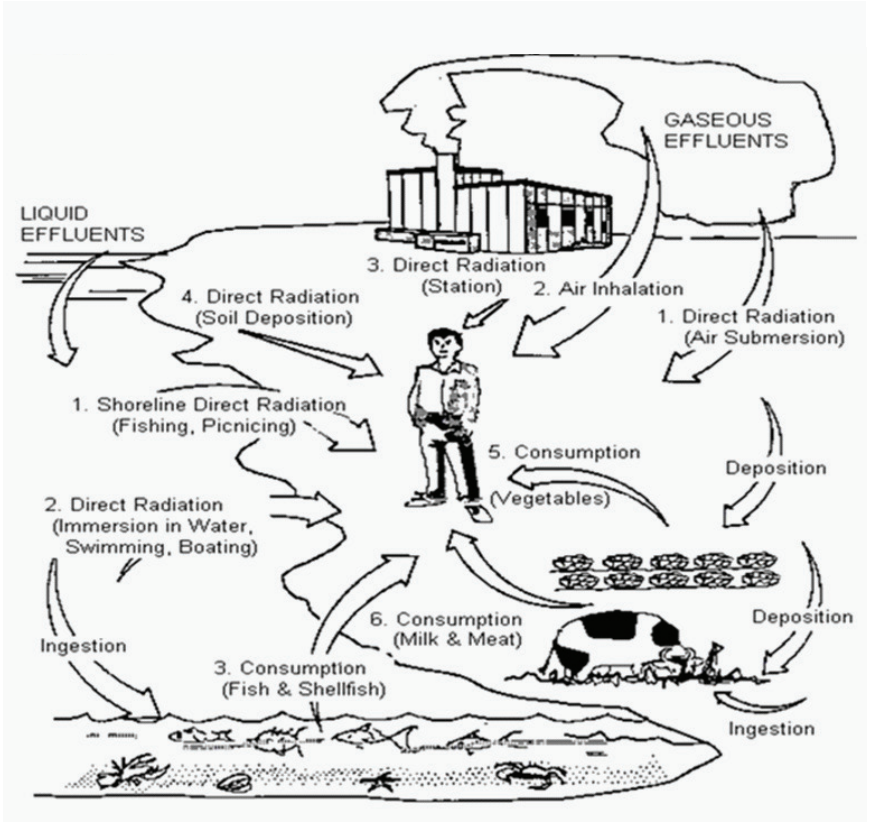

FIGURE 3.1 Pathways for exposure to radiation from effluent releases from nuclear plants and fuel-cycle facilities. SOURCE: Soldat et al. (1974).

from a facility, from radionuclides present in air, or from radionuclides deposited on the ground or in local water bodies. External exposure is usually the principal exposure route for radioactive effluent releases from nuclear plants.

- *Internal radiation* from radionuclides that are inhaled, ingested, or absorbed through intact or broken skin. Ingestion is usually the principal route of intake for radioactive effluent releases associated with nuclear fuel-cycle facilities.

Sophisticated computer models have been developed to reconstruct doses from exposures to external and internal radiation. To estimate external dose, transport calculations are carried out to determine atmospheric, water, and ground-surface concentrations of radionuclides at appropriate locations and times based on known or assumed meteorological and hydrological conditions. These quantities are then used to calculate the absorbed dose to individuals based on their locations relative to these radionuclide concentrations.

To estimate internal dose, the biokinetic models described in Appendix I are used to estimate the fate of radionuclides that are taken into the body by inhalation, ingestion, or absorption through skin. Radiation doses from internally deposited radionuclides are estimated by determining the spatial and temporal distribution of energy deposited in tissues and organs as a result of radioactive decay. Generally, this requires knowledge of the distribution of sources and targets in space and time. The source is the radionuclide of concern in a particular organ, tissue, or route of transit in the body. The target is the biological entity considered most relevant for determining dose and risk, which can range from molecules and cells for microdosimetry models to organs, tissues, or whole organisms. For radiation protection and epidemiologic studies, the level of averaging of radiation doses has consistently been at the tissue or organ level.

Retrospective dose assessments related to effluent releases of radioactive materials into the environment can be classified in two categories:

1. The dose assessments made for establishing *compliance* with standards or regulations. Usually, the calculated dose is much lower than the dose limit or standard. Under those conditions, the rationale is to show that the calculated dose is an overestimate. Upper bound values of parameters such as the time spent at the location of maximum exposure or the consumption rates of local foodstuffs are used to demonstrate that there is no doubt that the calculated doses are below the dose limits or standards and, therefore, that there is no need to evaluate the uncertainties in the calculated doses.

- The calculated doses are expressed in terms of equivalent dose (for specific organs or tissues) or effective dose (to take into account the irradiation of all organs of the body) in rem or in sievert because the dose limits or standards are expressed in those quantities.
- The equivalent dose per unit intake (for internal irradiation) or per unit exposure (for external irradiation) is the product of the absorbed dose per unit intake or exposure, which is a physical quantity, and a factor representing the biological effectiveness of the type of radiation that is considered. The value of this factor, called the "radiation-weighting factor" and denoted as w_R in ICRP Publications 60 and 103 (ICRP, 1991, 2007b), is based on experimental data for the relative biological effectiveness of various types of radiations at low doses, biophysical considerations, and expert judgment. The values for equivalent dose per unit intake and equivalent dose per unit exposure are set by the regulatory agency and, by convention, have no uncertainty.
- The dose limits or standards apply to equivalent doses due to 1 year of effluent releases. In the case of intakes of radionuclides with long biological times of residence in the body, such as strontium-90 or plutonium-239, the equivalent doses are still delivered many years after the year of intake. These "committed" equivalent doses are calculated for the entire period of time between the age at intake and age 70 and are not broken down on an annual basis.
- The dose limits or standards apply to the sum of the equivalent doses from all types of radiation. This means that the equivalent dose from high-LET (linear energy transfer) radiation, such as alpha particles, are not separated from the equivalent doses from low-LET radiations, such as photons and electrons.

2. Dose assessments made for *research* purposes, for example, in epidemiologic studies. For this application, the doses need to be calculated as realistically as possible and the uncertainties in dose estimates have to be evaluated. The dose estimates should have no bias (that is, they should not be overestimates or underestimates), implying that all parameter values should be chosen accordingly. This is particularly difficult when absorbed doses to specified individuals have to be calculated, but no interviews to those persons are feasible, thus precluding the knowledge of their lifestyle and dietary habits.

   - The calculated doses are expressed in terms of absorbed doses

to specific organs or tissues. The special name of the unit of absorbed dose is the gray, which is equal to 100 rad (see Table 3.1).

- The absorbed doses per unit intake (for internal irradiation) or per unit exposure (for external irradiation) are physical quantities. Their values may be adjusted to the individuals that are considered if there is justification for such adjustments. In fact, the absorbed doses per unit intake or exposure are often derived from the values recommended by the ICRP.
- The absorbed doses are calculated on an annual basis for each year of exposure, for example, from radioactive effluent releases. This means that in the case of intakes of radionuclides with long biological times of residence in the body, such as strontium-90 and plutonium-239, the absorbed doses must be calculated starting with the year of initial exposure and for each year afterward.
- The annual absorbed doses must be calculated separately for the low-LET and the high-LET radiation.

The focus of this report is on the second category of retrospective dose assessment.

3.2 REPORTED RADIATION DOSES AROUND NUCLEAR PLANTS

Reported radiation levels outside the property lines of nuclear plants are now (and have been in the past) low compared to natural background radiation exposure levels (see Section 3.5.1), which varies from plant to plant. Annual absorbed doses from naturally occurring terrestrial gamma sources and cosmic rays typically range from 50 to 100 millirad per year (mrad/yr) (free-in-air[6]). However, an individual living in close proximity to the property line (i.e., "fence line") of a nuclear plant might receive slightly elevated annual doses. Even during periods when nuclear plants released orders of magnitude more activity on average than currently (see Chapter 2), estimated external radiation doses to even the most exposed individual as a result of plant airborne effluent releases was likely only a fraction of the dose received from ambient natural background radiation.

TLD measurements at various locations at some nuclear plants suggest that the direct radiation dose from stored waste onsite and nitrogen-16 gamma rays (see Chapter 2) could have amounted to a significant fraction of the ambient natural background exposure level at plant fence lines. In fact, these exposures could have accounted for most of the dose to the MEI at these plants. However, the dose from direct radiation from stored waste and nitrogen-16 decreases rapidly with distance from the fence line

[6]That is, uncorrected for shielding by housing and indoor radiation sources.

and is generally an insignificant contributor to population exposures. For example, conservative estimates of doses from nitrogen-16 and stored waste at the Dresden plant (located in Illinois) were reported to result in an annual dose on the order of 8 mR/yr in 2009 to the MEI who was assumed to live in a home at the plant fence line and fish outdoors in an unshielded area for several hours per day (Exelon, 2010).

Most nuclear plant licensees use conservative assumptions in calculating annual doses to MEIs. For instance, some licensees assume that all effluent releases occur at ground level even though most airborne releases are made from tall stacks. This conservative assumption results in estimated maximum offsite dose levels that are much higher than would actually occur at any offsite location, particularly when averaged over a calendar quarter or year. Nevertheless, in recent years the estimated MEI doses are mostly less than 1 mrem/yr (Daugherty and Conatser, 2008), small fractions of ambient natural background radiation dose levels. However, doses in the 1970s and 1980s at some nuclear plants were higher, but even these doses were still much lower than doses from natural background radiation. Table 3.2 compares estimates of MEI doses for the early years of reactor operations at selected nuclear plants with estimates for more recent years.

The reported MEI doses shown in Table 3.2 are also generally consistent with independent measurements made at some of these sites. For example, the U.S. Department of Energy's Environmental Measurements Laboratory measured the integrated exposure from airborne radioactivity at a location 1.3 km from the Millstone-1 plant (a boiling-water reactor [BWR]) over a period of 500 days in 1973-1974 (Beck, 1975; Gogolak and Miller, 1974a,b). The absorbed dose in air was 3.5 mrad (0.035 mGy), in

TABLE 3.2 Comparison of Estimated Whole-Body Doses to the MEI from Effluent Releases and Direct Radiation from Selected Nuclear Plants

Plant (source)	Whole-Body Dose CED to MEI (mrem/year)	
Dresden (noble gases)	14 (1975)	0.9 (2009)
Dresden (liquid)	0.1 (1975)	1.0×10^{-4} (2009)
Dresden (direct)	—	8.4 (2009)
Oyster Creek (air)		0.0036 (2008)
Oyster Creek (water)	NA	NA
Millstone (air)	16 (1975)	0.33 (2010)
Millstone (liquid)	0.2 (1975)	0.0012 (2010)
Millstone (direct)	(incl. in air dose)	0.19 (2010)
North Anna (air)	1.3 (1984)	0.013 (2008)
North Anna (liquid)	4.0 (1984)	0.36 (2008)

NOTE: CED, committed effective dose; NA, not available.
SOURCE: Compiled from facility Radiological Environmental Monitoring Program reports.

reasonable agreement with what would be expected based on reported effluent releases over that time period, which ranged from 6 to 100 millicuries per second (mCi/s); the free-in-air natural terrestrial background radiation exposure at that site over the same period was 109 mrad. Comparisons of calculated and measured airborne exposures for other locations around the Millstone plant are shown in Table 3.3.

The Health and Safety Laboratory (now the Environmental Measurements Laboratory) also made similar measurements at a second BWR plant (Oyster Creek) over a period of several months in 1972. The maximum estimated offsite annual absorbed dose in air ranged from 10 to 15 mrad close-in with measurable levels out to 7 miles (~11 km) (Harold Beck, personal communication, unpublished).

The U.S. Environmental Protection Agency (USEPA) made similar measurements near several plant sites in the 1970s (Kahn et al., 1970, 1971, 1974). Measurements at the Prairie Island plant (a pressurized-water reactor [PWR] located in Minnesota) indicated a whole-body dose to the MEI of about 0.6 mrem/yr, excluding carbon-14. USEPA measurements at the Haddam Neck plant (a PWR located in Connecticut) in 1974 indicated a maximum annual dose of 0.9 mrem. Based on measurements at the Dresden plant in 1968, USEPA estimated a maximum annual dose of 14 ± 5 mrem. The total noble gas releases to the atmosphere during 1968 for Dresden were about 6 petabecquerels (PBq = 10^{15} Bq), comparable to the releases for 1975 when the facility estimated (conservatively) a dose to the MEI from noble gases of 14 mrem/yr.

As indicated in Chapter 2, the releases of carbon-14 are, as of 2010, included in the effluent release reports that are submitted by facility licensees. Table 3.4 provides the estimated carbon-14 releases and corresponding equivalent doses for a sample of reactors that supplied that information in

TABLE 3.3 Measured and Calculated Airborne Exposures at Seven Locations near the Millstone Plant

Location Distance (km) and Compass Direction	Length of Monitoring Period (August 1973 through March 1974) (hours)	Measured Absorbed Dose in Air (mrad)	Calculated Absorbed Dose in Air (mrad)
1.3 NNE	4727	0.312	0.342
2.6 ENE	4832	0.403	0.448
4.6 NNE	4254	0.080	0.100
4.6 E	4216	0.126	0.217
5.2 NE	4511	0.181	0.176
6.8 NNE	2919	0.046	0.055
8.0 ENE	4806	0.144	0.152

SOURCE: Gogolak and Miller (1974b).

TABLE 3.4 Carbon-14 Atmospheric Releases (Ci) and Equivalent Doses to MEI (mrem) Reported in Selected 2010 Annual Radioactive Effluents Releases Reports (ARERR)

Reactor Name	C-14 Release (Ci)	Fraction as CO_2	Estimation Method	Bone Equivalent Dose to MEI[a] (mrem)	Total-Body Equivalent Dose to MEI[b] (mrem)
BWR					
Brunswick	21	1	FSAR	2.4 (99%)	0.47
Cooper	11.6		5.1 Ci/GW_{th}-y	1.52 (99%)	
Dresden	20		5.1 Ci/GW_{th}-y	0.73	
Grand Gulf	9.5	0.95	FSAR	5.94 (94%)	
Nine Mile Point	9.16	0.95	5.1 Ci/GW_{th}-y	0.22	0.043
Pilgrim	8.54	0.99	Neutronic calculation	0.089 (80%)	0.018 (60%)
Susquehanna	24.5	1	EPRI (2010)	6.45 (96%)	1.29
PWR					
Beaver Valley	22	0.4	3.9 Ci/GW_{th}-y	5.63 (95%)	
Catawba	20.4	0.2	9.4 Ci/GW_e-y	4.78 (100%)	
Diablo Canyon	22.3	0.3	3.4-3.9 Ci/GW_{th}-y	0.37 (98%)	
H.B. Robinson	5.04		NUREG (1979)[c]	0.26 (76%)	0.052 (96%)
McGuire	20.2	0.2	9.4 Ci/GW_e-y	0.92 (98%)	0.44 (67%)
North Anna	17	0.3	EPRI (2010)	1.26 (98%)	
Palisades	7.69	0.3	Neutronic calculation	0.10	0.021
San Onofre	21.9			0.78 (90%)	
Sequoyah	19.2	0.2	3.9 Ci/GW_{th}-y	1.94 (96%)	
Waterford	19.2	0.2	FSAR	3.8 (98%)	
Wolf Creek	8.8	0.3	EPRI (2010)	1.3	0.26

NOTE: EPRI, Electric Power Research Institute; FSAR, Final Safety Analysis Report.

[a]The figure given in parentheses represents the percentage of the maximum organ equivalent dose from atmospheric effluent releases that is due to C-14.

[b]The figure given in parentheses represents the percentage of the total body equivalent dose from atmospheric effluent releases that is due to C-14.

[c]USNRC (1979).

their 2010 reports. Even though different assumptions were used by the facility operators to estimate both the releases and the equivalent doses, it is clear that carbon-14 is currently a major contributor to the equivalent dose to the MEI from atmospheric effluent releases. Not included in these estimates is the equivalent dose to the MEI from nitrogen-16 and stored wastes, which is, for some reactors, the most important contributor to the total equivalent dose to the MEI.

Pacific Northwest Laboratory (PNL)[7] has published estimates of col-

[7]PNL was renamed as the Pacific Northwest National Laboratory in 1995. This laboratory is located in Richland, Washington, adjacent to the Hanford Site.

lective doses[8] to populations living in the vicinity of operating nuclear plants in the United States resulting from airborne and waterborne effluent releases (NUREG/CR-2850[9]). Figure 3.2 shows PNL's collective dose estimates for persons living between 2 and 80 km from selected nuclear plants that have a range of effluent releases.[10] As can be seen from the figure, the total collective doses for some plants (e.g., Millstone and Dresden plants) were several orders of magnitude higher than for other plants (e.g., Fort Calhoun and Trojan plants). The estimated collective doses generally correlate with total noble gas effluent releases from the plants. Note that most of the collective dose for each site was usually delivered in only a few years (but not necessarily the same years) as shown in Figure 3.3. The 12 nuclear plants with the largest effluent releases accounted for over 75 percent of the total collective doses from all nuclear plants. Nuclear plants that have had high and low collective dose impacts over their operating histories are listed in Table 3.5.

Because the calculated collective doses are integrals over 2-80 km, they do not reflect the dose to MEIs or to populations living within 2 km of the plants. In addition, neither the doses resulting from atmospheric releases of carbon-14 nor the doses incurred prior to 1975 are included in the estimates shown in the table. Based on reported total effluent releases, the additional collective dose from operations prior to 1975 may have been comparable or greater at some plants, and the collective dose from atmospheric releases of carbon-14 may be a more significant contributor to the collective dose in more recent years as releases from other radionuclides have decreased dramatically (see Section 2.1 in Chapter 2).

For illustrative purposes, Table 3.6 lists the radionuclides that were reported by facility operators to make the highest contributions to collective doses from effluent releases (airborne and waterborne) in 1988 from 71 operating commercial nuclear plants. The relative contributions of each radionuclide to the total collective doses from all 71 plants are also shown in the table. It is clear that, at least in 1988 and probably since that time, tritium (hydrogen-3) has played an important role, both for airborne and waterborne releases. For airborne releases, isotopes of noble gases (krypton-88, xenon-133, and xenon-135) also contributed substantially to the collective dose, whereas iodine-131 was not a critical radionuclide for any of the

[8]These collective dose data are presented here because they are the only data the committee could find that provide some basis for comparing doses to populations living near different nuclear plants. As noted earlier in this chapter, collective dose is not an appropriate metric for epidemiologic studies.

[9]PNL issued a series of reports entitled Dose Commitments Due to Radioactive Releases from Nuclear Power Plant Sites that covered nuclear plant operations from 1977 to 1992. The first four reports in the series were issued as PNL-2439 (1977), NUREG/CR-1125/PNL-2940 (1979), NUREG/CR-1498/PNL-3324 (1980), and NUREG/CR-2201/PNL-4039 (1982). The remaining reports were issued from 1982 to 1996 as NUREG/CR-2850, vols. 1-14.

[10]As shown in Chapter 2, effluent releases among nuclear plants can vary substantially.

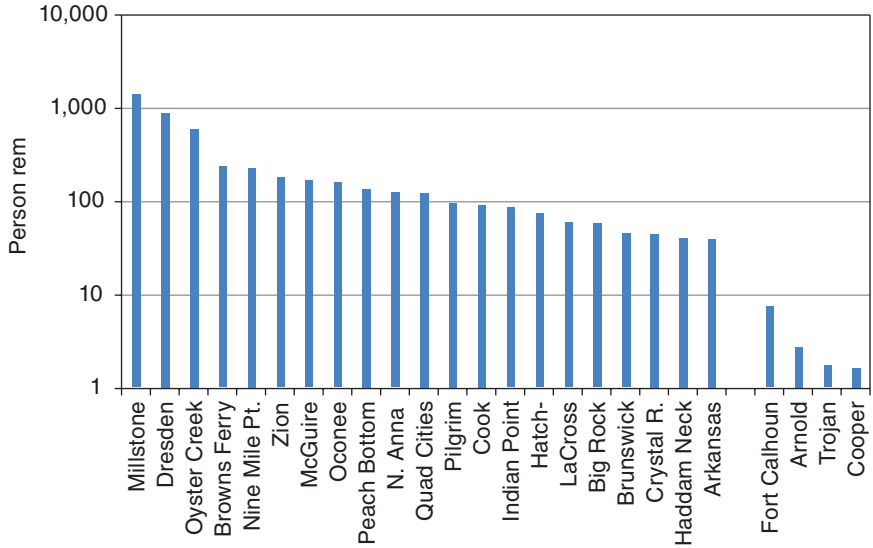

FIGURE 3.2 Collective doses to populations living between 2 and 80 km from selected nuclear plants. SOURCE: NUREG/CR-2850 (PNL-4221), vol. 14.

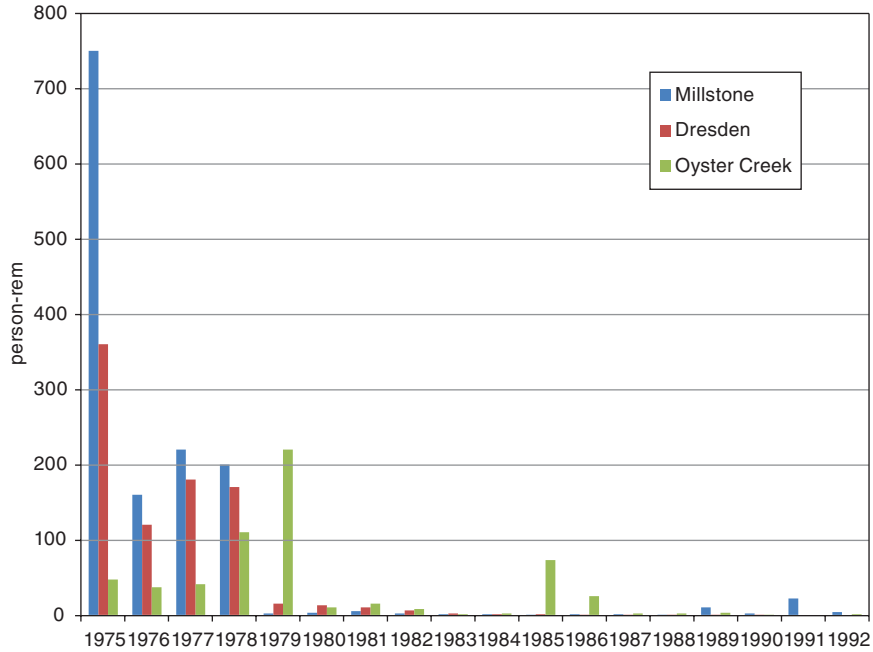

FIGURE 3.3 Collective doses to populations living between 2 and 80 km from the Millstone, Dresden, and Oyster Creek plants, 1975-1992. SOURCE: NUREG/CR-2850 (PNL-4221), vol. 1-14.

TABLE 3.5 Nuclear Plants with High and Low Collective Dose Impacts over their Operating Histories

Site (number and type of reactors)	State	Population within 5 Miles in 2000 (thousands)	Collective Dose[a] 1975-1992 (person-rem)	Maximum Annual Collective Dose (person-rem)	Maximum Dose Year	Total Noble Gas Releases 1975-1992 (GBq)
Plants Having High Dose Impacts						
Millstone (1 BWR, 2 PWR)	CT	53.3	1384	750	1975	1.80×10^8
Dresden (BWR)	IL	22.9	879	360	1975	1.20×10^8
Oyster Creek (BWR)	NJ	44.2	594	220	1979	9.40×10^7
Browns Ferry (BWR)	AL	6.1	232	106	1984	9.00×10^7
Nine Mile Point. 2(BWR)	NY	6.7	227	140	1979	5.50×10^7
Zion (PWR)	IL	—	177	34	1984	1.70×10^6
McGuire (2PWR)	NC	51.2	165	20	1984	1.10×10^6
Oconee (3PWR)	SC	15.6	157	38	1977	1.50×10^7
Peach Bottom (BWRs)	PA	11.3	134	30	1979	3.20×10^7
North Anna (2PWR)	VA	6.9	123	44	1984	3.30×10^6
Quad Cities (2 BWR)	IL	6.3	121	42	1980	1.20×10^7
Pilgrim (BWR)	MA	23.1	95	52	1977	3.10×10^7
Cook (2 PWR)	MI	17.0	90	40	1978	
Indian Point (1BWR, 2PWR)	NY	88.2	85	13	1977	5.00×10^6
Hatch (2 BWR)	GA	2.1	73	35	1977	6.90×10^6
LaCrosse (BWR)	WI	—	59	12	1976	9.00×10^6
Big Rock (BWR)	MI	—	57	10	1980	1.70×10^7
Brunswick (2 BWR)	NC	13.4	45	14	1982	8.20×10^7
Crystal River (3 PWR)	FL	6.1	44	19	1981	7.20×10^6
Haddam Neck (PWR)	CT	—	40	7.5	1980	2.30×10^6
Arkansas (2 PWR)	AK	14.2	39	4.7	1985	5.70×10^6
Plants Having Low Dose Impacts						
Fort Calhoun (PWR)	NE	9.3	7.3	1.9	1984	6.0×10^5
Duane Arnold (BWR)	IA	12.2	2.7	0.87	1978	1.0×10^6
Trojan (PWR)	OR	—	1.7	0.25	1991	4.1×10^5
Cooper (BWR)	NE	0.9	1.6	0.39	1976	4.6×10^6

NOTE: BWR, boiling-water reactor; PWR, pressurized-water reactor.

[a]For individuals living between 2 and 80 km of the plant boundary.

SOURCE: Population information from Table 1.3 in Chapter 1; other information from NUREG/CR-2850 (PNL-4221), vol. 14.

TABLE 3.6 Radionuclides with the Highest Contribution to Collective Dose from Effluent Releases (Airborne and Waterborne) in 1988 from the 71 Operating Commercial Nuclear Plants

	Airborne Releases		Waterborne Releases	
Radionuclide	Number of Nuclear Power Plants with the Highest Contribution to Collective Dose	Relative Contribution to the Total Collective Dose (Percent)	Number of Nuclear Power Plants with the Highest Contribution to Collective Dose	Relative Contribution to the Total Collective Dose (Percent)
Tritium	39	28	18	35
Carbon-14[a]	1	0.4	0	0.1
Manganese-54	0	<0.1	1	2.8
Iron-55	0	0	4	0.5
Iron-59	0	0	1	0.2
Cobalt-58	0	<0.1	1	1.1
Cobalt-60	2	1.0	3	2.3
Zinc-65m	0	<0.1	4	1.6
Krypton-88	8	7	0	0
Strontium-90	1	<0.1	1	1.4
Iodine-131	0	<0.1	1	0
Xenon-133	17	31	0	0
Xenon-135	3	13	0	0
Cesium-134	0	<0.1	11	28
Cesium-137	0	0.3	18	24
No release	0		8	
Total number of plants	71		71	
Total collective dose (person-rem)		9.6		65

[a]The collective dose from releases of carbon-14 was calculated for only two power plants.
SOURCE: NUREG/CR-2850, vol.10.

plants. With respect to waterborne releases, cesium-134 and cesium-137 were the two most important radionuclides, in addition to tritium.

It is worth noting that the collective dose from carbon-14 was apparently calculated for only two nuclear plants (Ginna and Yankee Rowe) and was found to be the highest contributor to collective dose from airborne releases for one of those (Yankee Rowe). Had the collective doses from carbon-14 releases been estimated and reported for the other nuclear plants, it is likely that it would have been found among the main contributors to the collective dose from airborne effluent releases, assuming that the results of Table 3.4 for the dose to the MEI can be translated in terms of collective dose.

Figure 3.4 shows the reported annual collective doses from airborne and waterborne radioactive effluent releases from all operating nuclear plants from 1975 to 1992. In the early years of operations when doses were highest, most of the collective dose was from exposure to airborne effluents. In contrast, most of the collective dose in recent years is from waterborne releases, but these collective doses remained fairly constant over time. The contribution to the collective dose from waterborne versus airborne releases differed at different sites depending on such factors as the presence of nearby recreational facilities (e.g., rivers and lakes).

External radiation exposures around nuclear plants would be expected to vary not only with distance from the plant site, but also with direction,

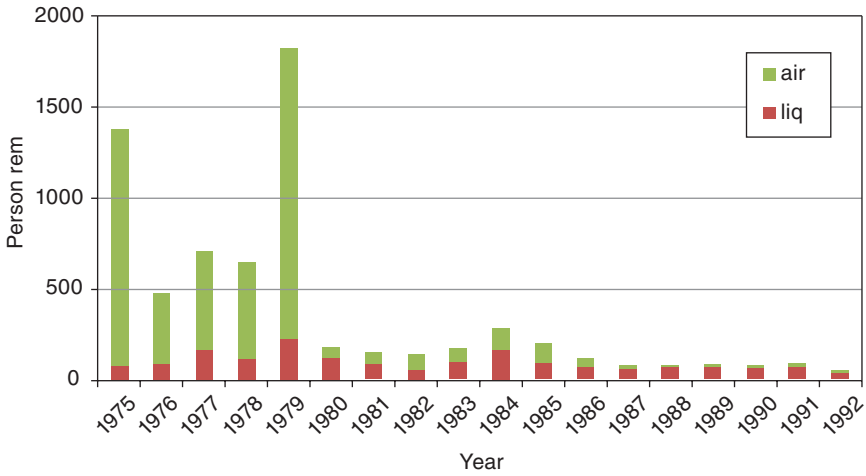

FIGURE 3.4 Collective doses from air and liquid effluents at all operating nuclear plants from 1975 to 1992. SOURCE: NUREG/CR-2850 (PNL-4221), vol. 14.

local topography, and stack height, particularly for sites where wind directions are not distributed isotropically. Consequently, spatial and directional variations in dose could be significant at some plant sites and could also vary with season. If so, the use of annual effluent releases and annual average meteorology to estimate doses would not reflect these spatial variations. This would be particularly true for plants that do not release effluents randomly in time such as PWRs, which release effluents in batches.

To illustrate, Figures 3.5 and 3.6 show the wind rose and calculated 1975 external doses around the Dresden plant. Both the wind rose and dose distributions display asymmetry. Residents living north of the plant received higher doses relative to residents living in other compass directions at a given distance from the plant site. It is likely that the asymmetry in calculated dose at some sites was even more pronounced.

With regard to waterborne releases, the degree of asymmetry is more difficult to predict. The degree of asymmetry depends to a large extent on the distribution of contaminated drinking water and contaminated foodstuffs (fish and invertebrates).

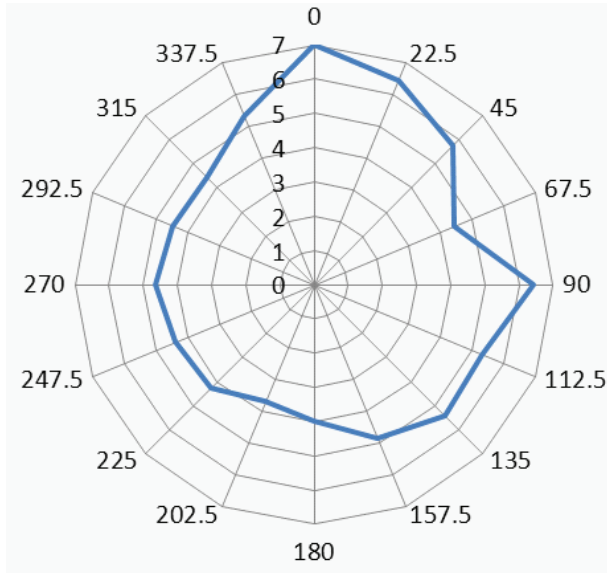

FIGURE 3.5 Annual wind rose for the Dresden plant for all stability classes and speeds combined at the height of the plant stack. The concentric lines indicate the percent time (from 0 to 7 percent) the wind was blowing. The radial lines show the compass direction that the wind was blowing. SOURCE: Commonwealth Edison (1976).

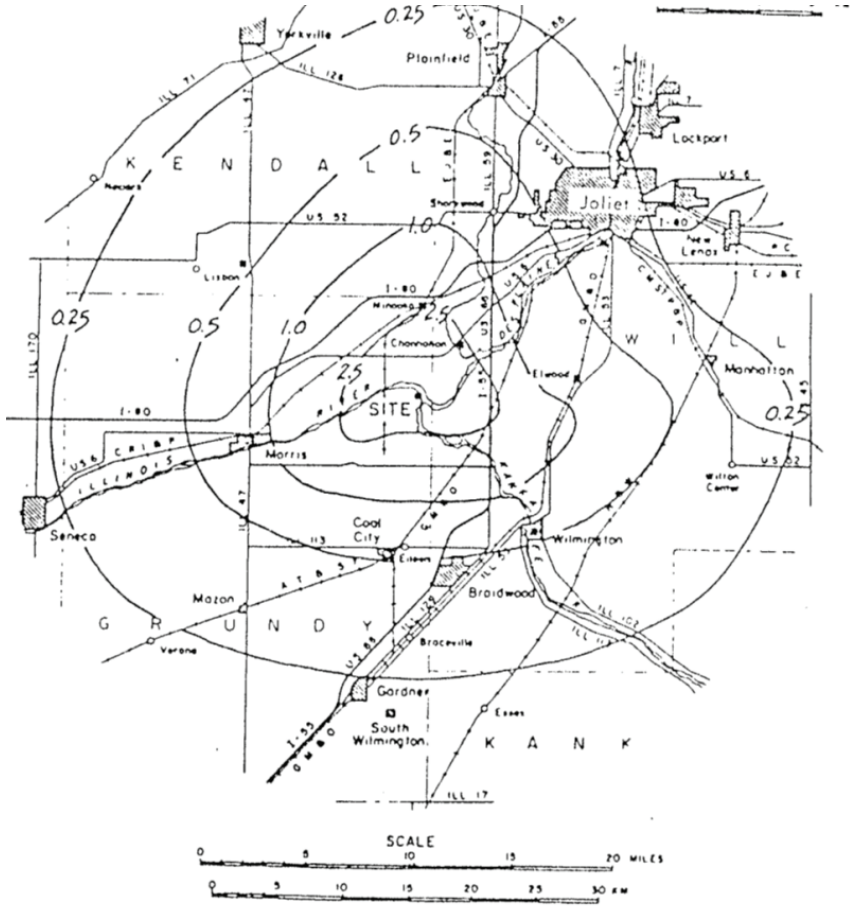

FIGURE 3.6 Calculated annual dose contours (rem) for 1975 at the Dresden plant from airborne effluent releases for comparison with the average wind rose (Figure 3.4). SOURCE: Commonwealth Edison (1976).

3.3 REPORTED DOSE ESTIMATES AROUND NUCLEAR FUEL-CYCLE FACILITIES

As is the case for nuclear plants (Section 3.2), doses to MEIs living near fuel-cycle facilities in recent years are very low. Some examples of MEI dose estimates for fuel-cycle facilities are shown below:

- Milling (Crow Butte): 0.68 mrem/yr in 2010 (Crow Butte Resources, 2010).
- Conversion (Honeywell): 0.57 mrem/yr in 2005 (Honeywell, 2006).

- Enrichment (Paducah): 0.94 mrem/yr in 2009 (Portsmouth, 2009).
- Fuel Fabrication (Nuclear Fuel Services): 0.002 mrem/yr in 2009 (NFS, 2009).

However, doses in early years of operation might have been significantly greater. The doses for various types of facilities are discussed below.

3.3.1 Mining and Milling Facilities

As noted in Chapter 1, the committee did not consider mining and milling facilities in this study because of their small surrounding populations (see Table 1.3 in Chapter 1). Because of the small populations, the collective doses to populations living within 80 km of these facilities have probably been small relative to collective doses to populations near nuclear plants. Doses in a recent year (2010) at a typical in situ uranium recovery facility (Crow Butte) to the MEI are estimated to be about 0.7 mrem/yr (0.5 mrem/yr from radon, the remainder from uranium). Doses in earlier years were much greater as shown in Table 3.7. External (direct radiation)

TABLE 3.7 Reported 50-Year Committed Doses to the MEI for 1979 or 1980 Effluent Releases from In Situ Uranium Recovery Facilities in the United States

Facility	Location	Whole-Body (mrem)	Bone (mrem)	Lung (mrem)
Atlas Minerals	Moab, UT	2.4	34.6	74.8
Bear Creek Uranium Co.	Converse Co., WY	0.486	6.14	0.782
Exxon Minerals Highland Mill	Converse Co., WY	0.847	12.2	13.9
Federal-American Partners	Gas Hills, WY	0.649	17.4	35.9
Energy Fuels Nuclear White Mesa	Blanding, UT	1.40	15.0	2.24
Minerals Exploration Co.	Sweetwater Co., WY	0.0081	0.0831	0.038
Pathfinder Mines	Gas Hills, WY	0.599	11.4	15.7
Pathfinder Mines	Shirley Basin, WY	1.61	18.0	6.56
Petrotomics Company	Shirley Basin, WY	0.696	9.75	9.58
Plateau Resources	Shootering Canyon, UT	0.135	3.60	6.63
Rio Algom Humeca Mill	La Sal, UT	0.528	11.0	23.5
Union Carbide Corp.	Gas Hills, WY	0.97	12.5	1.81
United Nuclear Corp. Morton Ranch	Converse Co., WY	0.08	0.34	0.28
Western Nuclear Inc., Split Rock	Jeffrey City, WY	2.0	24.2	11.5

NOTE: Committed dose is the total dose that would be received by an individual during a specified period (usually the 50-year period) following the intake of a radioactive material. The doses do not include contributions from radon because the dose criteria in 40 CFR 190 (Environmental Radiation Protection Standards for Nuclear Power Operations) do not apply to dose from radon and its short-lived decay products.
SOURCE: USNRC (1981).

whole-body doses result primarily from exposure to mill tailings. Bone and lung doses result from inhalation of airborne effluents. The United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR, 1982) estimated that organ doses from mining and milling operations were mainly from inhalation or airborne emissions of radon decay products, with additional contributions from uranium and thorium isotopes, radium-226, and lead-210. The highest doses were to the lung and bone.

3.3.2 Uranium Conversion Facilities

The only uranium conversion facility in the United States is the Honeywell plant, which is located at Metropolis, Illinois. The plant licensee estimated that the dose to the MEI in 2005 was 0.57 mrem (Honeywell, 2006). The MEI was located at the nearest residence, 564 meters (1,850 feet) north-northeast of the Metropolis facility. The MEI does not have a home garden; however, to be conservative, the ingestion pathway was included in the dose assessment. (The methodology, data, and assumptions used in the dose assessments were provided in Honeywell [2006]). Honeywell also estimated the annual collective dose to the population of about 517,000 people surrounding the facility as 0.0381 person-Sv (3.81 person-rem) per year.

The Paducah Gaseous Diffusion Plant is located near the Metropolis facility. Based on data reported by USEC, Inc., the radiation dose (TEDE[11]) to the MEI from atmospheric emissions from the Paducah Gaseous Diffusion Plant was estimated to be 3.54×10^{-4} mSv (0.0354 mrem) per year in 2004 (Honeywell, 2006). Therefore, the Paducah Gaseous Diffusion Plant would not contribute appreciably to the radiation dose for the Metropolis facility's MEI.

Although the radiological impacts from current normal operations are very small, doses in early years of operation might have been greater. The committee did not investigate data on estimated doses from conversion for early years of operation.

3.3.3 Uranium Enrichment Facilities

The maximum dose that a member of the public was estimated to have received from reported effluent releases from the Portsmouth enrichment facility in 2009 was 0.94 mrem: 0.024 mrem from airborne radionuclides, 0.037 mrem from radionuclides released to the Scioto River, 0.72 mrem from direct radiation from the depleted uranium cylinder storage yards, and

[11]Total effective dose equivalent. This is the sum of the effective dose equivalents from internal and external exposures.

0.16 mrem from exposure to radionuclides detected at offsite monitoring locations (DOE, 2011). This dose calculation used a worst-case approach; that is, the calculation assumes that the same individual is exposed to the most extreme conditions from each pathway. The maximum potential doses in 2004 and 2005 were 1.86 mrem (DOE, 2006) and 1.67 mrem (USEC, 2006), respectively. The 2005 estimate broke down as follows: 0.012 mrem from airborne radionuclides, 0.025 mrem from radionuclides released to the Scioto River, 1.1 mrem from direct radiation, and 0.53 mrem from exposure to radionuclides detected at offsite monitoring locations. The relatively high external (direct) exposure is primarily from tanks of depleted uranium.

The maximum effective dose equivalent to the MEI for the Paducah plant was reported as 0.0433 mrem/yr in 2002 (USEC, 2008). Based on estimated 2002 census data, the total committed effective dose equivalent (CEDE) to the 50-mile population (approximately 531,000 persons, including 36,500 within 10 miles (~16 km) of the plant and approximately 104,000 within 20 miles [~32 km]) was <0.2 person-rem.

The committee did not attempt to find data for very early years of operation at these facilities.

3.3.4 Fuel Fabrication Facilities

The committee reviewed reported dose estimates for recent years for two currently operating fuel fabrication facilities: Nuclear Fuel Services, Inc. (Tennessee) and Westinghouse Electric Company, LLC Columbia Fuel Fabrication Facility (South Carolina).

Doses related to Nuclear Fuel Services (NFS) Erwin plant operations are dominated by airborne effluent releases. In 2009 (NFS, 2009), the estimated dose to an MEI located 300 m north-northeast of the site was 0.0018 mrem; the maximum organ doses were 0.0068 mrem (spleen) and 0.0022 mrem (red bone marrow) (doses are expressed as CEDE). Doses were calculated using reported stack effluents and a 5-year average wind rose (Class D). For 2004-2007, doses (again expressed as CEDE) to the MEI averaged only 0.007 mrem/yr (NFS 2009 license renewal[12]).

Airborne effluents from NFS have been decreasing since 1989. In 1999, the maximum CEDE was 2.6 mrem/yr (2.4 from air, 0.5 liquid) and the maximum lung dose was 21 mrem (NFS, 1999). External (direct) exposure was generally negligible (inhalation dose $\times 10^{-6}$). Internal dose was mainly from technetium-99, thorium, and uranium. There were no reported drinking water impacts for that year (NFS, 1999).

The Westinghouse fuel production facility similarly reported that the

[12]http://pbadupws.nrc.gov/docs/ML0930/ML093010370.html.

critical dose pathway is inhalation (lung dose with an annual TEDE dose in 2002 of <0.4 mrem to an exposed individual living at the site boundary). The dose from liquid effluents was estimated as <0.0003 mrem/yr (Westinghouse, 2002).

3.4 APPROACHES FOR ESTIMATING DOSES FOR AN EPIDEMIOLOGIC STUDY

As discussed in Section 3.1, the use of an MEI dose is not appropriate for epidemiologic studies: MEI doses are calculated by facility licensees to demonstrate compliance with applicable regulations. It provides an estimate of dose at a single point and does not provide any information on the variation of dose as a function of distance and direction from a facility. Further, MEI doses are larger than would likely be received by any actual individual living around a nuclear facility as a result of radioactive effluent releases. More realistic estimates of individual dose as a function of distance and direction from the facility are needed to support an epidemiologic study.

Also as noted in Section 3.1, computer models have been developed to estimate absorbed doses in persons exposed to radiation through environmental pathways (see NCRP, 2009b). Such models could be used to estimate doses to individuals living near nuclear facilities to support an epidemiologic study. An existing computer model could be modified for this purpose, or a new model could be developed. Regardless of the approach used, it is essential that the computer model reflect modern practices for dose reconstruction.

Guidance provided in USNRC Regulatory Guides 1.109, 1.111, and 1.113 (USNRC, 1977a,b,c) is used by nuclear plant licensees to estimate equivalent doses to the MEI. This guidance can also be used to estimate equivalent doses to representative individuals in the vicinity of the nuclear plant. For example, a computer program was developed by PNL to estimate doses received via airborne and waterborne pathways by representative individuals living in the vicinity of operating nuclear plants from 1975 through 1988 (Baker, 1996). It is possible that this program (or similar more recent programs developed by the USNRC or other organizations) could be modified to obtain dose estimates to support the epidemiologic studies that are recommended in this report (see Chapter 4). The remainder of this section describes the modifications that would need to be made to make the PNL computer model usable for developing dose estimates to support an epidemiologic study.

It is not the intention of the committee to endorse the PNL model or to recommend its use. It is only for practical reasons that the PNL model and, by extension, the USNRC Regulatory Guides 1.109, 1.111, and 1.113 are used as a basis for the presentation of recommended modifications and

improvements. Namely, the PNL model was developed to use the effluent data that are reported to the USNRC by facility licensees. As noted in Chapter 2, these data represent summed quantities typically over periods of weeks to months.

The PNL model was used to estimate equivalent doses for representative individuals of population groups living in 160 segments around nuclear plants defined by 22.5-degree radial slices of the 16 compass points (i.e., N, NNE, NE, ENE, E, ESE, SE, SSE, S, SSW, SW, WSW, W, WNW, NW, NNW) and 10 concentric intervals from 2 to 80 km from the facility boundary (Table 3.8). The population was divided into four age groups: infants (<1 year), children (1-10 years), teenagers (11-17 years), and adults (> 17 years). Doses to selected organs (Table 3.9) were calculated for both airborne and waterborne pathways (Table 3.10) for 83 radionuclides (Table 3.11). The dose to a representative individual of a given age is assumed to be the same in any location within a given segment, except when the dose to the MEI was calculated.

TABLE 3.8 Concentric Intervals and Midpoints Used for Dose Calculations in the PNL Model

Distance Interval from the Plant Boundary (km)	Midpoint of Interval (km)
2-3	2.5
3-4	3.5
4-6	5
6-9	7.5
9-14	11.5
14-20	17
20-30	25
30-40	35
40-60	50
60-80	70

SOURCE: Baker (1996).

TABLE 3.9 Doses to Organs Estimated by the PNL Model

Airborne Pathways	Waterborne Pathways
Total body	Total body
Thyroid	Thyroid
Bone	Bone
Gastrointestinal tract	Gastrointestinal tract
Liver	Liver
Lung	

SOURCE: Baker (1996).

TABLE 3.10 Environmental Pathways Considered in the PNL Model

Airborne Pathways	Waterborne Pathways
Air submersion	Ingestion of drinking water
Ground irradiation	Ingestion of fish and invertebrates
Inhalation	Shoreline irradiation (for MEI)
Ingestion of foodstuffs and animal products	Ingestion of irrigated food products (for MEI)
Gamma and beta air dose (for MEI at site boundary)	

SOURCE: Baker (1996).

TABLE 3.11 Radionuclides Considered in the PNL Model

Noble gases: ^{41}Ar, ^{83m}Kr, ^{85m}Kr, ^{85}Kr, ^{87}Kr, ^{88}Kr,[a] ^{89}Kr, ^{131m}Xe, ^{133m}Xe, ^{133}Xe, ^{135m}Xe, ^{135}Xe, ^{137}Xe, ^{138}Xe[a]

Radioiodines and precursors: ^{132}Te,[a] ^{133m}Te,[a] ^{131}I,[a] ^{132}I, ^{133}I,[a] ^{134}I, ^{135}I[a]

Other radionuclides: ^{3}H, ^{10}Be, ^{14}C, ^{13}N, ^{18}F, ^{22}Na, ^{46}Sc, ^{51}Cr, ^{54}Mn, ^{56}Mn, ^{55}Fe, ^{59}Fe, ^{57}Co, ^{58}Co, ^{60}Co, ^{57}Ni, ^{63}Ni, ^{65}Ni, ^{64}Cu, ^{65}Zn, ^{69m}Zn,[a] ^{76}As, ^{82}Br, ^{88}Rb, ^{89}Rb,[a] ^{89}Sr, ^{90}Sr,[a] ^{91}Sr, ^{92}Sr, ^{90}Y, ^{91m}Y, ^{95}Zr,[a] ^{97}Zr,[a] ^{95}Nb, ^{97}Nb, ^{99}Mo,[a] ^{99m}Tc, ^{103}Ru,[a] ^{106}Ru,[a] ^{110m}Ag,[a] ^{115m}Cd, ^{115}Cd, ^{125}Sn,[a] ^{124}Sb, ^{125}Sb,[a] ^{134}Cs, ^{136}Cs, ^{137}Cs,[a] ^{138}Cs, ^{139}Cs,[a] ^{139}Ba, ^{140}Ba,[a] ^{140}La, ^{141}La, ^{141}Ce, ^{144}Ce,[a] ^{152}Eu, ^{154}Eu, ^{187}W, ^{232}Th,[a] ^{239}Np

[a]The dose calculation includes the contributions from the decay products.
SOURCE: Baker (1996).

The PNL model was developed about 30 years ago, and some of the approaches used to obtain dose estimates are outdated. Consequently, the model would need to be modified to make it useable in a modern epidemiologic study. Needed modifications are discussed below, using as a framework a general form of the calculation of the radiation dose, D, resulting from releases of radioactive materials into the environment (Till and Grogan, 2008):

$$D = (A \times T \times E \times K)_{u,v} \quad (2)$$

in which

A = radionuclide activity released into the environment;

T = environmental transport, resulting in estimates of radionuclide concentrations in air, soil, water, and foodstuffs;

E = exposure factors, resulting in estimates of doses in air and of radionuclide intakes of radionuclide-contaminated air, water, and foodstuffs;

K = conversion to organ or tissue dose;

u,v = uncertainty and validation, which should be taken into account throughout the dose estimation process.

The following subsections describe the needed modifications for each of these factors.

3.4.1 Dose (D)

Several factors are required to estimate the dose term (D) in the equation. These include the radionuclides that are released from the facility, their environmental pathways, the locations and ages of representative individuals who are exposed to these radionuclides, the specific organs exposed, and the type of dose that is estimated. These factors are described briefly in the following subsections.

3.4.1.1 Radionuclides

All of the radionuclides present in detectable quantities in the effluents released from nuclear plants appear to have been considered in the PNL model (Table 3.10). However, radionuclides released from fuel-cycle facilities, namely uranium-238 and its decay products, will also need to be included in the model if these facilities are considered in the epidemiologic study.

3.4.1.2 Environmental Pathways

The environmental pathways used in the PNL model (see Table 3.9) are adequate to estimate doses for an epidemiologic study. However, the underlying computer code would need to be modified to include doses received from direct radiation from onsite sources, from external irradiation from the shoreline of a contaminated water body, and from internal irradiation due to the consumption of irrigated food products where these doses comprise greater than 1 percent of the total dose.

3.4.1.3 Location of Representative Individuals

As noted previously, the PNL model estimates doses to representative individuals in each of 160 segments surrounding a nuclear plant. However, the spatial area of interest for an epidemiologic study (see Chapter 4) is the census tract, not the PNL segments. The PNL model could be modified to estimate doses in census tracts around nuclear facilities. For this purpose, a simplifying assumption could be made that the dose calculated at the centroid[13] of the census tract is representative of the dose received at any

[13]The centroid location could be determined geographically or based on population distribution.

location in that census tract. Alternative approaches employing modern Geographic Information System (GIS) methods could also be employed to generate predicted doses on a GIS grid.

3.1.4.4 Ages of Representative Individuals

As noted previously, four age groups were considered in the PNL model but no gender distinctions were made. With respect to the estimation of doses from external irradiation, data in ICRP Publication 74 (ICRP, 1997) indicate that differences of about 30 percent between external doses to infants and adults are plausible; such differences would need to be taken into account in an epidemiologic study. With respect to the estimation of doses from internal irradiation, age and gender groups considered by the ICRP (1990) could be used: newborn (<1 year), infants (1-2 years), young children (3-7 years), older children (8-12 years), teenagers (13-17 years), adult males, and adult females.

3.1.4.5 Organs

Because one of the committee's recommended epidemiologic study designs involves assessment of risks for all cancers (see Chapter 4), doses from internal radiation to all organs and tissues considered by the ICRP to be radiosensitive (i.e., adrenals, bladder, bone marrow, bone surface, brain, breast, esophagus, stomach, small intestine, colon, extrathoratic tissue, gall bladder, gonads, heart, kidneys, liver, lung, lymphatic nodes, muscle, oral mucosa, pancreas, prostate [males only], salivary glands, spleen, skin, thymus, thyroid, uterus/cervix [females only]) will need to be considered (ICRP, 2007b). With regard to the doses from external irradiation, the simplifying assumption could be made that all soft tissues of the body receive the same dose and that there is no age or gender dependency. However, special consideration would be warranted for red bone marrow or bone surfaces in case they are tissues of interest in an epidemiologic study.

3.1.4.6 Type of Dose

The PNL model estimates the committed equivalent dose per year of effluent release for representative individuals resulting from internal radiation. The dose of interest in epidemiologic studies is the annual absorbed dose by year of effluent release. This difference may pose a problem for long-lived alpha emitters that are released from fuel-cycle facilities because (1) the committed equivalent dose will have to be broken down into its yearly components, and (2) the dose from alpha particles will have to be separated from the dose from photons and electrons. Data files published

by the USEPA (USEPA, 2002) may be used to satisfy both purposes. In case of external irradiation involving gamma radiation, such problems do not exist. This problem can be avoided by modifying the model to estimate annual absorbed dose.

Doses for representative individuals are calculated using the simplifying assumptions that those individuals resided at the same place during the entire period of exposure. However, if a case-control study is carried out, doses will need to be calculated for specific individuals. It would then be important to gather information on their residential histories, at the census-tract level, of those individuals during the entire period of exposure.

3.4.2 Activities Released (A)

As noted in Chapter 2, nuclear plants and fuel-cycle facilities release different types of radionuclides and have different effluent release reporting requirements.

3.4.2.1 Nuclear Plants

As indicated in Chapter 2, the effluent releases of specific radionuclides are available on a monthly, quarterly, semiannual, or annual basis for any year since 1975. It is important to note for almost all reactors the released activities of carbon-14 are not included in the reports. Prior to 1975, when the released activities were much higher than in recent years, the information on released activities is more limited: it usually consists of total activities grouped into categories; the categories for airborne effluent releases are (1) noble gases and (2) iodine-131 and particulates with half-lives longer than 8 days. For waterborne effluent releases, the categories are (1) tritium (hydrogen-3) and (2) mixed fission and activation products. Information on the activities released for specific radionuclides appears to be only available for some reactors and some years of operation (see, for example, Logsdon and Robinson, 1971; BNL, 1979).

For the purposes of an epidemiologic study, it is essential to use reliable data for specific radionuclides. For most reactors and years before 2010, the airborne releases of carbon-14 in the form of CO_2 will have to be estimated, for example on the basis of the thermal power generated or according to methods developed by EPRI (2010) or the USNRC (1979). Because there is no easy way to trap CO_2, it is presumed that practically all of the carbon-14 activity that is produced as CO_2 is released into the atmosphere. For years prior to 1975, simplifying assumptions might have to be made to reconstruct the released activities of some radionuclides[14];

[14]For example, the activities of individual radionuclides might have to be estimated using radionuclide distributions and group activities.

the uncertainties attached to the estimates of reconstructed activities for specific radionuclides, which may be very large, will have to be evaluated.

Another consideration is the time period over which the activities are summed (i.e., monthly, quarterly, semiannually, annually, or by batch) for the purposes of dose estimation. The decision over which time period to select may vary from site to site and from year to year according to the availability of other data that are needed for dose estimation, such as meteorological data and river flow data. In any case, the doses of interest for the epidemiologic study are annual doses. Consequently, any doses estimated for any fraction of the year will have to be summed over the entire year.

It is worth noting that the doses from direct radiation due to nitrogen-16 contained in BWRs and radioactive materials stored onsite do not depend on the activities released, but rather on the shielding characteristics of the reactor and its procedures for storing waste materials. The corresponding doses will have to be based on site-specific measurements or on site-specific calculations.

3.4.2.2 Fuel-Cycle Facilities

At this time, the information that will be available for the entire period of operation of any fuel-cycle facility is unclear (see Chapter 2), as it seems that at least part of this information will have to be requested from the plant licensees. Annual releases of specific radionuclides would be needed to calculate doses using the PNL model.

3.4.3 Environmental Transport (T)

Environmental transport parameters link the radionuclide activities released to the concentrations of those radionuclides in environmental media (air, soil, water, sediments, and food products) at any time and location in the vicinity of a nuclear facility. A list of the main environmental transport parameters is provided in Table 3.12. Transport of airborne and waterborne releases are described in the following subsections.

3.4.3.1 Airborne Effluent Releases

The most important environmental parameter for airborne effluent releases is the atmospheric dilution factor, which is the quotient of the radionuclide concentration at the location of interest (expressed, for example, in Ci m^{-3}) and the release rate of that radionuclide (expressed, for example, in Ci s^{-1}). In the PNL model, atmospheric dilution factors are calculated as averages over 160 segments and also for specific locations near the plant site (site boundary, closest residence, closest garden, and closest pasture).

TABLE 3.12 Main Parameters Used to Estimate Dose per Unit Activity Released.

Pathway of Exposure	Environmental Transport (T)	Exposure Factors (E)	Conversion to Organ or Tissue Dose (K)
Airborne Effluent Releases			
Air submersion	Atmospheric dilution factor	Indoor shielding and occupancy factors	Dose coefficient (FGR 12)
Ground irradiation	Atmospheric dilution factor; dry deposition velocity	Indoor shielding and occupancy factors	Dose coefficient (FGR 12)
Direct radiation	Transport model	Indoor shielding and occupancy factors	Dose coefficient (ICRP 74)
Inhalation	Atmospheric dilution factor	Indoor shielding and occupancy factors; breathing rates	Dose coefficients (ICRP 71)
Ingestion	Atmospheric dilution factor; dry deposition velocity; transfer coefficients	Consumption rates; culinary factors; holdup times	Dose coefficients (ICRP 56, 67, 69)
Waterborne Effluent Releases			
Ingestion (water)	Aquatic dilution factor	Consumption rate; water treatment	Dose coefficients (ICRP 56, 67, 69)
Ingestion (fish and invertebrates)	Aquatic dilution factor; transfer coefficients	Consumption rates; culinary factors; holdup times	Dose coefficients (ICRP 56, 67, 69)
Ingestion (irrigated products)	Aquatic dilution factor	Consumption rates; culinary factors; holdup times	Dose coefficients (ICRP 56, 67, 69)
Shoreline irradiation	Transport model	Occupancy factor	Dose coefficient (FGR 12)

NOTE: FGR, Federal Guidance Report; ICRP, International Commission on Radiological Protection.

Several sets of atmospheric dilution factors are calculated according to the height of effluent release: ground, elevated, or mixed mode. Several assumptions are made about depletion[15] and radioactive decay.

[15]Depletion reflects the loss of activity in the radioactive cloud along its transport downwind as a result of radioactive decay and deposition on the ground via dry (sedimentation or impaction) or wet (rain or snow) processes.

The values for atmospheric dilution factors are derived from sets of meteorological data that are recorded by the licensee on an hourly basis: wind speed, wind direction, and atmospheric stability class. These meteorological data are averaged over a specific year (or over a period of time greater than one year) that differed from one plant to another to obtain annual joint frequency distributions.

For the purposes of the epidemiologic study, it seems sufficient for recent years of effluent release to use the annual average atmospheric dilution factors calculated for the appropriate release height(s) using the correction for depletion and decay according to the physical half-life radionuclide that is considered. For early years (prior to 1975), calculation of the atmospheric dilution factors over the year of release that is considered or averaged on a quarterly basis or for the time of the batch releases during that same year could be considered if the appropriate meteorological data are available. In case the meteorological data are not available for the year or time period of interest, data averaged over 5-year time periods representative of the time period or year of interest could be used.

As shown in Table 3.12, the atmospheric dilution factor is the only environmental transport parameter that is needed to calculate the doses resulting from air submersion and inhalation. With respect to the doses from ground irradiation and ingestion, the radionuclide activities deposited per unit area of ground (expressed, for example, in Ci m^{-2}) are needed. In the PNL model, activities on the ground are also derived from the annual joint frequency distributions, supplemented with values of dry deposition velocity (a quantity that relates the activity deposited on the ground to the ground-level air concentration, in the absence of precipitation). For the purposes of the epidemiologic study, the same procedure could be used. It is recognized that the influence of the precipitation events, which are more effective than dry processes in scavenging the radioactive materials from the atmosphere, would not be taken into account. This is deemed to be a reasonable simplification because deposition on the ground does not occur for noble gases and occurs by different processes for tritium and carbon-14, which are the most important contributors to the dose from airborne releases. Finally, the deposition on the ground is partitioned between the activity that is first retained by vegetation and the activity that falls directly on the soil.

With respect to ingestion of food products, the activity deposited on the ground must be related to the radionuclide concentrations in agricultural products (mainly milk, leafy vegetables, and meat). This is done by means of transfer coefficients. Those provided in Tables E.1 and E.2 of Regulatory Guide 1.109 (USNRC, 1977a) should not be adopted blindly: In the framework of an epidemiologic study, it would be important to carry out a thorough literature search, especially for tritium and carbon-14, which

seem to be the most important radionuclides with respect to intakes by ingestion, to determine which coefficients to use.

3.4.3.2 Waterborne Effluent Releases

Just as in the case of the atmospheric effluent releases, the most important parameter in waterborne releases is the aquatic dilution factor, which is the quotient of the radionuclide concentration at the location of interest (expressed, for example, in Ci m^{-3}) and the release rate of that radionuclide (expressed, for example, in Ci s^{-1}). The locations of interest are those where water is taken for drinking or irrigation purposes (for freshwater releases) and where fish and invertebrates are harvested (for saltwater as well as for freshwater releases). For releases into rivers, the aquatic dilution factor can be reasonably assumed to correspond to homogeneous mixing of the released activity into the entire flow of the river. For other types of releases (into lakes, estuaries, oceans, etc.), the aquatic dilution factors are site specific.

In the PNL model, the annual average values of the aquatic dilution factors are, whenever possible, taken from the environmental information provided by the licensees; when no information is available, the PNL model provides default values. For the purposes of the epidemiologic study, it also seems sufficient to use annual averages of the aquatic dilution factors. Whenever possible, site-specific values should be derived from a thorough analysis of the relevant documentation.

With respect to ingestion of fish and invertebrates, the radionuclide concentrations in those foodstuffs are derived from the radionuclide concentrations in water using transfer coefficients, for example expressed in Ci kg^{-1} or Ci m^{-3}. Element-specific recommended values of such transfer coefficients, in the absence of site-specific data, are listed in Table A.1 of Regulatory Guide 1.109 (USNRC, 1977a). If site-specific data are not available, more up-to-date transfer coefficients may be available from other sources.

3.4.4 Exposure Factors (E)

For each pathway, exposure factors, representing the usage that humans make of the environment and of its products, have to be taken into consideration. In the PNL model, site-dependent parameter values were taken from plant-specific environmental information whenever possible. However, site-dependent values were usually not available; in that case, the generic values recommended in Regulatory Guide 1.109 (USNRC, 1977a) and presented in Table 3.13 were used.

The values of the exposure factors presented in Table 3.13 for inhalation and external irradiation appear to be reasonable for use in an

TABLE 3.13 Generic Values of Exposure Factors Used in the PNL Model for Average Members of the Population

Pathway	Infant	Child	Teenager	Adult
Ingestion: milk (L yr^{-1})	170	170	200	110
Ingestion: meat and poultry (kg yr^{-1})	0	37	59	95
Ingestion: fruits, vegetables, and grains (kg yr^{-1})	0	200	240	190
Ingestion: water (L yr^{-1})	170	260	260	370
Ingestion: fish (kg yr^{-1})	0	2.2	5.2	6.9
Ingestion: invertebrates (kg yr^{-1})	0	0.33	0.75	1.0
Inhalation: breathing rate (m^{-3} yr^{-1})	1400	3700	8000	8000
External irradiation: shielding and occupancy factor	0.5	0.5	0.5	0.5

SOURCE: Based on Table A-1 in NUREG/CR 2850, vol. 1 (1982).

epidemiologic study. For ingestion, however, two important considerations are not taken into account: (1) the fact that water treatment and culinary processes may result in a decrease in radionuclide concentrations in the consumed water and food products, and (2) the dilution of contaminated water and food products due to consumption of water and food products from noncontaminated sources. These factors would need to be taken into account in the framework of an epidemiologic study.

3.4.5 Conversion to Organ or Tissue Dose (K)

The conversion factors used to calculate doses from the activity intakes of water and food products (in the case of internal irradiation), and, in the case of external irradiation, from the ground-level air concentrations weighted according to shielding and indoor occupancy (for air submersion), and from the radionuclide concentrations in soil and sediments (for ground irradiation and shoreline irradiation, respectively) are discussed in Appendix I. Generally speaking, the factors related to external irradiation appear to be adequate for use in an epidemiologic study, but those related to internal irradiation will have to be updated with data included in the publications of the ICRP-56 series (ICRP, 1990, 1992, 1995a,b). These ICRP data are in terms of committed equivalent doses per unit intake. Additionally, because it will be important to calculate annual absorbed doses for high-LET and low-LET radiations separately, it will be necessary, for radionuclides with long biological half-lives of residence in the body (e.g., strontium-90) and for all alpha emitters, to use data files published by the USEPA (USEPA, 2002) that provide the required information. For all other radionuclides, the committed equivalent doses per unit intake are numeri-

cally equal to the annual absorbed doses per unit intake, so that the data provided in the publications of the ICRP-56 series can be used without modification.

3.5 OTHER RISK FACTORS

Individuals living near nuclear facilities may be exposed to radiation from other sources besides facility effluent releases. The most significant sources of these other exposures are from natural background radiation, radiation from medical diagnostic procedures, and cosmic radiation from air travel. For the purposes of dose reconstruction, all radiation is equal: That is, a cell, tissue, or organ cannot distinguish between radiation received from USNRC-licensed facilities and radiation received from these other sources. In fact, these other sources of radiation exposure may result in doses that are much larger than those from facility effluent releases. If doses from these other sources are differentially distributed in individuals living near a nuclear facility (e.g., by distance or direction from a facility), they could confound the results of an epidemiologic study (see Chapter 4). Even if these doses are not differentially distributed, they would still produce "noise" that could swamp the "signal" resulting from exposures to facility effluent releases. In either case, these other sources of exposure are risk factors that need to be considered in dose assessment studies.

3.5.1 Natural Background Radiation

As noted in Section 3.2, reported annual whole-body doses from nuclear facilities were generally at most only 10-20 mrem/yr to the MEI (e.g., Table 3.2), even in early years of facility operations when effluent levels were much higher than at present. Reported average doses to populations living within a few miles of a plant were generally much less than 1 mrem/yr. These doses are much lower than annual whole-body absorbed doses received from natural background radiation.

The levels of terrestrial gamma radiation from naturally occurring radioactivity in soil and building materials and from cosmic rays vary widely across the United States (NCRP, 2009a). For example, free-in-air terrestrial gamma radiation levels measured at 210 sites in the United States averaged 61 mrad/yr with a standard deviation of 23 mrad/yr (Eisenbud and Gesell, 1997).

Cosmic radiation adds to natural background levels. Cosmic-ray levels vary with altitude from about 30 mrad/yr at sea level to over 50 mrad/yr at high altitudes (Lowder and Beck, 1966; NCRP, 2009a). Thus, direct external radiation doses to persons living near nuclear plants due to facility effluents were much less than the doses they received from ambient natural

background at most sites, even in the 1970s and 1980s. They were also generally much less than the spatial and temporal variations in natural background radiation from site to site.

The natural terrestrial background radiation level at any site in any annual quarter can vary by several mrad due to variations in rainfall (soil moisture), snow cover, and radon levels. Figure 3.7 illustrates daily variations in exposure rate measured at a site in New Jersey in 1979.

The natural background doses cited above are free in air (that is, uncorrected for shielding by housing and indoor radiation sources). The exact dose to any individual from facility releases would depend very much on their exact location when the releases occurred, type of housing (shielding), the fraction of time an individual spent in housing or away from the facility vicinity, and other factors. The doses cited above also do not include internal exposure from naturally occurring radionuclides in the body or exposure to indoor radon.

Background doses from terrestrial and cosmic-ray free-in-air external exposure have been estimated only for some selected facilities using those facilities' reported TLD monitoring data. The approximate annual terrestrial background exposures[16] are shown in Figure 3.8. These annual background doses often vary by more than a factor of 3, and they are one or more orders of magnitude higher than the estimated doses to the MEI discussed elsewhere in this chapter (e.g., Table 3.2).

The spatial variations in background can be significant even over relatively small distances. Figure 3.9 shows the spatial variation, based on annual TLD readings, around the Millstone plant in 2009 when external radiation exposures due to effluents from the facility in 1979 were essentially zero. Annual background radiation levels varied by over a factor of 2 and were higher west of the plant than north of the plant. Variations over shorter intervals were likely even greater.

Because the ambient background doses are so much higher than expected doses from facility effluent releases and vary both with direction and distance, the epidemiologic study will need to consider variations in background radiation not only from facility to facility, but also around each facility. By evaluating the reported quarterly TLD monitoring data from each facility for recent years (when facility contributions to dose were very low), reasonable estimates of average annual background doses as a function of distance and direction can be made for use in the epidemiologic study.

[16]Based on the facility TLD monitors (biased low due to partial shielding because TLDs are generally attached to telephone poles, trees, or buildings). Note that these "background" exposures do not include exposures from internal emitters or indoor radon.

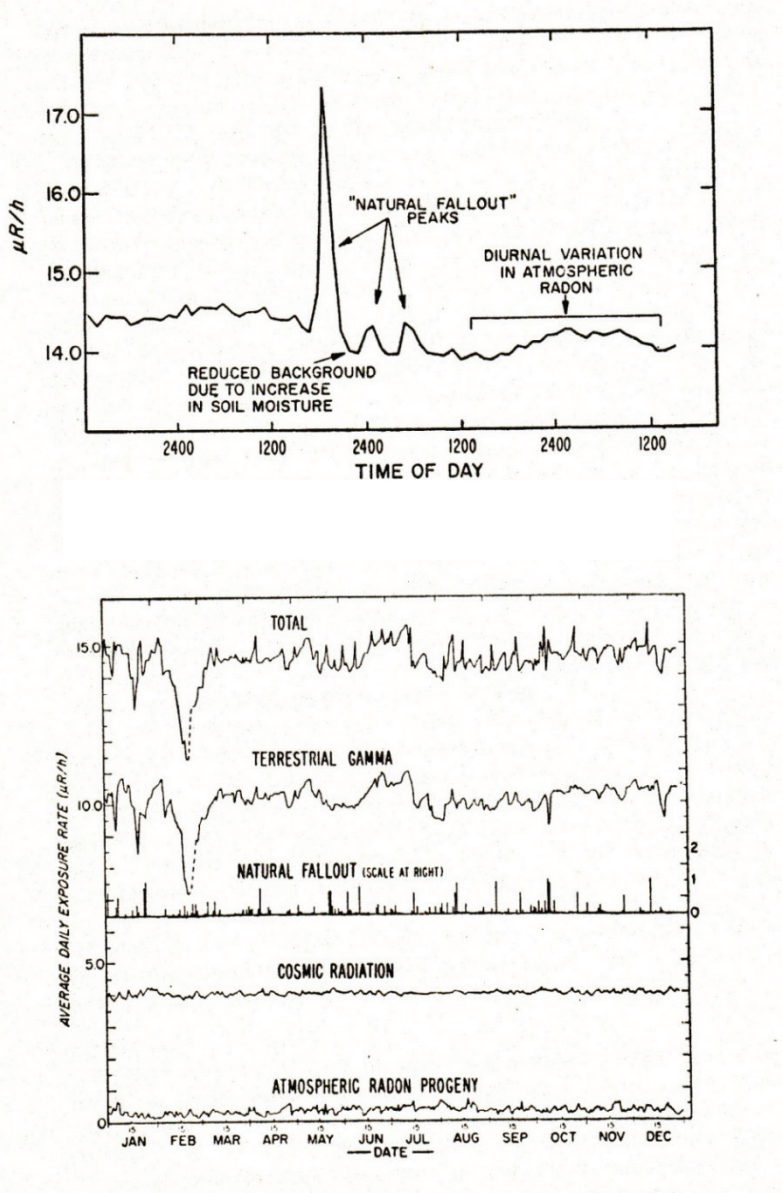

FIGURE 3.7 Daily variations in background radiation for a site in New Jersey. SOURCE: Beck and Miller (1982).

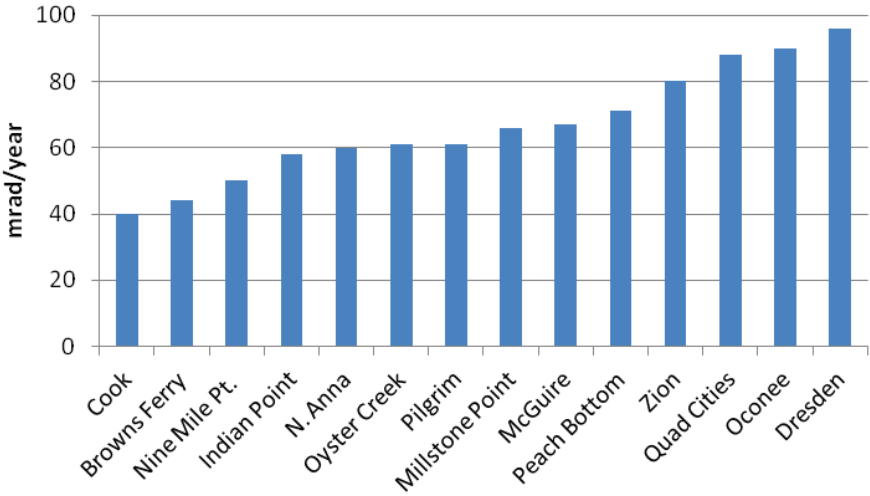

FIGURE 3.8 Variation in annual terrestrial free-in-air terrestrial and cosmic-ray natural background doses for selected facilities. SOURCE: TLD data from 2008-2009 radiological environmental monitoring reports for the plants shown in the figure.

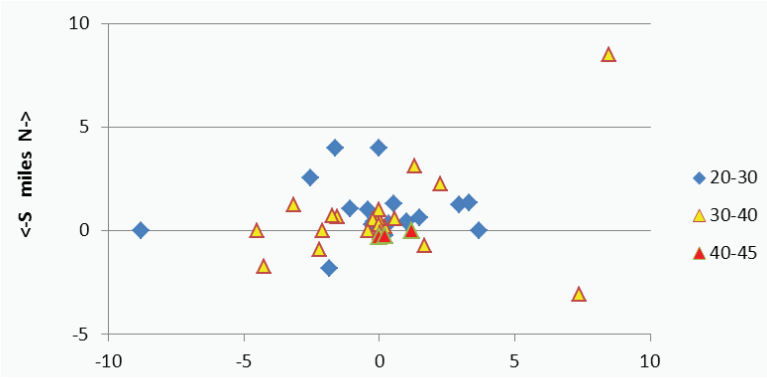

FIGURE 3.9 Variations in background radiation around the Millstone plant for 2009 based on TLD data. Note the relatively higher values near the fence line and variations with distance and direction. SOURCE: Dominion Nuclear Connecticut, Inc. (2009).

3.5.2 Other Sources of Radiation

Individuals living near nuclear facilities receive radiation from a number of other sources besides background radiation. Arguably, depending on age and lifestyle factors, the two largest of these may be radiation from medical diagnostic[17] procedures and air travel. These sources and their impacts on epidemiologic studies are described briefly in this section.

The NCRP estimates that the average person in the United States is exposed to almost as much radiation from medical procedures each year (~3 mSv annual effective dose) as from background radiation including radon (~3.1 mSv annual effective dose) (NCRP, 2009a). Radiation from medical procedures has increased more than seven times since the 1980s when the last NCRP report was published (NCRP, 1987), whereas radiation from natural background sources has remained unchanged. The most significant changes in medical imaging were attributed to rapid increases in usage of computed tomography (CT) and nuclear medicine procedures.

The exposures of particular individuals could be higher or lower than these averages depending on how many medical diagnostic procedures that use radiation they receive in any given year. There is no way to determine an individual's exposure to medical radiation without interviewing them, but even in these cases there are likely to be large uncertainties in estimated exposures. These uncertainties arise from recall bias (i.e., the individual's ability to recall the number, type, and dates of procedures) as well as the large variation in radiation doses that an individual receives from a given medical procedure depending, for example, on that individual's age and what body part is being irradiated.

Medical radiation could be a potential confounding factor in an epidemiologic study if individuals who live closer to nuclear facilities are exposed to radiation from medical diagnostic procedures at different rates compared to those who live farther away. This differential exposure could be due, for example, to differences in access to health care based on socioeconomic status. Confounding from medical radiation is likely to be less of a concern in epidemiologic studies that focus on children because they are less likely than adults to have received medical procedures involving high doses of radiation (e.g., CT scans, cardiac nuclear medicine procedures), although in utero exposure may be of concern (see, e.g., Table 3.14 in NCRP, 2009a).

Air travelers are also exposed to increased levels of radiation resulting from galactic cosmic radiation.[18] This radiation is primarily energetic pro-

[17]Exposure to radiation from radiation therapy is not discussed here. About 1 percent of individuals having diagnostic procedures are believed to be undergoing radiotherapy. The doses from radiotherapy are on the order of 5,000 to 50,000 times as large as diagnostic procedures (NCRP, 2009).

[18]Solar disturbances (e.g., solar flares) can also inject energetic particles into the Earth's atmosphere.

tons (i.e., hydrogen nuclei) and alpha particles (i.e., helium nuclei). These particles interact with air molecules in the atmosphere and generate additional ionizing radiations including neutrons, protons, muons, electrons/positrons, and photons. In general, the amount of radiation received during any particular flight depends on its altitude, latitude, and duration.[19] For example, a 13-hour one-way flight from New York to Tokyo flown at a maximum altitude of 43,000 feet is estimated to result in an effective dose of about 0.0754 mSv (i.e., 7.54 mrem).[20]

Radiation from air travel could be a risk factor in epidemiologic studies involving individuals who are frequent air travelers. There is no way to determine an individual's exposure to radiation from air travel without interviewing them, but even in these cases there is likely to be large uncertainties in estimated exposures owing to recall bias. Exposure due to air travel is likely to be less of a concern in epidemiologic studies that focus on children because they are less likely than adults to have undertaken extensive air travel.

3.5.3 Exposures to Other Hazardous Materials

Exposure to other hazardous materials, most notably toxic chemicals released from industrial facilities, can lead to a number of health outcomes including cancer (IARC, 2011; DHHS, 2011). Many of the front-end nuclear facilities discussed in Section 3.2 also release chemicals. Furthermore, it is well known that the chemical toxicity of some radioactive effluents such as uranium may be more deleterious than the low levels of radioactivity (Bleise et al., 2003). Consequently, chemical exposures could be an important risk factor in epidemiologic studies of populations that are exposed to both radiation and chemical hazards. This could be especially problematic if the epidemiologic study focuses on cancers that have both radiation and chemical etiologies such as bladder cancer and leukemia.

It will be important to identify major industrial facilities in the vicinity of nuclear facilities that are examined in the epidemiologic study. For example, the Metropolis, Illinois, conversion facility discussed earlier is co-located with a large chemical plant. The annual material releases from industrial facilities can be obtained from the USEPA[21] and assessed to determine their potential impact on the epidemiologic study. It might be

[19]The Earth's atmosphere and magnetic fields shield this radiation. As a consequence, less radiation is received at lower altitudes and at locations closer to the Earth's equator.

[20]See http://www.faa.gov/library/reports/medical/oamtechreports/2000s/media/0316.pdf.

[21]USEPA's Toxics Release Inventory program (see www.epa.gov/tri/) maintains a database on releases of over 600 toxic chemicals from facilities in the United States. Facility owners are required to provide information on their toxic releases to USEPA on an annual basis. The database was complete through 2010 when the present report was in development.

necessary to exclude particular census tracts or cancer types from the epidemiologic study in cases where there are substantial industrial releases. This will need to be handled on a facility-by-facility basis.

3.6 CHARACTERIZING AND COMMUNICATING UNCERTAINTIES

The uncertainties in dose estimates for an epidemiologic study are likely to be substantial. These uncertainties arise from uncertainties in source terms (i.e., reported effluent releases; see Chapter 2) and, usually to a greater extent, uncertainties in atmospheric transport and liquid dispersion models that relate these source terms to environmental concentrations, and also uncertainties in pathway models that relate environmental concentrations to dose. Uncertainties in dose estimates have the potential to mask the "true" dose-response relationship in an epidemiologic study. Consequently, understanding and characterizing these uncertainties is important.

The magnitude of dose estimate uncertainties is also likely to vary over time. Effluent release data for early years of facilities operations are of lower quality than more recent data (see Chapter 2). As a consequence, dose estimates based on earlier data are likely to be more uncertain than doses calculated for releases for more recent years. Moreover, because effluent releases in earlier years were much higher as a result of higher airborne effluent releases (see Chapter 2), uncertainties in airborne effluent releases are likely to be relatively more important than uncertainties in liquid effluent releases. The airborne effluent release uncertainties are a function of how representative the weekly grab samples[22] were with respect to the actual releases of specific nuclides, as well as to uncertainties in stack airflow rates, especially if they varied with time. There is much less uncertainty associated with the measured activities of the grab samples themselves. Furthermore, the use of an average quarterly value for batch releases rather than the actual values for each batch adds to the reported uncertainties and resultant dose estimates, particularly for PWRs.

Uncertainties in diffusion and dispersion models that relate source terms (effluent releases) to environmental concentrations as well as exposure pathway models relating environmental concentrations to doses can be high. Atmospheric dispersion estimates can also be very uncertain, particularly when releases are episodic, when there are terrain irregularities, and for locations that are distant from the facility fence line (Table 3.14). On sites with flat terrain, Gaussian plume models have been shown to provide reasonable estimates of air concentrations when integrated over a sufficient

[22]Effluent releases of specific radionuclides for continuous (as opposed to batch) releases are based on analyses of weekly grab samples rather than continuous monitoring. See Appendix H.

TABLE 3.14 Uncertainties in Gaussian Plume Models

Conditions	Range, Predicted over Observed Air Concentration (P/O)
Highly instrumented site; ground-level, centerline; within 10 km of a continuous point source	0.65 to 1.35
Specific time and location, flat terrain, steady meteorology, within 10 km of release point	0.1 to 10
Annual average, specific location, flat terrain, within 10 km of release point	0.5 to 2
Annual average, specific location, flat terrain, 10 m to 150 km downwind	0.25 top 4
Complex terrain or meteorology, episodic releases	0.01 to 100
Episodic, surface-level releases, wind speeds less than 2 m s^{-1}	1 to 100

SOURCE: Miller (1995).

time interval, although estimates for a shorter integration times can be very uncertain. Uncertainties increase for sites with complex terrain (e.g., sites with hills or valleys). Also, local meteorology at any particular time (wind speed, direction, and atmospheric stability) can vary significantly from annual averages and result in significant errors if the latter are used to estimate doses for batch effluent releases into the atmosphere.

Liquid diffusion models for effluent releases into estuaries, lakes, and oceans, as well as spills into surface and ground water, are very crude. Additionally, estimates of environmental usage of potentially contaminated water are also very crude in the absence of subject interviews. Thus, most estimated doses resulting from liquid effluents to representative individuals residing in specific locations are likely to be highly uncertain and will vary significantly from individual to individual and location to location.

As discussed in Chapter 2, effluent emissions varied widely over time and generally decreased rapidly with distance from the facility fence line. Exposed persons were not at the same place with respect to the facility at all times. Consequently, the dose to any particular individual will be even more uncertain than the dose to an unspecified individual at a particular location and time. For studies that are based on individuals (such as a cohort or a case-control study) that require individual dosimetry data, this uncertainty will depend on the ability to determine individual lifestyle behaviors.

Considering the complexity and range of uncertainties discussed above, a detailed quantitative analysis of uncertainty in an epidemiologic study is not practical, particularly for an ecologic study. An extensive quantitative analysis would require resources and effort not commensurate with the magnitude of the likely doses, the quality of the effluent release data, and the degree of complexity recommended by the committee for dose reconstruction. However, a quantitative or at least semiquantitative uncertainty

analysis could be performed, at least for a few facilities and years of operation, for the case-control study.

Nevertheless, at the very least, any epidemiologic study will need to address uncertainty, at least qualitatively. Such an analysis should:

- Identify, evaluate, and rank all potential sources of major uncertainty and identify site-to-site and temporal differences;
- Identify potential bias versus random errors in the dose calculations that could affect interpretation of the epidemiologic findings; and
- Identify shared errors[23] as opposed to stochastic variability to properly evaluate the risk from radiation exposure should any increased risk of cancer be identified.

Although the reported environmental monitoring data for almost all sites and times was either below minimum detectable levels or, for external radiation, not distinguishable from background, an epidemiologic study could still use these data to set upper limits on the reported effluents by back-calculating from the minimum detection levels. This would at least place upper bounds on effluent releases.

3.7 FINDINGS AND RECOMMENDATIONS

This chapter provides the committee's assessment of methodological approaches for assessing offsite radiation doses to populations living near nuclear plants and fuel-cycle facilities to support an epidemiologic study. Based on this assessment, the committee finds that:

1. *Absorbed dose*—the energy deposited by ionizing radiation per unit mass of tissue in specific organs of interest—is the appropriate dose quantity for use in an epidemiologic study. Other dose quantities, for example effective dose, equivalent dose, and collective dose, are designed for regulatory purposes and are not appropriate for epidemiologic studies (see Section 3.4.1). The dose to a maximally exposed individual (MEI) is also not an appropriate quantity for an epidemiologic study because it provides a high-sided estimate at

[23]As discussed in NCRP (2009b), uncertainties that are common to many individuals (for example, error in the amount of effluents from a facility) can introduce bias (systematic uncertainty) in estimated doses compared to uncertainties that are unshared and represent stochastic variability in true doses among individuals. When uncertainties are shared among individuals in a population, the degree of variability in true doses among individuals is less than would be estimated by assuming that uncertainties in each individual's dose are purely random. An overestimation of the variability in true doses among individuals results in a suppression of dose-response relationships derived in an epidemiologic study, i.e., the true dose response is flattened (Schafer and Gilbert, 2006).

a single spatial point and does not reflect the variation is dose with distance and direction from a nuclear facility.

2. Absorbed doses to individuals attributable to living near nuclear plants and fuel-cycle facilities are anticipated to be very low (see Sections 3.2 and 3.3), in most cases well below variations in levels of natural background radiation in the vicinity of the facility and from facility to facility. These doses are also anticipated to be lower than levels of radiation received by some members of the public from medical procedures and air travel. Consequently, dose estimates used in an epidemiologic study need to account for these other radiation exposures and other risk factors such as exposure to hazardous (and potentially carcinogenic) materials released from industrial facilities located near nuclear facilities (see Section 3.5).
3. Estimates of doses to individuals living around nuclear facilities will have uncertainties owing to facility effluent releases, dose models, and other risk factors. A detailed quantitative analysis of uncertainty is not practical. However, a qualitative uncertainty analysis can be performed for a few facilities and years of operation to estimate the probably magnitudes of these uncertainties (see Section 3.6). It will be important to communicate these uncertainties to stakeholders as part of the epidemiologic study.
4. Computer models have been developed to estimate absorbed doses in individuals exposed to radiation through environmental pathways. These existing models could be adapted or a new model could be developed to estimate doses to individuals living near nuclear facilities to support an epidemiologic study. Regardless of the approach used, it is essential that the underlying computer model reflect modern practices for dose reconstruction (see Section 3.4).

In light of these findings, the committee recommends that a pilot study be undertaken to demonstrate the feasibility of reconstructing absorbed doses for an epidemiologic study. This pilot study should:

1. Develop a computer model (i.e., by modifying or adapting an existing model or building a new model) to obtain estimates of absorbed doses to the whole body and individual organs resulting from airborne and waterborne effluent releases. This model should be similar in scope and complexity[24] to that used by the

[24]The committee uses the phase "similar in scope and complexity" to mean that the model should use the same general approach as the PNL model to estimate annual absorbed doses as a function of direction and distance from a facility based on effluent release and meteorological data averaged over daily to quarterly periods.

Pacific Northwest Laboratory (Baker, 1996) to estimate doses to populations living near nuclear plants in the 1970s and 1980s, but it should be updated as described in Section 3.4 to provide point and census-tract estimates of absorbed dose using modern dose reconstruction practices.

2. Demonstrate the utility of this model for dose reconstruction to support the epidemiologic study designs recommended in Chapter 4 (See Section 4.4 in Chapter 4) by:

   - Using the model to obtain dose estimates as a function of distance (0 to 50 kilometers [30 miles] from the plant) and direction for the six nuclear plants and one fuel-cycle facility subject to the pilot study in Chapter 2 (see Chapter 2, Section 2.5).
   - Developing a methodology to account for natural background radiation and, to the extent feasible, other sources of radiation in the dose estimates.
   - Undertaking an uncertainty analysis as described in Section 3.6.

The results of this pilot study should be used to inform decisions about any Phase 2 epidemiologic study effort.

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4

Epidemiologic Studies

This chapter addresses the second charge in the statement of task for this study (see Sidebar 1.1 in Chapter 1) on methodological approaches for assessing cancer risks in populations near U.S. Nuclear Regulatory Commission (USNRC)-licensed nuclear facilities. It is specifically intended to address the following issues:

- Different epidemiological study designs and statistical assessment methods.
- Geographic areas to use in the study.
- Cancer types and health outcomes of morbidity and mortality.
- Characteristics of the study populations.
- Availability, completeness, and quality of cancer incidence and mortality data.
- Approaches for overcoming potential methodological limitations arising from low statistical power, random clustering, changes in population characteristics over time, and other confounding factors.
- Approaches for characterizing and communicating uncertainties.

4.1 BACKGROUND ON EPIDEMIOLOGIC STUDIES

Epidemiology is the study of the distribution of diseases and other health-related conditions in populations, and the application of this study to control health problems. The purpose of epidemiology is to understand what risk factors are associated with a specific disease, and how disease

can be prevented in groups of individuals; due to the observational nature of epidemiology, it cannot provide answers to what caused a disease to a specific individual. Epidemiologic studies can be used for many reasons, commonly to estimate the frequency of a disease and find associations suggesting potential causes of a disease. To achieve these goals, measures of disease (incidence) or death (mortality) are made within population groups. Epidemiology is fundamentally multidisciplinary and it uses knowledge from biology, sociology, statistics, and other fields.

The four types of epidemiologic studies commonly used in radiation research are *cluster*, *ecologic*, *case-control*, and *cohort* studies. An additional approach for estimating risk in radiation research—although strictly not an epidemiologic study—is risk-projection models. These models are used to predict excess cancer risks by combining population dose estimates with existing risk coefficients to transfer risks across populations with different baseline rates. This type of modeling approach is not new; one of the earliest examples of its use was by the U.S. Federal Council Report, where 0 to 2000 leukemia deaths in the United States attributed to exposures to fallout from above-ground nuclear testing up to 1961 were estimated (Federal Radiation Council, 1962). As discussed in a comprehensive review (Berrington de González et al., 2011), recent applications of the risk-projection modeling have increased partly because of the publication of user-friendly risk estimates for U.S. populations in the BEIR VII report (NRC, 2005) and the increasing acceptance of the limitations of epidemiologic studies of low-dose radiation exposures, mainly owing to their limited statistical power.

The study designs described in this chapter can provide clues for potential associations between cancer and living near a nuclear facility. The first thing that the epidemiologist questions is whether any observed association is real, or if it is due to bias, confounding, or simply due to chance. "Bias"[1] is a general term related to error in the measurement of a factor and can arise from a variety of sources such as the method of selection of cases and controls, or exposed and unexposed (selection bias), or due to the inaccurate information regarding either the disease or exposure status of the study participants (information bias). On the other hand, confounding refers specifically to the existence of some third variable, the "confounder," that alters the degree of association between the exposure and the disease of interest. Confounding is a potential issue with all epidemiologic studies discussed here.

[1]The term "bias" when used scientifically does not necessarily imply the researcher's desire for a particular outcome, or any prejudice, as it is often implied with the conventional use of the term.

4.1.1 Cluster Studies

A *cancer cluster* is an aggregation of a relatively unexpected high number of cases. Clustering can be "spatial," when the disease in question has a higher incidence rate in some places than in others, or "temporal," when the incidence rate is higher at a specific time compared to other times. A disease cluster can also be "spatiotemporal." Testing involves comparing the observed number of cases with the number expected, based on the size and age composition of the population.

The scientific reason to examine disease clusters is to learn about the causes of the cluster and, by extension, gain insight toward the causes of disease. Epidemiologists and public health workers recognize the value of historic examples of cancer cluster examination which contributed to the recognition of human carcinogens in those situations. Typically, exposure was high, prolonged, and well defined. In contrast, most cluster reports involve exposures that are low and poorly defined, and the cases involved are a mix of unrelated, relatively common cancers. For these reasons there is skepticism regarding the scientific value of the investigation of reported clusters (Neutra, 1990; Rothman, 1990).

In a rather provocative summary of the reasons why—with a few exceptions—there is little scientific or public health purpose to investigate individual disease clusters, Rothman (1990) explains that the boundaries of the space and time that encompass the cluster should be clearly defined before examination of the cluster and should not be defined after the fact to capture a population that has experienced the high disease rate. This interpretation has been described as the "Texas sharpshooter's" procedure in which the shooter first fires his shots randomly at the side of the barn and then draws a bull's eye around each of the bullet holes. This kind of process tends to produce clusters of causally unrelated cases of no etiologic interest. As noted by Rothman (1990), assigning statistical significance to a reported cluster requires clear definitions of the populations, regions, and/or time periods under consideration, often a challenging undertaking.

4.1.2 Ecologic Studies

An *ecologic study* (sometimes referred to as a *geographic study* or *correlation study*) evaluates the relationship between an exposure and a disease in some aggregate group of individuals, but not specific individuals, such as those living in a country, a county, a community, or a neighborhood. This is in contrast to case-control and cohort studies where the unit of analysis is the individual. In an ecologic study, average measures of exposure and disease frequency are obtained for each aggregate, and the analyses focus on determining whether or not the aggregates with high levels of exposure also display high disease rates. For example, in a study that uses counties as the

unit of analysis, the data of interest are average values of exposure and aggregate counts of disease by county. However, the individuals who actually develop cancer in a county may be more or less exposed than the county average, so the association across county populations may not accurately reflect the association for the individuals who develop cancer. This issue is referred to as *ecologic fallacy* or *ecologic bias* and is the main limitation associated with ecologic studies. The magnitude of the ecologic bias is not measurable; therefore, conclusions need to be stated carefully and results interpreted with caution.

One of the causes of ecologic fallacy is that average levels of potential confounding variables across the geographic units may be subject to considerable measurement error, so trying to adjust for the geographically estimated confounding variables fails to control for confounding. This was illustrated in a study of the association of average county radon levels with lung cancer rates, with an attempt to characterize smoking levels by county (Cohen, 1995, 1997). The radon–lung cancer ecologic correlations were in the negative direction, whereas a series of studies using estimated individuals' radon exposure have shown positive associations (Darby et al., 2005). This poor control for confounding is important mainly for potential variables that have strong association with the target disease (e.g., smoking and lung cancer) and is of lesser concern for weak confounding variables. However, when expected effects of exposure are themselves quite weak, then good control for confounding variables becomes especially important.

4.1.3 Case-Control Studies

The aim of a *case-control study* is to determine whether the frequency of exposure to several possible risk factors is higher in the group of people with the disease of interest (cases) than in the group without the disease (controls). The proportion of cases with and without an exposure suspected to be linked with the disease is compared to the proportion of controls with and without the relevant exposure. If a certain exposure is associated with or causes a disease, then a higher proportion of past exposure among cases is expected compared to the proportion of past exposure among the controls. If the difference cannot be explained by chance, an association between the disease and the characteristic may be inferred.

Cases can be selected from hospitals, registries, or other relevant sources. However, cases based on hospitals may be a biased sample; for example, those cases seen at referral hospitals may represent more serious or unusual cases. Therefore, population-based case ascertainment is the preferred study design. This may be possible through a cancer registry if the registry can provide complete information on diagnoses of cases. Control selection requires equal thought and consideration, because the controls

must come from the same population base as the cases; subtle differences in the way cases and controls are selected may lead to selection bias. The major point is that the controls have to reflect the population from which the cases arose. For general-population case-control studies, various methods are used to identify controls for study as discussed in Section 4.3.4.

4.1.4 Cohort Studies

In a *cohort study*, the investigator typically selects a group of exposed and a group of unexposed individuals and follows both groups over time to determine disease occurrence in relation to the exposure. In the radiation epidemiology field, when individual exposures or doses are available, cohort studies typically examine gradients of exposure rather than just unexposed and exposed groups. The data necessary for assessing disease diagnosis can be obtained either directly by periodic examinations of individuals or by obtaining data from disease registrations, hospital records, and death certificates. For rare diseases or those that take a long time to become evident, such as cancer, the investigator needs to start with a large number of exposed and unexposed individuals and follow them for a long period of time. Study participants may be lost to follow up in a cohort study because they do not wish to take part in the study, because they cannot be located, or because they have died. Minimizing these losses is crucial because they reduce the number of participants being followed. Also, participants that are lost to follow-up may differ in characteristics from those that remain enrolled in the study. When reporting the study design, it is important to note the percentage of and any available demographic information on subjects that are lost.

A cohort study is considered to be a more scientifically rigorous study design compared to case-control, ecologic, or cluster studies. This is because cohort studies measure potential exposures before the disease has occurred and therefore can demonstrate that they may have caused the disease. Because cohort studies most often look forward to the future, they are also referred to as *prospective studies*. However, a cohort study can also be retrospective if both exposures and outcomes have already occurred and accurate historical data are available when the study begins. Studies on radiation effects are often jointly retrospective and prospective; exposures occurred mainly in the past and disease ascertainment includes both past and prospective follow-up.

4.2 STUDY DESIGNS CONSIDERED

Choosing from among different possible study designs to assess cancer risks in populations near nuclear facilities, or even deciding against mak-

ing a proposal for a particular study design, is based on answers to several difficult questions. Most of these questions are scientific, dosimetric, epidemiologic, and statistical, and require technical knowledge and expertise. However, some are less technical and involve public concerns and perceptions that may be difficult to quantify. The primary focus of this chapter is on technical issues, partly because they serve as a foundation for judgments that may involve additional public and stakeholder considerations.

The committee considered the following general approaches to an epidemiologic study of cancers that might be undertaken by the USNRC:

1. Risk-projection models.
2. An ecologic study based on estimates of exposure levels at the census-tract level.
3. Cohort studies tracking estimates of individual exposure levels and recording case incidence within the cohort. Variations considered include:
   - A prospective cohort study.
   - A retrospective cohort study.
4. Case-control studies comparing estimates of individual exposure levels between cancer cases and controls. Variations considered include:
   - A record-linkage-based case-control study with no direct contact with cases and controls or their proxies.
   - A de novo case-control study with direct contact with cases and controls or their proxies.
   - Building on existing studies and their associated data.

The discussions of these possible studies in the following sections are based primarily on the study characteristics summarized in Table 4.1. Section 4.2.1 of this chapter considers matters that affect most or all of these study designs; Section 4.2.2 describes each approach in some detail. These descriptions define the strengths and weaknesses of the recommended studies, summarized in Section 4.2.3. Section 4.3 provides a summary of data sources for population counts, health outcomes, and other information required for the execution of the studies considered and recommended.

4.2.1 Issues Affecting Several Epidemiologic Study Designs

In any of the studies considered, population sizes, estimated doses, and resulting risk estimates may be too low to demonstrate statistically significant increased cancer risks near nuclear facilities. As noted in Chapter 3, the dose received from living near a nuclear plant is estimated to be less than 0.01 mSv/yr (USEPA, 2007). This dose is much lower than doses from

TABLE 4.1 Summary of the Characteristics of the Studies Considered

	Risk Projection Models Theoretical Evaluation	Ecologic Hypothesis Generating	Case-Control Hypothesis Testing		Cohort Hypothesis Testing	
			Record Based	Subject Contact	Prospective (Subject Contact)	Retrospective (Record Based)
Outcome						
Incidence/ Mortality	Theoretical	GU-based rates	Individual level	Individual level	Individual level	Individual level
Time period	Past, current or future	Past and current	Fairly recent past and current	Recent past and current	Future	Fairly recent past and current
Number of Cases	N/A	Large, depending on availability of aggregated cancer incidence and mortality data	Limited to relatively recent cases, depending on available birth record and cancer incidence data	Limited to recent cases (and those that are alive), successfully traced, and willing to participate	Limited to future cases and subject to length of follow-up period	Limited to cases that are successfully linked via birth records
Cancer Types	All	All	Limited, primarily suitable for childhood cancers or those due to early exposures	Limited to one or a few types	Limited to a few relatively common types depending on follow-up period	Limited, primarily suitable for childhood cancers or early exposures
Age	All ages	All ages	Best for childhood cancers; limited for adults	Targeted ages	All or targeted ages	Best for childhood; limited for adult

continued

TABLE 4.1 Continued

	Risk Projection Models Theoretical Evaluation	Ecologic Hypothesis Generating	Case-Control Hypothesis Testing		Cohort Hypothesis Testing	
			Record Based	Subject Contact	Prospective (Subject Contact)	Retrospective (Record Based)
Nondiseased comparison group	N/A	Census denominators	Requires selection and study of a comparison group	Requires selection and study of a comparison group	Participants would be nondiseased at entry, and number of individuals developing disease during study period would be determined	Participants would be nondiseased at entry, and number of individuals developing disease during study period would be determined
Exposure						
Dosimetry	GU-based	GU-based	Individual location at birth	Individual locations	Individual	Individual location at birth
Lifetime exposure	Can be constructed for hypothetical individuals based on residential history	Approximate, without information about residential changes	Limited primarily to exposure at time of birth	Complete lifetime residential history derived from interview data	Complete lifetime residential history derived from interview data	Lifetime residential history derived from records, but realistically will be limited primarily to exposure at time of birth

Temporality	Can fully utilize historical variations in plant exposure levels prior to each year of interest	Can fully utilize historical variations in plant exposure levels prior to diagnosis dates	Restricted to exposure at birth location; limited because must use relatively recent cases	Restricted to exposure prior to diagnosis, limited because must use recent cases only	Can include all exposure prior to diagnosis, but does not address the higher past exposures	Restricted to exposure at birth location, dependent on how far back in time birth records with adequate information are available
Potential Confounders						
Natural background radiation	GU-based	GU-based	Residence based	Residence based and direct measurements possible	Residence based and direct measurements possible	Residence based
Socioeconomic status	GU-based	GU-based	Individual level through socioeconomic proxies insofar as available in records	Individual level via questionnaires	Individual level via questionnaires	Individual level through socioeconomic proxies insofar as available in records
Urban/rural/ mixed residence	GU-based	GU-based	Individual level at birth	Individual level complete history	Individual level complete history	Individual level at birth
Medical exposures	GU-based approximations	GU-based approximations	GU-based for individual birthplace	Individual level via questionnaire	Individual level via questionnaire	GU-based for individual birthplace
Other risk factors	GU-based	GU-based	Limited to information available on birth records	Individual exposures and risk factors via interviews	Individual exposures and risk factors via interviews	Limited to information available on birth records

continued

TABLE 4.1 Continued

	Risk Projection Models Theoretical Evaluation	Ecologic Hypothesis Generating	Case-Control Hypothesis Testing		Cohort Hypothesis Testing	
			Record Based	Subject Contact	Prospective (Subject Contact)	Retrospective (Record Based)
Biases						
Selection bias	In- and out-migration	In- and out-migration	Out-migration and unsuccessful linkage	Out-migration, unsuccessful linkage, and unlocatable study subjects	Lost to follow-up, study dropouts	Out-migration and unsuccessful linkage
Nonparticipation	None	None	None	Likely	Likely	None
Response	None	None	None	Possible over or underreporting	Possible over or underreporting	None
Assessment of causality	N/A	Requires confirmation using another study design	Considered	Considered	Considered	Considered

NOTE: GU, geographic unit such as census tract; N/A, not applicable.

natural background radiation and medical diagnostic procedures, which combined are estimated to be 6.2 mSv/yr for the average[2] person in the United States (NCRP, 2009). Consequently, the attributed risk to exposure from radiation from a nuclear facility, if any, would be a small increase above the baseline lifetime risk of cancer occurrence in the general population in the United States, which is considered to be 42 percent (NRC, 2005).

Statistical power calculations based on estimated exposure estimates indicate that extremely large sample sizes are required except under the following scenarios:

A. Routine releases from the operating facilities have been far greater than those reported to the USNRC, or
B. Sensitivity to radiation as characterized in most or all generally accepted risk models is either inappropriately low or simply irrelevant to the populations living near nuclear facilities in the United States.

Regarding scenario B, underestimation of risks associated with radiation could be perhaps a result of inaccurate models for interpolation to low doses. Translation of risk estimates from World War II atomic bombing survivors to the population in the United States may also be proven inaccurate, though there is reasonably good concordance of estimated risks for Japanese and Western populations (UNSCEAR, 2006, Annex A). Exceptions are a few cancer sites with disparate background rates, such as stomach and liver cancer. (These cancers are more common among the Japanese compared to Western populations due to differences in risk factors such as diet and rate of infections.)

Even if one or both of these scenarios are considered possible, the reliability of any proposed study still hinges on the technical issues of accurately characterizing doses received by the populations under study over the time of facility operations. Accurate estimation of those doses requires reasonably accurate measures of releases, modeling of exposure levels at various geographic locations, and biologic uptake and biokinetics for radionuclide exposures (see Chapters 2 and 3).

4.2.1.1 Questions Addressed by the Studies

Epidemiologic studies provide the most direct and relevant evidence for an association between a suspected risk factor and disease. Each of

[2]This dose to the average person in the United States includes people who never had a medical procedure that involves high-dose radiation, such as CT scan or a fluoroscopy procedure. For those individuals that have had such procedures, the annual dose is higher. For reference, the average dose received from a CT scan is 8 mSv.

the study approaches considered in this chapter might produce useful new information regarding the association between living near a nuclear facility and potential cancer risks. However, they are unlikely to contribute substantial scientific knowledge regarding low-dose radiation effects because exposure levels are uncertain and probably low, which produces risk estimates with large relative uncertainties. Moreover, each of the possible study approaches is subject to limitations in the types of questions that may be answered. The committee has framed three questions of primary interest based on its statement of task (see Sidebar 1.1 in Chapter 1):

1. Are any detectable cancer-related health effects, such as mortality and morbidity from any type of cancer, associated with living near a nuclear facility at present or in the past?
2. If so, what are the characteristics of the affected persons (such as age, gender, race/ethnicity)?
3. What are the factors that could (and should) be examined to help detect and adjust for possible confounding (such as smoking and exposure to medical diagnostic procedures)?

These questions are closely related, and cannot be fully investigated as if they were independent of each other. The second and third questions are of little interest if there is no health effect to be studied. Furthermore, the difficulties in deriving an unambiguous answer are so great that it seems unlikely that the other questions, as important as they are, can ever be answered with precision by epidemiologic studies of populations living near nuclear facilities. As a result, the committee focused most of its effort on evaluating approaches to address aspects of this first question. If an association between living near a nuclear facility and cancer risk is observed, a balanced "weight-of-evidence" approach needs to be applied to determine whether the association is real, and whether that association can be explained by the radioactive releases from nuclear facilities.

A plausible cause-effect relationship between radioactive releases from nuclear facilities and cancer cannot be established solely by examining risks in populations living near nuclear facilities through any of the study designs considered. Direct epidemiologic investigation of the exposures in populations near nuclear facilities is limited by small numbers, the presence of unmeasured risk factors and potential confounders, and/or uncertainty in the exposure estimation. For these reasons, understanding the carcinogenic effects of low-level radiation exposure requires a diverse body of evidence in addition to any epidemiologic findings. Such evidence includes the effects of radiation on cell culture systems and animal models where all conditions including dose and dose rate are easily controlled and measured and

therefore causal associations with disease outcome can be established. This is the focus of the Department of Energy's Low Dose Radiation Program.[3]

4.2.1.2 *Study Endpoints: Cancer Incidence and Mortality*

Fundamental to the assessment of cancer risks are the concepts of *mortality* and *incidence rates*, that is, numbers of cancer deaths or new cancer occurrences observed or expected per year in a population of a specified size (often presented per 100,000 persons in a population or per 100,000 persons of each gender in a population).

Incidence is a measure of disease burden, as it describes the occurrence of new cancer cases. Mortality can index a more severe form of disease burden provided that survival is the same in the groups being compared, as mortality reflects both incidence and survival probability. However, for cancers that are not commonly fatal, for example, thyroid cancer, the most useful end point of disease burden is incidence of the disease since in any given year mortality will represent both new and existing cases of disease. A mortality study of thyroid cancer would have restricted statistical power in testing increases in risk at a certain time and interpretation because most of the incident cases in a year would not be captured in the mortality statistics for that year, and many of the deaths in the mortality data for a given year would have been diagnosed many years earlier.

On the contrary, for highly fatal cancers such as lung and pancreatic cancers, mortality data would reflect cancer incidence quite accurately. For diseases that have a greater susceptibility to surveillance bias such as prostate cancer, mortality data may be useful because they are minimally affected by that bias.

In an ideal study, one would identify each newly diagnosed case of some cancer type in the population under study at or near the time it was diagnosed. This may be possible in states where cancer registries have been in place for the study period of interest and the data are complete and of good quality (see Section 4.3.2). However, many cancer registries were established after nuclear facilities began operations. The time-limited availability of some registry data would affect mortality studies that use aggregated data at small geographic units such as a census tract; however, it would not affect mortality studies that use aggregated data by county. County-level mortality data have been commonly used in the United States because of the ease of identifying cases nationwide over a long time period through the National Center for Health Statistics (NCHS) (see Section 4.3.3).

[3]http://science.energy.gov/ber/research/bssd/low-dose-radiation/.

Misdiagnosis of cancer is currently less of a concern than it used to be for both incidence and mortality studies; however, misclassification[4] of cancer types may occur. Moreover, incidence studies may lead to biased results when there are changes over time in the likelihood that a cancer was diagnosed, that it was diagnosed but not reported, or that the diagnostic criteria changed. The likelihood that a life-threatening cancer will not be diagnosed is small, but the prevalence of asymptomatic, undiagnosed cancers, especially in older persons, can be large. Changes in the intensity with which people are screened and cancers are reported and registered (for example, prostate cancer) can produce an appreciable artifactual trend in recorded incidence. Also, the reported site of a cancer may be incorrect, especially in earlier years. An example is the earlier misdiagnosis of metastatic cancers as primary in the brain, whereas newer imaging technologies continue to improve the classification of cancer to the correct primary site.

The detection of small, more indolent cancers and the appreciable variation within and between populations in the use of diagnostic tools can affect incidence data but may have little effect on mortality data. Variations in degree of cancer surveillance can be a concern for some cancers; uneven degrees of surveillance in populations in various geographic locales can artificially simulate or mask exposure-response relationships. The primary site of a cancer is more likely to be recorded accurately by a cancer registry than a death certificate (German et al., 2011). Also, trends in registration rates should not be biased by improvements of cancer therapy on patient survival. This problem is avoided by using data on deaths from registries with active follow-up of patients such as that implemented by the Surveillance, Epidemiology, and End Results (SEER) registries (see Section 4.3.2), although such studies would be limited to the states or regions covered by these registries and would not cover all areas near nuclear facilities.

For the reasons mentioned above, incidence and mortality studies provide complementary data, and both could provide potentially useful information. When the quality of the incidence and mortality data is high, the mortality-to-incidence ratio is related to case survival; when the quality of one or the other is not adequate, the ratio will deviate from the survival ratio. The value of either incidence or mortality registries increases when data from different times and locations can be compared because they are compiled according to agreed national or international standards. All cancer registries in the United States use classification schemes that are largely compatible with each other and with the classification for causes of death on death certificates.

Both risk of developing cancer and risk of dying of cancer are sub-

[4]Misclassification is the erroneous attribution of a cancer into a category other than that it should be assigned.

stantial public concerns. In an analysis of cancer risks near nuclear facilities, incidence and/or mortality data are linked with residence at the time of cancer diagnosis or death from cancer that is retrieved from medical records or death certificates, respectively. As cancers manifest themselves years or decades after the exposure (see discussion on latency period in a later paragraph of this section), for such inferences use of incidence data is somewhat preferable to mortality because residence at time of diagnosis is a better indicator of where the person may have lived at time of exposure compared to residence at time of death. Persons who lived in a particular area at time of death may not have been long-term residents of that area and, therefore, may not reflect the address at which the relevant exposure occurred, possibly many years earlier.

4.2.1.3 Selection of Cancers to Study

Radiation can cause cancer in almost any tissue in the body but some sites are more susceptible to radiogenic effects than others (UNSCEAR, 2006, Annex A). In general, it has been found that cell radiosensitivity is roughly proportional to the rate of cell division, so cells that actively divide are more radiosensitive (although there are exceptions to this).

Radiation-induced cancers, similar to cancers induced from other risk factors, manifest themselves years or decades after the exposure. The lag time between exposure to a disease-causing agent such as ionizing radiation and the clinical recognition of the disease is known as the *latency period*. The mean latency period per cancer type due to radiation has not been comprehensively summarized, partly because it varies by age at exposure to radiation (Preston et al., 2002; Ron et al., 1995), type of cancer, and especially duration of follow-up of the cohort. However, studies of the atomic bomb survivors in Japan have demonstrated that for most major cancers the latencies of individual cancer cases begin at some minimum period and extend for the rest of the lifetime. Epidemiologic studies that aim to link exposure to radiation and cancer often use a 2-year minimum latency period for leukemia and a 10-year minimum latency period for solid[5] cancers (Boice et al., 2011). For this reason, past exposures are more relevant than current exposures as potential causes of cancer.

Given that different segments of the public have concerns about a variety of cancers, study of a wide range of cancers may be necessary, but

[5]Often in radiation epidemiology nonleukemia cancers are grouped and analyzed together in a category named "solid cancers." This grouping may make only limited sense from a biological or medical point of view since cancers at different sites are too different to be grouped in terms of their causes, other risk factors including genetic effects, carcinogenesis stages (Trott and Rosemann, 2000), and possibly histology. However, because the numbers of cancers at individual sites are too small for a robust analysis, grouping is often a necessity.

particular attention needs to be given to the most radiosensitive cancer sites, including leukemia, female breast, bladder, thyroid, brain, and ovary.[6] Childhood leukemia is a "sentinel" cancer for radiation exposure and may merit separate, more detailed study with individual exposure information, as will be discussed in Section 4.2.2. Examining cancers that are presumably nonradiogenic in origin such as prostate cancer could serve as useful negative controls.

Much of what we know about tissue radiosensitivity comes from studies of the Japanese atomic bombing survivors, who generally received radiation exposure to the whole body. In that population, statistically significant excess risks have been shown for leukemia, non-Hodgkin lymphoma (males only), total solid cancer, and cancers of the oral cavity, esophagus, stomach, colon, liver, lung, skin (nonmelanoma), female breast, ovary, bladder, brain, and thyroid. These results are broadly confirmed by other studies (UNSCEAR, 2006, Annex A). For most other sites data suggest possible positive associations; however, a larger number of cases is needed to reach firm conclusions. The highest relative risks (RR; shown as the estimated RR at a 1 Sv dose at age 70 after exposure at age 30) in the atomic bombing survivors study were: leukemia (RR = 5.3), urinary bladder (RR = 2.2), female breast (RR = 1.87), lung (RR = 1.81), brain and central nervous system (RR = 1.62), ovary (RR = 1.61), thyroid (RR = 1.57), and colon (RR = 1.54) (Preston et al., 2007). For comparison, the risk estimate for total solid cancers was RR = 1.47 (90% confidence interval [CI]: 1.40, 1.54).

Two sites were notable for the fact that relative risk after exposure in childhood was much larger than that associated with exposure at age 30, namely, thyroid cancer (exposure at age 10 and age 30, RRs = 2.21: 1.57), and nonmelanoma skin cancer at high doses (greater than 1 Gy) (RRs = 3.28: 1.17) (Preston et al., 2007). Leukemia also showed a higher risk for those exposed in childhood, although the exact excess risk is difficult to estimate because of the complex temporal patterns of risk (Richardson et al., 2009) demonstrated in Figure 4.1. More specifically, excess risk for leukemia varies from >50-fold 5-10 years after exposure, to only roughly twofold by 30 years after exposure; therefore, an average estimate would not correspond to the estimate in various time periods.

An epidemiologic investigation of cancer risks due to radiation exposure is complicated by the lack of diagnostic tests, clinical or molecular, that can determine the cause of cancer in an individual. For this reason, it is important to collect, where possible, information on other risk factors

[6]The studies discussed in this report focus on first cancers only. Second primary and multiple primary cancers, that is, those cancers occurring in patients who were diagnosed with another cancer in the past, are not considered. A second primary is different from a cancer that reappears after treatment (recurrence) or is a result of the original cancer metastasizing to a nonadjacent organ. Recording of multiple cancers in cancer registries is discussed in Section 4.3.2.

FIGURE 4.1 Predicted excess relative risk (ERR) (see Appendix A, Sidebar A.1 for definition) at 1 Gy for leukemia (all types) as a function of age at exposure and time since exposure. SOURCE: Richardson et al. (2009).

linked with the cancer type in question so that investigators can exclude other possible reasons for the disease to have occurred. For some cancers, established risk factors can explain the majority of the observed cases. This is true for lung cancer as smoking causes 90 percent of the lung cancer cases. Given the strong smoking effect, analyzing lung cancer data in relation to low-dose radiation exposure would be fraught with potential problems that would be difficult or impossible to address without accurate historical smoking data for individuals in the study population. For other cancers, however, such as those of childhood, established risk factors that include specific genetic syndromes, prenatal exposure to ionizing radiation, infections, and demographic characteristics such as race/ethnicity, gender, and high birth weight collectively can explain only a small fraction of cases.

4.2.1.4 Defining Exposure: Lessons Learned from Past Radiation Epidemiologic Studies

With the possible exception of purely spatial or purely temporal cluster studies, all environmental epidemiologic studies require some assessment

of "exposure" to individuals or groups. This exposure is hypothetical and is used in a general sense (rather than specifically defined by radiation quantity) and could include simply categorizing study subjects into levels based on exposure surrogates as defined below. For studies of cancer in populations near nuclear facilities, there are many different options for exposure classification, ranging from simple proximity of residence at time of diagnosis to the facility to modeled dispersion of reported releases, but "exposure" in such studies has never included detailed personal measurement of radiation for every individual (as it does in occupational radiation monitoring). For details on the studies discussed here, see Appendix A.

Table 4.2 lists several definitions of exposure in the literature of radiation epidemiology on health risks of populations living near nuclear facilities. Using examples, the definitions are ranked from a less-defined to a better-defined characterization of exposure. The particular type of exposure used in the design and associated analysis defines the question(s) under study and provides an essential context for interpreting the results of any epidemiologic study. It is obvious that a study with well-defined, accurate exposure data can contribute the most to our understanding of the cancer-associated effects of radiation in the setting examined.

The national study conducted by the National Cancer Institute (NCI) and published in 1990 (Jablon et al., 1990; 1991) defined exposure as living in a county in which nuclear facilities are located. This definition is loose because—as pointed out by the investigators—many counties, especially in the West, are large and some are more than 80 km (50 miles) in diameter. For example, the San Onofre plant in San Diego County is located about 60 km (40 miles) from San Diego center. If there was indeed a risk associated with living near the San Onofre plant but the risk is limited to persons living in close proximity to the plant (say, 5 km), the effect would be impossible to detect in a county-based study. This is because the normal cancer rates

TABLE 4.2 Definition of Exposure in Selected Epidemiologic Studies

Definition of Exposure in Relation to the Nuclear Facility	Example
Countywide geographic area	(Jablon et al., 1991)
Zones of increasing distance	(White-Koning et al., 2004)
Zones of increasing distance, and continuous	(Kaatsch et al., 2008)
Distance-based theoretical exposure scores	(Bithell et al., 2008)
Zones of increasing distance, adjusted for dispersal directions of airborne emissions	(Spycher et al., 2011)
Zones based on models of dispersion of gaseous discharges	(Evrard et al., 2006)
Zones of increasing effective dose	(Nuclear Safety Council and the Carlos III Institute of Health, 2009)

in the large distant population in San Diego city would dominate the summary statistics for the count and dilute any local effect that might be there (Jablon et al., 1990).

An improvement to the 1990 NCI approach is that used in a study in France. Established zones of 20-km radius centered on the nuclear facilities, further subdivided into 0-5, 5-10, 10-15, and 15-20 km zones were used for analysis of cancer incidence in populations residing near the facilities (White-Koning et al., 2004). The German *Kinderkrebs in der Umgebung von Kernkraftwerken* (KiKK) study used distance of the family's place of residence from the chimney of the nearest nuclear power plant to define exposure. The distance measurements were established with a precision of about 25 m, although the investigators primarily used and highlighted a distance of ≤5 km for analysis (Kaatsch et al., 2008). An isotropic distribution of discharges was assumed (i.e., circular rings of equal exposure around the plant); a more accurate method would model releases according to local topography, wind direction, and precipitation.

More graduated rank-order measures of closeness were employed in a British study, using the distance of centroids of census wards from nuclear power plants to define several different types of distance scores as continuous exposure variables. No associations were observed to suggest increasing risk in relation to closer proximity to the plants (Bithell et al., 2008). A recent study in Switzerland (Spycher et al., 2011) also used distance of the family's place of residence (current or at birth of the index child) to the nearest nuclear power plant as a measure of exposure. Although no doses were actually estimated, an analysis was performed accounting for main dispersal directions of airborne emissions from the nuclear power plants. For this analysis, investigators redefined the exposure as living in a zone around a nuclear power plant that is equivalent in area to a circle with 5-km radius but extends to a distance proportional to the average duration of slow winds (<3 m/s) in a given direction (Spycher et al., 2011). Downwind concentration of radioactive particles has been found to be inversely correlated to wind speed.

Evrard et al. (2006) conducted a study using geographic zoning based on doses to the bone marrow estimated due to gaseous radioactive discharges using radionuclide discharge data, local climate data, and a mathematical model of nuclide transfers in the environment. The model was developed by the National Institute of Radiological Protection and Nuclear Safety in France (Morin and Backe, 2002). This ecologic study examined communes (small administrative divisions) located within a 40-km circle around the nuclear facilities in France. The communes were divided into five categories based on the estimated dose. The investigators noted that the categories defined by dose assessments differed from those defined by concentric circles around the facilities due to topographic and meteorologi-

cal characteristics. Although the estimated doses and distances were significantly and inversely correlated (Spearman's rank correlation coefficient $r = -0.58$, $p = 10^{-4}$), marked variability in the estimated dose within each concentric band remained. The contrast in the mean dose between the lowest and highest dose-based categories (range: 2.11 mSv/yr; ratio: 106) was much larger than the maximum contrast between the concentric bands 0-5 and 15-20 km (range: 1.16 mSv/yr; ratio: 30) (Evrard et al., 2006). This suggests that dose precision and probably statistical power are lost by using only crude distance-based surrogates for exposure levels.

The same model to estimate bone marrow doses associated with gaseous discharges from nuclear power plants was used in the recent investigation. This investigation further considered the risks around nuclear power plants in France and included a case-control analysis which had an ecologic element (Sermage-Faure et al., 2012): cases and controls were assigned a single exposure value estimated at the town hall of the commune of residence.

A study in Spain performed historical reconstruction of the exposure of the population in municipalities within a 30-km zone from the nuclear facilities or 50-100 km from the facilities as a result of the discharges of liquid and gaseous effluents from the facilities (Nuclear Safety Council and the Carlos III Institute of Health, 2009). Estimated effective dose of the populations of municipalities were reported. The investigators state that upon consultation with the International Commission on Radiological Protection, use of effective dose as an indicator of exposure (created for protective purposes and not for estimation of risk) instead of absorbed doses in individual organs and tissues was deemed acceptable for the epidemiologic study, provided that the uncertainties and limitations involved were clearly stated.

As demonstrated above, studies of cancer risks near nuclear facilities use differing estimates of exposure and commonly suffer from several weaknesses by not accounting for:

1. Prevailing wind directions and speeds or terrain factors, which may appreciably alter exposures to gaseous effluents.
2. Directionality and distance of exposures resulting from liquid effluents, the pathways for which may be narrowly focused geographically.
3. Differences in historic release levels of nuclear facilities, when the pure proximity approach is used and multiple sites are examined.
4. Temporal cumulative exposures or increases in nuclear facility–associated disease risks as the cumulative exposure increases.
5. Temporal and spatial variations in natural background radiation in the vicinity of each site as well as from site to site.

In principle, the pure proximity approaches of any study can be improved by incorporating dosimetry information into the risk analyses. Comparison of the study findings regarding the risks in a population using a pure proximity approach to those from an analysis that incorporates reconstruction of the doses received by the same population can prove informative. An example is the recent study in France that showed that children living within 5 km of nuclear plants are twice as likely to develop leukemia compared to those living farther away from the plants. However, analysis of the same population of children using a dose-based geographic zoning approach, instead of distance, did not support the findings. The absence of an association with the dose-based geographic zoning approach may indicate that the observed association of distance and cancer risk may be due to factors other than the releases from the nuclear power plants (Sermage-Faure et al., 2012).

4.2.1.5 Dosimetry Models for a Geographic Unit or Individuals

Dosimetry models for a geographic unit apply to ecologic studies, where an average exposure is assigned to a population residing in an area (for example, census tract) and every individual in that area is assumed to have experienced this exposure; typically, the smaller the geographic unit the less heterogeneity in exposure per individual, and the more precise the estimated exposure of the populations within that unit. Dosimetry information that takes into account the magnitude and temporal variations of annual releases and the factors that provide directionality and distance variations to those releases provide more accurate estimations of exposure. Operationally, for each geographic unit, an areal centroid can be calculated using Geographic Information Systems (GIS), and the estimated annual organ doses to representative individuals at that centroid point can be calculated. Either the population-weighted centroid or the geographic centroid can be used, depending on whether or not investigators want to adjust for a heterogeneous distribution of people within a given census area. One could use those imputed values in dose-response analyses of health outcomes, including appropriate summations of cumulative radiation dose specific to time, lag times, and age truncation.

The same methodology could be used to estimate the doses received by the individuals in a record-linkage-based case-control or cohort study. This implies that each individual is assigned the calculated dose for the census tract within which he or she resides. This leads to loss of statistical power compared to a study in which individual doses are assigned since variability in true dose is underestimated.

It is preferred, when possible, to calculate individual doses based on residential address at the time when exposure is likely to be most relevant,

such as residence at time of birth for the cases and controls. Calculating individual doses based on the address where the person lived at time of cancer diagnosis may also be relevant to where the person may have lived at time of exposure and likely more relevant than calculating doses based on residence at time of death. An analysis based on residence at time of death is the most likely to be affected by migration bias.

Individual dose reconstruction for members of a large case-control or cohort study could be time consuming, especially when the investigator wants to incorporate information on residential history of each individual if this is available through interviews or questionnaires. Information on the approaches for modeling dosimetry data in geographic units is described in detail in Chapter 3.

4.2.1.6 Statistical Power

Statistical power is the probability that a study of a specified size and design can detect a predetermined difference in risk in the absence of significant bias, when such a difference actually exists. While the computations can be complex, the concept is simple; higher power to detect effects is better, and if power is too low, a study is unlikely to find a difference of interest even when it actually exists, meaning the study can be shown to be uninformative before it starts and perhaps is not worth undertaking. Thus, a fundamental issue regarding the estimation of risks from low-dose studies is statistical in nature.

The sample size required to detect a significant association between dose and an effect is a function of the inverse variance of the dose distribution. In general, as the variance of the distribution of doses increases, the required sample size to detect a particular effect decreases proportionately. This implies that the required sample size (for the exposed group) varies approximately as the inverse of the square of the expected effect size (i.e., $N = k / (\text{Effect size})^2$, where k is some constant).

To illustrate this, consider the simple case where there is an exposed group, all with approximately the same degree of exposure, and a very large unexposed group for comparison, and one wished to determine whether there was a difference between the groups in the rate of colon cancer. In this case, variation in the sample size requirements in proportion to the inverse variance of the dose distribution implies that the needed sample size to achieve adequate statistical power (80 percent power is usually taken as adequate statistical power) to see a difference between the two groups varies approximately as the inverse square of the mean dose in the exposed group if the dose-response association is linear. For a hypothetical example, suppose the association between radiation dose and colon cancer risk is linear, and observation of 500 exposed persons for a given period of

time compared to a very large unexposed group is needed to have adequate statistical power to detect a radiation-associated colon cancer risk when the mean dose is 0.5 Sv. In the analogue of that scenario, 100 times as many (i.e., 50,000) exposed persons would be required to detect a risk if the mean dose were instead one-tenth as large (i.e., 0.05 Sv), and 5,000,000 exposed persons would be needed if the mean dose were 0.005 Sv. This is graphically illustrated in Figure 4.2, where dose (mGy) versus the required sample size is plotted (Brenner et al., 2003). For doses equivalent to those received by individuals that live near a nuclear power plant in the United States which are estimated to be <0.01 mSv/yr (USEPA, 2007) the numbers of exposed persons required to find a possible association would be truly enormous.

Having a range of doses tends to increase the dose variance, so a dose-response analysis would probably have somewhat better statistical power than the simple two-group comparison; but given the typically high correlation between the dose variance and the mean dose in the exposed group, the "inverse square of mean dose" relationship is still a rough rule of thumb that is easier to ascertain and conceptualize than the size of the dose variance.

Instead of statistical power to detect an effect, an investigator may want to set bounds on the magnitude of risk. In that case, two different purposes need to be distinguished:

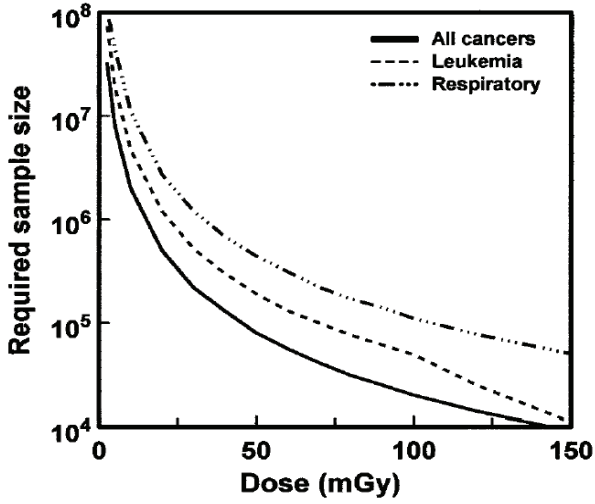

FIGURE 4.2 Size of a cohort exposed to different radiation doses, which would be required to detect a statistically significant increase in cancer mortality in that cohort, assuming lifetime follow-up. SOURCE: Brenner et al. (2003).

1. If the interest is to establish narrow bounds (i.e., narrow confidence intervals) on the magnitude of risk per unit dose, then a principle similar to that for mean dose and statistical power would apply—namely, a much larger sample size would be required to achieve a given tightness of the bounds on risk per unit dose when the doses are smaller.
2. If the interest instead is to "rule out" a certain magnitude of risk (for example, a 20 percent increase in risk in the exposed group) without reference to their estimated dose levels, then sample size calculations associated with finding a detectable risk per unit dose do not apply. Instead, the calculations involve an estimation of likely confidence bounds given the sample size and anticipated number of cases of the disease (Satten and Kupper, 1990). The latter is usually determined using available disease rates.

This second purpose, that is, to "rule out" a certain magnitude of risk, is how the committee based its power calculations. The committee's aim was to establish the minimum sample size required so that the investigation is reasonably likely to detect an effect of a given magnitude. A 20 percent increase in risk was used as a rough figure that would raise the level of concern in statistical terms (but other alternative scenarios of higher risks are also considered). Similarly, power calculations can be used to calculate the minimum magnitude of the change of risk that can be detected given a particular sample size.

To reiterate, calculations of required sample sizes based on current knowledge of the average population exposure of the people in the United States to radiation from the nuclear industry would lead to a small anticipated increase in risk that would require an enormous population size to detect with statistical precision. Even for leukemia, which is considered the most radiosensitive cancer, the expected increase in risk is small. The committee discussed that in the atomic bomb study the relative risk for leukemia was 5.3/Sv dose at age 70 after exposure at age 30. This means that the excess relative risk for leukemia is 4.3/Sv, which is equated to 1.43/100 mSv, 0.143/10mSv, or 0.0143 for 1 mSv. Therefore, the estimate of excess risk that one would be trying to detect in relation to exposures from nuclear facilities would be on the order of 0.000143 or smaller. Such a risk would be virtually impossible to detect for any cancer given the statistical and other variability on the baseline risk. As a result, precise computations of statistical power based on risks due to the expected doses would have little meaning; therefore, computations of statistical power are focused on the population sizes required to "rule out" larger risks. Arguably, the power calculations presented here are based on risks tied to exposures that are on the order of 0.5-1.0 Sv, which are much higher than those expected from the releases of nuclear facilities.

On the basis of demographic parameters specified by the committee (U.S. population in 2010 of approximately 300 million, about 15 percent live within 50 km [approximately 30 miles] and 0.3 percent live within 8 km [approximately 5 miles] of a nuclear facility, about 20 percent are children under 15 years of age), the committee calculated the power of several possible scenarios that apply to different study designs using distance from a site as a surrogate exposure measure. The choices of 8- and 50-km comparison zones are used solely to provide a frame of reference for the sample sizes required for adequate performance of an epidemiologic study. These reference scenarios are in general agreement with some published studies (see Table A.2), although often the "at-risk zone" in many of these studies is designed to be slightly closer to the facility (for example, 5 km). As described later in this section a gradient type of analysis rather than an analysis based on two categories is preferred.

The scenarios explored are the following: a case-control study with equal number of cases and matched controls (1:1 matching plan), a case-control study with 5 controls per case (1:5), and a case-control study with 100 controls per case (1:100). The latter could approximate the matching ratio of cases and controls of a large cohort study or an ecologic study; as is generally true for rare diseases, far more controls are available than cases in these two study designs.

For purposes of this discussion, risk estimations for the different scenarios are presented as relative risks (RR). The odds ratio (OR) calculated for case-control studies (see Sidebar A.1 in Appendix A) approximates the RR from a cohort study when rare diseases are examined. Reporting power calculations based on RR provides a more conservative assessment of power.

In these comparisons, the committee made several simplifying assumptions about the relationship between exposure and distance. The committee assumes that:

a. Distance to the nearest facility is classified into just two categories, for example, living within the 8-km zone (nearest category/exposed) versus living within the 8-50-km zone (farthest and larger category/unexposed) from the nuclear facility.
b. Two and one half percent of the population under study is in the exposed category and 97.5 percent in the unexposed category.
c. Risk in the exposed category is equal to RR × (baseline risk), where RR is relative risk due to being close to the nuclear facility and baseline risk is the risk in the unexposed category.
d. National rates provide the rates of cancer for the unexposed population in the regions under study.
e. Distribution of risk factors other than the exposure of interest is nondifferential between the two categories.

These assumptions need to be refined if a study is in fact undertaken.

Figure 4.3 plots detectable RR as a function of total number, *n*, of cases for each of the three matching scenarios (1:1, 1:5, 1:100). Detectable RR is defined to be the ratio of risk in the exposed category compared to the unexposed category, for which a study with a given number of cases, *n*, will have 80 percent power (usually taken as adequate statistical power) to detect the increase at the 5 percent level of significance (one-sided test; see Sidebar A.1 in Appendix A for definition).

The detection of RRs that are equal to 1.2 (a 20 percent increase in risk in the 2.5 percent of the study population nearest a facility) with acceptable power (80 percent power) requires that 7,000 to 14,000 cases be recruited (depending on the matching scenario). A 40 percent risk increase can be detected with about 3,800 cases for a 1:1 case-control study and about 1,800 with a case control or a cohort and ecologic study designs of 1:100 matching. Doubling of risk (RR = 2) can be detected with approximately 765 cases and controls for a 1:1 matched case-control study and with about

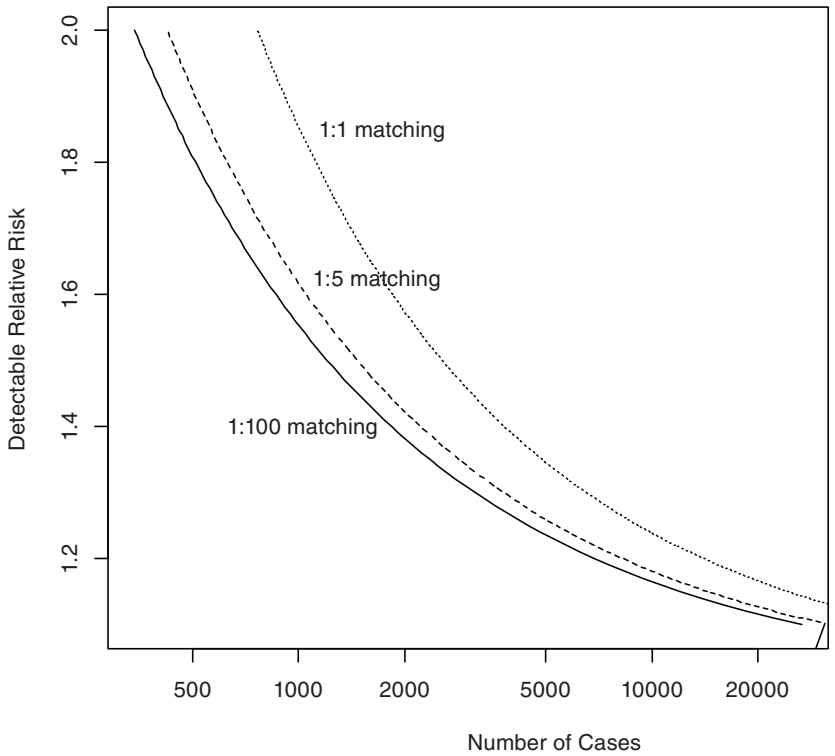

FIGURE 4.3 Detectable relative risk for a case-control study with 2.5 percent of subjects exposed.

345 cases with a case control or a cohort and ecologic study designs of 1:100 matching (see Table 4.3 for summary).

For rare cancers such as childhood leukemia where the observed number of exposed cases will be relatively small, multiple controls (for example, 5 per case) would help to increase the power of the study. However, the improvements diminish rapidly as the number of controls per case increases, so that 5 compared to 100 controls per case do not increase substantially the power to detect an increase in risk (see Figure 4.2).

Another consideration for the design of the study is the number of years of study needed to accrue enough exposed cases so that the study achieves 80 percent power to detect a 20 percent increase in risk of childhood leukemia among the "exposed." From Figure 4.3, a 1:1 matched case-control study would require about 14,000 cases within the overall study zone in order to have power to detect a 20 percent increase in risk. There are approximately 3,000 childhood acute lymphoblastic leukemia cases diagnosed per year in the entire United States (http://www.cancer.gov/cancertopics/pdq/treatment/childALL/HealthProfessional), 15 percent of which (450) would be in the study zone (50 km from a nuclear facility). Therefore, it would require 31 years of accrual before a study would reach acceptable power. Increasing the number of controls from 1:1 to 1:100 (as in a cohort or an ecologic study) would reduce the needed number of cases to roughly 18 years of accrual. Of course more extreme risks are detectable with much less study accrual time. For example, a doubling of risk could be detected with 350-765 cases or about <1 to 1.7 years of accrual for the 1:100 to 1:1 matched studies. A 40 percent increase in risk could be detected with 4 to 8 years of accrual for the 1:100 to 1:1 matched studies (see Table 4.3 for summary).

For most adult cancers the period of accrual required to detect relative-risk increases of these magnitudes is much shorter because of the higher prevalence of disease and the larger population numbers. For example, for

TABLE 4.3 Approximate Number of Total Cases and Years of Study Follow-Up

	1:1 Matching			1:5 Matching			1:100 Matching		
RR	Cases	$Years_L$	$Years_B$	Cases	$Years_L$	$Years_B$	Cases	$Years_L$	$Years_B$
1.2	14,000	31	2	8,200	18	1	6,900	15	1
1.4	3,800	8	<1	2,200	5	<1	1,800	4	<1
2	765	1.7	<1	425	<1	<1	345	<1	<1

NOTES: 1:1, 1:5, and 1:100 indicate matching scenarios of cases and controls; RR, relative risk; cases, total number of cases (any cancer) in the area under study to detect the indicated RR; $years_L$, $years_B$, years of study coverage required to accrue childhood leukemia ($years_L$) and breast cancer cases in women under 50 ($years_B$) in order to detect the indicated RR.

breast cancer in women under 50 years of age the national rates are approximately 43/100,000 person-years or about 40,000 women diagnosed per year. Since approximately 15 percent of these women (6,000) are expected to live within 50 km of nuclear facilities this means that it would take around 1-2 years of follow-up to detect an excess risk of 20 percent for this cancer, under the same assumptions as above.

The total number of cases and years of follow-up required for the different matching scenarios to detect a range of increases in risk following the assumptions stated above are summarized in Table 4.3.

The sample size computations provided here are the bare minimum of data to test the hypotheses at the specified level; thus, a sample size estimate is generally a lower bound on what will be needed, and actual requirements could be much larger. This is because the power calculations presented here are based on simplified models that ignore the effect of other risk factors that are largely unknown at the design stage. Internal pilot data are often used to better inform the power calculations and more reliably estimate the required sample size. Pilot data can account for the patterns of risk factors and potential confounders (if information is available) and the nature of confounding—whether it is positively or negatively associated with the exposure. Power calculations that have not accounted for the effects of risk factors may under- or overestimate the required sample size.

Modest improvements in the statistical power can be achieved by examining dose-response gradients, especially when the population under study is exposed to a range of doses (Shore et al., 1992). However, since the mean doses received by the populations near nuclear facilities are expected to be low and the associated risks, if any, are expected to be small, very large numbers of cases and controls would still be required in order for the study to be informative and useful. If the study intends to examine dose-effect relationships, improving the quality of the dosimetry can also afford gains in statistical power. Imprecise estimation of doses can be a source of error that increases the uncertainty in the estimated association, which tends to flatten the dose response and decrease the likelihood of finding a statistically significant association.

One way to improve statistical power is to increase the effective sample size. As the time since onset of exposures increases, the follow-up number of the exposed populations increases and the exposed population becomes older. Both of those serve to increase the statistical power to observe potentially elevated risks, the latter because much of a population's cancer risk is expressed at older ages as the disease rates increase. An additional method to increase sample size is to pool data across numerous studies or study sites. Bias, on the other hand, is not reduced by simply increasing sample size in the absence of other improvements; if larger samples mean that less attention can be given to quality of the individual observations, bias may even increase with sample size.

Another way to achieve a more statistically powerful study is to focus on radiation-sensitive end points, that is, those that have shown the largest association with radiation. Leukemia (except for chronic lymphocytic leukemia) has shown the highest radiation relative risks per unit dose of any malignancy in a number of studies, so it is a natural target for study. Other endpoints that show relatively high radiation relative risks are breast cancer in younger women, thyroid cancer in children, and bladder cancer. In mounting a study with an exposed group of a certain size, however, there may be a trade-off between the size of the relative risk and the baseline frequency of the disease in question. If a disease is very rare, even with a high relative risk there may not be enough disease cases to demonstrate an association. On the other hand, with a common disease a relatively low elevation in relative risk may be sufficient for statistical significance.

Another strategy to increase statistical power is to concentrate on a "sensitive" subgroup of the population, that is, a subgroup for whom any radiation-associated relative risk may be appreciably higher than for the population as a whole. Efforts are ongoing to try to identify genetically susceptible subgroups of the population and—not surprisingly—research indicates that the DNA repair and cell cycle control pathways may play an important role. To date, however, either the genetic variants are too rare to be studied separately (e.g., in the *BRCA1* and *BRCA2* genes; women carriers of mutations in these genes are at high risk of developing breast cancer) or to have much impact in general-population studies (Bernstein et al., 2010), or the susceptibility variants show only small elevations in risk and frequently are not replicable. A recent study that examined a set of genetic variants (haplotype approach), as opposed to each variant separately, showed that the risk of acute lymphoblastic leukemia associated with diagnostic irradiation is modified by variants in DNA repair genes (Chokkalingam et al., 2011). The WECARE[7] study is examining the interaction between radiation exposure and genetic susceptibility in the etiology of second breast cancer in women with radiation treatment for an initial breast cancer. For genetic sensitivity variables, thus far mostly only rather rare mutations have shown an appreciably heightened radiation effect, which means the number with such mutations among cancer cases nearby to nuclear sites would be very small and not promising for a study (Bernstein et al., 2010; Malone et al., 2010).

One sensitive subgroup clearly needs to be considered. A substantial amount of data supports the concept of greater radiation cancer risks after exposure in childhood than after exposure in adulthood. For example, the Japanese atomic bombing survivors data suggest this age differential for cancer mortality or incidence for total solid cancer, leukemia, and cancers of the stomach, breast, colon, bladder, thyroid, skin (nonmelanoma), and a

[7]Women's Environment, Cancer, and Radiation Epidemiology.

combined miscellany of other sites (Preston et al., 2003, 2007; Richardson et al., 2009). For total solid cancer and a number of the individual sites, the radiation relative risks are roughly 1.5 to 2 times greater for childhood exposures than adult exposures. For leukemia, thyroid cancer, and breast cancer the ratios of relative risks by age at exposure are even larger. In contrast to an investigation that focuses on exposure of genetically susceptible individuals, a study on childhood exposure would affect a significant proportion of the potential study population and therefore has good potential for a study (or for a focus within a broader study).

Since the risk of leukemia after radiation exposure at young ages is so pronounced for the first 15-20 years after exposure (Figure 4.1) (Richardson et al., 2009), a study focusing on those with potential exposure who develop leukemia at an early age (e.g., before age 15) might be a relatively powerful study if the doses are high enough. The 0-14 age group has been the target age group for many international studies (see Table A.2, Appendix A).

4.2.1.7 The Multiple Comparison Problem

The design of an epidemiologic study of cancer risks around nuclear facilities may include one or few a priori hypotheses to be tested. For example, an epidemiologic hypothesis may be that cancer (all types together or a specific type) occurs more often in populations that live near nuclear facilities than in populations that live further away. Stating the hypothesis precisely, with the method that will be used to test it, is important not only for the collection of the appropriate information, but also because standard statistical techniques require that each tested hypothesis be prespecified; otherwise statistical measures such as p values and confidence intervals lose much of their scientific meaning and become hard to interpret. Statistical issues aside, asking "Does this study yield *any* associations?" is a poor research strategy (Savitz and Olshan, 1995).

If a study has low statistical power and only a small number of disease outcomes is examined (i.e., only a small number of a priori statistical tests is performed), then null (negative) results would be the most likely outcome of those statistical tests. However, when a considerable number of different disease outcomes will be examined, the potential for one or more false-positive results (purely by chance) can become large. If two sets of statistically independent observations are available, each is testing a true null hypothesis, and each is tested at the usual 5 percent level, the probability that the first will be found significant is 5 percent and the same for the second. The probability that at least one will be significant by chance is $(1 - 0.95 \times 0.95) \times 100 = 9.75$ percent, almost twice the probability for either test alone. The probability increases further if there are more than two

hypotheses. For instance, for independent disease outcomes the probabilities of at least one false-positive result when 10, 20, or 30 outcomes are examined are about 40, 64, and 79 percent, respectively, while the respective probabilities of at least two false-positive results are 9, 26, and 45 percent.

In other words, the probability of one of many prior hypotheses yielding false-positive results increases with the number of hypotheses tested. Furthermore, when investigators also examine risks in various subsets of the data (e.g., dose, time, or age subgroups), this also will tend to increase the probability of false-positive findings, especially if particular subsets are chosen because of preliminary inspection of the data to identify "suspected differences."

With a substantially underpowered study, any "positive" finding usually has two characteristics. First, it is likely to be a false-positive finding. Second, it is likely that the risk estimate associated with that positive finding is a large overestimate of the "true" degree of risk (Land, 1980). This can be understood intuitively with a hypothetical, but possible, example. Suppose that, given the mean dose in some underpowered low-dose study, the expected true RRs for a series of health outcomes were about 1.1. However, because of the sample size, the RR would have to be about 2.0 to be likely to be detected as statistically significant. Due to sampling variability, by chance one out of the number of health outcomes might show a "statistically significant" RR of 2.0. The excess for the RR of 2.0 is on the order of 10 times larger than the true excess (that is by chance, an excess of 100 percent when the "true" excess is about 10 percent). In short, "statistically significant" results in low-dose studies where the true risk is small tend to provide falsely exaggerated estimates of risk. Accompanying that is often the common human tendency to focus on the "statistically significant" risks, which means that the false-positive results with large imputed risks get undue attention.

The multiple comparison issue would be particularly limiting in the interpretation of the results of an ecologic study in which multiple cancers are examined for individual facilities as well as combinations of facilities, different time periods, and different age groups. Positive associations found by chance are likely to be misinterpreted. In the 1990 NCI study, for example (Jablon et al., 1990, 1991), 3,090 comparisons were made for leukemia after startup of a nuclear facility for different areas and age groups. Nineteen were expected to have a probability below 0.05 by chance alone; the actual number observed was 18.

Statistical scientists have various ways of dealing with the multiple comparison problem. One strategy that is sometimes employed to guard against excessive false-positive (i.e., "chance") outcomes is to use a more stringent level for declaring that some difference is statistically significant. Two such commonly used procedures are the Bonferroni multiple com-

parison correction and the Benjamini and Hochberg method. However, increasing the stringency for declaring a statistical test as positive has the downside of decreasing the statistical power to detect a real effect. Another way is to examine the number of significant results and look for patterns in them (such as increases in cancer only around a certain type of facility, or in one type of cancer around a number of facilities). A third way is to reexamine the results of the significant tests, perhaps in light of additional data, to see whether there is reason to suspect a real effect. For example, was there a radionuclide released that tends to be carcinogenic to a certain organ, as in the case of radiostrontium and bone cancer? Is the association consistent with other studies of radiation effects and biological plausibility? For example, is an association for female breast cancer more plausible than one for male prostate cancer? None of these, applied in a mechanical fashion provide a sure procedure to distinguish real effects from chance (false-positive) associations, and in the end scientific judgment has to be applied based on such considerations as strength of the study methodology, ability to rule out biases and confounding, and biological plausibility.

4.2.1.8 Confounding

Confounding refers to an apparent change in the magnitude of the association between the exposure (e.g., radiation) and some outcome (e.g., lung cancer) that comes about because of associations with a third, "confounding" variable. Confounding variables might be exposures to toxic or preventive agents, lifestyle or dietary variables, or other disease risk factors. An important statistical concept regarding confounding is that the degree of confounding of the exposure-outcome association depends on the degrees of association of the potential confounder variable with both the exposure and the outcome, as well as the strength of the exposure-outcome relationship.

The term "confounding" is frequently used without careful consideration of the true definition to describe the differential distribution of characteristics of the groups under study (for example, between cases and controls, exposed and unexposed). So, for example, if there is an empirical association between the potential confounder and the outcome, but no association between the potential confounder and the exposure, there will be no confounding. Likewise, an association of the potential confounder with the exposure but not with the outcome will mean there is no confounding. (In actual studies it is typically not an all-or-none situation, but a matter of degree, depending on the magnitude of correlations of the confounder variable with the exposure and outcome variables.)

Issues of confounding are important in all epidemiologic studies with no exception, and they are particularly important in low-dose radiation

studies that examine rare diseases, as even a small degree of confounding can distort the study results substantially and produce incorrect results. An observed small relative risk such as 1.2 (a 20 percent increase in risk) is more likely to be a result of methodological flaws than a relative risk of 5 (fivefold increase in risk). Confounding can create erroneous risk estimates that either exaggerate or nullify the true degree of association. Studies of health effects associated with high levels of radiation exposure usually are not affected by major confounders, because confounding by other exposures or risk factors tends to be considerably smaller than the radiation effects in question. However, with low-dose studies in which the size of the radiation effect is expected to be small, the magnitude of potential confounding effects may be as large, or larger than the size of the radiation effect. In that circumstance, there is a potential for a substantial degree of confounding of the exposure effect. Insofar as studies do not have information with which to evaluate particular variables that might be confounders, potential confounding is a source of uncertainty that can make low-dose study effects difficult to interpret. When information on the potential confounders is available, adjustment[8] for them can be made in the statistical analysis to help remove their effects.

Smoking is an example of a serious possible confounder for lung cancer because of the very strong causal relationship between smoking and lung cancer. (Smoking can also be a confounder for other cancers such as bladder cancer.) Small differences in smoking habits can have a greater influence on lung cancer risks than do differences in exposure to low levels of radiation; the relative risk of lung cancer associated with cigarette smoking for moderate to heavy smokers generally exceeds 10, while the RR associated with exposure to high doses of radiation rarely exceeds 2 (Pierce et al., 2005). Therefore, collecting detailed information on the individuals' smoking history (number of cigarettes smoked per day, age of smoking initiation, years of smoking) is crucial as even slight variations in smoking patterns can bias the results. If the information is not available, it is almost impossible to determine that radiation exposure increases one's risk of developing lung cancer even if data suggest that.

An ecologic study that uses aggregate health survey data on smoking is not expected to provide adequate adjustment for potential confounding by smoking because it is unable to capture specific smoking patterns or the complicated interactions between smoking and socioeconomic factors. This inability of ecologic studies to properly adjust for confounding often leads to hesitation of the scientific community to embrace results and outcomes of these studies. An example already discussed is the large county-based

[8]Statistical procedure used to minimize the effect of differences in the composition of the populations or individuals compared.

ecologic study in which a decrease in lung cancer mortality was observed in association with increased radon exposure in sharp contrast to the increase expected from current knowledge (Cohen, 1995, 1997). Subsequent investigators who reviewed the data were skeptical as to whether confounding by smoking was properly adjusted for (Heath et al., 2004; Pawel et al., 2005). Indeed, a series of studies using estimated individuals' radon exposure have shown positive associations (Darby et al., 2005).

If the likely confounders have been measured in the study, one way to control for confounding in the design stage is to match[9] on one or more factors about which the investigator is concerned that would distort or confound the relationship between exposure and disease under study. Matching has been defined as "the process of making a study group and a comparison group comparable with respect to extraneous factors" (Last, 1995). This way, there will be identical confounder distributions among cases and controls or exposed and unexposed groups. Matching is more often used in case-control than in cohort studies and can occur at the level of the group and is then called *group* or *frequency* matching or at the individual level and is called *individual* or *paired* matching.

Although matching for factors may appear to be a tempting way of controlling confounding, adjusting for confounders inappropriately can result in "overmatching." Overmatching can occur when investigators match for a variable that is correlated with the exposure of interest or is connected with the mechanism whereby that exposure affects the disease under study.

If the confounding factors have not been measured, the data may be misleading and findings need to be interpreted with caution. If a confounder is measured imperfectly due to missing information, classification of the confounder is too broad, or the confounder is misclassified, confounding may still exist, and it is termed residual confounding.

4.2.1.9 Uncertainties

A valuable strength of an epidemiologic investigation of cancer risks that incorporates dose reconstruction stems from the fact that the population of interest is examined directly for cancer occurrence or death from cancer; no extrapolations are required from other human populations exposed to high doses, or acute doses, or from animal or cell studies that would add various uncertainties in the risk estimations. (The risk projection model described in Section 4.2.2 is not considered to be an epidemiologic investigation.) Still, any of the study designs considered would attempt to demonstrate very small radiation effects, if any, associated with low doses,

[9]There are other methods of controlling for confounding at the design phase such as *restriction*, or at the analysis phase by *standardization*, *stratification*, and *multivariate analysis*.

and would deal with particularly challenging problems related to uncertainty from various sources. These sources are more often discussed in the context of dose estimations (presented in Chapter 3) and include inaccuracy of measurements used to reconstruct radiation doses, lack of knowledge about true values of dosimetric parameters, and inappropriate assumptions in dosimetric models used to calculate radiation doses to the populations under study. Uncertainty related to the epidemiologic study design itself is often discussed in terms of limitations of the design, analysis, and subsequent interpretation of the findings.

Almost any conceivable epidemiologic study must base its analysis on incomplete or imperfect information regarding the population under investigation. Furthermore, some potentially incorrect assumptions, small or large, will be needed, for example, because data are not available or because clarifying the assumptions is not possible. The unknown effects of the necessary assumptions made in analysis contribute to uncertainties in the results. In this section uncertainties are discussed in terms of:

a. Completeness of cancer case ascertainment. Cancer risk estimates are based on disease rates obtained from cancer registries and vital statistics offices. Although well-organized means of assessing the quality of cancer registration are in place, at least for the more recent years (see Section 4.3.2), registration is not 100 percent complete or free of errors such as diagnosis misclassification. However, if the frequency of these errors is not large, and not different in exposed versus unexposed areas, the random misclassification should have little effect on the identification of any increased risk.
b. Population mobility. Inability to retrieve information on residential history and duration of residence at each location is a major source of uncertainty in the epidemiologic investigation of cancer risks near nuclear facilities. In most such studies investigators estimate the exposure of the individuals or the populations based on one time point: place at time of diagnosis, or at time of death (and the equivalent for controls), or at time of birth. The assumption is that the exposures relevant to the disease occurred while living at that location and that individuals remained at the location of exposure for the period of interest. The issue with this assumption is not only that is likely not true, but also that the results of the study are sensitive to the driving forces that cause people to migrate. Social and economic factors (such as education, job opportunities, and housing) often drive migration and also affect disease outcomes. If migration patterns differ between cases and controls (or between exposed and nonexposed), then the results from the study could be biased.

Although it may be possible to quantify the uncertainty introduced by in- or out-migration, exposure from the releases of the nuclear facilities may not be relevant to place of residence but more to place of employment for the adult working population. As an example, take a person that lives 60 km away from a nuclear facility (outside the zone of interest of 50 km that has been discussed in this report) but works 10 km from a nuclear facility or in a nuclear facility. This exposure misclassification is impossible to capture without enquiring detailed information on both residential and employment history through interviews and questionnaires.

A study of young children (for example, 0-14 years of age) is likely the least affected by the issues related to migration and/or place of exposure misclassification. Young children would not only have less opportunity to migrate, but they would also tend to spend more of their time at home compared to adults whose work or other activities may be taking them elsewhere. Additionally, a study of young children where analysis is based on birthplace (rather than place of diagnosis or death and the equivalent for the controls) could capture exposures of the child's early life and exposures of the fetus during pregnancy, two periods during which humans are particularly sensitive to the effects of ionizing radiation (Pierce et al., 1996). This said, studies of young children are not immune to the impact of mobility or exposure misclassification. A surprising number of families move during pregnancy (Fell et al., 2004) and more than 50 percent of children ages 3-6 are enrolled in center-based care (http://www.childstats.gov/americaschildren/famsoc3.asp). Arguably, a study of the cancer risks of populations near nuclear facilities (especially of the older populations) that is based on place of death is more affected by migration bias. There are, however, good reasons to perform combined analyses of mortality and incidence for reasons described in Section 4.2.1.

c. Variability in risk factors. There is inherent variability in the characteristics of the populations in an epidemiologic study that include variability in their genetic make-up, susceptibility to cancer, lifestyle factors, and personal habits. These factors are not easily measurable even if detailed interviews are conducted and/or biological samples are taken. In a low-dose epidemiologic study, the magnitude of the variation in these unmeasured factors may surpass the expected effect from radiation released by the nuclear facilities and therefore obscure any actual effect attributed to the radiation. The variability in population characteristics would not have as profound of an effect in a high-radiation-dose epidemiologic study because the excess risk tends to be greater than most variation in the baseline risk.

d. Inability to distinguish risks from different sources of radiation. Similar to the "noise" on baseline cancer risk that arises from the variability of risk factors such as those discussed above, variability in exposure to other sources of radiation is difficult to measure with accuracy. An increasing source of radiation dose to the population in the United States is from exposure to medical diagnostic procedures, which accounts for almost half of the annual dose that the population receives (NCRP, 2009). In the current context, collecting information on frequency of high-dose procedures such as computed tomography (CT) exams or doses received from these procedures is important as these doses are much higher than those expected to be received from routine operations of the nuclear facilities.[10] In the absence of a national system that tracks population utilization and exposure to medical procedures that involve radiation use, retrieving the information on medical imaging utilization is not possible unless medical charts are reviewed or personal interviews are conducted; then the potential for collection of inaccurate information or recall bias is a concern. As the methods to obtain organ dose are not fully developed yet, calculating the doses to the exposed populations per imaging modality, if possible, would introduce additional uncertainty.
e. Potential confounding. A risk factor such as smoking or exposure to medical diagnostic procedures has to be formally tested to assess whether it is a true confounder or not under specific circumstances. Smoking is of particular interest because as discussed in the previous sections it has the potential to be a serious confounder for lung cancer and other cancers such as bladder cancer. However, it is often not possible to collect accurate and detailed information to fully test for confounding.
f. Synergistic and antagonistic effects with radiation. Collecting information on lifestyle factors and exposure to agents such as toxic substances is also important for the examination of synergistic or antagonistic effects with radiation. A collaborative multicountry study in Europe aimed to determine the risk of lung cancer associated with exposure to radon at home. Results demonstrated that residential exposure to radon among smokers and recent former smokers increased the risk of lung cancer compared to individuals who did not smoke currently or in the near past (Darby et al.,

[10]Radiation doses are much higher during radiation therapy, often on the order of 5,000 to 50,000 times as large (NCRP, 2009), but only a small fraction of the population undergoes radiation therapy, primarily as part of a cancer treatment plan. As discussed in Section 4.2.1 only the first primary cancers are considered for inclusion in the analysis; therefore, secondary cancers attributed to therapeutic radiation are not taken into account.

2005). Similar interactions may exist between radiation and inherent characteristics of the individuals such as genetically based inability to repair damage from the exposure. A review of the literature on the interaction between genetic susceptibility and radiation on cancer risk is presented elsewhere (UNSCEAR, 2006, Appendix A).

g. Use of proxies. Although proxy measures in general are often accepted indicators of an exposure and can prove informative, there is uncertainty as to whether the exposure of interest has been sufficiently investigated by the use of that proxy. The uncertainty varies with the degree of "closeness" between the proxy and the real measure. For example, high socioeconomic status and educational level are often used as a proxy for a healthier lifestyle and access to health care. Birth order[11] and day care use during infancy (Law, 2008) are often used to measure frequency of infection in children. These proxies have been used by a recent study of risks in populations near nuclear facilities (Spycher et al., 2011) to adjust for confounding linked with the "population mixing hypothesis" that has been applied to explain observed leukemia clusters around nuclear facilities in Europe, such as that around Sellafield in Britain (Kinlen, 2011). According to this hypothesis, childhood leukemia is a rare response to common infection, which may be introduced to a previously isolated rural community by sudden in-migration and changes in the dynamics of infectious diseases. Simply, when a population is mixed with another population that has not previously been exposed to the infectious agent (yet to be identified), individuals in the previously unexposed population may develop the disease.

h. Statistical uncertainty. There are inherent statistical variations in fitting dose-response models. It is important that uncertainties be incorporated properly into risk calculations and be communicated clearly. Interpretation of risk estimates is also based on uncertainties from less than perfect knowledge of the effects of low-level radiation on human health. The value of a study increases if it is performed in the context of existing investigations, and if its results are supported by other studies in the field.

[11]In a strictly demographic definition, birth order is based on the ordinal number of live births.

4.2.2 Descriptions of the Study Designs Considered

4.2.2.1 Risk Projection Models

To evaluate the potential cancer risks associated with living near a nuclear facility directly requires very large-scale studies (Land, 1980) and still it would be extremely difficult to estimate the health effects by studying the exposed populations alone. This is because at very low doses, the radiation-related excess risk tends to be buried under the noise created from statistical and other variation in the baseline lifetime risk of cancer which in the population of the United States is estimated to be 42 percent (NRC, 2005). A more timely risk assessment can be obtained using risk-projection models.

Risk-projection models would involve using dose data related to the exposures of individuals living near nuclear facilities and quantifying the risk by transferring that observed in other exposed populations. Data from the Japanese atomic bombing survivors' cohort are most often used for the purposes of assessing the risks arising from exposure to radiation. This is because this cohort has the most detailed information available for most cancer sites. The models for breast and thyroid cancer are often based on pooled analyses of the Japanese and Western populations such as those that were medically and occupationally exposed (see Appendix A for literature review). These models would calculate a theoretical excess risk of cancer for the populations near the facilities by using the most relevant risk estimates and interpolation models, as well as population characteristics like age structure and population mobility. Then one can produce estimates of changes in risk, or demonstrate that any increase is smaller than some upper limit. If the upper limit is an "acceptable" level, then the true level of risk associated with living near a nuclear facility which by definition is lower than the upper limit is unlikely to be unacceptable (Land, 2002).

Such a method was used to project the cancer risks associated with exposure to radiation from other sources such as the use of CT scans and to assess which age groups were associated with the highest risks (Berrington de González et al., 2009). Organ-specific doses and frequency of CT use were derived from national surveys. The investigators discuss that they used this indirect modeling approach to provide more timely risk projections; otherwise, long-term follow-up of very large populations would be required.

There are limitations associated with the use of risk-projection models to transfer risks from more heavily exposed populations such as the Japanese atomic bombing survivors to the populations in the United States that receive much lower doses estimated from reported releases from each facility to be studied.

First, the baseline cancer rates of the comparison population (i.e., Japa-

nese atomic bombing survivors) are often different from that of the population of interest (i.e., residents around nuclear facilities in the United States), and for a few cancers such as breast and stomach cancer the relationship between radiation-induced and baseline risk may differ (UNSCEAR, 2006, Annex A). For example, the age-adjusted incidence rate for breast cancer is 34 per 100,000 per year for Japanese women and 90 per 100,000 per year for the women in the United States (Parker et al., 2002). Breast cancer has occurred in excess among women survivors of the atomic bombings in Japan and among those exposed over many years to medical radiation in the United States. The excess relative risk of breast cancer incidence in the Japanese atomic bombing survivors, however, is significantly higher than that of medical radiation patients in the study in the United States (Little and Boice, 1999) and the best estimate of the ratio of the excess relative risk coefficients for the Japanese and U.S. cohorts is about 2. However, this higher relative excess risk is attributable to the lower baseline risk of breast cancer among Japanese women compared with the women in the United States. The excess absolute breast cancer risks in the two populations are statistically indistinguishable (Little and Boice, 1999). Related to this difference in baseline cancer rates and the relationship between radiation-induced and baseline risk is the question of whether relative or absolute transfer of risks between populations is the most appropriate (see Sidebar A.1 in Appendix A for discussion on risk measures).

Second, additional assumptions are required in risk-projection modeling, which are major sources of uncertainty: sampling variability in parameter estimates in the risk models; the choice of adjustment factors (known as the dose and dose rate effectiveness factor) to use for interpolation from high-dose-rate exposure to much lower dose rates resulting from prolonged releases; and accounting for differences in relative biological effectiveness between different types of ionizing radiation (known as the radiation effectiveness factors).

As a standalone study, a risk-projection model would provide less information than the other study designs considered by the committee and described below. A serious problem with such a study is one of public credibility: the calculated dose distribution by necessity must be based on the reported release data—which if drastically wrong, would provide misleading results. Simply said, the accuracy of the risk-projection models is entirely dependent on the accuracy of the reporting of the releases.

Noting the concerns above, the committee notes that risk-projection models could provide useful background information in conjunction with the empirical epidemiologic studies discussed in this chapter to provide guidance for dose assessment and to aid in the interpretation of such studies.

4.2.2.2 *Ecologic Study*

A main reason why investigators may choose to perform an ecologic study rather than an individual-based study is that the necessary data—depending on the level of aggregation—are routinely available from relevant cancer registries and census bureaus. Hence, it is easier and faster to obtain the aggregated data than it is to collect individual data, the release of which from cancer registries and other relevant offices often involves demanding approval procedures. Because of the relative ease of accessing aggregated data (which is highly dependent on the level of aggregation), multiple disease endpoints in a range of age groups can be studied at once. Despite their inherent limitations, ecologic studies based on cancer incidence or mortality data, even those that focus on large geographic areas such as counties, have proved to be of value in suggesting avenues of research. Ecologic studies are considered as "hypothesis generating" investigations and a finding with possible public health impact will require more rigorous testing using a different study design.

As discussed in earlier sections, radiation is associated with elevated risk for a large number of different cancer types and leukemia, female breast, bladder, thyroid, brain, and ovarian cancers are considered the most radiogenic. Given that different segments of the public have concerns about different cancers, an ecologic study that examines the risks associated with a wide range of cancers may be necessary, but particular attention needs to be given to the most radiogenic types. It is important that ecologic studies are conducted using reliable methods and the susceptibility of their research to the ecologic fallacy is clearly described when results are reported. Recent analysis showed that this is often not the case, and the quality and clarity of some publications on ecologic studies is compromised (Dufault and Klar, 2011).

The NCI reported an ecologic study of cancer mortality across all nuclear facilities that began operations prior to 1982 and for cancer incidence for two states (Jablon et al., 1991). For the NCI study, the rates observed in the population living in a county containing a nuclear facility or an adjacent county that contained more than 20 percent of the area within a 16-km radius of a facility (exposed) were compared to the rates observed in counties not containing a nuclear facility (unexposed). For every exposed county, three unexposed counties were selected to match on certain demographic factors: percentages of persons in the population over age 25 that were white, black, American Indian, Hispanic, urban, rural, employed in manufacturing, and high school graduates; mean family income; net migration rate; infant death rate; and population size.

The analysis assumed that populations living closer to a nuclear facility would receive higher doses of radiation. However, no data regarding

radiation exposures or measured releases from the facilities were used in the analysis. That is, the NCI study, similar to other studies of proximity, was not a direct study of health effects of radiation released from nuclear facilities, but rather a study of the health effects of the collection of factors differentiating populations residing near the facilities from those farther away. This includes exposure to radiation but can also include the demographics of the nuclear workforce and the population-mixing hypothesis discussed earlier (Kinlen, 2011). This context is important when considering the role of dosimetry based on reported radiation releases and monitored values from nuclear facilities, especially since the reported doses in recent years fall well below exposures that have been directly shown to cause cancer.

The primary analysis in the NCI study compared the ratios of standardized mortality ratios or standardized incidence ratios before and after the date a facility began operation, with the same measures for the matched unexposed counties. Hence, the values were not mutually standardized and are, at best, generic rate ratios. The main focus of the NCI report was on the ratio of pre- and postoperation cancer mortality ratios since appropriate incidence data were only available for two states with long-standing cancer registries (Connecticut and Iowa).

Several changes could be made to update and improve the 1990 NCI study design and analysis. Here we discuss five:

1. Reduce the size of the geographic units in the analysis.
2. Use the current nuclear facility inventory.
3. Include years of mortality and incidence data that are relevant to the years of exposure.
4. Incorporate estimated exposure levels for each geographic unit.
5. Use stronger analytic methods that permit direct adjustment for possible confounding variables, and incorporate population mobility and temporal changes in the sociodemographic characteristics of the populations under study.

For the first change, reducing the geographic unit to be considerably smaller in terms of physical size, but also in population, for example, using census tracts, allows for a finer distance-based exposure characterization as well as better characterization of the populations that reside within these units such as age, gender, and race/ethnicity structure, and socioeconomic status. As an example of the magnitude of reduction of the geographic size, the U.S. Census Bureau defined 628 census tracts in San Diego County for 2011. This may be one of the most important of these five ways to improve on the NCI study. This approach would also facilitate analyses of risks at a range of distances. Using smaller geographic units in an ecologic study is also a potential strategy to reduce the impact of the ecologic fallacy.

Although groups are rarely completely homogeneous, smaller geographic groups can be more homogenous with respect to the exposure under study and possibly other risk factors and potential confounding factors. The strategy of reducing the size of the geographic unit for analysis to reduce ecologic fallacy can also lead to another problem, greater migration between groups (Rothman and Greenland, 1998).

For the second change, the inventory of the nuclear facilities in the United States has changed substantially since the NCI analysis; therefore, estimated risks associated with facilities in that study may not be relevant to those operating today. Many nuclear facilities have started operations since 1982[12] (as the total number of currently operating reactors has increased from 80 to 104), but in some cases these are located at the sites of existing plants within which reactors may have been decommissioned since 1982. Some states that did not have nuclear power plants in 1982 now do (Arizona, Kansas, Louisiana, Mississippi, Missouri, New Hampshire, Texas, and Washington), and some other states that had an operating power plant pre-1982, now do not (Colorado, Maine, Oregon) (see Table 1.1, Chapter 1).

For the third change, the follow-up in the NCI study was through 1984 and included facilities that were in operation by 1982. There was very little follow-up time beyond a presumed minimum latency period of 10 years for most solid cancers. (Only with the passage of some years from the year that a facility started operation is it expected that populations living near the facility have accumulated sufficient exposure to develop cancers because of the releases from these facilities.) A current analysis of risks could add 25 or more years (1984-2009) of follow-up. However, an important limitation is the lack of mortality data at the census-tract level: Mortality data that could be readily geocoded to census tract (i.e., addresses are available electronically) do not exist for early years, although data summarized at the county level do exist (see discussion in Section 4.3.3). This recognized limitation of the census-tract-level ecologic design considered here is balanced with the possible gain in statistical power due to the more relevant geographic classification and follow-up period.

Many of the 117 plants that are examined in this study (currently operating and decommissioned; see Table 1.1, Chapter 1 for the list) began operations in the 1970s (45 percent) or early 1980s (37 percent), so if mortality data by census tract exist from the mid 1980s onward (with significant variation across states), some 25 years of follow-up would be possible (in some states follow-up would be much shorter, in some longer). Whereas a large fraction of the observation time in the NCI study predated a minimum latency period (of perhaps 10 years after exposure), most of

[12]The NCI study included facilities that were in operation by 1982.

the observation time in this study would occur after the minimum latency period has elapsed. As incidence data in only two states were examined in the NCI investigation (Connecticut and Iowa), the improvements in the incidence analysis are more clear. Moreover, as the year that mortality and incidence data in a state become available varies, the two approaches would provide complementary time coverage.

For the fourth change, the level of exposure of populations in specific locations around a nuclear facility is dependent on the magnitude of the releases from the facility, the distance of the population from the facility, the mix of wind directions and velocities, and variations in terrain (for gaseous releases), and the locations and directional flow of liquid releases. All these factors are incorporated in dosimetric models that could be used by epidemiologists to calculate cumulated exposure levels for any given geographic unit, such as a census tract within the 50-km radius from the facility, for each year and perform "dose-response"-type analyses of health endpoints. This would be a substantial improvement over most previous approaches, such as examining a 5-km radius around the facilities.

For the fifth change, an overall modeling framework for the analysis of the ecologic data is to develop an extended cross-classification table, each cell of which contains a count of the incident or fatal cases of interest, an estimate of the person-years at risk, and the appropriate estimated exposure quantity and values for other covariates of interest. The cross-classification would be according to geographic unit (for example, census tract, which itself implies the particular nuclear facility under study), calendar year, age, gender, and race/ethnicity. For example, cancer registration of a 50-year-old African American woman, diagnosed with breast cancer in 2005, living in census tract *X* at the time of diagnosis, would contribute a case count to the cell which records the number of African American women in tract who in 2005 were 50 years old. Census data would be used to estimate the total number of African American women aged 50 years who were living in census tract *X* in 2005 so that rates can be computed. Other variables available for this census tract at this time would include a calculated dose estimate or dose surrogate, as well as other census data, or data integrated from other sources with census data. These may include estimates of socioeconomic conditions prevailing in census tract *X* in 2005 or at some other time, based on data about education, land use, and home ownership rates. Information about these and other variables may be important because they could act as confounders in the dose-response analysis. For example, breast cancer risk is influenced by factors such as age at first birth, hormonal use and other factors, all of which may depend to some degree on socioeconomic conditions. Poisson regression techniques (described in more detail in Appendix J) would relate the dose surrogates available to the rate of cancer seen in each census tract, after stratifying on race/ethnicity, age, and

calendar year, and adjusting for socioeconomic or other variables available at the census-tract level.

As population distributions change with time, an ecologic study needs to account for such changes. In the 1990 NCI study, matching of exposed and unexposed counties was based on data for the years 1979 and 1980 (the latest years included in the analysis) and did not consider county characteristics in the 1950s and 1960s, which were likely different from those in 1979. An improvement over the 1990 NCI study would be to allow for differences in cancer rate (incidence or mortality) between geographic regions (census tracts) to depend upon distance or dose as well as time, while adjusting these for the changes in various socioeconomic variables and other risk factors.

In addition, dose surrogates will change over time depending on the total cumulative dose that someone living in a given census tract would receive, so that this dose surrogate increases in time as releases accumulate, and the dose surrogate level is specific for time, nuclear facility, census tract, and age (e.g., persons at age 10 in 1990 would not have been exposed to transient plant releases in the 1970s, whereas those at age 30 would have been). The flexible manner of dose assignment to specific cells in the projected analyses could take into account these variations. In census tracts judged to be stable demographically (with few people moving in or out) this could be the most relevant dose function. In other census tracts (with higher in-migration or turnover) early doses may be regarded as less relevant than later doses, and this could be taken into account in various ways.

As discussed in Section 4.2.1, dealing with the comparison issue and the expected false-positive findings is especially challenging in ecologic studies where each of the thousands of risk estimations is subject to statistical tests to assess whether any observed association occurred by chance or not. At the end, scientific judgment based on biological plausibility and current knowledge are needed to interpret the findings.

Investigators of the 1990 NCI study who based their analysis primarily on a pre- versus post-facility-operation comparison of risks in counties with or without a nuclear facility were able to interpret and communicate the appearance of false-positive findings rather effectively. Data were presented in support of the fact that many statistically "significant" increases in risk in relation to nuclear facilities were found for the period before facilities started operation; these risks could not possibly be attributed to releases from the facilities but are rather statistical effects (Jablon et al., 1990, 1991). The pre- versus postoperation analysis was possible using county-level data as they are available uniformly across the United States and are of good quality. However, reducing the geographic unit to be considerably smaller than a county, which is considered one of the most important ways to improve on the NCI study, comes with the trade-off that risks before the

operation of the nuclear facilities can only be estimated for a small number of facilities. These are the facilities that are in states where long-standing cancer registration and mortality data with available information on geocoded address are available for many years.

4.2.2.3 Cohort Studies

In a cohort study, a defined population is followed forward in time to examine the occurrence of many possible health outcomes. Cohort studies may be either prospective, focused on health outcomes occurring after the start of the study, or retrospective, using existing data in registries to construct a cohort and follow it forward to the present and sometimes beyond. Disease incidence in individuals who are "exposed" are compared to those who are "unexposed."

Prospective Cohort Study

Prospective cohort studies in which participants are recruited, data on residence locations and various potential confounder variables are collected, and then participants are followed for incident disease occurrence are generally thought to provide the most reliable information about disease risk in relation to a risk factor. The major advantage is that the study can be carefully planned in advance to include such things as individual exposure assessment (e.g., using dosimeters) and other covariate data. Since the exposure data are measured before the cancer occurs, some kinds of biases are reduced or absent, so this cohort design is generally preferred over others for making causal inferences. However, prospectively followed cohorts must generally be observed over a very long time (decades) before enough cases of most diseases are available for statistical analysis. To give one example, atomic bombing survivors, exposed in 1945, were initially interviewed around 1950 and have been followed for mortality outcomes since that time and for incident cancer since 1958. It was not until the 1960s (about 15-20 years after the atomic bomb exposure) that the first statistically significant findings emerged of an increase in solid tumor mortality in exposed survivors (Socolow et al., 1963; Wanebo et al., 1968).

A cohort study of the future cancer outcome of individuals near nuclear facilities would involve enormous logistical problems in order to follow individuals for decades into the future. The study would not be able to evaluate past exposures, and this may be a serious problem because the highest radiation exposures may have been in the early years of the nuclear facilities' operations. Far more individuals than are typically needed for a case-control study would have to be interviewed initially and then tracked in the future for cancer incidence and mortality. Population mobility would mean that such tracking would involve large-scale regional or country-wide efforts. Additionally, to follow a population for many decades in the

future as needed in a prospective cohort study relies on long-term institutional commitment that may be difficult to sustain. However, prospective monitoring of populations living around nuclear facilities would provide more accurate estimates of ongoing exposures than those reconstructed retrospectively based on modeling of reported releases from the nuclear facilities. It would also provide data regarding the cancer risks associated with exposures in the future.

Retrospective Cohort Study

Retrospective cohort studies, when feasible, are more efficient than prospective studies because the follow-up period is in the past. A retrospective cohort study identifies a group of people at a time in the past for which exposure estimates exist or can be constructed, and follow-up extends from that time to the present. Such designs are commonly used in occupational epidemiology in which workers employed at a particular facility during specific time periods and meeting other inclusion requirements are followed forward to the present for disease incidence or mortality using existing mortality information or cancer registry information. A retrospective study requires that systematic exposure information at the beginning of and during the follow-up period be available from existing records. Exposure information that might be available from company employment records is related to disease or mortality using statistical methods appropriate for time of event analysis (often Cox regression). Other retrospective studies are based on the follow-up of defined birth cohorts and record linkages used to establish both follow-up and exposure. For example, a recent retrospective cohort study of childhood cancer in Switzerland linked birth records with cancer registration data across the country and used the birth and current residential records to determine proximity to nuclear power plants as a risk factor (Spycher et al., 2011).

The feasibility of a retrospective cohort study depends upon the ability to define a cohort that will include both exposed and unexposed individuals, to estimate appropriate exposure information passively (that is, without the aid of patient or family contact) from existing records, and to link, also passively, the cohort to cancer registration or mortality records from the time that an individual entered the cohort (e.g., time of birth for a birth cohort) until the end of follow-up.

The committee carefully considered the feasibility of a retrospective cohort study of cancer incidence in and around states with nuclear facilities. For the reasons outlined below, only studies of childhood cancers were considered for such a study.

- Children and fetuses, due to their rapidly dividing cells during development, are typically more sensitive to environmental effects than adults.

- Pediatric cancers have been the focus of many studies, some of which found a positive association between proximity to a nuclear facility and cancer risk. Leukemia is recognized to be the "sentinel indicator" for radiation effects, occurring with a shorter time latency following exposure than for solid tumors and with a clear dose-risk relationship (experience from atomic bombing survivors).
- The minimum latency period for leukemia in children is lower compared to that in adults. Associations of childhood cancer risk and radiation releases from nuclear facilities, if any, are probably less affected by co-carcinogens compared to adults, where smoking, occupational exposure, and other established lifestyle risk factors play an important role. Nevertheless, there may be still some risk factors and potential confounders in the development of a cancer during early years of life that are presently unknown.
- Mobility (in- and out-migration) of young populations is less frequent; therefore, observed associations of cancer risk with residence at birth and at diagnosis (often the basis for dose estimations) are more relevant compared to those in more mobile adult populations.
- Children typically spend more time at place of residence compared to adults, whose work may take them elsewhere.
- Societal concerns regarding the radiation health effects of children are the most frequently expressed.

Pediatric leukemia warrants particular attention in the analysis for the reasons summarized at the second bullet point. Similarly, brain cancer, which is the most common solid cancer in children, needs to be given particular attention. Radiation exposure is one of the few established risk factors for this disease. Although all pediatric cancer types can be examined individually, because of the rarity of cancers in children and expected loss in precision in risk estimation it may be needed to create case subgroups based on homogeneity of disease manifestation, etiology, or other categories.

The outlines of the study considered are as follows. All reports of childhood cancer in all available cancer registries over a fixed time period would be linked to birth records from states that contain nuclear facilities or are adjacent to nuclear facilities. Nearness to nuclear facilities (or doses from nuclear facilities estimated by the reported releases) at the time of birth would be established using the residential addresses recorded in the birth records. The entire birth cohort would be linked to all cancer registries, not only in the state of residence at time of birth, but also to other state registries, to capture the mobility of the population. Ideally, changes in residence (and hence changes in potential exposures) would be obtained by linkage to databases providing address histories. Dose surrogates would be constructed starting from the time of birth according to residential loca-

tion. These dose surrogates and cancer incidence data would be analyzed to investigate whether residence patterns that indicated a potential for higher exposure are associated with increased rates of childhood cancers.

Although simple to describe, there are many practical difficulties with performing such a study in the United States. These include:

1. Low coverage of cancer registration before about 1992 for most states.
2. The size of the birth cohort required to have adequate power.
3. Lack of information concerning residence changes following birth.
4. Administrative difficulties accessing state birth records databases and cancer records.

For more details regarding the first difficulty, see Section 4.3.2.

Regarding the second difficulty, Figure 4.3 and Table 4.3 indicate that for a cohort study with a large fraction of unexposed subjects it would take about 1,800 cases in order to have good power to detect a 40 percent excess cancer risk (RR = 1.4) and would require approximately 4 years of incidence data. For example, if all childhood cancers among children aged 0-14 diagnosed in the 4-year time period 2006-2009 were to be targeted in the study (a time when almost all states have working cancer registries), then this would involve linking 18 years of birth records (all children born between 1992 and 2009) to some or all of the cancer registry cases. If we assume that approximately one-fifth of the 4 million births taking place each year in the United States are to women who have home residences within 50 km of a nuclear facility, then this would mean that approximately 14 million birth records would need to be accessible.

For the third difficulty, while there are many ways to try to trace people as they change residences (see Section 4.3.5), no comprehensive databases are available, and ad hoc searching for residence changes on a cohort-wide basis (for millions of birth records in numerous states with disparate sources of residential information) appears on its face to be prohibitively impractical. This means that the only consistently available dosimetry information would be for the period at time of birth. After that, residential changes would gradually degrade the applicability of individual exposure information, such as estimates of cumulative dose. If one assumed that all individuals remain in the same residence as at birth, then cumulative dose calculations are easy to perform, but developing a more realistic model for the accumulation of dose would involve population-based estimates of the probability of mobility. This may lead to some minor improvements in dose estimation, but the fundamental problem, that it is impossible to trace large numbers of individuals from residence to residence, remains. Despite the inadequacies in the use of birth place as the point of exposure over

the follow-up period of interest, it is widely thought that children are the most sensitive to dose received in early childhood or in utero (Pierce et al., 1996), so birthplace may be a more relevant dose surrogate than would be residence at time of diagnosis, as discussed, for example, in the ecologic study. As birth place is defined by maternal residence at time of delivery of the index child it can be used as the point of in utero exposures as well as early life exposures. The mobility of the population during pregnancy remains an issue (Fell et al., 2004).

For the fourth difficulty, birth records and cancer registries are typically managed within each state. However, as shown in Figure 4.4d, many nuclear facilities in the United States are located near state boundaries, and populations of interest often reside in more than one state. In addition, the mobility of the population in the United States may also necessitate linkage of registry data across additional states. While not impossible, access to records will require approval from all states involved, creating a logistical barrier to implementation.

Going further, although linking birth record data across states may be technically possible, there are anticipated difficulties due to the differences of state statutes governing cancer and birth registration, support to research activities, and concerns about privacy following release of information. All these could decrease the quality of the linkages, lead to failure of linking data across states, and delay completion of the study.

The retrospective birth cohort study is judged by the committee to have high scientific merit. However, there are some feasibility concerns at a nationwide scale. A modification of the retrospective cohort study that may be more efficient would be to conduct a record-linkage-based case-control study that is nested in a restricted retrospective cohort study.

4.2.2.4 Population-Based Case-Control Studies

A case-control approach may be appropriate if efforts are directed to selecting just one or two major diseases that may appear in populations around nuclear sites or are restricted to a specific age group. For example, it may be relevant to focus efforts on studying the risks associated with pediatric cancers developing in young residents close to nuclear facilities or more specifically look at risk factors involved in childhood leukemia developing in this group. The German KiKK study and some other studies have suggested a possible increase of this type of childhood cancer, though many other studies have not replicated this observation (see Section A.4.1 in Appendix A for literature review).

Case-control studies using incident (newly diagnosed) cancer cases with data from several registries must consider the years in which registry data are available; the period of inclusion of the cases and controls can

be defined once the quality of cancer registration is found to be adequate. Moreover, a case-control study that requires contact with the study participants that is restricted to recent cases (e.g., those diagnosed within the past 5 years) minimizes potential selection biases due to differential disease severity or availability for interview and/or data collection for nonsurvivors.

In a case-control study, cases are generally matched to appropriate controls either individually or according to a categorization of variables (often age, gender, race/ethnicity; this is known as frequency matching). In either individual or frequency-matched studies investigators need to determine the ratio of the number of controls to the number of cases, a decision generally driven by calculations of statistical power, and the number of cases expected. For rare cancers such as childhood leukemia, the observed number of cases will be relatively small, and multiple controls (two to five per case) would help to improve the precision of results. However, the improvements diminish rapidly as the number of controls per case increases, and more than five controls per case is not likely to be helpful (see Figure 4.3). It is critical that the number and nature of matching criteria be considered carefully. Overmatching must be avoided; for example, matching closely on place of residence or distance from a nuclear facility may constitute overmatching. That is, investigators "force" the cases and controls to be too similar in the exposure under investigation; therefore, the effect of the exposure on disease cannot be investigated.

Obtaining accurate information on past exposures (predating the occurrence of the cancer, or an equivalent time point for controls) can be problematic. If information is to be obtained from existing records, it may be only partly suited to the desired study information. For example, data on smoking might be obtained from employment health records, but the smoking information may be incomplete or too cursory for the need (e.g., "Do you smoke?" rather than detailed information on duration and frequency of smoking, and information may vary across time periods and employers). Records relevant to some exposures would have been generated for administrative rather than medical purposes and therefore might be poor surrogates for the desired information.

The information for cases and controls must be collected by the same approach in order to limit bias related to quality of information or extent of detail of the data collected in different administrative files or medical records, or due to differential interviewing. Residential history, socioeconomic characteristics of the parents, infections, exposure to radiation in utero or as a child, and parental smoking are some of the factors previously associated with childhood leukemia and such information, if available, can be included. Birth order is of interest because it has been implicated as a risk factor for leukemia and may be a marker of exposure to infectious agents, with later-born children presumed to be exposed more often and

at earlier ages from their older siblings. Therefore, birth order could be used as a proxy to examine the postulated population mixing hypothesis and infectious etiology for childhood leukemia (Kinlen, 1988). According to this hypothesis, childhood leukemia is a rare response to common infection, which may be introduced to a previously isolated rural community by sudden in-migration and changes in the dynamics of infectious diseases.

Record-Based Case-Control Study

As stated earlier, the retrospective birth cohort study was judged by the committee to have high scientific merit but involves logistical and administrative barriers. A record-linkage-based case-control study that uses data on cancer registration and birth records to identify cases and controls and relevant information is an alternative to the retrospective birth cohort design.

In a record-linkage-based case-control study, children diagnosed with cancer at age 0-14 years are identified from population-based cancer registries of states that have or have had a nuclear facility or are adjacent to such a facility. Cancer cases identified among children in the registry are linked to birth records within the respective state(s). Those born within the area of interest (e.g., 50 km around a nuclear facility) are eligible cases. One or more controls are randomly selected from birth records restricted to those born within the 50-km zone from the facilities, with matching to cases on year of birth at minimum, and if possible month of birth, race/ethnicity, and gender. The 50-km zone provides a wide range of potential exposures for controls but keeps controls in similar regional settings. Children diagnosed with cancers but who were born outside the study area could be excluded from the control group; however, the likelihood of them being selected randomly as controls is very small as indicated below.

The record-linkage-based case-control study of pediatric cancers differs from the retrospective cohort in some important issues that enhance its feasibility by:

1. Restricting the linkages to within state instead of across states. Rather than considering (for example) all of the 3,000 childhood leukemia cases per year that are expected nationwide for linkage to birth registry information for all states with or proximal to nuclear facilities, cases would be identified from state cancer registries with or near facilities, and linkages would occur only within the respective states as opposed to between states. This should reduce considerably the number of birth records that need to be searched for each cancer case included. Also, as a consequence of restricting the cases to those born and diagnosed in the same state, the record-linkage-based case-control study focuses on the more residentially stable children (although arguably the children and their families may have moved within the state in which the child was born).

2. Limiting the number of cases and controls that would be followed to update residential history, or dropping the requirement. As a relatively small number of controls for each study case would be selected for analysis along with the cases (since many fewer study subjects would be involved than in the retrospective cohort study) it may potentially be more feasible to follow these forward and retrieve residential information than it would be to follow an entire birth cohort forward to look at changes in residence, in order to refine dose estimates. This effort still, however, could be substantial and may be worth doing only for a relatively small number of cases and controls in order to give estimates of overall rates of out-migration and loss to follow-up. Dropping the requirement of following the subjects forward in time via records, the overall efforts required to conduct the study are substantially reduced.

As with the retrospective cohort design, cases as well as controls are required to be born within a fixed region (e.g., 50 km from a nuclear facility). For the record-linkage-based case-control design more selective targeting schemes could be considered, such as requiring the cases selected for study to be residents of a 50-km proximity zone at the time of diagnosis. It must be kept in mind, however, that as further restrictions for selecting eligible cases apply, the potential for loss of study power increases if large numbers of cases were excluded from consideration. Additionally, as the design does not rely on follow-up of the controls to establish if they also remained at the 50-km zone from birth to the time that the cases were diagnosed, the potential for selection bias increases and false relationships between case status and distance could appear if the probability of moving versus staying within the same region is inhomogeneous with respect to distance from nearest nuclear facility. Results from regions with high in- or out-migration of children would be less reliable than those from regions with less population mobility.

The design could be extended as far back as registries with good quality data exist and birth years of cases and controls would co-extend with good practices of registry operation. A study that includes subjects that were born before the state's cancer registration is of acceptable quality could appreciably increase the number of eligible cases at the older targeted ages, and it also could assess exposures in earlier years when the exposure levels were likely higher. Inclusion of these subjects can be achieved as follows: For cancer cases at each age X, the birth records for up to Y years before the beginning of good quality cancer registration could be used. For instance, if the year of good quality cancer registration data is 1996, the birth records from 1990, 1991, or 1992 could be used to include cancer cases and controls of ages 6 or older, 5 or older, 4 or older, respectively. While this approach might introduce slight bias as those who developed

cancer at earlier ages would not be eligible, for all practical purposes the study could be regarded as unbiased on that respect.

An advantage of either the record-linkage-based case-control approach or the retrospective cohort study is that certain relevant characteristics of the parents and infant are available on birth records and, depending on the year and state, would include: mother's address; duration of residency at that address, parental age, race/ethnicity, educational level; and date of birth, gender, weight, and order of birth of the index child. Additional information on the birth certificate such as substance abuse by the mother (including smoking and alcohol) does exist in certain cases but will have varying reliability and completeness depending on the state (Spector et al., 2007). The above-mentioned data elements are included on the 2003 national standard certificate of live birth; however, the certificate was not implemented systematically. As described elsewhere, 2 states implemented use of the certificate in 2003, 7 additional states in 2004, and cumulatively 15 states used it in 2005 (Kirby and Salihu, 2006). Information on abnormal conditions of the infant such as Down's syndrome and other congenital anomalies of the newborn can be used to exclude cases and controls from subanalysis.

Regarding these issues, in a five-state pooled analysis study of parental age (available from birth records) and risk of childhood cancer (Johnson et al., 2009) which used the methodology described here, diagnoses went back to 1980 in Washington State, 1985 in New York State, 1988 in Minnesota and California and 1990 in Texas. The analysis from five states comprised approximately 30 percent of the U.S. pediatric population. Using probabilistic record linkage, the linkage success of cancer registry and birth records data within a state was 88 percent for leukemia cases age <5 years in California (Reynolds et al., 2002), 87 percent for hepatoblastoma cases age <5 years in New York (McLaughlin et al., 2006), and 82 percent for cancer cases age <15 years in Minnesota (Puumala et al., 2008). The information was not reported for Washington (Podvin et al., 2006) or Texas (Walker et al., 2007). Although the authors did not provide a breakdown of the possible reasons for unsuccessful linkage, these may include in-migration (children born elsewhere moved to the reference state and were diagnosed there), rather than flaws in the linking methodology.

A 17-county study of childhood leukemia (age <15 years) in California demonstrated that a small percentage of cases (12 percent) were not born in the study area; approximately 5 percent were born in other counties in California and 7 percent outside of California (Ma et al., 2004). The recent study in Switzerland, a country where populations are likely less mobile than in the United States, demonstrated that 68 percent of pediatric cases had not moved between birth and diagnosis, 22 percent had moved once, 6 percent three times, and 4 percent three times or more. Although in-migration is expected in all states under study and appears to be some-

where between 10 and 20 percent for children 0-14 years, it is expected to be lower for children 0-5 years old (Ma et al., 2004), which is also the age range in which most leukemia cases are expected (peak for acute lymphoblastic leukemia is 2-4 years old).

It may be possible to estimate in- and out-migration of subjects based on census data and to describe the characteristics of the cases who migrate based on cancer registry data such as age, year of diagnosis, and race; correction for selection bias may be possible if probabilities of exposure can be stratified by these same variables.

Study controls in the record-linkage-based case-control design are randomly selected from each state's birth registry. The matching ratios for the pooled analysis of the five states mentioned above differed by state from 1:1 to 1:10 (Johnson et al., 2009). A concern is that children identified by the birth registries as eligible controls may have been diagnosed with cancer in a different state. However, given the rarity of childhood cancers (about 4.8 per 100,000 children will be diagnosed by age 15 with leukemia or brain cancers, the two most common cancers in children), this issue should have essentially no effect on the power of a study, but might nevertheless have some unknown potential to introduce bias, since controls but not cases may have migrated from the state and such migration might reflect socioeconomic or other differences that affect childhood cancer risk.

Feasibility of the record-linkage-based case-control study depends on availability and release criteria of the information on both birth and cancer registration information that may involve demanding Institutional Review Board (IRB)[13] or equivalent body approvals. Release of the required information may not be possible in all states under investigation, or in rural areas within the states for reasons of subject protection or because linkage capabilities are not in place. For these reasons, it may not be possible to include all of the states of interest in the analysis.

Part of the predicted feasibility and practicality of this study lies in the fact that it can be based on and expand on existing studies and ongoing efforts to link state cancer registry records with birth records, by partnering with the appropriate investigators. Such linkages are established statewide within Washington, New York, Minnesota, California, and Texas. Similar linkage analyses have been performed in metropolitan regions and surrounding counties of Seattle, Washington; Detroit, Michigan; and Atlanta, Georgia, as well as statewide in Utah (Mueller et al., 2009), to investigate pregnancy outcomes in female childhood and adolescent cancer survivors.

[13]The term IRB describes the standing committee in a medical or research institution, hospital, or other health care facility, whose task is to ensure the safety and well-being of human subjects and privacy of any information retrieved from those subjects.

De Novo Case-Control Study with Patient or Family Contact

The committee also considered the development of a new case-control study. To illustrate, a study of childhood cancer might begin with definition of a reference population of children less than 15 years old, living in the vicinity of nuclear facilities. Controls would be children of the same age and gender who lived in the same general area with the cases at the time the cases were diagnosed. Contact with children or families would be used to define residential history and therefore the study is not dependent on assumptions about continued nearby residence from birth until time of diagnosis.

The challenges of selecting appropriate controls through random-digit dialling, school records, or friend controls and the emerging use of birth record controls are discussed in Section 4.3.4. It is important that controls be selected in a way that does not bias the basic comparisons that are the object of the study. In particular, controls must represent the distribution of distances from the nearest nuclear facility for the same population from which the cases are being drawn.

Within a case-control study, investigators would usually choose the recent cases (for example, those diagnosed during the period 2005-2010) and appropriate controls and trace individuals for interviews in order to collect information on residential history and other risk factors and refine the exposure of the individuals. Tracing recent cases tends to be more successful than tracing past cases as the more recent cases would have less opportunity to move, would be easier to find, and are more likely to be alive. Children with cancer would be traced through the treating institution as identified from cancer registration files or other means and they and/or their parents contacted in order to obtain additional information regarding residential history and a list of known or putative risk factors for childhood cancer. If the identified cases who were children at diagnosis and are adults at the time of interview are those providing the information, their responses may differ from those of the parent, and many now-adults may not know answers to questions about childhood residential history or early life care. (Cancer registries may require that contact with the now-adult is established first to obtain permission to be a study subject and to allow parental contact.) Depending on the method selected for control identification, tracing for controls may also be required (see Section 4.3.5).

Even when tracing is successful, collection of detailed information by interviews or by questionnaires will face issues of nonparticipation. As nonparticipation rates are often considered an indicator of the potential for selection bias, it is important that they are kept as low as possible; individuals (or parents) who refuse to participate in the study may differ in relevant ways from those who are willing to participate, and this may affect the study outcome. Controls often are more likely not to participate than cases, and participation rates of controls have declined in recent years,

regardless of source (Bunin et al., 2007). One survey estimated the decline of population-based controls to be –1.86 percent per year (Morton et al., 2006). Low participation rates or differential participation rates between cases and controls can introduce bias, when willingness to participate is related to exposure and this tendency is stronger (or weaker) in cases than in controls (Hartge, 2006).

Differences in the accuracy and detail of answers provided need to be minimized. Focus groups and pretests of questionnaires and procedures may help to establish a well-designed questionnaire for the specific study scope. To avoid bias associated with information given during an interview or when filling out a questionnaire, one useful approach is to not inform interviewers whether a specific subject is a case or a control; this can limit the bias that an interviewer might unconsciously inject into the information, though information on case or control status may often come out during the interview. In contrast, a patient (or proxy) cannot be kept in ignorance of his or her status, so an additional concern is "recall bias," under which controls may have given less thought or pay less attention to past exposures (such as infections, medical imaging, and other) and underreport them, thus introducing a bias. For example, a mother whose child has died of leukemia may be more likely than the mother of a healthy living child to provide more complete and accurate information on past experiences such as x-ray exposures when the child was in utero (see Section A.4.6.2, Appendix A). This recall bias could artificially suggest a relation between x rays and leukemia.

Moreover, the information that individuals give may be affected by unconscious biases; this is particularly true if a study has been widely publicized and subjects are aware of reported health effects and what exposures are suspected to cause these effects. A well-designed questionnaire may minimize these biases by carefully wording the questions, often requesting the same information by two questions phrased differently to identify inconsistencies and judge the reliability of the information, or simply by forcing the individual to think more carefully. Telephone interviewing may be a better approach than interviews in person, especially when questions touch on sensitive matters such as possible exposures during pregnancy.

In a study of childhood leukemia the questionnaire is likely to contain details on lifestyle, socioeconomic status, residential history, occupational exposure of parents at the time of conception of the child and during pregnancy, medical radiation exposure during pregnancy and early childhood, infectious diseases during early childhood, contact with other children during first years of life, nursery care, birth order, and number of children in the family as well as questions specific to milk consumption to better estimate individual exposure. As most risk factors for leukemia are still unknown, it may be necessary to consider trade-offs between collecting a large amount of information per subject and the number and geographic

source of subjects. Experience from previous studies in similar populations and areas often provides useful lessons learned.

As shown in Section 4.2.1.6 a study which would have good power to detect 20 percent increases in cancer risk for a relatively rare exposure (RR = 1.2, assuming 2.5 percent of subjects are exposed in the calculations in Figure 4.3) would have to be extremely large (thousands of cases and at least as many controls). For rare cancers (such as childhood leukemia) this would involve decades of accrual in regions near sites; while much larger relative risks could be detected far more easily, the expectation is that 20 percent increases are extremely large relative to the cancer risks expected based on reported releases. For more common cancers, while the rates of case accrual are larger, the expectation is for even weaker dose-response relationships. Thus, the power of any feasible case-control study (one that could be completed in years rather than decades) is likely to be extremely low.

For reasons primarily related to considerations of both statistical power and logistics, combined with the fact that only relatively recently diagnosed cases could be included and the potential for participation (and possible information) bias, a de novo case-control study and the associated efforts required to collect additional information on potentially confounding factors may not be justified over the record-linkage-based case-control approach.

Building on Existing Studies

As discussed earlier in this section, it may be possible to partner with investigators who are already using linkages between cancer registration and birth records to perform the record-linkage-based case-control study. As these linkages exist in at least six states, representing more than 30 percent of the U.S. pediatric population, using existing data, if possible, would reduce substantially the overall efforts required to conduct the record-linkage-based case-control study.

Several recent or ongoing case-control studies, cohort studies, and clinical trials could be useful in developing a new case-control study with contact of individuals or their proxies. The advantage of working with existing studies is that cancer cases and controls have already been identified, the initial contact has been established, and collected information related to the original study may be useful. Participants or their proxies can be recontacted and additional relevant information can be requested such as residential history and potential confounders. In certain instances it may be possible to find existing data about residential history passively (from old city directories, for example), without individual participant contact. Here, however, we assume that (as for most studies) individual exposure and covariate data are obtained directly from participants or their families. The requirement for direct contact would seem to require that the existing study contains recently diagnosed cases and that patients or families be

contacted soon after diagnosis. This limits the number of existing studies that would be useful as partners.

Most existing large studies are focused on adults, and often for populations with specific characteristics and outcomes to serve the specific research focus of the study. A few such examples are the Women's Health Initiative, a study of more than 160,000 generally healthy postmenopausal women, designed to test—among other issues—the effects of postmenopausal hormone therapy on breast and colorectal cancer (Hays et al., 2003), and the Nurses' Health Study, a study of about 238,000 female nurses, focused primarily on cancer prevention (Willett et al., 1987). For rare cancers such as pediatric cancers, investigators have realized that individual large cohort studies are unable to examine the effect of different exposures on the disease due to inadequate sample size. For that reason, multiple large children's cohorts have joined to establish national or international consortia such as the Pediatric Brain Tumor Consortium and the International Childhood Cancer Cohort Consortium.

Even if existing studies include the age group and cancer outcome of interest, the biggest issue is that, since only a relatively small fraction of the U.S. population overall lives quite near a nuclear facility (about 0.3 percent within 8 km and 15 percent within 50 km in 2010; see Tables 1.3 and 1.4 in Chapter 1), existing studies probably do not cover enough persons living within the 0-50-km zone to provide statistical power for the study of the relation between residential history and/or individually estimated exposures and cancer occurrence. The possibility of using an existing study to build a contact-based case-control study was not considered further, since no known studies that would meet the necessary criteria were identified.

4.2.3 Recommended Studies

Of the several studies considered, two epidemiologic study designs were judged by the committee as suitable to have scientific merit and address the nonscientific issues that they must deal with for assessing cancer risks in populations near nuclear facilities: the ecologic and record-linkage-based case-control studies. A summary of the strengths and limitations of the recommended studies is presented here.

4.2.3.1 Summary of Strengths and Limitations

1. Ecologic study

Description

The study design investigates incidence and mortality rates for all common cancers identified at the census tract within which cases reside at the

time of diagnosis or death from cancer, respectively. The study is restricted to census tracts within a fixed distance (perhaps 50 km) of a facility which represents a range of potential exposures from the highest to essentially no exposure. Cancer rates among census tracts are compared by average estimated levels of exposure.

The question such a study can answer

Are observed cancer incidence and/or mortality rates higher in census tracts with higher estimated exposures (as estimated from reported releases from the nuclear facility)?

Feasibility[14] depends on

a. Availability and release of aggregated cancer registry and mortality information at the census-tract level, according to age, gender, race/ethnicity, and cancer site.
b. Availability of population structure and size (also by age, gender, race/ethnicity) data from the U.S. census, with interpolation for noncensus years.

Strengths

a. Has the ability to look at all potentially radiosensitive types of cancers and for all age groups.
b. Examines both incidence and mortality, which provide complementary data and can be mutually supportive.
c. Can examine past outcomes and therefore can examine risks at times when releases were higher and more likely to cause cancer.
d. Only cancer registries and/or vital statistics offices of those states that have or have had a nuclear facility or which contain populations within the study distance of a nuclear facility need to be contacted.
e. Provides results relatively quickly as information comes mostly from existing databases.
f. No issues related to control selection appropriateness or feasibility.
g. Does not rely on recruitment of study participants.
h. IRB or equivalent body approvals for cancer incidence and mor-

[14]The committee judges that a study is feasible if it satisfies the following criteria: (a) it is based on existing data for cases, the at-risk population, and common confounding factors; (b) it meets the criteria regarding release of those data for research purposes; and (c) it considers knowledge and experience from studies in the field including anticipated participation of subjects.

tality data will possibly be needed, but procedures are likely to be undemanding (possible exceptions are procedures for data release from rural areas where only a few cases reside within a census tract).

Limitations

a. Subject to ecologic fallacy and has limited ability to conclusively establish or refute a relationship between radiation and cancer because exposure information on actual cancer cases is not obtained; might be subject to biases that cannot be taken into account. Is considered hypothesis generating.
b. Study type has been criticized. It may be viewed as an easy, quick, and least expensive study, bound to give inconclusive results because:
   - It is particularly subject to multiple comparison problems as numerous cancer types and age groups will be examined.
   - It can control for confounding only by using aggregate census-tract data. The registry and census data do not include specific lifestyle factors.
c. Can only examine associations based on residence at diagnosis or death rather than place of birth or place of relevant exposure. Associations based on place of death may only partially reflect past exposures due to population mobility.
d. Can only estimate average in- and out-migration rates, with no information on the residential history of actual cancer cases.

2. Record-linkage-based case-control

Description

Children diagnosed with cancer (in the period of reliable cancer registration) in states that have or have had a nuclear facility or are within a fixed distance (for example, 50 km) of a nuclear facility are linked to the birth records of the respective states to identify those children that developed cancer and were born within a fixed distance from the facility (for example 50 km). Controls are children identified from birth records to be born in the same general study area as cases and matched at minimum to cases on year of birth (birth month and gender where possible).

The question such a study can answer

Among children born within 50 km of a nuclear facility, are pediatric

cancers associated with higher exposure at maternal residence at time of birth?

Feasibility depends on

a. Availability of maternal residence at the time of delivery in the birth records.
b. Within-state linkage capability of cancer registration with records kept in vital statistics offices that will provide information on births (and possibly deaths) in the areas around the facilities.
c. Availability and release of linked data at the individual level.
d. Accrual of enough childhood cases during the times in which cancer registries are of reasonable quality to have power to detect disease patterns related to estimated exposure levels.
e. Ability to obtain birth record information on all births in the relevant risk sets (e.g., all those born within 50 km of the nuclear facility in each of the relevant birth years) in order to define an unbiased set of geographic controls.

Strengths

a. Provides individual risk estimates rather than estimates based on geographic units.
b. Examines associations relevant to early life exposures (birth place) which can be considered more relevant than those later in life as would be captured in a study based on place of residence at time of cancer diagnosis or death from cancer and the equivalent for the unexposed.
c. Can be considered an objective study as it does not rely on contact of individuals or interviews and therefore is not subject to selection or possible information bias related with subject participation and collection of information on risk factors.
d. Does not need to be restricted to very recent cases, as cases and controls are not traced to be interviewed.
e. Provides results relatively quickly as information comes from existing databases and requires linkage only between cancer and birth registration data.
f. Information on certain relevant covariates is available in the birth certificates and can be adjusted for.
g. Because the study is focused on children, uncertainties sourcing from population mobility or lifestyle choices are less of a concern.
h. In-migration of cancer cases (but not controls) can be estimated.

Limitations

a. Restricted to a specific age group and few cancer types (i.e., childhood cancers). Hence, it may not address many of the concerns of the public stakeholders.
b. Restricted to recent cases, therefore
   - Harder to accrue large numbers of cases (and hence statistical power may be limited).
   - Risks associated with higher releases in the past cannot be examined.
c. Cannot estimate the frequency of, or the altered exposures and effect estimates due to, out-of-state migration of cases or any migration of controls.
d. Linkage of birth and cancer registry records may not be possible (or permitted) in some states.
e. IRB or equivalent body approvals for data release of birth and cancer registration will be required.

4.2.3.2 Approaches for Conducting the Recommended Studies

The recommended studies are complementary in that each addresses different aspects of cancer risks:

- The ecologic study would provide an assessment of risks for a variety of cancer types over longer operational histories of nuclear facilities for which effluent release and cancer mortality and incidence data are available.
- The record-linkage-based case-control study would provide an assessment of cancer risks for childhood exposures to radiation during more recent operating histories of nuclear facilities.

The recommended studies are mutually independent, and could be carried out individually or together. The decision on which of the recommended studies to carry out and their order of execution involves a host of policy and other considerations that are beyond the scope of this Phase 1 project. These include, for example, considerations such as the following:

- Which age groups and cancer types are most important to address in the epidemiologic study or studies?
- How much time is available to carry out the study or studies?
- How much funding is available to carry out the study or studies?
- Which public concerns are most in need of help with addressing?

4.3 DATA SOURCES AND METHODS

4.3.1 Population Data

Each of the approaches considered requires some knowledge about the size and demographic characteristics of populations living close to a nuclear facility, and this information must be on a suitable time scale. The committee is convinced that the information should be for geographic areas smaller, perhaps much smaller, than counties.

Population counts for small areas are available from the U.S. Census.[15] Every 10 years, in years ending in "0," the Bureau performs the official count of people living in the United States. The Bureau of the Census supplements the decennial census on a continuing basis by the sample surveys and statistical models that make up the American Community Survey (ACS[16]), which provides more data on social and economic characteristics than does the decennial census. The ACS sends surveys to approximately 3 million housing units and group quarters in the United States in every county, so detailed information on a small geographic scale may be sparse. In 2009, completed ACS interviews represented 66.2 percent of the housing units initially selected for inclusion in the sample.

The decennial census reports show aggregate population demographic data for a standard set of geographic regions defined by state, county, census tract, block group, and block. Blocks are small geographic areas bounded by visible features such as streets and railroad tracks and by nonvisible boundaries such as property lines or county boundaries. Block groups consist of collections of blocks and are typically defined to contain 600 to 3,000 people. Census tracts contain several block groups and typically contain 1,200 to 8,000 people (with a target of 4,000 people) (www.census.gov). While the typical and target population sizes generally hold, there is wide variation across the country and some tracts contain population counts well below or above the example ranges stated here. The spatial size of the census tract also varies widely across the country. Census tracts were not fully defined until the 1980 Census. The 1970 Census had tracts for some areas, but not the entire country. Enumeration units at one level do not cross those at higher levels so, for instance, a census-tract boundary does not cross a county boundary. This nested hierarchy ensures that counts are "upward compatible." County boundaries rarely change over time, and state boundaries do not change at all. If an analysis requires attention to these changes, the Geography Division of the Bureau of the Census may be able to help.

Census Summary File data from each household include information

[15] http://www.census.gov/.

[16] http://www.census.gov/acs/www/.

regarding the population (such as gender, age, self-reported race and ethnicity, household relationships). Questions about race and ethnicity have evolved rapidly and substantially over recent censuses, so comparability across time may be an issue. The 2000 census tabulates 171 population items and 56 housing items at the block level and an additional 59 population items at the census-tract level. At various times the data available at the census-tract level have included race-specific tabulations of other variables such as counts of age by gender by race, and household characteristics by race. From a one-in-six sample weighted to represent the county's population, to which the "long form" was distributed until the 2000 census, more detailed population data exist, including, for example, place of birth, education, employment status, commuting distance to work, school enrollment, and income as well as housing data such as value of housing unit, telephone service, plumbing, vehicles available, and year structure built. The unpopularity of the "long form" led to its replacement by the ACS (www.census.gov/acs).

The ACS began collecting data in four test counties in 1995. National data were first released in 2001 (with data for 2000) and the ACS was fully implemented by 2006. Each year it publishes three sets of estimates: estimates based on the most recent 1 year of survey data for geographic areas of 65,000 and larger, 3-year average estimates for geographic areas of 20,000+, and 5-year estimates for all geographic areas down to the block group.[17] ACS data are summarized for 5 years (for example, 2005-2009). The ACS has a rather short history but might be combined with data from the "long form" to provide useful information for long-term studies of health risks.

While the state-county-tract-block group-block hierarchy defines the primary framework for U.S. Census geography and aggregate data releases, data are aggregated in a variety of other ways. These include congressional districts and school districts, which need not follow block, block group, tract, or county boundaries. The U.S. Postal Service (USPS) defines ZIP code units for mailing addresses. ZIP codes are designed primarily to serve the needs of the USPS in management tasks related to local post offices. Some records (such as billing records and birth certificates) can easily be aggregated by ZIP code. While geographic areas are associated with ZIP codes, these areas rarely match block, block-group, or census-tract boundaries and, at times, even cross county and state boundaries. Compared to census tracts, ZIP codes are not only typically larger but also less homogenous aggregate units. In addition, ZIP code areas are modified as needed by the USPS, unlike census regions, which are updated only following a decennial census, to address in- and out-migration. As a result, direct linkage between

[17]http://www.census.gov/geo/www/.

ZIP code areas and census summary data is challenging, especially over long periods of time. As a compromise, the Bureau of the Census provides ZIP Code Tabulation Areas with summary data from block units combined to match ZIP code areas as closely as possible.

For both the census and the ACS, a number is not published when the number of persons in a cell of a table is small (often five or fewer), as a way to maintain the confidentiality of individually reported data. This can be a serious limitation in using the ACS but may be less serious for analyses based on the decennial census.

It is recommended by the Census Bureau that ACS data not be used below the census-tract level because the margins of errors on block-group estimates are generally high. These data are made available primarily to allow users to add block groups to create estimates for custom geographies.

Accounting for migration is important in studying the risks of living near a nuclear facility, but it is also challenging, particularly when smaller geographic units are analyzed. The decennial census and the ACS track migration, but in different ways. ACS asks individuals where they lived a year earlier and monitors place-of-residence changes if across county or state boundaries, but not smaller geographic units. If a person has moved multiple times within a year, the ACS captures only the earliest move in the prior 12 months. The decennial census has tracked migration by asking the individuals where they lived 5 years earlier. The 2010 Census did not collect information on migration.

Migration statistics from the Bureau of the Census are tracked every 10 years; this implies that any trends within the 10-year period are not captured. Models for migration into regions can be incorporated; for example, if it is known that a given locality has had much recent migration this can be used to modify (down-weight) the dose-surrogate variable under an assumption that migrants are unexposed prior to their move, thus reducing the average time-weighted dose value for that unit. Generally this would be done in a time and possibly age-dependent fashion allowing for migration patterns to vary over time and by age.

Pretabulated data are available for all levels of geographic units and would cover 100 percent of available data. The microdata file that is available for public use includes 40 percent of the data for geographic units that include at least 100,000 persons. For non-Census employees, gaining full access to the microdata is possible in special cases but requires substantial paperwork, including permissions and background checks, and the investigator would need to work in or with a designated Research Data Center to retrieve the information.

To appreciate the size of the populations residing near the nuclear facilities, the committee estimated the number of individuals that reside within the census tracts at 0-8- and 0-50-km radii around currently operating nu-

clear facilities. The numbers are presented in Tables 1.3 and 1.4 of Chapter 1. For demonstration, the 2010 census data were used, although it is clear that recent census data may not be relevant to risks associated with early operations of facilities. The committee used the geographic information system ArcGIS to draw circles around the facilities at 8 and 50 km. As the radius around a plant would cut through census tracts, the map assigned a share of each census tract's population to the circle based on the percentage of the tract's land area that falls within the circle. If the circle would intersect, for example, 30 percent with a census tract, then 30 percent of the census-tract population would be included in the circle; this assumes homogeneity in population density within the census tract. Such population size estimates are attractive and appear very precise, but they can be sensitive to the choice of map projection (Figures 4.4a-4.4d are based on a conic Lambert projection) and to the assumption that the proportion of area is an accurate reflection of the proportion of individuals residing in a portion of a census tract. In some cases, small changes in these two issues (map projection and proportional-to-area assignment) can result in changes in population estimates in the hundreds or even thousands of individuals.

In summary, in 2010, approximately 47 million people (15 percent of the population in the United States) lived within 50 km of an operating nuclear facility and 1 million (0.3 percent of the population in the United States) lived within 8 km of an operating nuclear facility. The series of regional maps (Figures 4.4a-4.4d) highlight different challenges that need to be considered when evaluating the risks of the populations around the nuclear facilities and these are discussed here.

The population size residing near (e.g., within 50 km of) a nuclear facility varies considerably across the facilities. As an example, approximately 2,400,000 people live within 50 km of the San Onofre Nuclear Generating Station located in the San Diego County, California, indicated by the red circle, while only 54,000 people live within 50 km of the Cooper Nuclear Station located in Nemaha County, Nebraska. This can be visualized in Figure 4.4a, by the much smaller but denser (darker brown) census tracts that are around the San Onofre plant compared to the Cooper plant. Inner black circles indicate the boundary of the 8-km radius.

There is often overlap in the populations that reside within the 50-km radius from two or more nuclear facilities due to the proximity of the sites in some areas of the country. For example, approximately 143,000 residents of Illinois reside within the intersection of the 50-km radii of Dresden, LaSalle, and Braidwood plants combined (Figure 4.4b); in an epidemiologic investigation of cancer risks, these residents would be considered to be exposed from all three plants and doses would be estimated using an additive model.

Exposure estimations may be further complicated if the facilities that

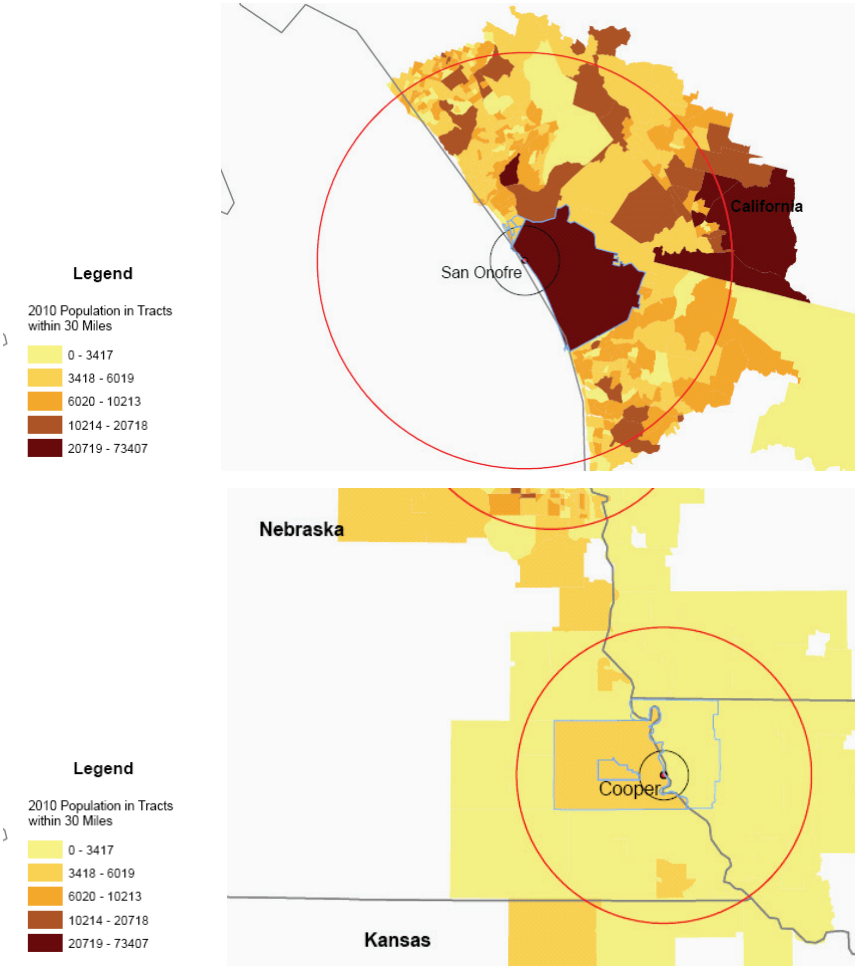

FIGURE 4.4a Size differences in the populations near nuclear facilities.

share the population around them are of different type; therefore, the radioactive release content or pathways of exposure may be different. An example describing such a situation is the conversion facility in Metropolis, Illinois, operated by Honeywell International, Inc., and the uranium enrichment facility in Paducah, Kentucky, operated by USEC Inc. These two types of facilities are in such close proximity that there is an almost complete overlap of the exposed population within the 50-km zone (Figure 4.4c).

The above-mentioned example is also an example of facilities being located at or near the border of two or more states; hence, the population

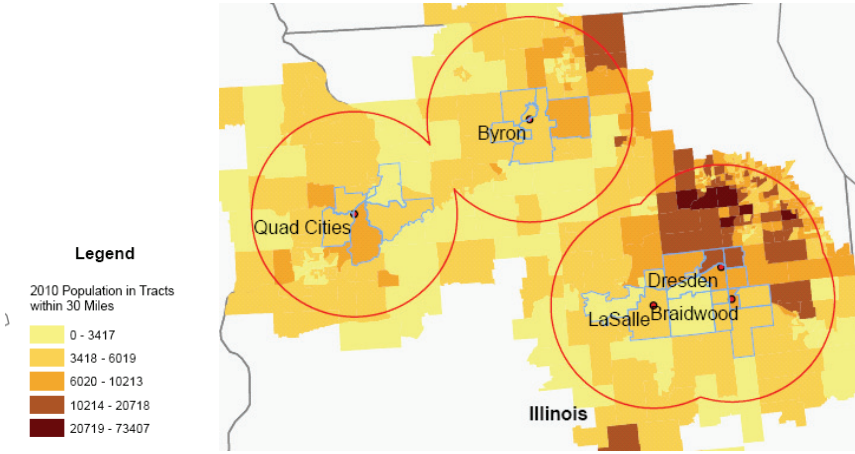

FIGURE 4.4b Population overlap among nuclear power plants.

within 50 km of the facility is shared between two, three, or four states. Figure 4.4d illustrates some of the many power plants whose populations in close proximity reside not only in the state where the plant is located but also in neighboring states. For example, the populations living within 50 km of the Vermont Yankee plant in Vermont reside in Vermont, Massachusetts, and New Hampshire. Similarly, the populations living within 50 km of the Seabrook Station in New Hampshire reside in New Hampshire,

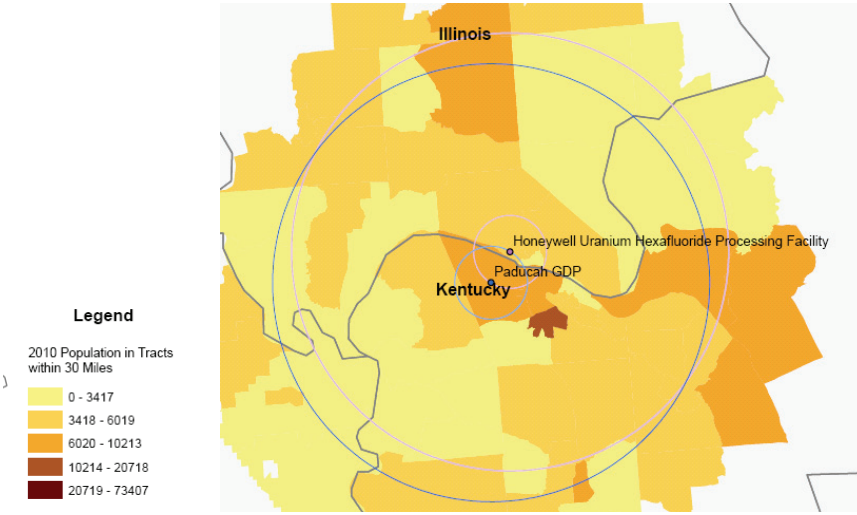

FIGURE 4.4c Population overlap between different types of facilities.

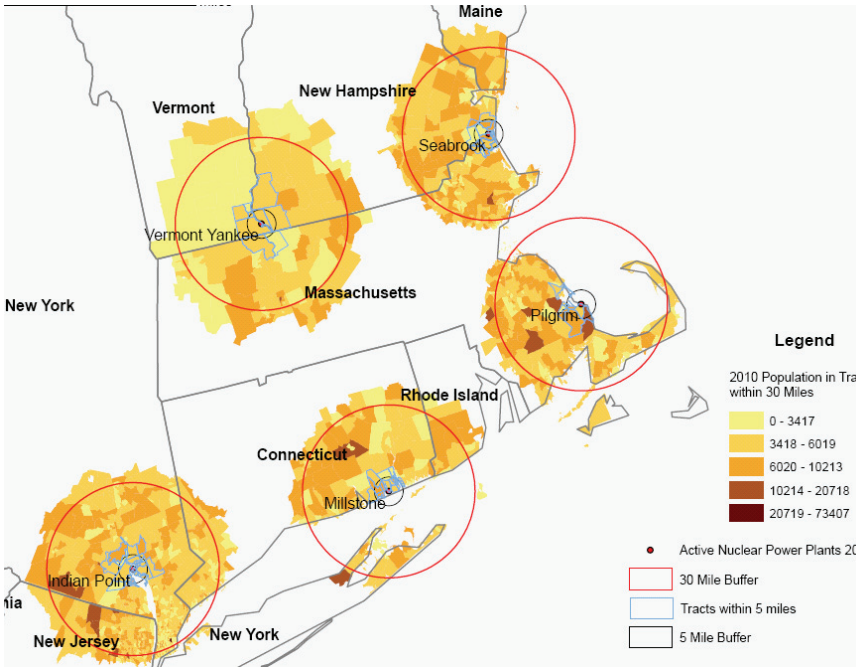

FIGURE 4.4d Exposed population from a nuclear power plant crossing state boundaries.

Massachusetts, and Maine. This means that a requirement of a study that investigates the cancer risks of populations 50 km around the Seabrook plant is that it gains access to cancer registry data from New Hampshire, Massachusetts, and Maine. This has the potential to create logistical challenges in access to state-level administrative and health outcome data.

4.3.2 Cancer Registration Data

In theory, a cancer registry includes all cases of cancer in a defined population over a defined time period (such as all cases with a diagnosis after January 1, 1990). In practice there is always a cutoff date as well (such as diagnosis before January 1, 2009). Registries also have rules about what constitutes date of diagnosis to deal with such problems as a clinical suspicion of cancer, followed by an imaging study, followed by a positive biopsy. Such information is needed for any incidence- or mortality-based ecologic study, any cohort study that compares cancer rates in different areas, or a case-control study that estimates associations.

It takes time, typically 1-2 years after the occurrence of the cancer, to get registry files that are virtually complete. Connecticut was the first state

to create and continuously run a population-based cancer registry; the data begin in 1935. In 1973, NCI established the SEER program, which now covers a sociodemographically diverse segment of 28 percent of the population in the United States. In 1992, the U.S. Congress expanded cancer surveillance to all states by establishing the National Program of Cancer Registries (NPCR), administered by the Centers for Disease Control and Prevention (CDC). In 2003, SEER and NPCR together provided 100 percent national coverage for cancer incidence reporting, with some overlap (see Figure 4.5). Cancer incidence reporting is accomplished through individual state mandates that are not entirely uniform.

4.3.2.1 SEER

The SEER program is the primary source of historical information on cancer incidence and survival in the United States. Starting in 1973, SEER originally included geographic areas comprising about 10 percent of the U.S. population. SEER expanded in the early 1990s and again in 2001 and 2010 to cover 14, 26, and 28 percent of the U.S. population, respectively. SEER currently collects and publishes cancer incidence from 15 population-

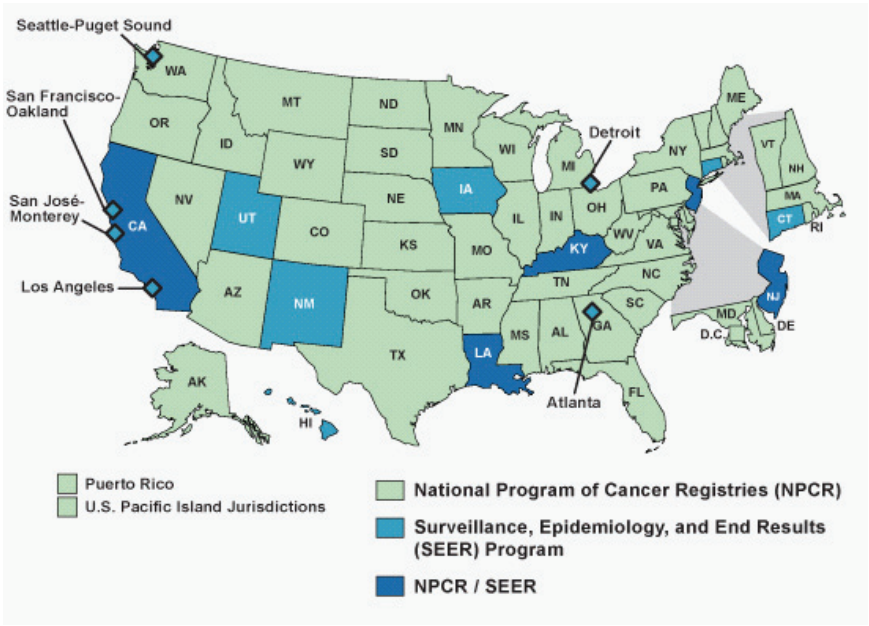

FIGURE 4.5 Cancer registration coverage within the United States. SOURCE: NPCR.

based cancer registries and is the source of much of the survival data. Incidence reporting is based on residency in a SEER-covered geographic area at time of diagnosis. Registries have data-sharing agreements with neighboring states. This is important because residents of a state may seek medical diagnosis and treatment in a state other than the one where they reside, and thus have all of their medical records elsewhere. Also, states with many part-time residents (e.g., Florida) may experience reporting delays and extra work to consolidate records. The SEER program registries collect data on patient demographics, primary tumor site and morphology, stage at diagnosis, first course of treatment, and follow-up of vital status.

The registries in SEER collect information on address, state, county, and ZIP code, and derive the census tract. The registries send geographically coded ("geocoded") county, census-tract, and census-tract certainty code[18] to SEER, but addresses are not reported to SEER and if needed must be requested from the individual state registries. Census-tract certainty of at least 90 percent is required for urban areas and at least 80 percent for rural areas for SEER participation. Census-tract variables together with other identifiers are removed from the SEER public-use research file to protect the confidentiality of data for persons in small areas.

Although the studies considered here focus on the risks of developing first cancers only, this paragraph describes the registries' regulations of recording multiple cancers, mostly to clarify that second or multiple cancers of an individual are recorded separately from the first. The SEER rules for classifying multiple primary cancers are followed by all registries in the United States (that is from all SEER and NPCR registries) and can be accessed at http://seer.cancer.gov/tools/mphrules/index.html. In general, all cancers that occur 2 or more months after the diagnosis of the first cancer are considered as separate primaries, unless the pathology report indicates that the cancer is due to recurrence or metastasis. Classification of multiple primary cancers depends on the cancer site of origin, date of diagnosis, histology, tumor behavior, and laterality of paired organs. Advances in the diagnosis and treatment of cancer leads to a rising number of cancer survivors who are at risk of developing new primary cancers.

A recent survey aimed to characterize the site-specific risks of second cancers and to provide clues to the underlying causal factors including the carcinogenic potential of treatment modalities such as chemotherapy and radiation, and/or the combination of the two treatments (SEER registries collect data on the first course of treatment of the cancer such as surgery, radiation therapy, chemotherapy). The survey used data from nine cancer registries participating in the SEER program from 1973 to 2000. Two mil-

[18]A code provided by the geocoding vendor service that indicates the quality of assignment of census tract for an individual record; address scores higher than residence ZIP code, which scores higher than ZIP code of P.O. box.

lion cancer survivors who survived at least 2 months and developed a new malignancy were included in the analysis; nearly 390,000 cases survived at least 10 years and 76,000 cases survived 20 or more years (http://seer.cancer.gov/publications/mpmono/MPMonograph_complete.pdf). About 9 percent of the survivors developed a second cancer and the risk of developing a second malignancy was dependent on multiple factors including smoking, alcohol use, viral infections and immunosuppression, genetic susceptibility, and prior cancer treatment, particularly the combination of radiotherapy and chemotherapy. The risk of developing a new malignancy was six times higher among childhood cancer survivors compared to adult survivors (SEER, New Malignancies Among Cancer Survivors: SEER Cancer Registries, 1973-2000). This finding is in agreement with previous studies of childhood cancers, which have implicated initial therapy and genetic susceptibility as major risk factors for cancers later in life (Neglia et al., 2011).

4.3.2.2 NPCR

CDC provides support for states and territories to maintain registries that provide high-quality data through the NPCR. NPCR collects data on the occurrence of cancer, including the type, extent, location, and first course of treatment. Follow-up is not included except as noted below. Before NPCR was established in 1992, 10 states had no registry and the data collected by most state registries were incomplete. Today, NPCR supports central cancer registries in 45 states, the District of Columbia, Puerto Rico, and the U.S. Pacific Island Jurisdictions. The state registries' year of operation and entry to the NPCR program is presented in Table 4.4. The NPCR data cover 96 percent of the population in the United States. Sources of information on cancer incidence are hospitals, laboratories, radiation therapy centers, medical oncology facilities, outpatient centers, and physicians' offices; the last three are regarded as less complete reporting systems but the entire data set (1995 and forward) is resubmitted each year and completeness improves over time. Data items reported are age, race, gender, state, county, ZIP code and census tract, date of diagnosis, primary site, histology, staging, and follow-up information that includes vital status by linkage with the National Death Index. Census tract has been a required field since 2003.

4.3.2.3 North American Association of Central Cancer Registries (NAACCR)

NAACCR is an oversight group established in 1987 to set uniform standards for cancer registration as well as electronic data record struc-

TABLE 4.4 State Registries' Year of Operation and Entry to the NPCR Program

State	Registry Year	NPCR Year	State	Registry Year	NPCR Year
Alabama	1996	1996	Montana	1979	1995
Alaska[b]	1996	1996	Nebraska	1987	1995
Arizona[b]	1981	1995	New Mexico[a]	1966	N/A
Arkansas	1996	1996	Nevada	1979	1995
California[b]	1988	1995	New Hampshire	1986	1995
Colorado	1968	1995	New Jersey[a]	1979	1995
Connecticut[a]	1935	N/A	New York	1940	1996
Delaware	1972	1997	North Carolina	1986	1995
District of Columbia	1987	1996	North Dakota	1997	1997
Florida	1981	1995	Ohio	1992	1996
Georgia[b]	1995	1995	Oklahoma[b]	1997	1997
Hawaii[a]	1960	N/A	Oregon	1996	1996
Idaho	1969	1995	Pennsylvania	1982	1995
Illinois	1985	1995	Rhode Island	1986	1995
Indiana	1987	1995	South Carolina	1996	1996
Iowa[a]	1973	N/A	South Dakota	2001	2001
Kansas	1968	1995	Tennessee	1986	1999
Kentucky[a]	1991	1995	Texas	1976	1995
Louisiana[a]	1974	1995	Utah[a]	1966	N/A
Maine	1983	1995	Vermont	1992	1996
Maryland	1982	1996	Virginia	1970	1996
Massachusetts	1982	1995	Washington[b]	1992	1995
Michigan[b]	1981	1995	West Virginia	1993	1995
Minnesota	1988	1995	Wisconsin	1976	1995
Mississippi	1996	1996	Wyoming	1967	1996
Missouri	1972	1996			

NOTES: N/A, states are not part of the NPCR program. Registry year is based on year of operation of the registry. NPCR year is based on first diagnosis year for which cancer cases were reportable to CDC.

[a]Entire state is part of the SEER program.

[b]Part of the state or a selected population within the state is part of the SEER program.

SOURCE: NPCR (for NPCR year) and Betsy Kochler, Executive Director, NAACCR (for registry year).

ture. CDC, NCI, and other sponsoring organizations support it. All NPCR and SEER registries are members of NAACCR. NAACCR develops and promotes uniform data standards for cancer registration; provides education and training; certifies population-based registries; and aggregates and publishes data from central cancer registries. Data down to county level are released by NAACCR beginning in 1995, when NPCR started. Census-tract or address data for any year, or county data prior to 1995, must be requested from individual states. A major role of NAACCR is to provide state certification for quality of cancer registration.

4.3.2.4 *Assessing the Quality of Cancer Registration: National and International Efforts*

The utility of cancer incidence data for research depends on the quality of the data. Researchers want to ensure that the data they use for their studies meet the highest standards of quality and reliability and therefore can have faith in their analyses. The two main factors that define the quality of a cancer registry are the completeness of case ascertainment and the accuracy of the details retrieved for each case. Cancer incidence data quality varies by state.

CDC has established standards for quality and completeness for NPCR registries. Data are evaluated each year and only data from those registries that meet NPCR standards are used for reporting of cancer incidence. The standards are presented in Table 4.5.

Data in the SEER and NPCR data sets are combined to produce the United States Cancer Statistics (USCS) data set. The data set is produced by NCI and CDC in collaboration with NAACCR. Only cancer registries that demonstrated that cancer incidence data were of high quality are included in the data set. The criteria for USCS publication are also presented in Table 4.5. Data from all states and the District of Columbia met the USCS data quality criteria for 2008, but data from only 44 states and three U.S. Census regions (covering 90 percent of the U.S. population) met these criteria for the entire period 1999-2008 (Centers for Disease Control and Prevention, 2011).

In 1998 NAACCR developed a set of data standards for cancer registration and certified data quality beginning with 1995 data. NAACCR independently reviews the data from member registries for their completeness, accuracy, and timeliness and provides silver or gold registry certifications

TABLE 4.5 Summary of Data Quality Criteria and Standards

	NAACCR Registry Certification			
Criteria	Gold	Silver	NPCR	USCS
Completeness	≥95%	≥90%	≥95%	≥90%
% Passing EDITS[b]	100%	≥97%	≥99%	≥97%
Death certificate only cases	≤3%	≤5%	≤3%	≤5%
Duplicate reports	≤1/1,000	≤2/1,000	≤1/1,000	N/A
Missing data field				
age, gender, county	≤2%	≤3%	≤2%	≤3%[a]
race	≤3%	≤5%	≤3%	≤5%

[a]Completeness of county is not part of the criteria in the USCS data sets.
[b]http://www.cdc.gov/cancer/npcr/tools/edits/editintr.htm.
SOURCE: NAACCR, NPCR, and USCS.

(Table 4.6). States that do not meet the standards are uncertified. Nearly all states in the United States have received a silver or gold certification for the most recent years. The data quality criteria and standards for 2011 are presented in Table 4.5.

A cancer registry may not be able to collect complete information on all the incidence cancer cases within the timeframe for submission of the data to NAACCR, or may not be able to collect the information at all. (Of course, the actual number of incident cancer cases that a registry should have captured is an unobserved quantity that can be estimated by available data. The methodology used by NAACCR is described elsewhere [Das et al., 2008]). Having a high proportion of cases identified only by death certificates suggests that the procedures and sources used for case finding are inadequate or that matching to other sources is incomplete.

Similarly, a high proportion of duplicate reports suggests that the data "cleaning" processes are insufficient. NAACCR has been criticized for looking at the accuracy and timeliness of data at a single time point; recertification based on correctness of initially reported data has been suggested (Das et al., 2008).

Using cancer registration data for the years during which states had compromised quality of data is problematic because data quality may vary from place to place within the state. This may lead to bias and errors in comparing cancer frequency in these areas; the scope for such errors is reduced when data quality for the state as a whole is high.

It is not always clear how investigators can assess the quality of cancer registration for data prior to the NAACCR certification system (1995 data). Since the 1960s, the International Agency for Research on Cancer (IARC) publishes cancer incidence data from populations all over the world for which good quality data are available. The purpose of the publication is to compare rates of cancer incidence from different populations and draw conclusions on differences between and changes in cancer patterns by geographic area and formulate hypotheses about causes of cancer. The most recent publication (Volume IX) covers the period 1998-2002 and presents statistics from 60 countries and 225 registries, of which 54 are in North America (Curado et al., 2007). The publication provides a comprehensive summary of the participating states in the United States and includes information on the registration area covered, cancer care facilities that provide the cases' information, registry structures and methods, and use of the data (for example, annual publications, support to researchers or policy makers, and intervention efforts). The publication also includes a table with the geographic coverage in the nine successive volumes of cancer incidence in the five continents which has been replicated here to present the data for the United States (Table 4.7).

As within the United States, the cancer registry certification system did

TABLE 4.6 Summary of State Cancer Registries' Data Quality by NAACCR Certification Methods

State	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008
Alabama	U	U	U	U	U	S	S	G	S	G	G	G	G	G
Alaska	U	U	C	G	G	G	G	G	G	G	G	G	S	G
Arizona	C	G	C	G	U	S	S	S	U	U	U	U	S	S
Arkansas	U	U	U	U	U	U	S	U	S	S	S	G	G	G
California	C	G	C	G	G/S	G/S	G/S/U	G/S	G	G	G	G/S	G	G
Colorado	C	G	C	G	G	G	S	G	G	G	G	G	G	G
Connecticut	C	G	C	G	G	G	U	G	G	G	G	G	G	G
Delaware	U	U	C	S	G	U	U	S	G	G	G	G	G	G
District of Columbia	U	U	C	G	G	G	G	S	G	S	U	S	S	U
Florida	U	U	C	S	S	G	G	G	G	G	G	G	G	G
Georgia	C/U	G/U	C/U	S/U	G/U	G	G	G	G	G	G	G	G/S	G
Hawaii	C	G	C	G	S	G	G	G	G	G	G	G	G	G
Idaho	U	G	C	G	G	G	G	G	G	G	G	G	G	G
Illinois	U	G	C	G	G	G	G	G	G	G	G	G	G	G
Indiana	U	U	U	U	U	S	G	G	S	G	S	G	G	G
Iowa	C	G	C	G	G	G	G	G	G	G	G	G	G	G
Kansas	U	U	C	S	S	G	G	U	G	G	G	G	G	G
Kentucky	C	G	C	G	G	G	G	G	G	G	G	G	G	G
Louisiana	C	S	C	G	G	G	G	G	G	G	G	G	G	G
Maine	U	U	U	U	U	U	G	G	G	G	G	G	G	G
Maryland	U	U	C	G	G	G	U	G	G	U	U	U	G	G
Massachusetts	U	U	C	G	G	G	G	G	G	G	G	G	G	G
Michigan	C/U	G	C	G/S	G	G	G	G	G	G	G	G	G	G
Minnesota	U	S	C	U	G	G	G	G	G	U	G	G	G	G
Mississippi	U	U	U	U	U	U	U	U	U	S	S	G	G	G
Missouri	U	U	U	S	S	G	S	G	G	G	G	G	G	G

continued

TABLE 4.6 Continued

State	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008
Montana	U	U	C	S	S	S	S	S	G	G	G	S	S	G
Nebraska	C	G	C	G	G	G	G	G	G	G	G	G	G	G
Nevada	U	U	U	U	U	G	G	G	G	G	G	G	U	G
New Hampshire	U	U	C	G	S	G	G	G	S	G	G	G	G	G
New Jersey	C	G	C	G	G	G	G	G	G	G	G	G	G	G
New Mexico	C	G	C	G	S	S	S	G	U	S	G	S	S	S
New York	U	U	C	G	G	G	G	G	G	G	G	G	G	G
North Carolina	U	U	C	G	S	G	S	S	S	G	S	G	G	G
North Dakota	U	U	U	G	G	S	U	G	G	G	G	G	G	G
Ohio	U	G	U	S	S	S	S	S	U	S	U	U	S	S
Oklahoma	U	U	U	U	U	U	G	G	G	G	G	G	G	G
Oregon	U	G	C	G	S	G	G	G	G	G	G	G	G	G
Pennsylvania	U	U	C	G	G	G	U	G	G	G	G	G	G	G
Rhode Island	U	G	C	G	G	G	G	G	G	G	G	G	G	G
South Carolina	U	U	C	G	G	S	S	G	G	G	G	G	G	S
South Dakota	U	U	U	U	U	U	U	U	G	S	G	S	G	G
Tennessee	U	U	U	U	U	U	U	U	U	U	G	G	G	G
Texas	U	U	U	U	U	U	U	U	G	G	G	G	G	G
Utah	C	G	C	G	G	S	G	G	S	G	G	G	G	G
Vermont	U	U	U	U	U	U	U	U	S	G	G	G	G	G
Virginia	U	U	U	U	U	U	U	U	S	S	G	G	G	U
Washington	C/U	G	C	G/S	G	G	G	G	G	G	G	G	G	G
West Virginia	C	S	C	G	G	G	G	G	G	G	G	G	G	G
Wisconsin	C	G	C	G	G	G	G	G	G	U	U	U	S	U
Wyoming	U	U	C	G	G	G	S	U	U	U	G	G	S	G

NOTE: C, certified; S, silver; G, gold; U, uncertified; multiple certifications if more than one registries exist within the state.
SOURCE: NAACCR.

TABLE 4.7 Geographic Coverage in the Nine Successive Volumes of IARC's Cancer Incidence in Five Continents

	Vol. I	Vol. II	Vol. III	Vol. IV	Vol. V	Vol. VI	Vol. VII	Vol. VIII	Vol. IX
Alabama									1998-2002
Black									1998-2002
White									1998-2002
Alaska									1998-2002
Arizona									1998-2002
California									1998-2002
Asian and Pacific Islander									1998-2002
Black									1998-2002
Hispanic White									1998-2002
Non-Hispanic White									1998-2002
California, Alameda County:									
Black		1960-64	1969-73	1973-77	1978-82	1983-87			
White		1960-64	1969-73	1973-77	1978-82	1983-87			
California, Central Valley:									
Hispanic							1988-92		
Non-Hispanic White							1988-92		
California, Greater San Francisco Bay Area:									
Black			1969-73	1973-77	1978-82	1983-87	1988-92	1993-97	1998-2002
Chinese			1969-73	1973-77	1978-82		1988-92		1998-2002
Filipino					1978-82		1988-92		1988-2002
Hispanic White							1988-92	1993-97	1998-2002
Non-Hispanic White							1988-92	1993-97	1998-2002
Japanese				1973-77	1978-82		1988-92		1998-2002
White			1969-73	1973-77	1978-82	1983-87			

continued

TABLE 4.7 Continued

	Vol. I	Vol. II	Vol. III	Vol. IV	Vol. V	Vol. VI	Vol. VII	Vol. VIII	Vol. IX
California, Los Angeles County:									
Black				1972-77	1978-82	1983-87	1988-92	1993-97	1998-2002
Chinese				1972-77	1978-82	1983-87	1988-92	1993-97	1998-2002
Filipino					1978-82	1983-87	1988-92	1993-97	1998-2002
Hispanic White							1988-92	1993-97	1998-2002
Non-Hispanic White							1988-92	1993-97	1998-2002
Japanese				1972-77	1978-82	1983-87	1988-92	1993-97	1998-2002
Korean					1978-82	1983-87			
Spanish-surnamed White				1972-77	1978-82	1983-87			
Other White				1972-77	1978-82	1983-87			
Colorado									1998-2002
Connecticut	1960-62	1963-65	1968-72	1973-77			1988-92	1993-97	1998-2002
Black					1978-82	1983-87	1988-92	1993-97	1998-2002
White					1978-82	1983-87	1988-92	1993-97	1998-2002
District of Columbia									1998-2002
Black									1998-2002
White									1998-2002
Florida									1998-2002
Black									1998-2002
White									1998-2002
Georgia									1998-2002
Black				1975-77	1978-82	1983-87	1988-92	1993-97	1998-2002
White				1975-77	1978-82	1983-87	1988-92	1993-97	1998-2002
Idaho									1998-2002
Illinois									1998-2002
Black									1998-2002
White									1998-2002

Indiana								1998-2002
Iowa		1969-71	1973-77	1978-82	1983-87	1988-92	1993-97	1998-2002
Kentucky								1998-2002
Louisiana								1998-2002
Black								1998-2002
White								1998-2002
Louisiana, Central Region:								
Black						1988-92	1993-93	
White						1988-92	1993-97	
Louisiana, New Orleans								
Black			1974-77	1978-82	1983-87	1988-92	1993-97	1998-2002
White			1974-77	1978-82	1983-87	1988-92	1993-97	1998-2002
Maine								1998-2002
Massachusetts								1998-2002
Michigan								1998-2002
Black								1998-2002
White								1998-2002
Michigan, Detroit:								1998-2002
Black		1969-71	1973-77	1978-82	1983-87	1988-92	1993-97	1998-2002
White		1969-71	1973-77	1978-82	1983-87	1988-92	1993-97	1998-2002
Missouri								1998-2002
Black								1998-2002
White								1998-2002
Montana								1998-2002
Nevada	1959-66							
New Jersey								1998-2002
Black							1993-97	1998-2002
White							1993-97	1998-2002

continued

TABLE 4.7 Continued

	Vol. I	Vol. II	Vol. III	Vol. IV	Vol. V	Vol. VI	Vol. VII	Vol. VIII	Vol. IX
New Mexico						1983-87		1993-97	1998-2002
American Indian			1969-72	1973-77	1978-82		1988-92	1993-97	1998-2002
Hispanic			1969-72	1973-77	1978-82				
Hispanic White							1988-92	1993-97	1998-2002
Non-Hispanic White							1988-92	1993-97	1998-2002
Other White (Anglo)			1969-72	1973-77	1978-82				
New York City					1978-82	1983-87			
New York State									1998-2002
Black									1998-2002
White									1998-2002
New York State (less New York City)	1959-61		1969-71	1973-77	1978-82	1983-87		1993-97	
Black								1993-97	
White								1993-97	
NPCR									1998-2002
Black									1998-2002
White									1998-2002
Ohio									1998-2002
Black									1998-2002
White									1998-2002
Oklahoma									1998-2002
Oregon									1998-2002
Pennsylvania									1998-2002
Black									1998-2002
White									1998-2002

Rhode Island								1998-2002
SEER (9 Registries)								1998-2002
Black					1983-87	1988-92	1993-97	1998-2002
White					1983-87	1988-92	1993-97	1998-2002
SEER (14 Registries)								1998-2002
Asian and Pacific Islander								1998-2002
Black								1998-2002
Hispanic White								1998-2002
Non-Hispanic White								1998-2002
South Carolina								1998-2002
Black								1998-2002
White								1998-2002
Texas								1998-2002
Black								1998-2002
White								1998-2002
Texas, El Paso								
Latin	1960-66	1968-70						
Other than Latin	1960-66	1968-70						
Utah		1966-70	1973-77	1978-82	1983-87	1988-92	1993-97	1998-2002
Vermont								1998-2002
Washington								1998-2002
Washington, Seattle			1974-77	1978-82	1983-87	1988-92	1993-97	1998-2002
West Virginia								1998-2002
Wisconsin								1998-2002

SOURCE: IARC's Cancer Incidence in Five Continents, Volume IX.

not exist until the 1995 data; the IARC judgment for "good quality" could be potentially used to select registries prior to 1995 that can be included in an epidemiologic study.

Independent of the certification of cancer registries by NAACCR or other systems, the quality of cancer registration will need to be judged following close examination of the data for each state cancer registry individually.

4.3.2.5 State Registries

Collecting and maintaining high-quality cancer incidence data requires time and experience, and data in the first few years of a new registry need to be viewed with caution. Individual state cancer registries collect information on state, county, ZIP code, and address and derived census tract. Accessing cancer registry data for research, in particular for multistate data, is complicated and challenging because procedures for data use and confidentiality vary by location. On September 2010, CDC launched Cancer Registry Data Access (CRDA). The purpose of CRDA is to (a) provide understanding of comprehensive requirements and barriers of cancer registry data access for research, (b) identify optimal state and registry rules and policies, (c) investigate methods for streamlining the IRB processes and pilot test the best methods, and (d) assist researchers in managing the process. Basic and special requirements for data access vary substantially among states. The initial summary of information is expected to be completed September 2013 and will continue as needed.

To better understand what data are available in individual cancer registries for the immediate need of this study, the committee requested information regarding cancer incidence from the states that have or have had a nuclear facility. A letter template is presented in Appendix K. A summary of the results is presented in Table 4.8. Briefly, data were requested from 38 states, and 31 states responded (81 percent). The median year for which complete incidence data exist is 1992; cancer registration goes as far back as the 1970s for three respondent states, and to the 1980s for eight respondent states. All states that responded to the request had complete cancer registration by 1999. For convenience, Table 4.8 also summarizes availability of cancer mortality data, which is further discussed in Section 4.3.3.

The letter responses received from the cancer registries and vital statistics offices identified several potential problems related to the availability and release of data. Although not strictly quantitative, examples of these obstacles are discussed here.

As expected, the year that complete data are available in a registry and year that the registry started operation may be different. For example, the cancer registry in New Mexico was established in 1966 and initiated state-

TABLE 4.8 Availability of Cancer Incidence and Mortality Data of States that Have or Have Had a USNRC-Licensed Nuclear Facility

State	Information Received from		First Year Data Are Available and:				
	Cancer Registry	Vital Statistics Office	M=mortality I=incidence	Complete	Address Is Present	Census Tract Is Present	In Electronic Format
Alabama		✓	M	1991	1991	—	1991
	✓		I	1996	1996	—	1996
Arizona			M	1970	1989	1975	1970
			I	1995	1990	1995	1990
Arkansas			M				
	✓		I	1997	1997	1997	1997
California	✓		M	1988	1988	1988	1988
	✓		I	1988	1988	1988	1988
Colorado		✓	M	1975	1975	1990	1975
	✓		I	1988	1988	1995	1988
Connecticut			M				
			I				
Florida	✓		M	1970	1970	1991	1970
	✓		I	1981	1981	1981	1981
Georgia	✓		M	1980	1980	1995	1980
	✓		I	1998	1998	1998	1998
Illinois			M	1950	2008	1979	1970
	✓		I	1986	1986	—	1986
Iowa			M				
	✓	✓	I	1973	1973	1990	1973
Kansas		✓	M	1995		—	1995
			I				
Kentucky			M				
	✓		I	1995	1995	1995	1995
Louisiana	✓		M	1969	1969		1969
	✓		I	1988	1988	1995	1988
Maine			M				
	✓		I	1995	1995	—	1983
Maryland	✓		M	1970	1987	1995	1970
	✓		I	1992	1992	2000	1995
Massachusetts	✓		M				
	✓		I	1982	1982	1982	1982
Michigan			M	1970	2000	2000	1970
	✓		I	1985	1985	1985	1985
Minnesota			M				
	✓		I	1988	1988	1988	1988
Mississippi			M				
			I				

continued

TABLE 4.8 Continued

State	Information Received from: Cancer Registry	Information Received from: Vital Statistics Office	First Year Data Are Available and: M=mortality I=incidence	Complete	Address Is Present	Census Tract Is Present	In Electronic Format
Missouri			M				
			I				
Nebraska			M				
		✓	I	1995	1987	1990	1995
Nevada			M				
		✓	I	1995	1995	—	1979
New Hampshire			M				
	✓		I	1990	1990	1990	1990
New Jersey	✓		M	1979	1979		1979
	✓		I	1979	1979		1979
New Mexico			M	1965	1980	2006	1980
			I	1973	1966	1973	1966
New York			M				
	✓		I	1976	1995	1995	1976
North Carolina		✓	M	1913	2000	2001	1956
	✓		I	1990	1990	1990	1990
Ohio	✓		M				
	✓		I	1996	1996	1996	1996
Oregon		✓	M	1971	2006	2007	1989
	✓		I	1996	1996	1996	1996
Pennsylvania			M	1959	1979	—	1959
			I	1985	1985	2000	1985
South Carolina	✓		M				
	✓		I	1996	1996	1996	1996
South Dakota			M				
			I				
Tennessee			M				
			I				
Texas			M				
	✓		I	1995	1995	1995	1995
Vermont	✓		M	1985	2008	—	1985
	✓		I	1994	1994	2001	1994
Virginia			M				
	✓		I	1999	1990	1998	1990
Washington		✓	M			1980	
	✓		I	1992	1992	1992	1992
Wisconsin			M				
			I				

SOURCE: Based on responses to the letter shown in Appendix K.

wide coverage in 1969; the most reliable data in accordance with standards set by the SEER program are for 1973 onward. Similarly, the cancer registry in Virginia started operation in 1979, but complete data are not available until 1999. In Nebraska, Maine, and Nevada the first years of complete data are 1995-1996, which coincides with the year the registries joined the NPCR program. Using cancer registry data prior to NPCR involvement requires further examination for consistency and comparability with the data collected post NPCR who implemented uniform rules across states. For New York, statewide data are available from 1976; however, the reference year is 1996 for the NPCR program. When the registry became part of NPCR it adopted the SEER multiple primary rules which are considered the national standard; previously the state was using the IARC rule for counting primary tumors which allows only one primary per site per person per lifetime. This change is important for the interpretation of cancer incidence statistics. The extent of the effect for each cancer site depends on the site-specific probability of multiple primaries.

Address at time of diagnosis is being collected widely at all times. However, for many rural residents, residential information may be expressed as P.O. boxes and rural route numbers. This may influence the quality of geocoded data in these areas and it likely is a problem throughout the United States particularly when going back in time. Indeed, Boice and colleagues have emphasized that mailing addresses in small rural areas may not always reflect actual residences, and validation by contacting area postmasters and using Census Bureau geocoding information may be necessary to prevent misleading conclusions (Boice et al., 2003).

Census tract became a required field by NPCR in 2003. However, some states were recording this information before it became a required field. For example, Iowa has recorded census-tract information since 1990. Some states that are part of the NPCR program, such as Maine and Alabama, do not collect census-tract information. As the Maine Cancer Registry director informed the committee, although NPCR made census tract a required field, it is not enforced. Since the decennial census may lead to changes in census tracts, reconstructing census tract from address is not straightforward and would require expertise in geocoding addresses; such expertise is available from some contractors and GIS professionals.

Several cancer registries noted the importance of knowing and understanding the methodology used to construct census-tract data. For example, to create the census-tract data for 1998, an investigator may have used population data from the 1990 census as it would have been available in 1998, or recalculated retrospectively by using the 2000 census data when those became available. The Massachusetts cancer registry noted that for the 1982 cancer registration data, the 1990 tracts were used, since in the 1980 census not all Massachusetts counties had defined tracts. In New

Mexico, for incident cancer cases diagnosed in the calendar years 1973-1977, the 1970 census-tract boundaries would be assigned; for 1978-1987 the 1980 census-tract boundaries; for 1995-2000 incident cases would be assigned both the 1990 and 2000 census-tract boundaries. The quality of the census-tract determination depends on the availability of residential information in source records and as mentioned earlier this may influence the quality of geocoded data in rural areas.

The data item "census-tract certainty" documents the quality of residential information that was used to assign census tract for each case. The State of Illinois emphasized that the registry would not release census-tract data information for research, and thus they were reluctant to inform the committee when the registry started collecting the information, or if the information exists. However, if justified by research needs, address information from the Illinois cancer registry may be released upon review and approval of the application. Interestingly, although generally census-tract data exist for cancer registries, mortality data have not been routinely geocoded. Some vital statistics offices have data only for recent years while others (for example, Pennsylvania) will start in the near future.

Although cancer registries attempt to collect information on place of birth (and in the context of this study, one may need the information to make assumptions as to whether the person lived in the same place since birth), the information is largely missing from the medical record, which is the primary source of cancer diagnosis. For example, for the state of New York, birth place is missing for 26 percent of cases diagnosed in the period 1995-2008; for Texas birth place is missing for 42 percent; and for Illinois for 75 percent. Some states reported that the information often becomes available from death certificates. When it is available at all, place of birth is poorly reported and is coded only to the state level (or the national level for persons born outside the United States).

When states were asked about the quality and completeness of the data, they commonly referred to the certification received by NAACCR. Although "missing county" is a criterion for data quality, missing address is not and this may be a problem when data in small geographic units are needed for analysis.

Active follow-up for vital status is performed only by SEER registries. There is some passive follow-up in all states queried, commonly through linkages with the state's vital records office, national death index, and social security death index. For states with more than one cancer registry, such as Washington, active follow-up is performed for the SEER registry only. More specifically, of the 39 counties within Washington, active follow-up occurs in the 13 counties that comprise the Washington SEER registry, while passive follow-up alone occurs in the remaining 26 counties.

All states that responded to the request for information on procedures

for release of the data reported that approval is required following submission of a detailed study protocol that may include data elements requested, analysis plan, and plan for reporting and dissemination. (The committee was advised to use the NAACCR data element code book for communication of variables requested as it is a uniform language among all states.) More than one level of approval may be required from some states. For example, for investigators outside the University of New Mexico, which maintains the cancer registry for the state, additional approval must be obtained from the senior leadership team at the cancer registry (i.e., Principle Investigator, Medical Directors, and Program Manager), the New Mexico Department of Health-Office of the State Epidemiologist, and the Office of Human Research Protections at the University of New Mexico Health Sciences Center. Application forms are available on each of the states' websites. Review processes vary with the protocol and the frequency IRB or other equivalent committees meet, but a decision within 1 to 6 months seemed to be the general rule. Alabama, Louisiana, and Tennessee place a limit on the studies the cancer registries support either due to staffing shortages or to minimize the patient burden when patient contact is required.

Table 4.9 summarizes the information on approval requirements for cancer registries (document Cancer Registry Data Access for Research was created January 11, 2012, by CDC). According to the CDC document on IRB requirements for central cancer registries, all states but Wisconsin permit the release of state resident's identifiable data to researchers, but three states (Georgia, New Mexico, and Hawaii) require sponsorship from a local researcher. Special requirements such as parental and/or physician consent and a more difficult approval process exist for release of information for pediatric research in 15 states. For research projects that require patient contact and consent for release of confidential data, the contact (or initial contact) is required to be established by registry in some states and by the researcher in other states.

Time and cost for release of the data are dependent on what is being requested and staff availability; data submission to NAACCR is the priority. Some states including Washington, Maryland, Massachusetts, Virginia, and Arizona do not charge for data release, although that is subject to policy changes. From those that charge for data release, different methods for estimating costs are in place. Oregon State charges $55 per hour, and Vermont charges $34 per hour. North Carolina charges a standard fee of $1,000 for a file that includes up to 50,000 records and an additional $100 for each additional 10,000 records. According to Illinois, data sets prepared for analysis can run anywhere from $5,000 to $10,000. Registries that are understaffed such as Maine (reported 50 percent staffing level, including no registry-based epidemiologist) would need to contract an epidemiologist to work on the data request. Currently the hourly rate is $75.00 per hour.

TABLE 4.9 Cancer Registry Research Approval Process

Level of Complexity[a]	State	Approval	Pediatric Special Requirements	Fee	Timeframe (months)
3	AK	GROUP	Yes	Yes	Varied
3	AL	CR Director/Group/CR IRB	Yes	Yes	Varied
1	AR	Epidemiologist/Group			<2
2	AZ	CR IRB			Varied
3	CA	CR IRB	Yes	Yes	<2
3	CO	CR IRB			2-6
2	CT	CR IRB			<2
2	DC	CR IRB			<2
2	DE	Epidemiologist/Group/CR IRB			2-6
1	FL	Group/CR IRB		Yes	<2
3	GA	CR IRB			<2
3	HI	Group/CR IRB		Yes	<2
1	IA	Epidemiologist/CR IRB/Group			<2
3	ID	Group	Yes	Yes	<2
2	IL	CR IRB		Yes	Varied
1	IN	Group			<2
2	KS	Group/CR IRB/CR IRB/Group		Yes	Varied
1	KY	Group		Yes	<2
3	LA	Group/CR IRB		Yes	<2
2	MA	Group/Commissioner			Varied
2	MD	CR Director/Officials/CR IRB/Dept Health Sec			Varied
3	ME	CR Director/Group/CR IRB	Yes	Yes	2-6
3	MI	Group/Group/Dept Health Director	Yes	Yes	2-6
3	MN	Group/Group	Yes	Yes	2-6
3	MO	Group/CR IRB/CR IRB		Yes	2-6
1	MS	Group		Yes	<2
1	MT	Group/Bureau Chief/Admin/Group		Yes	<2
1	NC	Group/Group		Yes	<2
2	ND	Group/Group			2-6
3	NH	Group/CR IRB	Yes		<2
2	NJ	Group/CR IRB		Yes	Varied
3	NM	CR Director/CR IRB		Yes	Varied
2	NY	Group/CR IRB		Yes	2-6
3	NV	CR Biostatistician/CR Manager/ Bureau Chief	Yes	Yes	2-6
2	OH	Group/CR IRB			2-6
2	OK	CR/CR IRB/Commissioner			Varied
3	OR	CR Director/CR IRB/Group	Yes	Yes	2-6
1	PA	Group		Yes	<2
3	PR	CR Director&Coord/CR IRB/ Group	Yes		2-6
3	RI	CR Director/CR IRB	Yes		2-6

Required			Physician/Patient Auth by		
Sponsorship	Human Subject Protection Training	Patient Contact Studies Allowed	CR	Researcher	Limit Number of Studies
		No	N/A	N/A	
			Physician		Yes
			Pt		
				Physician	
				Pt	
	Yes		Physician		
	Yes			Physician	
				Pt	
	Yes			Pt	
Yes			Physician/Pt		
Yes	Yes		Pt		
	Yes		Pt		
			Physician/Pt		
			Pt		
				Physician	
	Yes		Pt		
			Pt		
	Yes				Yes
	Yes		Pt		
			Pt		
				Physician	
	Yes		Physician/Pt		
			Physician/Pt		
	Yes		Pt		Yes
			Pt		
				Pt	
			Physician	Pt	
	Yes		Pt		
Yes	Yes		Physician/Pt		
			Physician/Pt		
	Yes		Physician/Pt		
			Physician/Pt		
				Pt	
	Yes			Physician	
				Pt	

continued

TABLE 4.9 Continued

Level of Complexity[a]	State	Approval	Pediatric Special Requirements	Fee	Timeframe (months)
1	SD	CR Director/Group/Group		Yes	<2
2	SC	Group/CR IRB		Yes	2-6
3	TN	Group/CR IRB			Varied
3	TX	CR Director/CR IRB/Group/ Commissioner	Yes		<2
3	UT	Group/CR IRB	Yes	Yes	Varied
1	VA	CR IRB /Commissioner			<2
2	VT	CR Director/Group/CR IRB		Yes	2-6
2	WA	CR IRB/Asst Sec			2-6
3	WV	CR Director/Group			2-6
3	WI[b]				
3	WY	Group/CR IRB	Yes		<2

NOTE: Group refers to committee, board, or review group: CR, cancer registry, CR IRB, IRB(s) affiliated with the cancer registry.

[a]1, less complex process; 3, more complex process.

[b]Confidential data release policy under development; currently data linkage only.

SOURCE: CDC, communication with Christie Eheman, Chief, Cancer Surveillance Branch.

4.3.2.6 Pediatric Cancer Registries

In contrast to cancers in adults, cancers in children are rare, making up less than 1 percent of all cancers diagnosed each year. About 11,200 children in the United States under the age of 15 will be diagnosed with cancer in 2011. Leukemia is the most common childhood cancer, accounting for about one-third of all cancers in children. Brain and other nervous system tumors, the second most common cancer in children, make up about 27 percent of childhood cancers (American Cancer Society, http://www.cancer.org/).

Many childhood cancers are curable with modern therapy. Five-year survival rates for all stages and all sites of cancer for children, aged <15 years, diagnosed from 1999-2006 was 82 percent (http://seer.cancer.gov/). Overall, this is great success compared to the 1970s, when the 5-year survival rate was less than 50 percent. This improvement in survival mostly reflects the improved leukemia treatments. For brain tumors, the 60 percent 5-year survival rate has improved slightly in the past 25 years.

Pediatric cancer incidence can be derived for any site or age group from individual state cancer registry data and from SEER. Unlike the situation

Required			Physician/Patient Auth by		
Sponsorship	Human Subject Protection Training	Patient Contact Studies Allowed	CR	Researcher	Limit Number of Studies
				Physician	
			Physician/Pt		
	Yes		Pt		Yes
	Yes			Pt	
	Yes		Pt		
	Yes				
	Yes		Physician/Pt		
			Pt		
					Yes

in some European countries, such as Germany and Switzerland, there is no national population-based childhood cancer registry in the United States. The closest approximation of a pediatric cancer registry is the Childhood Cancer Research Network (CCRN), which is built on the Children's Oncology Group (COG), an NCI-sponsored clinical trials cooperative group comprising more than 200 institutions, mostly in the United States and Canada, which collectively see and treat upward of 80 percent of children under the age of 15 with cancer (Steele et al., 2006).

The CCRN potentially could provide a resource for identification of cases for an epidemiologic study. The CCRN, after years in development and planning, was launched in 2001, as a pilot with funding from the NCI and participation by 23 COG institutions (Steele et al., 2006). The pilot experience showed roughly 96 percent patient and/or parent agreement to participate, after IRB approval and informed consent, for release of personal identifiers and possible future contact. Since completion of the pilot in 2007, the CCRN has been expanded groupwide with 100 percent participation of about 200 institutions obtaining IRB approval and roughly 20,500 cases enrolled as of April 2011. However, CCRN has definite limitations, including variation in registration rates by institution, geography, age, and

cancer type. There is also the problem that not all children and adolescents with cancer in the United States are seen at a participating COG member institution. A collaborative study of COG investigators and SEER analyzed 10,108 cases of cancer in children under the age of 20 years and reported to 11 SEER registries between 1992 and 1997; of these, 5796 (57.5 percent) were registered with COG. Rates varied by geographic region and by age, with rates found to be highest for children <5 years (74.3 percent). Rates were also higher for children with more advanced disease (Liu et al., 2003). Thus, while the CCRN and COG institutions provide a framework for collection of cases and obtaining informed consent, ascertainment of cases would be biased and incomplete. As the formation of the CCRN is relatively recent, the data could not be used for study of childhood cancer cases diagnosed prior to 2001.

4.3.3 Cancer Death Data

Over the years, the most common and routinely collected cancer data are related to mortality. Kelsey et al. (1996) have comprehensively described the process of reporting the event of death to the national statistics and their summary is presented here. After completion of the death certificate, the funeral director or other person in charge of interment is responsible for completing the parts of the death certificate that require personal information about the deceased and for filing the certificate with the local registrar of the district in which the death occurred. A physician must complete and sign the medical certification section and enter the cause of death. If a physician has not been in attendance or the cause of death is thought to be the result of an accident, homicide, or suicide, the medical examiner or coroner must sign the certificate. The local registrar verifies that the death certificate has been completed, keeps a copy, and sends the certificate to the state registrar. After querying the local registrar about any incomplete or inconsistent information, the state registrar keeps one copy and sends another copy to the National Vital Statistics System of the NCHS. The NCHS is a division of the CDC and as such is under the U.S. Department of Health and Human Services. Death registration is considered virtually complete.

NCHS then summarizes the mortality data and documents the health status of the population in the United States. NCHS provides access to its data but does not release data for geographic units smaller than county; the vital records office of each state needs to be contacted for access to more geographically precise data. At NCHS, county-level data are available for 1968 to the present. Data are also available for 1959-1967 but have not gone through rigorous checks, and some gaps may exist for the period 1957-1967.

In 1979 NCHS established the National Death Index (NDI), a central computerized index of death record information for the entire country.

Death records are added to the NDI database annually and become available approximately 12 months after the end of a particular calendar year. Personal identifiers such as name of deceased, father's name, date of birth, social security number (SSN), and other variables, can be used to determine whether a person has died anywhere in the United States. NDI can provide a death certificate number for further linkage to the NCHS database to determine cause of death. However, NDI does not contain the address of the deceased individual.

Release of address information for mortality data can only be achieved by contacting the vital statistics offices of the state in which death occurred. Although death registration has existed for many decades, this has not always been done electronically. The committee requested information from 38 states that have or have had a nuclear facility on electronic availability of cancer mortality data; 17 states responded to the request (45 percent) (see Table 4.8). Complete mortality data have been available since 1970 in most states but subject address at time of death is not captured until much later in some states. A striking example is cancer death registration in Illinois. Death from cancer information is available since at least 1950, but only exists electronically since 1970, and address is included in the records only since 2008.

These delays and gaps appear because the primary purpose of the vital statistics offices is to provide documentation of death, not to support research. Although this view may be changing slowly, adding addresses for past years retrospectively requires an enormous amount of work and is not feasible in many states. The lack of address information accompanying cancer death registration is problematic for a study of cancer risks in populations near nuclear facilities as investigators are unable to assess risks related to the early operational years of the nuclear facilities, for example, the 1960s (when cancer registration efforts were nonexistent in the majority of the states). It was anticipated that mortality data at a geographic level smaller than county, such as census tract, would go further back in time than incidence data for the same geographic unit, or at least address would be available electronically and could be used to geocode the data. However, this is not generally the case. Of course, address at time of death is present in the hard copies of the death certificates, though an effort to retrieve the information from those in an ecologic study would be impractical.

In contrast to cancer incidence data, geocoding addresses to census tract is not common practice for mortality data (see Table 4.8). For example, cancer mortality data for Arizona are available since 1970, but census tract of reported deaths is available only since 1995 and is not complete. In Illinois, census tract is geocoded only for Chicago, roughly from 1979. Alabama does not geocode the data and, as it is very rural in some parts, even aggregated data have small counts of cancer deaths and will not be released.

Finally, in contrast to the cancer registry that has information only on

state at birth (and even that is incomplete), the mortality database may contain city of birth.

4.3.4 Methods for Control Selection

In a case-control study the challenge is to identify individuals that are similar to the cases in all relevant respects except the exposure under study (controls). Random-digit dialing (RDD) has often been the preferred source of identifying population-based controls and it worked well until the mid 1990s. A 2.5 percent annual decline in the RDD response rates from 1982 to 2002 has been reported (Bunin et al., 2007). The increasing use of cellular phones, caller identification, and multiple telephone numbers for a given household are a few of the emerging problems with RDD as a source of control selection today, and the potential exists for RDD control samples to be biased with respect to socioeconomic status and population characteristics (Bernstein, 2006; Ma et al., 2004). An additional concern directly relevant to the design of the cancer risk assessment in populations near nuclear facilities is the fact that the population under study (cases and controls) will need to be geographically defined (residing within a specific distance from the nuclear facilities), which also makes RDD less appropriate.

Town records could be used, but these are not uniformly available across the country. The relevant Department of Motor Vehicles (DMV) is a possible source for control identification, but the files are restricted to those that drive. Thus, they do not include individuals who are not old enough to have a driving license and do not completely cover older populations. As a result, DMV records would not be useful for a study of childhood cancers. Alternative control identification methods such as use of a friend, neighborhood, family, or school controls have limitations that affect their appropriateness in a study of cancer risks in populations near nuclear facilities, including a high risk of overmatching on exposure and geographic location. Additionally, school controls would be appropriate only for studies of school-age children, and their use is likely to be administratively difficult in a multistate study (Ross et al., 2004).

Investigators, including those involved in multistate studies of childhood cancers, are exploring the feasibility of using birth certificate files to select controls in studies of childhood cancers. This strategy has the advantage of collecting data that facilitate matching on factors such as age and gender, but also data on risk factors of childhood diseases such as birth weight, and age and educational level of the mother. Birth registration is considered virtually complete and data on birth records are fairly complete, although the quality of information deserves consideration (Kirby and Salihu, 2006), and the use of these data eliminates the problem of recall bias as they are not self-reported after a diagnosis of cancer. In contrast

to RDD, this method for control selection allows characterization of nonparticipants. Although birth records have been used successfully in many epidemiologic studies (see, for example, Ma et al., 2002; Rosenbaum et al., 2000; Von Behren et al., 2011), their use presents challenges in nationwide studies, as investigators need to receive approvals for data release from many state health departments and the requirements for release of the information differ by state. Obtaining IRB approvals for each state may require modifications to the general protocol. Moreover, the standard certificate for live births has not been implemented fully across the United States (Kirby and Salihu, 2006), so achieving consistency of the format of the data retrieved from the different state birth registries is complex and necessary before a study database is ready for analysis and research. However, it has been demonstrated that birth registries may be used to select controls for pediatric studies on a national scale, even if information to locate potential control subjects is requested (Spector et al., 2007).

The reproductive statistics branch of the NCHS holds electronic birth registration data since 1968. Similar to the release restrictions for mortality data, NCHS cannot release data on births for geographic units smaller than a county. Investigators will need to contact the vital records office of each state (same office that releases mortality data) to obtain addresses or tabulations by census tract or other smaller geographic units. In an effort to identify the release criteria of birth registration data and the potential of linkage of birth records with cancer registries within and across states, the committee sent a letter to the 38 states that have or have had a nuclear facility. Of the 38 offices surveyed, 12 responded to the request for information (31 percent). A letter template is presented in Appendix L. Overall a detailed research protocol is needed before the offices could comment on the feasibility of any research activities requiring data on birth registration. However, some general guidelines were provided: the office of vital records in New York explained that data with personal identifiers are not released; and in Alabama and Michigan individual birth records cannot be released without permission of the individuals involved or the parents. In some settings, the Health Insurance Portability and Accountability Act of 1996 requires that geographic location at resolution smaller than three-digit ZIP codes be considered a personal identifier that cannot be released without special permission. Illinois reported that currently researchers' requests for data are not accepted.

4.3.5 Record Linkage and Individual Tracing Methods

Record linkage refers to the task of searching two or more files for records that belong to the same individual, such as a birth certificate and a medical record. Historically, most record linkage was performed by clerks,

who reviewed lists and made linkage decisions for scenarios for which rules had been developed. Nowadays, linkage that involves large files is generally computerized in order to reduce or eliminate manual review and make the results more easily reproducible. Computerized linkage is also faster, matching decisions are more consistent, and quality controls are better (Winkler, 1995). Common record linkages in epidemiology are between birth records and state cancer registries to identify individuals who developed the disease of interest or with mortality data to determine who has died.

Successful linkage requires that the various data sources share one or more common identifiers—referred to as the matching or linking variables—such as name and date of birth of the index individual. Many times, two or more individuals share the same linking characteristics, and unavoidably registries contain administrative coding errors or double entries which complicate the one-to-one linkage process and may lead to a true match erroneously being designated as nonlink or to the true match being one of many possible matches. The ideal linkage variable was described as the one that has many different values, all having about the same frequency of occurrence, contains no missing data or errors, and has not changed in value over time. The higher the number of matching variables, the better the ability to distinguish matches (Winkler, 1995).

The two main methodologies used for record linkages are *deterministic* and *probabilistic*. Deterministic record linkage links pairs of records on the basis of whether they agree exactly on specific identifiers. Such record linkage is often feasible in countries with a long-standing tradition of a unique identifier at birth, such as the personal registration number used in Denmark or the identification numbers given to all residents in Sweden and Norway (Tromp et al., 2006). In the United States and other countries where such a unique identifier is not established at the time of birth, linkage is less straightforward and the probabilistic record linkage methodology is often used. This method uses probabilities to determine whether a pair of records refers to the same individual (Machado, 2004; Tromp et al., 2006). More specifically, the probabilistic record linkage method assigns a weight of (dis)agreement for the linking variables based on the probability that a variable agrees among matches and the probability that a variable agrees among nonmatches, this way defining the error rate and discriminating power of the linkage (Tromp et al., 2006).

The committee requested information from the states' Departments of Vital Statistics on linkage capabilities (letter template is presented in Appendix L). Linkage of birth registration and cancer data within states is routine in many states (for example, California, Minnesota, Michigan, Arkansas, and Colorado). However, no state from those that responded to the committee's request for information reported existing methods for linkage across states. One obstacle is lack of consistency across states on

variables used for linkage. For example, Minnesota reported that currently records are linked on name, date of birth, and SSN. In North Carolina, SSN is not available in birth records. A second obstacle is that, even if such a nationwide linkage is technically possible, differences among state statutes governing cancer and birth registration would likely not support such a project.

Investigators can use record linkage to retrieve current information of the populations under study, and in this way attempt to trace and recruit them. This is not an easy task, as often the information available to start the tracing process is limited, and often a long time has elapsed since some of the information was current. Inability to recruit individuals may both reduce the power of the study and introduce bias in the results. For that reason, ensuring that tracing of individuals is done with success is key to the strength of any record-based study. Tracing of individuals for cohorts identified retrospectively is challenging and time consuming. Essential components described to contribute to successful efforts to track or retain study subjects include (1) attention to staff training and support, (2) effective tracking system, (3) incentives, (4) establishing rapport with participants, (5) ensuring confidentiality, and (6) use of a combination of contact means as appropriate (Hunt and White, 1998; McKenzie et al., 1999).

Tracing has been done successfully in the past. One example is the Hanford Thyroid Disease Study conducted in the 1990s, a retrospective cohort study of the effects of exposure to atmospheric radioactive releases from the Hanford Nuclear Site in southeastern Washington State in the 1940s-1950s (Davis et al., 2008; study is discussed in Appendix A). The study identified more than 5,000 cohort members using Washington state birth records from 1940 to 1946. The limited information contained in the birth records was used to trace more than 94 percent of the cohort members, nearly 50 years later. Tracing was conducted in two phases: a feasibility study to test the methodology proposed and to develop specific operational procedures, then a five-step approach to locate cohort members, beginning with the most readily available and least costly steps as described:

1. Computer matching to state records: birth records, DMV records, death records.
2. Readily available lists of individuals: telephone directories, post office forwarding, city and reverse directories, existing high school reunion lists, voter records, utility records.
3. Readily available, labor-intensive lists of individuals: neighborhood searches, former school teachers, old newspaper searches for death, birth and marriage announcements, other historical records.
4. Limited availability, labor-intensive lists of individuals: agricultural, civic, religious and veterans organizations, labor unions.

5. Available, costly contact of individuals: locating services, public appeal.

Motor vehicle licensing records and directories proved the most useful in tracing individuals. The investigators note that, at the time their study was conducted, the use of internet and email was not as widespread as it is today. These two options could potentially improve the tracing response rate. As methods of recruiting participants are also relevant for retaining participants in a longitudinal study, research on retaining participants emphasizes this point (Davis et al., 2008; Robinson et al., 2007). An average of five sources was required to locate an individual. An extensive effort was required before a cohort member was declared "unlocated" by the team of supervisory staff.

Another example of a study with satisfactory response rate of 75 percent used 14 sources to locate 230 parents of sudden infant death syndrome infants and 255 parents of healthy living infants in Southern California (Klonoff-Cohen, 1996). Possible reasons for the lower success rate compared to the Hanford study is that case parents were relatively young and transient without an established credit history and, therefore, harder to be traced through tax assessor records, and the fact that the Human Subjects Committee required at least a 1-year waiting period to contact the parents of the deceased infant, during which period the parents may have moved. The Northern California Childhood Leukemia Study, which enrolled birth registry controls aged 0-14 years reported a contact rate of 80 percent (Ma et al., 2004). A case-control study of birth defects based in seven Texas counties aimed to contact mothers and interview them by telephone 4 years after the births of their children. Case mothers were more likely than control mothers to be located (44 percent versus 30 percent, respectively) and, of those that were located, to be interviewed (43 percent versus 31 percent, respectively). Young maternal age and black race decreased the likelihood of locating mothers (Gilboa et al., 2006). Nationwide studies include the Pregnancy Risk and Monitoring System, which contacts mothers between 2 and 6 months after giving birth in 23 states. The study achieved a contact rate of 82 percent in 2001 (Shulman et al., 2006). As expected, age affects the effort required to trace children, with less efforts needed for birth certificate controls aged 0-4 years than for those aged 5-14 years (Ma et al., 2004).

4.3.6 Data on Population Characteristics

As discussed in Section 4.3.1, the U.S. Census is a source for information regarding the population characteristics such as age, gender, and race/ethnicity. Surveillance systems that collect information on population char-

acteristics over time, including lifestyle factors, are important for tracking such things as chances in the incidence of cancer or other chronic disease, and risk behavior prevalence. In the context of this report, surveillance systems are important as they could be a source of information on the characteristics of the populations compared and thus provide clues on potential confounders in an ecologic study. The committee found that three national surveillance systems might be relevant: The National Health Interview Survey (NHIS), the National Health and Nutrition Examination Survey (NHANES), and the Behavioral Risk Factor Surveillance System (BRFSS). All three surveys are managed by CDC. However, none of these surveys are directly applicable for the present task, as they do not contain information about behavioral data at the census-tract level. Technical and methodological details for the surveys are available online and briefly summarized here. Sources of health care information are also discussed, but again information from these sources is not directly applicable for the present task.

4.3.6.1 The National Health Interview Survey (NHIS)

The NHIS is a large-scale face-to-face household interview survey of a random sample of households in the United States. The main objective of the NHIS is to monitor the health of the population in the United States and track progress toward national health objectives. Interviewers of the U.S. Census Bureau have conducted the survey for the NCHS continuously since 1957. Each year, interviewers visit 35,000 to 40,000 households across the county and collect data for about 75,000 to 100,000 individuals. The annual questionnaire consists of three components: the family core, the sample adult core, and the sample child core. The family core collects information on everyone in the family, including family composition, and basic demographic characteristics such as age, race, gender, income, and health insurance coverage. In addition, one adult and one child, if applicable, from each household are randomly selected and information on each is collected. In 2007, participation rates for the survey were 68 percent. As noted above, the goal of the NHIS is to collect summaries of health at the national, and perhaps state level, not at the fine geographic scale of census tracts.

4.3.6.2 The National Health and Nutrition Examination Survey (NHANES)

NCHS also conducts NHANES, a survey that aims to assess the health and nutritional status of adults and children in the United States. The NHANES program began in the early 1960s. In 1999 the survey became a continuous program that examines a nationally representative sample of about 5,000 persons each year. Although substantially smaller than either

NHIS or BRFSS, NHANES is unique because it combines information from interviews with a physical examination and some laboratory tests. The NHANES interview includes demographic, socioeconomic, dietary, and health related questions while the physical examination component consists of medical and dental measurements. In the 2005-2006 survey, participation rates were 80 percent. Again, the goals are estimates at the national and perhaps state level, not at the fine geographic resolution desired for the studies under consideration. NHANES, like NHIS, is based on cluster sampling.

4.3.6.3 The Behavioral Risk Factor Surveillance System (BRFSS)

In 1984, the CDC recognized the importance to disease prevention of monitoring personal health behaviors in the general population and established the BRFSS in 15 states. A decade later, this system was in place nationwide. In contrast to NHIS and NHANES, BRFSS is a telephone-based survey conducted by state and territorial health departments with technical and methodological assistance provided by the National Center for Chronic Disease Prevention and Health Promotion of CDC. Each state works with CDC to develop a sampling protocol to select households and one adult (age >18 years) is selected from each household and is interviewed. BRFSS is the only one of these three surveillance systems that can generate state- or territorial-based estimates on a variety of health measures. BRFSS collects data from approximately 210,000 people in 50 states, the District of Columbia, Puerto Rico, the U.S. Virgin Islands and Guam. Self-reports of health-related variables (e.g., weight) have not matched measurements from the other surveillance systems that do not rely on self-reports (Carlson et al., 2009). Perhaps the largest challenge in using BRFSS data is that the response rates for BRFSS have declined from 72 percent in 1993 to 51 percent in 2007. The low, and apparently biased, participation rates produce different estimates in some outcome measures compared to NHIS and NHANES, both of which have higher participation rates. The consequences have been estimated to be minimum in some cases and unknown in others (Fahimi et al., 2008). Finally, BRFSS provides design-based state and national estimates and some research has considered extensions to county level. However, the data are not sufficient to support design-based estimates at the census-tract level.

4.3.6.4 Health Care Surveys

NCHS performs the National Health Care Survey to answer questions on the use and quality of health care, the impact of medical technology, and disparities in health care services provided to population subgroups in the

United States. The National Health Care Survey is built upon the merging and expansion of separate record-based surveys:

- National Ambulatory Medical Care Survey
- National Hospital Ambulatory Medical Care Survey
- National Survey of Ambulatory Surgery
- National Nursing Home Survey
- National Hospital Care Survey
- National Nursing Assistant Survey
- National Home and Hospice Care Survey
- National Home Health Aide Survey
- National Survey of Residential Care Facilities

The combined surveys use provider-based information which depending on the setting in which the care is delivered, may come from a record of the patient's most recent visit, the hospital discharge form, or review of the entire medical record. Information on the sample design for each of the component surveys can be found at http://www.cdc.gov/nchs/dhcs.htm. Overall, the design is such to permit monitoring of the delivery of specific health care services and understanding the characteristics of the patients that receive different types of services. The National Hospital Discharge Survey (NHDS) is briefly described here as an example to demonstrate the relation of the different health care surveys and the potential for linkage with other national data sets.

NHDS is a national probability survey that was initiated in 1965 and was the first survey of medical care delivery conducted by the NCHS to collect information on inpatient use of short-stay nonfederal hospitals in the United States (Dennison and Pokras, 2000). The survey was redesigned in 1987 to improve on its sampling and link with the design of NHIS and to use automated retrieval of data, among other reasons. In 1988 the survey collected data on diagnoses, procedures, length of stay, and patient characteristics from a sample of approximately 250,000 discharges from over 500 hospitals. NHDS was conducted annually since its inception until 2010, when it was integrated into the National Hospital Care Survey together with data from the emergency department, outpatient department, and ambulatory surgery center data collected by the National Hospital Ambulatory Medical Care Survey (NHAMCS). (NHAMCS was conducted since 1973 and data were collected from the physician who would be randomly be assigned a 1-week reporting period.) The integration of these two surveys along with the collection of patient identifiers will permit linkage of care provided in different departments. It will also be possible to link the survey data to the NDI and Medicaid and Medicare data to obtain a more complete picture of patient care.

Important to the committee's task and many times reiterated is the need for a source of information on medical diagnostic procedures that use radiation, especially those that use high doses such as CT scans. The main data source for aggregate counts on medical diagnostic procedures that involve radiation by body part is IMV.[19] IMV is a market research and database provider founded in 1977 which, using a variety of survey methods, tracks diagnostic medical procedures. While IMV surveys have high participation rates and cover a large number of imaging facilities (IMV data were the main source for the NCRP Report 160 [NCRP, 2009]), they do not have a detailed categorization of procedures and therefore are unable to capture the variation in radiation doses and protocols. Detailed data on counts of procedures for large populations are also available from administrative claims such as Medicare. However, information is restricted to those that are age 65 or over and use this social insurance program. Neither IMV nor Medicare data are directly applicable for the present task, as they do not contain information about medical diagnostic imaging at the census-tract level.

4.4 FINDINGS AND RECOMMENDATIONS

This chapter provides the committee's assessment of methodological approaches for carrying out a cancer epidemiology study. Based on this assessment, the committee finds that:

1. The statistical power of an epidemiologic study of cancer risks in populations near nuclear facilities is likely to be low because (a) the size of the estimated risks from the reported radioactive effluent releases from nuclear facilities is likely to be small and (b) the size of the populations most likely to be exposed (that is, those in close proximity to a nuclear facility, for example, within an 8-km radius) is relatively small. This implies that a large-scale multisite study with as many years of observations as possible is needed to reliably assess the potential risks.
2. Centralized cancer registries such as SEER and NPCR (for cancer incidence) or national offices such as NCHS (for cancer mortality) can only release data that are aggregated across geographic areas such as counties. Cancer incidence and mortality data for more refined geographic areas can be released only by individual states upon submission and approval of a research proposal. In general, cancer mortality data are available since about 1970, but individual address at time of death is not captured until much

[19] http://www.imvinfo.com.

later in some states. Moreover, mortality data are not consistently geocoded at the census-tract level. Cancer incidence data of known quality are available from about 1995. These data include address at time of diagnosis and have been widely geocoded.

3. Large-scale studies that rely on contacting individuals are likely to be subject to selection and information biases due to difficulties related to tracing individuals, low (and declining) participation rates of cases and especially controls in epidemiologic studies, and the risk of collecting inaccurate information via interviews and questionnaires. Alternatively, studies that rely on information in existing records are more practical and free of the biases mentioned above, although other limitations exist.
4. Studies of pediatric cancers could take advantage of existing linkages of cancer registration and birth records in at least six states that include more than 30 percent of the U.S. pediatric population.

In light of these findings, the committee recommends that, should the USNRC decide to proceed with an epidemiologic study of cancer risks in populations near nuclear facilities (Phase 2), two studies be carried out to assess cancer risks in populations near nuclear facilities: (a) an ecologic study of multiple cancer types that would provide an assessment of cancer incidence and mortality in populations living within approximately 50 km of nuclear facilities and (b) a record-linkage-based case-control study of childhood cancer that would provide an assessment of early life exposure to radiation during more recent operating periods of nuclear facilities. The strengths and limitations of the recommended studies are described in Section 4.2.3. Specifying up front the hypotheses to be tested and the analysis plan is the responsibility of the Phase 2 committee.

The committee judges that additional information and analyses beyond the scope of this Phase 1 activity are needed to assess the feasibility of carrying out the recommended studies that could be performed by a pilot study. The purpose of the pilot study is to evaluate the feasibility of the methods proposed, and to develop the specific operational procedures and data collection methods needed for a full study. The purpose of the pilot study is not to perform a small-scale preliminary assessment of risks, the results of which would be used for or against moving forward with the full study.

As discussed in Chapter 3, seven facilities were selected collaboratively by the dosimetry and epidemiology experts of this committee and include Dresden (Illinois), Millstone (Connecticut), Oyster Creek (New Jersey), Haddam Neck (Connecticut), Big Rock Point (Michigan), San Onofre (California), and Nuclear Fuel Services (Tennessee). The reasons of selection of these facilities with regards to dosimetry are discussed in Chapter 3. These facilities are also good candidates to evaluate the feasibility of the studies

from the epidemiologic perspective as they represent both currently operating and decommissioned facilities in six states, that started operation in different time points and with some variation in (a) the population size in close proximity, (b) quality and maturation of the state's cancer registration, and (c) level of complexity for registry's research approval processes and research support. Actions specific to the recommended studies to be taken during the piloting activity are the following:

- Retrieve cancer incidence and mortality data at the census-tract level within 50 km of selected facilities to assess feasibility of the recommended ecologic study.
- Confer with investigators conducting linkages of cancer and birth registration data to identify eligible cases of pediatric cancers and matched controls to assess feasibility of the recommended record-linkage-based case-control study in the selected facilities. In states with the necessary capabilities, but without such linkages in place, link birth registration and cancer incidence data.

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5

Risk Communication and Public Engagement

In carrying out this Phase 1 study, the committee quickly came to understand that the technical issues that it was being asked to address (see Sidebar 1.1 in Chapter 1) have important social overtones. There is public concern and a lack of social trust[1] on the key question underling this study: Namely, is it "safe" to live near a nuclear facility? As was noted in Chapter 1, the U.S. Nuclear Regulatory Commission (USNRC) has been using the results of the 1990 National Cancer Institute study (Jablon et al., 1990) as a primary resource for communicating with the public about cancer risks associated with the nuclear facilities that it regulates. The committee assumes that the studies recommended in this report, if carried out, would be used by the USNRC for this same purpose.

Although public engagement was not an explicit part of the task statement for this Phase 1 study (see Sidebar 1.1 in Chapter 1), the committee recognized that effective public engagement would be essential to the success of a Phase 2 study. The Phase 2 study must not only be scientifically sound to be perceived as credible by the scientific community, it must also be perceived as credible by the public audiences for which it is intended. Additional steps beyond those typically followed in a scientific study will need to be taken to achieve such credibility.

This chapter is intended to provide basic information about risk and risk communication for the benefit of nonexpert audiences and to identify some key elements of a stakeholder engagement plan for a Phase 2 study.

[1]Social trust is defined as the willingness of the public to rely on experts and institutions in the management of risks and technologies (Earle and Cvetkovich, 1995).

5.1 PUBLIC PERCEPTIONS ABOUT NUCLEAR POWER

The public's perceptions about nuclear power have been shaped to some extent by its associations with other nuclear technologies, particularly nuclear weapons, and also by the occurrence of high-profile accidents at nuclear plants: Three Mile Island (TMI) in 1979, Chernobyl in 1986, and Fukushima in 2011.[2] Less serious incidents that resulted in unintended and unmonitored releases of radioactive materials from operating plants (e.g., releases of tritium from operating nuclear plants; see Chapter 2) have reinforced these perceptions. Although nuclear accidents are uncommon occurrences, they can have very severe consequences. Moreover, they suggest to some that nuclear technologies are poorly understood and unpredictable and that the nuclear industry and its regulator cannot be trusted to protect the public from these technologies.

The question "Is it safe?[3]" is perhaps of greatest concern to individuals who have experienced cancer or have family members or neighbors who have experienced cancer. Reassurances by the nuclear industry and its regulator that facility operations are "low risk" are not always seen as credible. In fact, the USNRC has sponsored the present study in an effort to address such concerns. Engaging with members of the public in a Phase 2 study will be important for understanding their concerns about cancer risks.

5.2 RISK AND COMMUNICATION

The risk assessment community usually defines risk in terms of the following three questions, referred to as the risk triplet (Kaplan and Garrick, 1981):

What can happen (i.e., what can go wrong?)?
How likely is it that that will happen?
If it does happen, what are the consequences?

Scientists and policy makers usually view risk in terms of the likelihood of harm from a hazard. In other words, the definition of risk is intertwined with the notion of *probability*. Technical experts may use probability estimates (for example, one-in-a-million chance of harm) to convey the risk of dying from cancer. However, public perceptions of risks are not shaped solely on the endpoint of a technical analysis, such as the number of cancer deaths in a population near a nuclear plant. Some members of the public

[2]The Three Mile Island accident resulted in no discernible health effects from radiation releases, but it nevertheless served to galvanize opposition to the expansion of nuclear power (Walker, 2004).

[3]The term "safe" has different meanings to different people. Some people view safety in terms of probability and consequences, whereas others view safety in terms of whether an organization responsible for controlling a hazard is trustworthy.

may personalize the risk—that is, to see a potential harm as affecting someone they care for such as their spouse or child. Ultimately, each person decides how much risk is acceptable; the decision will be based on several factors, some of which are personal.

Some individuals and groups question the value of technical risk assessment. A survey of environmental groups in the United States suggested that "environmentalists resent the technocratic, exclusionary nature of risk assessments that undermine democratic participation in local environment decisions" and view risk analysis as a waste of resources, while little is done to reduce the risk (Tal, 1997). Part of the public frustration often originates from the fact that current policies in the United States appear to be more reactionary than precautionary in the way they manage risk (Kriebel et al., 2001).

There are many subjective dimensions to risks that are unrelated to its technical definition. These include such things as lack of understanding or familiarity with the mechanisms underlying a technology; whether a threat is invisible, manmade, or potentially catastrophic; whether exposure is involuntary, beyond the public's control, or unfairly distributed; and whether a risk affects children (Fischhoff et al., 1981). Other societal concerns such as environmental health and food safety, property values, and decline in community image (Kasperson et al., 1988) may be hidden within the overall public perception of risk. Individual differences in risk perception and risk tolerance can also affect people's willingness to receive information. There is also an obvious relationship between perceived risk and unfavorable mass media coverage. For example, media stories that thoroughly document accidents and threats may influence how audiences think, feel, and behave when they receive information (Slovic, 2000).

Public perceptions of risks associated with the nuclear industry are perhaps unique among advanced technologies. This is demonstrated in a 1978 study (Fischhoff et al., 1978), still relevant today, in which participants were asked to compare technologies based on nine dimensions of risk. These included whether the risk was involuntary, familiar, controllable, has potential for catastrophic consequences, immediacy of those consequences, and the extent to which scientists and the public understand those consequences. Nuclear power, non-nuclear electric power, and x-rays were scored (numerical values from 1 to 7) on these risk dimensions. As shown in Figure 5.1, nuclear power was judged to have a much higher risk than x-rays. Also, nuclear power was perceived as markedly more catastrophic and dreaded compared to other technologies that produce energy.

5.2.1 Communicating About Risk

Understanding how nontechnical audiences perceive risk is an important first step in successful risk communication. The failure to accept that

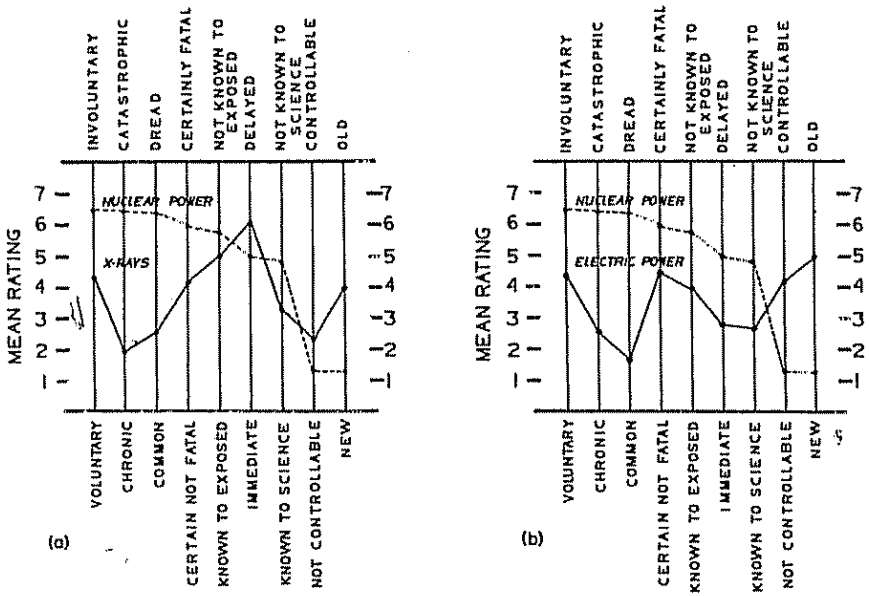

FIGURE 5.1 Qualitative characteristics of perceived risk for nuclear power compared to x-rays and other non-nuclear power technologies. SOURCE: Fischhoff et al. (1978).

many variables influence risk perceptions in a community, or labeling these perceptions as irrational, is guaranteed to raise hostility between community members and agency representatives (Slovic, 1987).

Historically, technical and policy experts have often performed and communicated the results of risk assessments to the public in a unidirectional manner. The assessments themselves often involved little or no public input. Experts would convey risk information that they deemed to be important, and risk communicators would clarify or simplify messages by translating technical jargon. However, the public no longer accepts expert judgments without question, especially when these judgments affect their lives. Indeed, Frewer (2004) suggests that there has been a refocusing of the primary goals of risk communication: initially from an effort to change public views about risk, later to gaining public acceptance for the sources of risk and their management, and more recently to building trust.

Successful risk communication now involves sustained, two-way communication and information exchanges between technical and policy experts and the public. Risk communication combines elements of conflict

resolution with the ultimate goal of solving problems rather than trying to "educate" the public. Even if problems are not solved, an interactive risk communication program can help to reduce unwarranted fear and distrust (Aakko, 2004). A recent paper (Aakhus, 2011) examines ways to improve interactivity in public communication.

Many federal government agencies recognize the importance of communicating with the public about risk. The U.S. Environmental Protection Agency (USEPA) took the lead in developing a two-way risk communication strategy in 1987. The Agency for Toxic Substances and Disease Registry (ATSDR), which has a mission to prevent harmful exposures and health effects related to toxic substances, has increased its capabilities for risk communication. The U.S. Department of Energy now trains its health officials in risk communication (Chess and Salomone, 1992). The USNRC has developed a handbook on effective risk communication (USNRC, 2004a) as well as other materials related to this topic (e.g., USNRC, 2004b, 2011).

Conveying technical information to nonexpert audiences needs to be done in a language that these audiences understand, and the content of the messages that are communicated has to fit the audience's needs (NRC, 1989). Matching content to needs can be particularly challenging when communicating about complex scientific and technical concepts, for example, radiation cancer epidemiology: Radiation terminology is specialized, concepts in cancer biology are complicated, and health effects at low radiation levels, if any, are generally small, often delayed, and therefore difficult to assess in an epidemiologic study.

It can be particularly difficult to communicate with nontechnical audiences about the scientific challenges of establishing a causal relationship between radiation and cancer. Nontechnical members of the public frequently associate "correlation" and "association" with "causality." Because proof of causality is scientifically demanding, scientists are usually cautious about making causal inferences. For example, if an association between living near a nuclear facility and cancer risk is observed, a plausible cause-effect relationship cannot be established solely by examining the risks in the communities around the facility. A conclusion about cause and effect will require additional information, including extrapolations from higher-dose human exposures and other types of studies.

Although it is important to help the public understand the science behind risk assessment, public audiences are often less interested in technical and methodological issues and more interested in issues such as trust, credibility, fairness, and empathy (Covello et al., 1987). Communication can be considered successful only if those inquiring about the risk are satisfied that they are being accurately informed and appropriately engaged (NRC, 1989).

Communicating about uncertainties associated with technical risk as-

sessments is an increasingly important and inseparable component of risk communication. Until recently, there has been little discussion of uncertainty communication by risk communication professionals because they assumed that the public was unable to conceptualize uncertainty (Wynne, 1992) or that admitting uncertainty could be seen as a sign of incompetence (Johnson and Slovic, 1995). The historic lack of communication about uncertainties has increased public distrust in the motives of regulators and scientists (Frewer, 2004).

All risk assessments are based to a certain extent on unproven assumptions and incomplete knowledge that limit the precision of risk estimates. This is certainly the case for assessments of cancer risks in populations near nuclear facilities, because data on exposures and disease occurrence may not be complete (see Chapter 3 and 4). Although uncertainties can be reduced by obtaining additional data, such acquisition can require great effort and can result only in marginal gains in precision.

Describing the uncertainties in a risk analysis can enhance the understanding of risk estimates. In describing uncertainties, it is important to separate known and speculative uncertainties and to identify areas of disagreement among experts. This helps others to make informed independent judgments about the meaning of the risk estimates.

In cases where scientific findings are ambiguous, communication may take place in an environment marked by disagreements, misunderstanding, and suspicion. Communicators must diagnose these difficulties, find ways to create trust and credibility to overcome them, and deepen understanding (Rowan, 1994). Creating trust, based on the expectations that the communicator is competent and well meaning, is probably the priority of a risk communication plan. People are generally uninterested in understanding a subject or taking any sort of action if they do not trust those who are communicating with them.

5.3 PUBLIC ENGAGEMENT IN PHASE 1 STUDY

Although this Phase 1 study did not involve a formal assessment of cancer risk, the committee understood the importance of engaging with the public to understand their views and concerns. The project sponsor (USNRC) also encouraged the committee to engage with the public during this Phase 1 study and provided funding to make this possible.

The committee judged that public engagement would improve the outcome of this Phase 1 study, particularly in helping the committee to identify Phase 2 study designs that could help to address public concerns. The committee membership includes experts in risk communication and public health (see Appendix B); these experts helped the committee to engage with the public during this Phase 1 study.

5.3.1 Outreach to Public Audiences

The committee used several processes to engage interested members of the public in this Phase 1 study. Two of these processes are legally required, as noted below, but most were implemented by the committee to enhance its efforts to inform and engage the public.

- Committee meetings were announced in advance through the National Academies website[4]; additionally these announcements were shared with news outlets.
- A study-specific website (www.national-academies.org/nrsb/CancerRisk) was developed to supply information about the study, for example, background materials on the project and meeting information, including copies of meeting presentations.
- An interested-parties listserv was created and maintained to communicate about upcoming committee meetings and other project-related activities.
- A project email address was established that could be used by anyone with access to email to submit information and comments to the committee. The committee also encouraged the submission of written comments at its meetings. Materials received from outside the National Academies are maintained in a Public Access File for the project.[5] Anyone can examine this file and request copies of materials.
- The committee met in different geographic regions of the United States, primarily near USNRC-licensed facilities, to afford opportunities for interested members of the public to attend and interact with the committee (see Appendix C). Public comment sessions were scheduled at all of the committee's public meetings.
- The information-gathering meetings of the full committee were webcasted, and the webcasts were archived on the project website (referenced above) to allow for later viewing.

The committee received a large number of comments from outside groups and individuals during this Phase 1 study. The committee found these comments to be useful for:

- Understanding public concerns about the study.
- Uncovering data sources and documents unknown to the committee.
- Identifying study issues that require clarification.

[4]This notification is required by Section 15 of the Federal Advisory Committee Act.

[5]Maintenance of a Public Access File is required by Section 15 of the Federal Advisory Committee Act.

- Receiving recommendations on study design.
- Receiving preliminary data on suspected cancer clusters near nuclear facilities.

The comments received from the public during this Phase 1 study covered many subjects. However, some common concerns emerged, including the following:

- The USNRC is sponsoring the Phase 1 study.
- The USNRC relies on the nuclear industry to self-report radioactive effluent releases; measurements and summaries of these data should be provided by independent sources and be made available to the public.
- Allowable radioactive effluent release limits are too high.
- There are multiple historic instances of leaks of radioactive materials at nuclear facilities, not always reported at the time of the release.
- Releases (routine or accidental) may be higher than those reported; therefore, associated risks may be higher than those conveyed.
- The high number of cancer cases in the communities around the nuclear facilities should be evidence of the risk.

Many of these comments appear to reflect public distrust of the nuclear industry and its regulator.

The committee also received some recommendations for study design, including the following:

- Widen the study scope; include non-USNRC-regulated facilities, and examine noncancer effects such as birth defects, cardiovascular disease, and infertility.
- Include multiple cancers and age groups in the analysis, with a special focus on susceptible populations such as young children and those exposed in utero.
- Consider current and past routine releases, accidental releases, and releases from spent fuel stored in the facilities.
- Find alternate ways to investigate risks in states where cancer registration is not adequate.
- Independently investigate the type and amount of radioactive releases from nuclear facilities.
- Do not rely solely on distance from a facility as a measure of exposure, but incorporate wind direction and water sources in the models.
- Include other plants that produce energy, such as coal-fired plants, as a comparison group.

- Use biomarkers to measure damage due to radiation to increase sensitivity of the study.
- Communicate with the public with clarity about the progress of the study.

5.3.2 Outreach to State Public Health Departments

To understand the concerns of individuals who live near nuclear facilities and collect information on past risk assessments, the committee contacted the Departments of Public Health in states that are now hosting or have previously hosted a USNRC-licensed nuclear facility to request information on the following issues:

- Reports from members of the public about health concerns[6] or suspected health effects related to nuclear plants or nuclear fuel-cycle facilities in their communities.
- Reports from physicians or other healthcare providers concerning suspected disease clusters that could be related to radioactive releases from these facilities.
- Assessments of cancer risks in association with nuclear facilities that were carried out by the department.
- Other individual or organized activities that have been undertaken by the department in response to environmental monitoring or health surveillance programs.
- Interactions between departments and communities around nuclear facilities to solicit feedback on potential health concerns.

The letter template is provided in Appendix M, and responses are tabulated in Table 5.1. Of the 38 state Public Health Departments contacted, 31 (81 percent) responded to the committee's request for information. Of these, 15 stated that no relevant concerns[7] had been reported. States to which health concerns were reported followed up with some investigation or analysis of cancer rates in counties at issue. Inconclusive results that required further investigation were reported from a few states, including Michigan, New York, and Virginia.

Departments heard concerns about or received requests for examination of potential cancer clusters from various sources including the public, news media, oncology practices, and elected officials. A typical examination

[6]The committee provided no guidance to health departments on what constituted a "health concern," leaving that determination instead to the professionals who responded to the committee's inquiry.

[7]In the absence of a clear definition of what constitutes a "health concern," the reader should be cautious when making judgments about the significance of the responses.

TABLE 5.1 Reported Health Concerns Associated with USNRC Licensed Nuclear Facilities

	Reported Health Concerns		
State	Reported inquiries	Year	Facility Implicated
Arizona	0	—	—
Arkansas	0	—	—
California	1	2008	Diablo Canyon San Onofre Humboldt Bay Rancho Seco
Connecticut	6	1987	Haddam Neck, Millstone
		2000	Haddam Neck
		2004	Millstone
		2007	Millstone
		2011	Indian point Millstone
Florida	not routinely	1996	St. Lucie
Georgia	0	—	—
Illinois	Multiple	2000-today	Dresden Braidwood
Iowa	0	—	—
Kentucky	2	2002	Paducah
		2007	Paducah
Louisiana	0	—	—
Maine	1	1989	Maine Yankee
Maryland	0	—	—
Massachusetts	Multiple	1980-today	Vermont Yankee Pilgrim
Michigan	4	1994	Fermi
		1999	Fermi
		2005	Fermi
		2009	Fermi
Minnesota	Multiple	1994	Monticello Prairie Island
		2000	Prairie Island
Mississippi	0	—	—
Nebraska	0	—	—
New Hampshire	1	2009	Vermont Yankee
New Mexico	0	—	—
New York	multiple	Major	
		1980s	Indian Point
		1990s	Indian Point Ginna Nine Mile Point FitzPatrick
		1995	Ginna Nine Mile Point FitzPatrick
		2002	Nine Mile Point
		2007	Ginna

TABLE 5.1 Continued

State	Reported Health Concerns		
	Reported inquiries	Year	Facility Implicated
		2003	Indian Point
		2008	Indian Point
North Carolina	0	—	—
Ohio	2	2011	Davis-Besse
		2009	Perry
Oregon	0	—	—
Pennsylvania	1	1979	Three Mile Island
South Carolina	0	—	—
Tennessee	2	2009	NFS
		2010	
Texas	0	—	—
Vermont	Routinely		Vermont Yankee
Virginia	2	2001	North Anna
			Surry Power
		2009	North Anna
			Surry Power
Washington	0	—	—
Wisconsin	0	—	—

NOTE: NFS, Nuclear Fuel Services.
SOURCE: Based on responses to the letter shown in Appendix M.

by a health department involved calculating incidence rates and case counts for areas at issue for a specific period by county, city, census tract, or ZIP code. The assessments were often performed by agencies or universities other than the health departments.

For example, in 2002 a public health assessment was conducted by the ATSDR in Kentucky. The assessment encompassed both radiological and nonradiological hazards related to the Paducah Gaseous Diffusion Plant. In 2007, the University of Kentucky's Kentucky Water Resources Research Institute produced an assessment on behalf of the Kentucky Radiation Health Branch addressing radiation dose and risk assessment attributable to surface waters near the plant.

Commonly, the concerns reported to the state health departments would be for noncancer health concerns related to nuclear facilities, such as Down's syndrome prevalence (Massachusetts Health Department), infant death (Illinois and New York health departments), and low birth weight (New York Health Department). Nonhealth issues were also reported, such as a claim regarding elevated radiation levels in goat milk samples in Connecticut and decreased productivity of livestock and crops in Kentucky. Some states reported that they received phone calls from concerned citizens

related to radiation risks from the recent disaster in Japan (North Carolina, Massachusetts). Health concerns resulting from 1979 TMI incident were reported by the Pennsylvania Department of Health. Following that incident, the Department received state funding to conduct multidecadal health-related studies.

The number of concerns received by the public health departments may not be an accurate estimate of overall community concerns. For example, although the Tennessee Department of Public Health reported that it has been contacted by only two members of the public in 2009 and 2010 with concerns about the Nuclear Fuel Services facility located in Erwin, Tennessee, the study committee is aware that a group of citizens in Erwin have filed a class-action lawsuit against Nuclear Fuel Services, claiming that releases from the facility are to blame for high rates of cancer. The Health Department of Georgia reported that it has not received any relevant health reports; however, members of the public voiced health-related concerns during the committee meeting in Atlanta, Georgia. It is possible that some members of the public are unaware of state health department reporting systems, or they lack confidence to report concerns or that their concerns will be investigated.

Some states, such as Kentucky and Oregon, noted that they do not have a formal database for tracking complaints. Instead, public complaints are addressed individually and followed up as deemed appropriate by the specific departments devoted to radiation health. It is possible (as stated by the New York Department of Health) that the records and recollections from staff are incomplete.

Finally, one state department of public health may receive public requests about facilities in neighboring states if the facility is close to the state border. For example, health departments in Massachusetts and Connecticut have received concerns about facilities in Vermont (Vermont Yankee) and New York (Indian Point), respectively.

5.4 PUBLIC ENGAGEMENT IN PHASE 2 STUDY

The committee judges that public engagement will be an import element of any Phase 2 study. Engagement needs to be designed to address the needs of the broad public population, which may not be coincident with the population that is targeted by the epidemiologic study. Although there is no checklist for sucessful engagement, previous National Research Council (NRC) reports can be used to identify important plan elements. Such reports include *Improving Risk Communication* (NRC, 1989), *Science and Judgment in Risk Assessment* (NRC, 1994), *Understanding Risk* (NRC, 1996), and the more recent *Science and Decisions* (NRC, 2009). The

objective of public engagement is to improve the Phase 2 study, particularly with respect to addressing public concerns, and to build trust and credibility in the study results.

5.4.1 Goal Setting

Public engagement requires the exchange of information among interested parties. Engagement efforts that have (and demonstrate commitment to) defined goals are more likely to be successful than those that do not. Goal setting is important to encourage realistic expectations and to clarify motives and objectives. For example, although public participation in any Phase 2 epidemiologic study is essential for its success, the scientific aspects of the study remain the responsibility of the experts who are carrying out the study. To avoid misunderstanding and false expectations, the limits of participation need to be made clear from the beginning. Moreover, goals may need to be adjusted based on new information, feedback from stakeholders, or a goal evaluation process. Having a schedule for goal accomplishment and a set of measures for evaluating effectiveness in achieving those goals can help to ensure communication program effectiveness.

5.4.2 Stakeholder[8] Identification

This Phase 1 project has already identified some key stakeholders. These include participants at the Phase 1 public meetings and users of the project listserv. A Phase 2 study could include other interested members of the public who live near the nuclear facilities to be studied as well as state and local officials and other community leaders. Although not formally stakeholders, the media and related intermediaries can help ensure that messages reach intended stakeholder audiences and are accurate.

By identifying key stakeholders, one can better select the appropriate communication channels and develop effective engagement strategies and tools. These strategies and tools may need to be tailored for different audiences, and it is important that this tailoring be easily seen and understood. Attention is often paid to the characteristics of the stakeholders when tailoring such strategies. Such characteristics include culture, language, knowledge and resources, and attitudes toward the nuclear industry and regulators. Stakeholders will have differing levels of participation and interest, but engagement needs to be consistent and ongoing, even if no new information is available.

Learning about the concerns of the stakeholders is important for effec-

[8]Stakeholders are defined as "interested and affected parties" (NRC, 1996).

tive engagement. Effective tools for gathering information about such concerns include interviews, surveys, informal discussions with small groups or community opinion leaders, and focus groups. Focus groups, if representative of the community, are particularly helpful for identifying obstacles to effective communication because they allow for social interaction and can surface issues that a structured questionnaire or interview would miss. Moreover, focus groups establish a basis for dialogue and generate active involvement, so participants view themselves as providers of useful information rather than as passive receivers (Johnson, 1993). Stakeholder views can change over time; focus groups can also be a way to monitor these changes.

5.4.3 Competence and Expertise

Credible and trusted sources can improve the perceived accuracy of communications with public audiences. Trust and credibility can usually be improved by engaging subject-matter experts (for a Phase 2 study, such experts would include epidemiologists and statisticians, for example) in the communication effort. Experts need to be able to demonstrate that they do not promote any particular interests and that they produce accurate and independent assessments. A distrusted information source that is perceived to promote a particular view may be perceived as deliberately biased or inaccurate. In some instances, partnering with a person or organization that stakeholders find credible, for example an organization that has strong ties to the community, can improve public trust. Moreover, periodic independent reviews of the study by scientists who are not involved in its conduct and are in part selected by stakeholders may increase credibility.

5.4.4 Transparency

Transparency is characterized by open and honest communication with stakeholders. It requires that information be accessible to the public when legal considerations permit, and also that information be presented with clarity. For example, background documents, conceptual information about the study design, sources of information used in the study, study results and uncertainties, and study progress reports can be shared.

Transparency also gives the communicator an opportunity to receive information from stakeholders. Affected parties have important perspectives that can help inform the Phase 2 study; it is important to demonstrate openness to receiving information and being clear about how such information is being used in the project.

5.5 RECOMMENDATION

The Phase 2 study should include processes for involving and communicating with stakeholders. A plan for stakeholder engagement should be developed prior to the initiation of data gathering and analysis for this study.

Stakeholder engagement is an essential element of any risk assessment process that addresses important public interests. Several approaches were used in this Phase 1 study to engage with stakeholders. The Phase 2 study can build on these Phase 1 efforts to achieve effective collaboration with local people and officials and increase social trust and confidence. To this end, the Phase 2 study should develop and execute an engagement plan that includes processes to:

- Identify key stakeholders and stakeholder groups with whom engagement is essential.
- Assess stakeholder concerns, perceptions, and knowledge.
- Communicate the questions that the Phase 2 study can address and its strengths and limitations; communicate the results from the Phase 2 study in forms that are useful to different stakeholder groups.
- Make the information used in the Phase 2 study publicly accessible to the extent possible.

It is important that the engagement plan be developed prior to the initiation of data gathering and analysis to ensure early engagement with stakeholders in the Phase 2 study. It will also be important to monitor how stakeholder views and concerns change during the study in response to external events. Adapting the plan to changing events can improve the success of engagement efforts.

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Appendixes

A

Radiation as a Carcinogen

A.1 RADIATION AS A CAUSE OF CANCER

At low doses of radiation, cells may be damaged. The main initiating event by which radiation damages the cells in the long term is damage to DNA in the cell nucleus. With well-orchestrated and efficient mechanisms, cells respond to the induced damage and attempt to repair it, but sometimes the damage cannot be repaired or is misrepaired, which may lead to mutations. The modifications induced by low levels of radiation dose may be transmitted to daughter cells and may lead to uncontrolled cell growth and consequently cancer, the health effect of primary concern in the context of radiation. Exposure to radiation is not the only way in which the DNA within a cell can be damaged and become cancerous. In fact, DNA damage can occur spontaneously or due to a number of other stressors such as chemical exposure (for example, smoking and lung cancer) and infectious agents (for example, hepatitis B virus and liver cancer). In other words, as ionizing radiation exposure induces DNA damage to the tissue, that tissue will already carry some damaged cells from other stressors.

Although small increases in the chance of developing cancer is the main health effect of low levels of radiation, such effects in individuals are probabilistic and known as stochastic effects. In other words, there appears to be no threshold below which effects do not occur, but the greater the exposure, the higher the probability that they will occur. Severity of the effects does not depend on dose. This is in contrast to the "deterministic" or "nonstochastic" radiation effects of high doses of radiation, that is, doses of several sieverts that can kill enough cells to cause injury such as skin red-

dening, burns, organ damage, radiation sickness, and even death. Patients receiving radiation treatment for cancer often experience controlled acute radiation sickness because they receive relatively high levels of radiation. Infertility and cataract are two other examples of nonstochastic effects of radiation; cataract may not occur until several years after exposure. Doses to people near nuclear facilities are far below levels that would cause deterministic effects.

In the case of the effects of exposure to low levels of radiation (less than 0.1 Gy, or 100 mSv effective dose), the scientific uncertainty of radiation-induced cancer is considerable as there is little or no empirical knowledge. Despite the uncertainty, decisions are needed for use in setting standards for protection of individuals against the side effects of low-level radiation. Based on current scientific knowledge (or lack thereof), regulatory agencies in the United States currently use a model that describes radiation injury as a linear function of radiation dose that has no threshold; this is called the linear no-threshold (LNT) model. According to LNT, if a dose equal to 1 Gy gives a cancer risk *X*, the risk from a dose of 0.01 Gy is *X*/100, the risk from 0.00001 Gy is *X*/100,000, and so on. Thus, the risk of health effects including cancer risk is not zero regardless of how small the dose is.

In the LNT model, data from high levels of exposure where radiogenic cancers have been observed are used to extrapolate risks at lower doses where cancers have not been observed, and if they exist they are beyond the current science to observe and measure. One result of following the LNT model is that a very small estimated risk, when multiplied by a large number such as the population of the United States, results in an estimate of a substantial number of cases or deaths, which in reality may not exist.

Scientific groups such as the International Commission on Radiological Protection (ICRP), the National Council on Radiation Protection and Measurements (NCRP), and the National Research Council Committee on the Biological Effects of Ionizing Radiation (BEIR), repeatedly review and endorse the use of this model for assessing risk, which is used to set radiation protection standards and operating policies, such as the "as low as reasonably achievable" (ALARA) policy. This approach is often considered to be conservative and gives emphasis to public health. Data provided by the updated report of the atomic bombing survivors in Japan continue to be in support of the LNT model across the entire dose range. However, a concave curve was the best fit for data restricted to doses of 0-2 Gy. This resulted because risk estimates for exposure to 0.3-0.7 Gy were lower than those in the linear model (Ozasa et al., 2012). The finding was not explained.

Not all countries support the LNT model at this time, but in general it is perceived that with so much uncertainty about the effects at low doses, it is appropriate to continue with the LNT model that has been in place for several decades for purposes of radiation protection.

A.2 BIOLOGICAL RESPONSES AT LOW DOSES

A variety of different biological responses have been identified at low doses of radiation, although it is difficult to identify effects at doses that are close to those encountered from natural background radiation. It is highly unlikely that epidemiologic studies of populations around nuclear facilities will contribute toward knowledge of the effects of radiation at very low doses. Because of the epidemiologic limitations, efforts are directed toward improving understanding of the effects, response, and defense mechanisms to low-dose radiation at the cellular and molecular levels. The Department of Energy's Low Dose Radiation Program is focused on understanding the effects of doses of radiation under 100 mSv by supporting research of the molecular and cellular responses to very-low-dose exposures. Some scientists have argued that DNA repair capabilities are effective at low doses, preventing the accumulation of DNA damage and mutations following low-dose exposures, while others have argued that low doses may be even more damaging per unit dose than high doses.

Major discussion on the biological consequences of low-dose radiation despite being controversial has also led to the identification of pathways of radiation damage that are evident at low doses but difficult to measure at high doses in light of overwhelming DNA damage. Among these is the adaptive response, which would tend to dampen the potential adverse effects and perhaps even provide a beneficial (or hormetic) effect of radiation exposure at low doses. In most studies of adaptive responses, cells in vitro are given a "tickle" low dose of radiation (for example 20 cGy or 0.2 Gy) followed by a high dose of radiation (1 Gy). The administration of the "tickle" dose prevents some of the damaging effects of the high dose, including cell killing and chromosomal injury. In animal models a variety of investigators have documented that low doses of radiation can enhance immune responses (Cheng et al., 2010).

There are also several damaging responses observed at low doses, including the bystander effect and delayed genomic instability. The bystander effect is defined as genetic changes (chromosome damage, mutations) induced in cells that are not directly hit by the radiation beam. The exact mechanism by which the bystander effect occurs is unclear, although data support both transmission of a factor either in conditioned medium (Sowa Resat and Morgan, 2004) or through gap junctions (Gaillard et al., 2009). Recent studies have documented that such bystander effects may occur in vivo as well (Singh et al., 2011). Delayed genomic instability has also been identified in irradiated cell populations where mutations do not occur in the irradiated cells themselves but rather in the progeny of these irradiated cells sometimes up to 13 generations later (Little et al., 1997; Morgan, 2003).

Another detrimental effect of low-dose exposures (mostly in the cGy range) is low-dose hypersensitivity in which some cells in culture show an

enhanced response to the killing effects of x-rays at the very low doses (10-60 cGy) than they do to slightly higher doses (1 Gy, for example). Whether this is really a low-dose hypersensitivity or an induced radiation resistance at the slightly higher doses (1 Gy) is not clear, and the mechanism for it has not been defined, although some attribute it to the need for a threshold number of double-strand breaks to induce cell-cycle arrest (Marples et al., 2004).

Dose-rate factors are also important in considering the effects of low-dose radiation. Most studies have documented that low-dose-rate exposure is less damaging than similar doses administered at high rates, although these studies are limited, difficult to conduct, and predominantly in animal populations (Brooks, 2011; Vares et al., 2011). In long-term animal studies carried out at Argonne National Laboratory in 1960-1990, dogs and mice were exposed to doses of radiation daily with very low doses per day and equal doses given in a single exposure; these studies revealed that life shortening and cancer incidence was significantly higher for animals given the high-dose-rate compared to the low-dose-rate exposures (Carnes and Fritz, 1991; Carnes et al., 1998). In other mouse strains (AKR), a lower incidence of cancer-induced thymic lymphoma was also found in mice exposed to low-dose-rate compared to high-dose-rate radiation (Shin et al., 2011), suggesting that there are significant differences in biological consequences (Uehara et al., 2010).

Radiobiological data, some based on animal experiments, have been the basis of the dose and dose-rate effectiveness factors (DDREFs), that is, factors used to convert risk estimates from populations exposed in larger acute doses such as the atomic bombing survivors to populations who are exposed to lower low-rate doses. The ICRP derived estimates of the excess cancer risk after low-dose exposures and after exposures with higher doses but low-dose rates by reducing the corresponding risk value for the atomic bombing survivors by a DDREF of 2.0 (ICRP, 2007). The BEIR VII Committee used a DDREF of 1.5 (National Research Council, 2005). It has been speculated that these DDREFs underestimate the risks from low-dose-rate exposures. For example, in a recent paper by Jacob et al. (2009), comparisons of risks of radiation workers who receive chronic exposures with those of the atomic bombing survivors who received acute exposures indicated that risks among workers tended to be higher, contrary to expectations.

A.3 BIOMARKERS

Most individuals exposed to radiation do not wear physical dosimeters such as film badges or thermoluminescent dosimeters; therefore, reconstructing their exposure requires collecting information through interviews and available models and thus estimated exposures often contain a high

level of uncertainty. In an attempt to overcome this problem, biological markers are being developed as a useful tool for estimating the exposure and the effects of, or the response to, radiation. A biomarker is in general an end point that is objectively measured and can be used as an indicator of a biological state. Studies have highlighted the importance of biomarker research in radiation epidemiology specifically in assessing occupational exposure (Schneider et al., 1999), exposure following industrial accidents (Menz et al., 1997), as well as response to radiation therapy (Wickremesekera et al., 2001). Two types of purpose-oriented categorizations of irradiation biomarkers have been proposed. Brooks segregates them into markers of exposure, sensitivity, and disease (Brooks, 1999), while others mention predictive, prognostic, diagnostic, and dosimetric markers (Okunieff et al., 2008). A single biomarker can often fit into several of these categories which serve different purposes. For example, biomarkers of effect measure the biological responses in individuals who have been exposed to an agent (and also include elements of individual sensitivity to that agent); markers of exposure, on the other hand, do not necessarily indicate effects. A methodology-focused categorization of radiation biomarkers would separate them into cytological and molecular markers, both with numerous subcategories. In addition, while cytological markers in radiation research are often very specific, molecular-based radiation biomarkers are often compendia of molecules rather than isolated molecular species. Today, the use of biomarkers in epidemiologic studies of low doses is unlikely to help with dose reconstruction, as the variability of the assays within a person and between persons is a major problem. However, the rapid advances in the research on biomarkers may in the future provide more sensitive tools that may also prove useful for epidemiologic purposes and significantly reduce the uncertainties related with current dose reconstruction models.

A.4 EPIDEMIOLOGIC STUDIES OF IONIZING RADIATION

A.4.1 Studies of Residents near Nuclear Facilities

A British television program in 1983 reported a cluster of childhood leukemia in Seascale, a village 3 km from the nuclear fuel reprocessing facility Sellafield on the Cumbrian coast, then known as Windscale. The television team discovered seven childhood leukemia cases over the previous 30 years, while less than one case was expected (Urquhart et al., 1984). Given the proximity of the village to the nuclear reprocessing plant, and in the absence of any other obvious causative agent, a direct effect of environmental pollution with radioactive waste was hypothesized. The British government appointed an independent advisory group to investigate the claims. The group produced its report within seven months (Black, 1984), confirming

the TV broadcast, but could not explain the finding in terms of radioactive discharges. In response, a governmental Committee on Medical Aspects of Radiation in the Environment (COMARE) was set up in 1985 and over the past 25 years has published several reports using data from the national registry of children's tumors. The reports include an extensive investigation of the Sellafield area (COMARE, 1996) and the sites of Dounreay in Scotland (COMARE, 1988), Aldermaston in Berkshire, and Burghfield in North Hampshire (COMARE, 1989). Reviews by COMARE of the discharges from the nuclear installations showed that the doses that the general public residing in the area were likely to have received were far too small to have caused increases in childhood leukemia (COMARE, 1988, 1989, 1996). In 2011, COMARE published an update on the issue as its fourteenth report (COMARE, 2011), undertaking a further review of the issues addressed in the tenth report that covered the years 1969-1993 (COMARE, 2005). The latest report covered the period 1969-2004 and found no significant evidence of an association between risk of childhood leukemia and living in proximity to a nuclear power plant (COMARE, 2011).

The sequence of cluster or ecologic studies finding excess cancers around a nuclear site and more detailed examination following to confirm the findings and research the associations has been a common approach for many years. Studies from Great Britain, Germany, France, and the United States contribute the most to the literature. Childhood leukemia is primarily investigated as it is recognized to be a "sentinel indicator" for radiation effects occurring with a shorter time latency following exposure and with a stronger dose-risk relationship. Although initially mortality data were used to evaluate the possible impact of living near nuclear facilities under normal operating conditions, it was soon realized that, given the advances in cancer treatment and consequent improvements in survival, incidence data (the number of newly diagnosed cases in a given period of time) could provide more relevant estimates.

Studies on the cancer risks associated with living near nuclear facilities have come to different conclusions, with some suggesting a positive association between living in proximity to a nuclear facility and cancer risk and others suggesting that there is not a risk, or that the risk is too small to be detected with the methodology used. The power of a study to detect an effect, if there is one, depends highly on the hypothesized strength of the association to be detected and the sample size. Neither of these variables is likely to be high in an epidemiologic study of cancer risks in populations near nuclear facilities:

a. The size of the estimated risks from reported radioactive effluent releases from nuclear facilities is likely to be small. Consequently,

epidemiologic studies have a limited ability to discern associations between radiation exposure and cancer risk in these populations.

b. The size of the populations most likely to be exposed (that is, those living in very close proximity to a nuclear facility, for example within a 5-10-km radius) is relatively small. This limits the expected number of informative (exposed) incident cases or deaths that will be available for study, especially for rare cancers such as those of childhood.

Study conclusions are based on a very small local population size, which makes the risk estimations statistically unstable because a single additional case, or one less case, can change the rate estimate dramatically. For example, in the study in Germany with 23 years of follow-up, out of the 593 leukemia cases in children under 5 years old diagnosed in the study area, only 37 cases (6 percent) were observed in the risk zone (≤ 5 km from a facility) (Kaatsch et al., 2008). Similarly, in the recent COMARE report (2011) with 35 years of follow-up, out of the 430 leukemia cases in children under 5 diagnosed in an area up to 25 km from the nuclear power plants in Britain, only 20 (5 percent) were in the risk zone (Table A.1). It is expected that a study in the United States would contain a larger number of exposed individuals than those in the European studies because the number of nuclear power plants in the United States is larger than that in any of the European countries.

For this and other reasons related to differences in study design or analysis stages (results may be influenced, for example, by unrecognized bias in the data, the effect of other relevant factors, or by chance variation; these need to be discussed by the investigators even if they cannot be quantified), interpretation of epidemiologic findings is not always easy and there are often subjective elements to their interpretation that experts may disagree upon. Evaluating well-designed studies that do not suggest the existence of an association between a factor and a disease is equally important

TABLE A.1 Number of Cases in the At-Risk Zone (≤ 5 km from a facility) in European Studies of Pediatric Cancers (children < 5 years old)

Country	Reference	Study Years	End Point	Cases (≤ 5 km)
Germany	Spix et al., 2008	23	all cancers	77
	Kaatsch et al., 2008		leukemia	37
France	Sermage-Faure et al., 2012	17	leukemia	24
Britain	COMARE, 2011	35	leukemia	20
Switzerland	Spycher et al., 2011	24	all cancers	18
			leukemia	8

to evaluating studies that show an association. However, it is often harder to convince stakeholders of the validity of the so-called "negative" studies especially if there are flaws or inefficiencies in their design, methods, or analysis. A better term for flawed studies would be "uninformative."

In absence of biological plausibility, a positive or somewhat positive association may be underinterpreted. In studies that assess cancer risks associated with releases from nuclear facilities, there are examples where investigators are hesitant to conclude that evidence supported the hypothesis when they find a positive association between risk and exposure associated with nuclear facilities (Baker and Hoel, 2007; Hatch et al., 1990; Kaatsch et al., 2008; Nuclear Safety Council and the Carlos III Institute of Health, 2009), even though direct radiation measurements were not made. This phenomenon has led a researcher to emphasize the importance of having explicit study hypotheses (Wing et al., 2011) and to the question, "Why conduct a study if the results cannot be interpreted as providing evidence in support of the hypothesis?" (Wing, 2010). Of course, there is the opposite error, too—that of overinterpretation. A balanced "weight-of-evidence" approach is the most appropriate.

It is important to be open to new information or novel interpretation and alternative hypotheses that can impact assumptions about exposure effects. A recent study from France demonstrated that children living in very close proximity to nuclear power plants are twice as likely to develop leukemia compared to those living farther away from the plants. However, analysis of the same population of children using a dose-based geographic zoning approach instead of distance, did not support the findings. The absence of any association with the dose-based geographic zoning approach may indicate that the observed association with distance may be due to factors other than the releases from the nuclear power plants (Sermage-Faure et al., 2012). Among such potential factors are population mixing (Kinlen, 2011a), a hypothesis that could not be evaluated in this study, and exposures to agents including natural or manmade exposures to radiation not modeled in the study.

From the reports published the past 4 years alone from Germany (Kaatsch et al., 2008), Finland (Heinavaara et al., 2010), Great Britain (COMARE, 2011), Switzerland (Spycher et al., 2011), and France (Sermage-Faure et al., 2012), it is obvious that additional scientific resolution to the question of whether living near a nuclear facility increases one's risk of developing cancer remains. Authors have called for collaborative analysis of multisite studies conducted in various countries (Sermage-Faure et al., 2012). Similarly, the need for a well-conducted meta-analysis that would provide a more precise estimate of the risk remains.

Two meta-analyses were conducted recently in an effort to provide more precise estimates of the possible risks associated with living near a

nuclear facility (Baker and Hoel, 2007; Greiser, 2009). Baker and Hoel combined and statistically analyzed studies of childhood leukemia around nuclear facilities published until 1999, but only included studies that calculated standardized incidence ratios (SIRs) or standardized mortality ratios (SMRs) (see Sidebar A.1 for risk measures) for individual facilities. Studies that calculated rates for multiple sites or those that did not distinguish leukemia from lymphoma were excluded. Seventeen published studies (out of 37 individual studies published at the time) addressing 136 nuclear sites in 7 countries (Great Britain, Canada, France, United States, Germany, Japan, and Spain) met the criteria. Due to variability between study designs, eight separate analyses were performed stratified by age and zone. Meta-SMRs and meta-SIRs were all greater than the reference group, implying an increase in risk. More specifically, the overall estimated relative risk was 1.22 (95% CI=1.05-1.41) and the 0-9 age group accounted for the majority of the excess cases and deaths. Excluding the Aldermaston nuclear weapons plant and Amersham plant that produces radioisotopes (both in Britain) reduced the overall estimate to a nonsignificant 14 percent increase in risk (RR=1.14, 95% CI=0.98-1.33). The authors discuss that although the meta-analysis showed an increase in childhood leukemia near nuclear facilities, it "does not support a hypothesis to explain the excess" (Baker and Hoel, 2007).

The meta-analysis by Baker and Hoel was criticized by authors of the German *Kinderkrebs in der Umgebung von Kernkraftwerken* (KiKK) study (Spix and Blettner 2009). The first issue they identified with the meta-analysis was the general problem of combining heterogeneous data such as different age groups (0-9 years or 0-25 years), the different types of nuclear facilities (nuclear power plants and other facilities), and the different exposure zone definitions (<10 km or county). Beyond that, there was criticism over the completeness of the publication search and lack of justification for excluding the 20 studies which were identified but did not fit the criteria for inclusion; possible selection bias resulting from the exclusion of sites with zero observed leukemia cases or deaths from leukemia; and a methodological problem with the confidence intervals presented in the forest plots which should be symmetric on a logarithmic scale but, contrary to expectation, were skewed (Spix and Blettner, 2009).

The meta-analysis by Greiser included data from 80 nuclear power plants in five countries (Germany, France, Great Britain, United States, and Canada). Data were retrieved in the literature but also from cancer registries. (Rather than relying on the data used in the Jablon et al. 1991 analysis of risks in nuclear facilities in the United States, the author retrieved cancer incidence data from cancer registries of Illinois, Pennsylvania, and Florida.) The incidence of leukemia was estimated to increase by 13 percent (95% CI = 10%-17%) relative to the corresponding average national or regional rate (Greiser, 2009). The latest COMARE report (2011) discusses the key

SIDEBAR A.1
Risk Measures, *P* Values, and Confidence Intervals

Several types of estimates of relative risk (RR) are used in epidemiologic studies. RR is generically defined as the ratio of the risk of developing the disease or of dying of the disease among an exposed population compared to an unexposed population. A simple type of estimate of the RR is the standardized incidence ratio (SIR) or standardized mortality ratio (SMR) for the exposed group. An SIR is the ratio of the number of cases observed in the exposed group in some time period to the number of cases expected if the group had the same disease occurrence rates as a standard population. The standard population is often the general population or a large reference population with characteristics similar to the study group except for the exposure of interest, and comparisons typically are based on cancer rates from population cancer registries. The ratio of observed to expected cases is often multiplied by 100 to yield results without decimals. Thus, an SIR of 100 indicates that the observed number of cases is the same as that expected in the standard population. Thus, an SIR of 140 indicates that incidence is 40 percent higher than expected, while an SIR of 80 indicates 20 percent fewer cases than expected.

SIRs should be interpreted with caution as their significance partially depends on the number of cancer cases in the exposed group. Imagine a situation where 5 cases were expected and 6 were observed and a second situation where 500 cases were expected and 600 were observed. In both instances the SIR is 120; however, because in the second scenario the SIR is based on a greater number of cases, the estimate is more precise, and hence more meaningful. In other words, although the one excess case could have occurred due to chance alone, it is highly unlikely that an excess of 100 incident cases has occurred by chance. This is a common issue in interpreting studies of risks in populations near nuclear facilities where the number of excess cancers in the exposed region is particularly small when rare diseases such as childhood leukemia are examined (see Table A.1).

The SMR is similar to the SIR, except it is based on deaths due to some cause rather than cancer occurrences to draw conclusions regarding whether there is excess mortality. As age is one of the main determinants of mortality, and other factors such as gender and racial composition may influence the mortality or tumor rates, SMRs and SIRs are usually calculated by summing the observed and expected numbers of deaths or cancers across categories of gender, age, and sometimes race with the expected numbers calculated separately for each category.

Results from cohort and ecologic studies are sometimes described in terms of SMRs or SIRs, but other techniques are often preferred which permit comparisons of disease rates (often called rate ratios) between exposed and unexposed study groups, usually with adjustment for gender, age, and perhaps other factors. More advanced techniques use some type of "regression analysis" to estimate exposure-effect associations, with study subgroups or individuals defined according to graded amounts of exposure.

Case-control studies (which compare exposures observed in cases to those observed in control subjects) are typically unable to calculate actual disease rates since they lack appropriate population denominators, which means that SIRs, SMRs, and rate ratios cannot be used. However, for case-control studies the odds ratio (OR) can be calculated. The OR and relative risk are closely related (and are nearly identical for "rare" diseases). The OR indicates the ratio of the probability of exposure to the prob-

ability of nonexposure among those with the disease of interest divided by the similar ratio of probabilities among those without the disease. A value greater than 1 means that the odds of disease are greater among the exposed than the unexposed. A value less than 1 means that the odds are higher in the unexposed than in the exposed. Similar to all the other statistics mentioned, the number of disease cases with exposure has a major influence on the precision and statistical significance of the OR.

A useful measure of risk in epidemiologic studies is that of "excess" risk associated with an exposure. Excess risk can be expressed as excess relative risk (ERR) or excess absolute risk (EAR). The ERR and EAR in principle are estimates of the amount of risk due to the exposure of interest when the effects of other risk factors are removed. Statistically, ERR = RR-1 and EAR = $R_E - R_U$, where R_E is the rate of occurrence of disease or death in the exposed group in a specified period, and R_U is the corresponding rate of occurrence in the unexposed group, which is the baseline rate. In contrast to ERR, which represents the ratio of the excess rate associated with exposure to the baseline rate, the EAR represents the additional rate of a disease due to the exposure of interest over a given period of time. As baseline disease rates depend on a number of factors, excess risks can vary not only with radiation dose but also with age at exposure, time after exposure, age at risk (attained age), gender, and other factors such as smoking. Therefore, risk estimates are usually reported for a specified combination of these factors. ERR and EAR estimates can best be calculated in a cohort study, although ecologic studies can sometimes permit such estimates to be made. A statistic analogous to the ERR estimate can be calculated as OR-1 for case-control studies, but usually EAR estimates cannot be obtained from a case-control study due to the lack of population denominators.

By describing the excess number of people affected by the disease of interest, EAR is a better descriptor than the ERR of the public health impact that an exposure may have in a population. For example, in the Life Span Study (LSS) follow-up of the Japanese atomic bombing survivors the ERR for leukemia is the highest among the various cancer effects of radiation exposure (RR approximately 5 for a dose of 1 Gy, which translates into an ERR of about 4), and the total number of radiation-related cases of leukemia among the LSS survivors is estimated to be about 90-100. In contrast, the ERR for solid cancers is much smaller (RR approximately 1.5 at 1 Gy, or an ERR of about 0.5), yet the total number of LSS survivors who have developed solid cancers due to the bombing is estimated to be about 850. This is because of the relative rarity of leukemias compared to the group of cancers described as solid cancers. Common cancers may appear to have a low ERR in an epidemiologic study, but the risk may translate to a large number of cases, or a large EAR. One can say that the ERR is an appropriate measure to assess disease etiology, whereas the EAR is useful for estimating the extent of a health problem.

Applying ERR or EAR estimates derived from individuals in one population to those in another population sometimes has substantial uncertainties. Since most types of cancer vary substantially in their baseline frequency according to age, both ERR and EAR estimates can be affected by differences in the age distributions of populations being compared. For instance, it would be inappropriate to compare radiation-related leukemia risk of children in one population with adult leukemia risk in another population. Sometimes there also are differences in the baseline rates of disease in different populations even with the same age distributions. For example, the Japanese have historically had much higher rates of stomach and liver cancers than in the United States. It is therefore uncertain as to how to extrapolate stomach or liver cancer ERR

continued

SIDEBAR A.1 Continued

or EAR risk estimates from the Japanese atomic bomb survivors to the U.S. population. Careful analysis and interpretation is required in making projections of risk across populations.

By itself a point estimate whether it is an SMR, SIR, OR, or RR is difficult to interpret because it does not indicate the extent to which chance may have played a role. This additional information regarding the reliability of an estimate is provided by calculating the *confidence interval*. A confidence interval with a particular confidence level, commonly set up at 95 percent, is intended to give the assurance that, if the statistical model is correct, the true value of the parameter is within the range indicated. If the 95 percent CI range does not include 1, then the estimated risk is significantly different from that of a comparison group. For example, if the risk ratio of a smoker being diagnosed with lung cancer is estimated to be 10 when compared to the risk of a nonsmoker and the 95 percent confidence interval (CI) is 8.6-12.7, then the investigator can conclude that the risk ratio is significantly higher than 1 as there is less than 5 percent chance that the observed difference is the result of random fluctuation. The width of the CI is also very important as it indicates the precision with which the risk is estimated. Narrow estimation indicates a fair level of certainty that the calculated estimate falls within a narrow range. A wide interval makes the estimation "imprecise" and leaves considerable doubt as to the accuracy of the estimate. However, confidence intervals do not account for the uncertainty resulting from bias in exposure estimates,

problems with the analysis, which are both methodological and also relate to lack of justification for excluding studies from the meta-analysis (for example, data from Japan).

The limitations of the two meta-analyses discussed here defeat their purpose, which is to estimate the effect size with higher precision than the single studies which are often underpowered. In addition, the selection of data to be included or excluded from the meta-analysis can influence the results. Although meta-analyses often suffer from the general problem of summarizing heterogeneous data and the possibility of "publication bias"[1] (studies that find a positive association are more likely to be published compared to studies that find no association), they ought to be based on a thorough literature search that identifies relevant studies and to clearly state the criteria and justify excluding studies from the analysis.

A review of the literature that includes all cancer types and all ages is presented here. Table A.2 summarizes information from selected multisite leukemia studies of children that investigated place of residence at time of

[1]Negative studies often do not interest the publishers, who may be biased in favor of positive or promising results (Simes, 1986), or the researchers themselves fail to write them up and submit them for publication (Angell, 1989). The results from the meta-analysis would then be skewed toward a positive association.

or from confounders that investigators were not able to fully adjust for, or confounders that were unidentified.

The *P* value is a statement of the probability that the association observed could have occurred by chance under the assumption that the null hypothesis is true. Traditionally, a *P* value < 0.05 is considered as sufficiently unlikely for the association to have occurred by chance and justifies the designation "statistically significant." The smaller the *P* value, the less likely the observed association could have occurred by chance under the null hypothesis. *P* values can be either two-tailed (also called two-sided) or one-tailed (or else one-sided) depending on the alternative hypothesis tested. The one-tailed test provides more power to detect an effect in the direction tested and should be used only after considering the consequences of missing an effect in the untested direction. The KiKK study, for example, used a one-tailed test and limited attention to identification of increases associated with living near a nuclear facility (Kaatsch et al., 2008).

Inferences about an association between a disease and an exposure are considerably strengthened if information is available to support a dose response in the relationship between the degree of exposure and the disease. In that case, risks are estimated for every category of exposure and a *P for trend* is estimated (that is, the alternative hypothesis reflects a trend of effect across exposure values rather than an increase or decrease for particular ranges of exposure).

diagnosis or death, or place of birth in relation to nuclear facilities as a risk factor for the disease.

A.4.1.1 Great Britain

In Great Britain the first multisite study came immediately as a response to the reported cluster in Sellafield. In 1984, Baron examined cancer mortality trends for the small areas around 14 nuclear installations in England and Wales using census and survey data for the years 1974-1979 (Baron, 1984). In the short period of observation, the data did not indicate any increase in mortality in areas around the major nuclear facilities examined. A year later, a preliminary report on the incidence of leukemia for the years 1972-1984 in children with age equal to or less than 9 years living near two nuclear establishments, the Atomic Weapons Research Establishment at Aldermaston and the Royal Ordnance Factory at Burghfield in the West Berkshire District Health Authority, showed that the incidence among those 0-4 years of age increased 60 percent (Barton et al., 1985). The study did not include children residents of the West Berkshire District Health Authority who were referred elsewhere for diagnosis and treatment. An updated and extended study that included incident cases diagnosed in 1985, those aged 10-14 years and residents in the above-mentioned district and neigh-

TABLE A.2 Selected Multisite Studies of Leukemia among Young People Living near Nuclear Facilities

A. Ecologic studies

Country	Reference	No. of Sites	Study Period	Age	I/M	Exposed Areas	Comparison Areas	No. Cases	SIR or SMR
Britain	Forman et al., 1987	14	1959-1980	0-24	M	10 km	Control local authority	44	2
Britain	Cook-Mozaffari et al., 1989a	15 (+8 possible)	1969-1978	0-24	M	16 km	Other districts	635	1.15
United States	Jablon et al., 1991	62	1950-1984	0-9	M + I	107 counties	292 counties	1,390	1.03
Canada	McLaughlin et al., 1993a	5	1950-1987	0-14	M + I	25 km	Province rates	54	1.17
Germany	Michaelis et al., 1992	20 (+6 possible)	1980-1990	0-14	I	15 km	30-100 km	274 13	1.06 7.09 (children <5 yr)
Britain	Bithell et al., 1994	23 (+6 possible)	1966-1987	0-14	I	25 km	National rates	4,100	0.98 (for NPPs) 1.02 (possible sites)
Scotland	Sharp et al., 1996	7	1968-1993	0-14	I	25 km	National rates	399	1.99 (reprocessing plant) 0.90 (for NPPs)
Germany	Kaatsch et al., 1998	20	1991-1995	0-14	I	15 km	30-100 km	550	1.05
France	White-Koning et al., 2004	29	1990-1998	0-14	I	20 km	National rates	670	0.92
Japan	Yoshimoto et al., 2004	44	1972-1997	0-14	M	10 km	10-80 km	473	1.01
France	Evrard et al., 2006	23	1990-2001	0-14	I	40 km^2	national	750	0.94
Britain	Bithell et al., 2008	13	1969-1993	0-4		5, 10, 25 and 50 km	National rates	409	1.36 (<5 km) 0.90 (<10 km) 0.97 (<25 km)

Finland	Heinavaara et al., 2010	2	1975-2004	0-20	I	15 km	Stratum-specific incidence rates	16	1.01
Britain	COMARE, 2011	13	1969-2004	0-4	I	25 km	National rates	511	1.01 (<5 km) 1.01 (<10 km) 1.00 (<25 km)

NOTE: I, incidence; M, mortality; SIR, standardized incidence ratio; SMR, standardized mortality ratio; OR, odds ratio; RR, relative risk; NPP, nuclear power plant.

B. Case-control studies

Country	Reference	No. Sites	Period of Diagnosis	Age	Area Examined	No. Cases	OR
Germany	Kaatsch et al., 2008	16	1980-2003	0-4	≤5 km ≤10 km	593	2.19 1.33
Finland	Heinavaara et al., 2010	2	1975-2004	0-14	5-10 km vs ≥30 km	16	0.7
France	Sermage-Faure et al., 2012	19	2002-2007	0-14	≤5 km vs ≥20 km	2,753	1.9 with distance 1.0 with dose-based zoning

C. Cohort studies

Country	Reference	No. Sites	Period Examined	Age	Area Examined	No. Cases	RR
Finland	Heinavaara et al., 2010	2	1975-2004	0-14	<15 km vs 15-50 km	16	1.0
Switzerland	Spycher et al., 2011	5	1985-2009	0-14	≤5 km vs >15	953	1.24

boring districts that may have been referred elsewhere for diagnosis, was conducted (Roman et al., 1987). Among the 60,000 children residents within a 10-km radius of a nuclear establishment, the recorded incidence was three cases per year while two cases per year were expected.

In 1986 a cluster of leukemia among children was reported around the area of the Dounreay nuclear reprocessing plant in Scotland (Heasman et al., 1986). In 1987 and 1989 two reports were published of an increased rate of leukemia in children under 15 years of age that reside within a 16-km (10-mile) radius of the nuclear weapons plants in Aldermaston and Burghfield (Forman et al., 1987; Roman et al., 1987) and the Hinkley Point nuclear power station in Somerset, England (Cook-Mozaffari et al., 1989a; Ewings et al., 1989). This later cluster was not confirmed by follow-up studies (Bithell et al., 1994). In 1992, a fifth cluster was reported in Britain among children under 10 years of age near the Amersham plant that produces radioisotopes (Goldsmith, 1992). Again the increased incidence was not confirmed by others (Bithell et al., 1994).

Using more comprehensive data sets and analyses, Draper and colleagues (1993) aimed first to reappraise the original report of possible excess of childhood leukemia incidence and non-Hodgkin's lymphoma in areas around the Sellafield nuclear installation and second to determine whether the excess incidence persisted in the years following the original report. All ages and other cancers were included. The authors confirmed an increased incidence in cancer, especially leukemia in young people. Cook-Mozaffari et al. (1989b) analyzed data on mortality for 400 districts of England and Wales where there was an existing nuclear installation or the construction of nuclear installations had been considered or occurred at a later date. The authors report an excess mortality due to leukemia in young people who lived near potential sites similar to that in young people who lived near existing sites, implying the presence of unidentified risk factors associated with the sites where nuclear stations reside or are selected to reside but not associated with the nuclear installations themselves.

A study aiming to examine the contribution of potential risk factors to the observed excess of childhood leukemia (< 25 years of age) and lymphoma near the Sellafield nuclear plant in Cumbria, England, was conducted, this time using a case-control design (Gardner et al., 1990). Fifty-two cases of leukemia, 22 cases of non-Hodgkin's lymphoma, and 23 Hodgkin's disease patients diagnosed in the period 1950-1985 and 1001 controls matched on sex and date of birth were compared. Antenatal abdominal x-ray examinations, viral infections, behavioral data, lifestyle factors, and parental employment at Sellafield were examined as potential risk factors. The authors concluded that there is an association between childhood leukemia and paternal exposure before conception to relatively high doses of radiation. More specifically, the relative risk for paternal

estimated dose of ≥100 mSv before the child's conception was 8.4 (95% CI: 1.4-52.0 based on 4 exposed cases). However, the relative risk for the next-highest paternal preconception dose category of 50-99 mSv was only 0.78 (CI: 0.1-7.8 based on 1 exposed case), which was not very supportive of a dose-related risk. When doses received 0-6 months before conception were examined, the relative risks for the highest (≥10 mSv) category was 8.2 (CI: 1.6-42 based on 4 exposed cases) and for 5-9 mSv was 3.0 (CI: 0.3-33, 1 exposed case). The authors speculate that radiation exposure during work may have an effect on the father's germ cells, producing genetic changes in sperm that may be leukemogenic in the offspring. The evidence, however, seems mixed and subsequent independent investigations in England, France, Scotland, and Canada did not support this association (Draper and Vincent, 1997; Draper et al., 1997; Kinlen et al., 1993; McLaughlin et al., 1993b; Pobel and Viel, 1997).

Bithell and colleagues (1994) performed the largest (at the time) incidence study for all of England and Wales and examined the relationship between the risk of childhood leukemia (<15 years of age) and non-Hodgkin's lymphoma and proximity of residence to 23 nuclear installations for the period 1966-1987. The authors investigated regions of 25-km radius and six control sites that had been considered for generating stations but were never used. Observed and expected numbers of cases were calculated and analyzed by standard methods based on ratios and by linear rank score test. Overall, there was no evidence of an increase of childhood leukemia or of non-Hodgkin's lymphoma around nuclear installations. The only significant results for the linear rank score test were for Sellafield and a weaker but significant association for Burghfield. The authors noted that a more appropriate analysis would be one based on place of residence at birth as an analysis based on place of diagnosis may fail to detect the effect of prenatal or preconception factors. A year later, a mortality study investigated seven districts near the sites of Harwell, Aldermaston, and Burghfield for the period 1981-1995, among children younger than 15 years. Excess leukemia deaths were reported in two districts (Newbury, 11 deaths observed, 5.7 expected; South Oxfordshire, 12 deaths observed, 4.9 expected) (Busby and Cato, 1997). However, the ranking of the seven districts by incidence rates for the period 1969-1993 did not agree with that for mortality and no excess of leukemia cases existed (Draper and Vincent, 1997). In Scotland, Sharp and colleagues carried out a similar study of the seven nuclear sites for the period 1968-1993. The only significant observation was the reported excess around the Dounreay reprocessing plant (Sharp et al., 1996).

The reported cluster around Dounreay, Scotland, was referred to COMARE for consideration and the committee recommended further epidemiologic investigations, including a cohort study of the incidence of leukemia among children born locally and those who attended school in the

area but were born elsewhere (Black et al., 1992) and a case-control study to examine possible risk factors for leukemia (Urquhart et al., 1991). The aim of the cohort study was twofold: (a) to determine whether the excess of leukemia and other cancer cases occurred in children born to mothers that were residents in the Dounreay area or in children who moved to the area after birth and (b) to determine whether any leukemia cases occurred in children born near Dounreay who may have moved elsewhere. The cohort included 4,144 children born in the area in the period 1969-1988 and 1,641 children who attended local schools in the same period who had been born elsewhere. Cancer registration records were linked to birth and school records and observed rates were compared to national rates. The authors showed that the incidence of leukemia and non-Hodgkin's lymphoma was raised in both the birth and school cohorts with observed-to-expected ratios of 2.3 and 6.7, respectively, suggesting that the place of birth was not a more important factor than place of residence in the series of cases observed near the Dounreay area. No cases were found in children who were born in Dounreay and moved elsewhere (Black et al., 1992).

The excess incidence of leukemia and non-Hodgkin's lymphoma in children and young adults in the area less than 25 km from the Dounreay nuclear installation was later reexamined for the period 1968-1991 and was found to continue to be a matter of concern (Black et al., 1994). In the case-control study, the study participants were 14 cases of leukemia and non-Hodgkin's lymphoma occurring in children aged less than 15 years diagnosed in the area between 1970 and 1986 and 55 matched controls. Antenatal abdominal x-ray examination, drugs taken, and viral infections during pregnancy were examined as potential risk factors by interviews and structured questionnaires. Given the findings of Gardner et al. (1990), who reported a possible association between paternal employment and development of leukemia by the offspring, detailed information on father's occupation, father's employment at Dounreay, and radiation dose preconception exposure to nonionizing radiation of the father was collected. The study in Dounreay did not provide any evidence of father's employment as a risk factor for childhood leukemia. (However, a possible but weak association between the children's use of local beaches and risk of leukemia was identified.) The paternal preconception exposure theory of genetically transmitted disease was also rejected by Doll, who published a commentary entitled "Paternal exposure not to blame," emphasizing the fact that the hypothesis that irradiation of the testes causes any detectable risk of leukemia in the offspring does not agree with what is known of radiation genetics or of the heritability of childhood leukemia (Doll et al., 1994).

A year earlier, Kinlen et al. (1993) also argued that paternal exposure as a risk factor for childhood leukemia would not explain the excess. Kinlen speculated that nuclear plants that were built in unusually isolated places,

for example, Dounreay and Sellafield in Britain, led to large influxes of people such as construction workers, scientists, and "nuclear" employees in the 1950s to those areas. Indeed, the development of the Dounreay plant, which started its operations in 1958, raised the population in the area of nearby Thurso almost 150 percent between 1951 and 1961. This or similar situations (irrelevant to the radiation industry) may result in bringing into contact susceptible and infected individuals for some unidentifiable transmissible agent whose route and nature of the infection remain unknown. Infected individuals could have been present in any of the groups and given a sufficient population density could have caused outbreaks (Kinlen, 2011a; Kinlen et al., 1995). The theory of population mixing was originally applied on the North Sea oil industry in Scotland (Kinlen et al., 1993) and was also tested later on the Nord Cotentin region in France, which shares some characteristics with the Sellafield and Dounreay regions in terms of population influx between the years 1978 and 1992 with the construction of the La Hague nuclear waste reprocessing site and the Flamanville nuclear power station (Boutou et al., 2002). Although the hypothesis of an infectious agent has some plausibility, the studies assessing the hypothesis are ecologic and have inherent limitations that would not allow them to prove a causal relationship between the unknown infectious agent and the disease. Still, the Kinlen hypothesis of population mixing is well perceived today and, although it has not been explicitly examined, it is part of the discussion of the studies on cancer risks in populations around nuclear facilities published the past 2 years (COMARE, 2011; Sermage-Faure et al., 2012; Spycher et al., 2011).

Following the publication of the results from the KiKK study showing an increased risk among children 5 years of age or younger that live within the 5-km radius from German nuclear power plants (Kaatsch et al., 2008; Spix et al., 2008), Bithell et al. (2008) conducted a study to reexamine the incidence of childhood leukemia around nuclear power plants in Britain. The main reason was that results from Germany did not support those of COMARE published in 2005, and this discrepancy could be accounted for by methodological differences, especially those related to the distances from the power stations and the ages of the children investigated. Bithell and colleagues used the same data as considered by COMARE's tenth report and modified the methodology to apply as similar of an approach as possible to that of the KiKK study. The incidence of childhood leukemia observed (18 cases against 14.58 expected within the 5-km zone) was not significantly raised. The original paper (Bithell et al., 2008) made no adjustments for demographic characteristics to resemble the methodology of the KiKK study. Follow-up analysis (Bithell et al., 2010) adjusted for population density at the ward level without altering the overall conclusions.

The latest report from Britain and the fourteenth in series by COMARE

presented a new geographic data analysis on the incidence of leukemia in children under 5 years of age, living in the vicinity of 13 nuclear power plants (COMARE, 2011). The investigators used cancer registration data for the period 1969-2004 extending the previous analysis presented in COMARE's tenth report for 1969-1993. The report concluded that there is no evidence to support an increased risk of childhood leukemia and other cancers in the vicinity of nuclear power plants due to radiation effects. COMARE recommended that monitoring of liquid carbon-14 discharges from the plants continues, as this radioactive isotope of carbon is a major contributor to the radiation doses which the public receive from discharges. Moreover, the report recommends that research continues for all possible causative mechanisms of leukemia, including the role of infectious agents. An extensive review of the KiKK study as well as useful unpublished analyses of the data are presented in the report.

A.4.1.2 Germany

An excess of childhood leukemia cases in the small rural community of Elbmarsch in Northern Germany, close to the Krümmel nuclear power plant, was first reported in the early 1990s (Schmitz-Feuerhake et al., 1993). Between 1990 and 1995, six cases of childhood leukemia were diagnosed, five of whom resided within a 5-km radius from the plant (Hoffmann et al., 1997). The cluster persisted until at least 2005 (Grosche et al., 1999; Hoffmann et al., 1997, 2007), and together with that of Sellafield and Dounreay (both fuel reprocessing plants) was a confirmed cluster of childhood leukemia near nuclear facilities (Laurier et al., 2008b). The modestly elevated levels of cesium detected in rainwater and air samples led to postulations that there was an accidental release of radionuclides from the nuclear research facility near the community (Schmitz-Feuerhake et al., 1997).

An ecologic study that compared disease rates within 15 km of German nuclear plants with those in control areas was designed following an approach almost identical to the British studies (Michaelis et al., 1992). The German study was based on 1,610 childhood malignancies identified from the country's childhood cancer registry including leukemia cases that were diagnosed before the child's fifteenth birthday from 1980 to 1990. An increased risk of all cancers or leukemia within the 15-km zone was not confirmed. However, exploratory analysis indicated that in children younger than 5 years old living within the 5-km zone, the increase in leukemia risk was statistically significant. A second study was undertaken to validate the results of the previous exploratory analysis and include independent data for the period 1991-1995 (Kaatsch et al., 1998). Results did not support the original hypothesis or the exploratory findings from the 1980-1990

period, although a tendency toward an increased risk estimation for leukemia to occur in children younger than 5 years within the 5-km vicinity persisted. Although the authors concluded that at that point no further investigations were necessary in Germany, discussions on the potential elevated risk of cancer in populations living near nuclear facilities under routine operation did not cease. This led the German federal government to start a case-control study, the third one in a series of corresponding investigations which differs from the previous ecologic studies that were based on aggregate data. The case-control study investigated exact information on distance of the family's place of residence at the time of diagnosis to the chimney of the nearest nuclear power plant with a precision of 25 m (Kaatsch et al., 2008).

The study is known as the KiKK study and was carried out by researchers from the German Childhood Cancer Registry in Mainz, on behalf of the Federal Office of Radiation Protection. Control subjects were randomly selected from the records of the appropriate registrar's office and matched to cases for the date of birth, age, gender, and nuclear power plant area. Five hundred and ninety-three leukemia cases and 1,766 matched controls were included in the study; however, only 37 cases lived within the 5-km zone, the most important number to assess the meaningfulness and strength of the observed association. Analysis indicated a statistically significant odds ratio (OR) of 2.19 [lower limit of the 95% confidence interval (CI) = 1.51] for residential proximity within 5 km of one or more of the 16 nuclear power plants compared to residence outside these areas for children aged less than 5 years. No effect was seen for the distance 5-10 km from a plant (OR = 1.09, based on 58 cases). A negative trend for distance was identified; the farther the residence was from the nuclear power plant, the lower the risk. No association between distance to the nuclear power plants and risk of developing leukemia was observed when children aged 0-15 years were examined together. The investigators attempted to collect data on exposures, residential history, and other potential confounders such as socioeconomic characteristics, pesticides, and immunological factors by administering questionnaires to a subset of the study participants. Because the response rates varied remarkably with distance to the plants (total response was 78 percent for cases, 61 percent for controls; response in the inner 5-km zone was 63 percent for cases, 45 percent for controls), the results were not summarized due to the high risk of selection bias. In the absence of a questionnaire survey, potential confounders could not be investigated; therefore, the study overall did not differ substantially from ecologic studies. Still, the study was associated with wide publicity (http://www.bfs.de/en/bfs/presse/pr07/pr0712, http://www.bfs.de/en/kerntechnik/kinderkrebs/statement_kikk_en.pdf) and some have argued that the sponsoring body made extravagant claims of its importance (Kinlen, 2011b).

The study has been criticized for potential defective control selection (COMARE, 2011; Little et al., 2008a), but also for the misleading presentation of study findings by zone, time period, and malignancy subtype (Kinlen, 2011b). As discussed in a recent critical review (Kinlen, 2011b), some 10 percent of community registrars tasked with control selection declined to cooperate, the proportion being higher within the 5-km zone (16 percent). Moreover, some registrars did not follow instructions regarding matching criteria of cases and controls, selecting potential control children for an inappropriate calendar year, that is, not for the year the matched case was diagnosed. Moreover, the increased risk was driven by risks associated with early operational years: The data from the most recent 8 years (1996-2003) were suggestive of a trend, though the association was not as strong as the earlier period (1980-1995, OR = 1.8, 95% lower bound of the CI: 0.99). Additionally, results seemed to be driven by the notable excess of cases of childhood leukemia around the Krümmel plant in northern Germany, an analysis that was not undertaken by the original authors but by COMARE (COMARE, 2011).

The same group published results from a larger population (1,592 cases and 4,735 controls) that included all other childhood cancers and concluded that leukemia was driving the positive association of cancer risk and living near the installations (Spix et al., 2008).

The Northern Germany Leukemia and Lymphoma (NLL) study is a population-based case-control study that preceded the KiKK and was designed to address the risk associated with three environmental exposures simultaneously: ionizing radiation released from nuclear power plants, electromagnetic fields, and pesticides (Hoffmann et al., 2008). In contrast to the KiKK study, which relied on distance to the residence as a surrogate of exposure, the NLL study reconstructed radiation doses arising from routine discharges of radioactive material from four nuclear power plants by extracting relevant information obtained from questionnaires. Exposure to ionizing radiation due to medical diagnostic or therapeutic radiation was also assessed. The NLL study did not find an elevated risk with the radiation doses assessed to have been received as a result of routine discharges from the nuclear power plants.

A.4.1.3 France

Following the cancer mortality study around nuclear installations in Great Britain (Forman et al., 1987), Hill and Laplanche (1990) reported the results of a similar study for the population residing around six nuclear installations in France, four of which were nuclear power plants. In the period 1968-1987, the number of leukemia deaths among children and young adults aged 0-24 was 58, compared to 62 in control areas. In the

same period, two studies examined mortality from leukemia among those aged 0-24 near the La Hague reprocessing plant in Nord Cotentin, a region with particularly high density of nuclear installations. No findings of excess mortality were reported (Dousset, 1989; Viel and Richardson, 1990,1993). An extended multisite study that included observed leukemia deaths for the years 1988 and 1989 around 13 nuclear installations, of which 11 were nuclear power plants, also showed no excess in mortality (Hattchouel et al., 1995).

In 1993, Viel et al. published the results of a study of the incidence of leukemia among persons up to 24 years of age living within 35 km of the La Hague nuclear reprocessing plant in the region of Nord-Cotentin in France and diagnosed between 1978 and 1990 (Viel et al., 1993). Twenty-three cases were diagnosed, giving an incidence rate of 2.99 per 100,000, which is close to the expected rate. Two years later, the same group continued their initial survey by including data through 1992 (Viel et al., 1995). Although the study did not show excess of leukemia for the zone as a whole, a non-statistically significant increased risk was observed if analysis was restricted to an administrative unit in the 10-km zone around the plant (four cases observed over 15 years while 1.4 were expected). These studies together with a third study on cancer incidence that covered the period 1978-1996 (Guizard et al., 1997) led to the conclusion that the potential elevated risk associated with living near the La Hague site should be kept under review. A follow-up ecologic study of incidence using zones defined according to their distance from the La Hague site (0-10, 10-20, and 20-35 km) was conducted to describe the occurrence of leukemia for each age group and cytological type for the period 1978-1998. The highest SIR was observed in the 5-9-year-old group (SIR = 6.38, 95% CI = 1.32-18.65) within the 10-km zone from the plant (Guizard et al., 2001).

Pobel and Viel (1997) reported the first case-control study in France. The study was undertaken within a 35-km radius of the nuclear waste reprocessing plant of La Hague. The aim was to investigate the association between childhood leukemia (<25 years of age) and established risk factors or other factors related to the plant. Twenty-seven cases of leukemia diagnosed during the period 1978-1993 and 192 matched controls were investigated, and information on antenatal and postnatal exposure to x-rays and viral infections, occupational exposure of parents, and lifestyle of parents and children was extracted through administered questionnaires and face-to-face interviews. A threefold increased risk of developing leukemia and frequent use of local beaches was found. Consumption of local fish and shellfish also showed an increased trend with risk. No association with occupational radiation was observed. The authors suggest an environmental route of exposure of children to radioactive material associated with certain lifestyle risk factors. These findings have been debated especially concerning

control selection, possible recall bias, multiple comparisons, and biological plausibility of the causal associations inferred (Clavel and Hemon, 1997; Law and Roman, 1997; Wakeford, 1997).

To respond to public concerns, the French government commissioned complementary epidemiologic investigations and also requested an analysis to be carried out by the Nord-Cotentin radioecology group to estimate the local population's exposure to radiation. No risk associated with radiation-induced leukemia was found (Rommens et al., 2000). More recent multisite studies in France like the one by (White-Koning et al., 2004) examined childhood leukemia (<15 years of age) incidence rates within 20 km of the 29 nuclear sites in the period 1990-1998. Comparison of the observed rates in areas surrounding the sites to expected rates based on national registry data did not provide any evidence of an excess leukemia in those areas. Results from intermediate analyses performed at the time of the White-Koning study that focused on leukemia incidence among children less than 5 years of age—to resemble the KiKK study in Germany (Kaatsch et al., 2008)—did not show an association (Laurier et al., 2008). However, the number of cases within the 5-km zone was small (5 observed cases compared to 5.2 expected from national rates).

The above-mentioned studies, as the majority of studies of incidence of leukemia around nuclear facilities, use distance to the site as a surrogate for radiation dose exposure, assuming an isotropic distribution of discharges. Evrard et al. (2006) investigated for the first time the incidence of childhood leukemia (<15 years of age) around 23 French nuclear installations (18 nuclear power plants, 2 nuclear fuel-cycle plants, 1 nuclear fuel reprocessing plant, 2 research centers) using a geographic zoning based on estimated doses to the bone marrow due to gaseous radioactive discharges. Direct radiation and liquid discharges were not considered. Compared to the study period of the previous report (White-Koning et al., 2004), this one included 3 additional years of observation (study period was 1990-2001). Risk was estimated for each of the five zones defined on the basis of estimated exposure levels, and trends of increasing risk with increasing exposure were recorded. Analysis showed no evidence of general increase of risk or trend in the incidence of childhood leukemia according to the zoning method developed in the study. More specifically, for the nuclear power plants, 242 cases were observed over the study period against 253 expected (SIR = 0.96), with no observed trend with dose. When the other nuclear facilities were included, the SIR was 0.94. Further analyses for the individual diagnosis age groups, 0-4-, 5-9-, and 10-14-year-olds, also did not show any significant trends by estimated exposure categories. Specifically, for the ages 0-4 years, the SIRs for the two highest exposure categories were 0.92 (based on 19 cases) and 0.93 (based on 5 cases), compared to the total SIR of 0.95 (based on 395 cases) for that age group. This study is notable in

that it was the first multisite study to conduct analyses based on estimates of exposure levels, although those estimates did not consider liquid discharges.

An updated study with an additional 5 years of observations (2002-2007) that used both a case-control and an ecologic approach showed that for the recent years, children living within 5 km of nuclear power plants (14 cases) are twice as likely to develop leukemia compared to those living 20 km or farther away from the plants. However, analysis of the same population of children using a dose-based geographic zoning approach, instead of distance, did not support the findings. The authors discuss that the absence of any association with the dose-based geographic zoning approach may indicate that the observed association of distance and cancer risk may be due to factors other than the releases from the nuclear power plants (Sermage-Faure et al., 2012). Among the potential factors are population mixing (Kinlen, 2011a) (a hypothesis that could not be evaluated in this study) and exposures to agents including natural or manmade exposures to radiation not modeled in the study. At least two additional aspects of this study are worth emphasizing: (a) While the KiKK study showed a doubling of risk in childhood leukemia only in children less than 5 years of age that live close to a nuclear power plant in Germany, the observed increase in leukemia incidence in this study was not restricted to the very young children but to all children ages 0-14. (b) The risk estimations from the case-control and ecologic approaches were in high concordance (OR = 1.9, 95% CI = 1.0-3.3 and SIR = 1.9, 95% CI = 1.0-3.2, respectively).

A.4.1.4 United States

In 1990, a national study by the National Cancer Institute (NCI), and the broadest of its kind ever conducted, investigated the potential excess of cancer deaths in 107 counties containing or closely adjacent to 62 nuclear facilities (Jablon et al., 1990, 1991). Three comparison counties were selected for each study county matched to study counties by the percent of persons in the population over 25, race, household income, and population size among other characteristics. The facilities included in the study were 52 nuclear power plants, nine Department of Energy (DOE) research and weapon plants and one commercial fuel reprocessing plant; all had begun operation before 1982 (Jablon et al., 1991). The survey examined 16 types of cancer that included those of the stomach, colorectal, primary liver, lung, female breast, and especially focused on leukemia. SMRs were calculated within "exposed" counties before and after the plant started operation and between "exposed" and "unexposed" counties both before and after plant startup. Over 900,000 cancer deaths occurred from 1950 through 1984 in the counties examined. The study results were essentially negative. No general increase in cancer mortality was found in counties with or near nuclear power plants and, unlike some reports in Britain (Black, 1984; COMARE,

1988, 1989; Heasman et al., 1986), no excess incidence of leukemia was found in children who lived near reprocessing and weapons plants. At the time the study was designed, county was the smallest geographic unit for which nationwide data on mortality could be quickly evaluated. However, it is well recognized that this was a limitation of the study because a county may be too large to detect risks present only in limited areas, which results in a dilution of any dose-associated effect. The limited incidence data available from two states (Iowa and Connecticut) provided inconclusive results.

Boice and colleagues (2005, 2006, 2007a,b) extended by 16-17 years the 1990 NCI study results at St. Lucie nuclear power plant in Florida, the Department of Energy's Hanford nuclear facility in Washington, and the uranium mining and milling facilities in Montrose County, Colorado. The team investigated cancer mortality rates among residents of counties near the facilities and found no evidence for increased risk compared to control counties that could be attributed to radiation exposures. Cancer mortality and incidence were also investigated in counties near the Apollo-Parks former nuclear materials processing facilities in Pennsylvania (Boice et al., 2003a,b). Although there was no observed increase in risk as measured by either mortality or incidence rates, the authors emphasize that mailing addresses in small rural areas may not always reflect actual residences, and validation by contacting area postmasters and using Census Bureau geocoding information may be necessary to prevent misleading conclusions. An update of the study showed consistent findings of lack of evidence for increased incidence near the former Apollo-Parks nuclear facilities (Boice et al., 2009).

Cancer risks were also investigated among residents in relation with the uranium milling and mining operations in Grants, located in Cibola County, New Mexico. Cancer mortality data were analyzed for the period 1950-2004 and cancer incidence data for the period 1982-2004 (Boice et al., 2010). Lung cancer mortality and incidence were significantly increased among men (SMR = 1.11, 95% CI = 1.02-1.21; SIR = 1.40, 95% CI = 1.18-1.64) but not women. Analysis among the population of the three census tracts near the Grants Uranium Mill revealed a higher risk for lung cancer among men (SMR = 1.57; 95% CI = 1.21-1.99). The authors discuss that etiologic inferences are not possible because of the ecologic study design. However, the excess in lung cancer among men is likely to be due to previously reported risks among underground miners from exposure to radon and its decay products, coupled with heavy smoking and possibly other factors.

Mortality among residents of Uravan, a company town built around the uranium mill in Montrose County, Colorado, was investigated in more detail using a retrospective cohort study design (Boice et al., 2007b). The study population was originally identified from worker and community

records (Austin, 1986). Workers at the Uravan mill and nearby uranium mines, their spouses and children, and other workers in the town such as teachers and postal clerks were included in this study. Approximately 1,900 men and women who lived in Uravan for at least 6 months within the period 1936-1984 and were alive after 1978 were included in the study. Results showed that among the approximately 450 residents who had worked in underground uranium mines, a significant twofold increase in lung cancer was found. No significant elevation in lung cancer was seen among the female residents of Uravan or the uranium mill workers. The excess of cancer among uranium miners was attributed to the historically high levels of radon in uranium mines of the Colorado Plateau, and heavy smoking among the workers (Boice et al., 2007).

Previous smaller studies of mortality or incidence in the United States, such as that around the San Onofre power plant in California (Enstrom et al., 1983), the Rocky Flats nuclear weapon production facility in Colorado (Crump et al., 1987), and Hanford and Oak Ridge in Washington State and Tennessee, respectively (Goldsmith, 1989), showed no evidence of increased risk. Mangano (1994) concluded that between 1950-1952 and 1987-1989, cancer risk from all types of cancer and all age groups increased significantly around the Oak Ridge site; however, a radius of 160 km was analyzed as a whole. An excess of incident leukemia across all age groups reported by Clapp and colleagues (1987) for the period 1982-1984 in Massachusetts seemed to be counterbalanced by a lower-than-expected incidence of cases the 2 following years (Poole et al., 1988; Wilson, 1991).

State health departments have also specifically addressed concerns of their communities on increased cancer rates around nuclear facilities. Such an example is the recent publication from the Illinois Department of Public Health, which analyzed childhood cancer rates in the vicinity of the plants in the state (Ma et al., 2011).

One of the largest and most comprehensive studies conducted in the United States regarding the risk of cancer near a nuclear facility, in this case thyroid disease, is the Hanford Thyroid Disease study. The Hanford Nuclear Site in southeastern Washington State was established in 1943 to produce plutonium for atomic weapons. In the mid 1980s it was revealed that during the 1940s and 1950s of plutonium production at Hanford, large amounts of gaseous and vaporized radionuclides were released into the atmosphere including about 740,000 Ci of ^{131}I resulting in estimated mean dose to the thyroid of 174 mGy. In response, the U.S. congress mandated the Hanford Thyroid Disease study in 1988 to investigate the widespread concerns among people living near the site that such releases may have increased their risk of developing thyroid disease. The primary analyses focused on living participants who received medical examinations to detect thyroid disease, and for whom thyroid radiation doses were estimated using

the dosimetry system developed by the investigators; dose reconstructions were based on environmental measurements and personal interviews (Davis et al., 2004). The investigators concluded that there was no evidence of a relationship between Hanford radiation dose and thyroid cancer incidence or other thyroid diseases. In an attempt to reconcile the study results with the evidence for thyroid disease that has been reported for the Chernobyl accident (see Section A.4.3), which also includes exposures primarily to ^{131}I, the investigators suggest that differences in the dose and dose rates delivered may account for the differences in observed risks. Other investigators recommend that the results are interpreted as inconclusive (rather than negative) because of possible inadequate power to detect an effect due to uncertainties associated with the models and assumptions used for individual dose reconstruction (Hoffman et al., 2007).

Potential health effects associated with the 1979 accidental releases of the Three Mile Island nuclear plant in Pennsylvania have been examined and have been a subject of controversy. Immediately after the accident, a presidential commission expressed confidence that the maximum external radiation dose to a person in the general population was less than the average background (~ 1 mSv) and that no health effects would be detectable and that the sole health consequence for the population in close proximity to the installation was mental distress (Kemenu et al., 1979). Karl Morgan, one of the founders of the field of radiation health physics, estimated that there would be 50 excess cancer cases in the area surrounding the plant, a presumptive risk characterized as "exaggerated" based on current knowledge of radiation effects at the doses surrounding populations would be exposed (Upton, 1980).

The initial cancer risk survey was conducted by Columbia University for the period 1975-1985 and was supported by the Three Mile Island Health Fund, which was created and governed by a court order (Hatch et al., 1990, 1991). Estimates of the emissions delivered to small geographic study zones were derived from mathematical dispersion models. Although the data provided hints of increased risk of leukemia and lung cancer in the surrounding areas, they were interpreted as not convincing based on the assumption that the doses were too low to produce a measurable effect (Hatch et al., 1990). Given the "mental distress" health consequence that the government reported for populations that lived near the facility when the accident happened, a study was conducted to test whether mental distress could be linked with the somewhat elevated cancer incidence in the area (Hatch et al., 1991). Stress following local community disasters has been linked with increased cancer in early studies (Bennet, 1970; Janerich et al., 1981); however, studies on the relationship between psychological stress and cancer have revealed conflicting results (Garssen, 2004), although it is known that stress can affect the immune system (Segerstrom and Miller, 2004). In the absence of individual and direct measures of stress, residential

proximity to the site was used as a surrogate (Hatch et al., 1991). Using this crude test of an accident-stress hypothesis, a 40 percent increased risk between postaccident cancer rates and proximity was estimated. The authors state that radiation emissions as modeled mathematically did not account for the observed increase (Hatch et al., 1991).

The topic of health effects related to the Three Mile Island accident reappeared in 1997 when attorneys representing more than 2,000 area residents asked epidemiologist Stephen Wing from the University of North Carolina to examine the original work. The examination, with severe criticism on the study approach followed by Hatch and colleagues, reanalyzed and reinterpreted exactly the same data. The claim was that the original study may have been biased, as analysis was driven by the belief that no association could exist at low exposures. The new analysis showed that incidence of leukemia and lung cancer following the accident increased more in areas estimated to have been in the pathway of radioactive plumes compared to areas outside the pathway (Wing et al., 1997a). An exchange of published responses between the Columbia team and Wing followed (Hatch et al., 1997; Susser, 1997; Wing et al., 1997b). To this day, Wing's article remains the only one to present original health data supporting an association between releases from the Three Mile Island accident and cancer.

A.4.1.5 Canada

A case-control study by McLaughlin and colleagues (1993b) of workers at nuclear facilities in Ontario, Canada, can possibly be directly compared with that of Gardner et al. (1990) because it tested the hypothesis of an association between childhood leukemia and the occupational exposure of fathers to ionizing radiation before a child's conception. In this study, cases (n = 112) were children (<15 years of age) who died or were diagnosed with leukemia in the period 1950-1988 and were born to mothers living near one of the five operating facilities under investigation (one research development facility, a uranium refinery, a uranium mining and milling facility, and two nuclear power plants). No association with paternal occupational exposure was found in the analysis (McLaughlin et al., 1993b). Also, an ecologic study examined the mortality and incidence of childhood leukemia for the period 1950-1987 among children less than 15 years of age living in the vicinity of the Ontario nuclear facilities (McLaughlin et al., 1993a). Overall, the observed number of leukemia deaths (O = 54) was slightly greater than expected (E = 46.1) during the period when the facilities operated, but the difference was not statistically significant (O/E = 1.17, 95% CI = 0.88-1.53).

A.4.1.6 Spain

Lopez-Abente and colleagues (1999) studied the mortality due to hematological tumors in towns lying within 30 km of seven nuclear power plants and five nuclear fuel facilities during the period 1975-1993. No study area yielded evidence of a raised risk of leukemia mortality among persons under the age of 25. A recent updated ecologic study that included all nuclear power plants and other nuclear fuel facilities in the country, regardless of whether they are in operation, studied mortality due to different types of cancer including leukemia in municipal areas within a radius of 30 km around the facilities and in control counties (50-100 km). The study period was 1975-2003. The main original contribution of the study was the reconstruction of the exposure of the population in each municipality accounting for both liquid and gaseous discharges from the facilities, described as means of effective dose (Nuclear Safety Council and the Carlos III Institute of Health, 2009). The spatial distribution of the data by the different dose categories differs from the radius pattern produced by distances used in most previous studies, since specific characteristics of each site, including land and water use, have been incorporated in the models. The dose estimates are conservative, constituting the upper limit for the exposures actually received by the populations.

Risk estimates were adjusted for natural radiation and other covariates. The investigators interpret their findings as there being overall no association of living near the nuclear facilities and cancer mortality. Increases in risk such as those observed for lung and bone cancer around specific nuclear fuel-cycle facilities were interpreted as inconsistent, as they were not replicated across the facilities examined and cannot be attributed to the effect of the doses generated as a result of their operation, primarily because the releases are too low to have an impact.

A.4.1.7 Sweden

The existence of leukemia clusters among those less than 15 years of age living near four nuclear facilities was examined for the period 1980-1990. No consistent evidence was found for childhood leukemia clusters associated with living in the proximity of nuclear power plants (Waller et al., 1995).

A.4.1.8 Finland

A recent multiapproach investigation in Finland (ecologic, case-control, and cohort studies) suggests no association of leukemia and vicinity to the two nuclear power plants (Heinavaara et al., 2010). However, the 5-km zone around the nuclear plants was not investigated.

A.4.1.9 Switzerland

The results of the Childhood Cancer and Nuclear Power Plants in Switzerland (CANUPIS) study were recently published (Spycher et al., 2011). CANUPIS was a large census-based cohort study that analyzed distance of residence at birth as well as distance of residence at diagnosis to determine if children who grew up near the country's five nuclear power plants had an increased risk of developing childhood cancer. Children aged 0-15 years born in Switzerland from 1985 to 2009 based on the 1990 and 2000 Swiss censuses and identified cancer cases from the Swiss Childhood Cancer Registry were included in the study. Completeness of registration was greater than 90 percent. In the study period, 2,925 children were diagnosed with cancer, 953 of whom had leukemia. The number of diagnosed children that lived within the 5-km zone was small: 18 and 31 children at ages 0-4 and 0-15 years, respectively, were diagnosed with cancer overall, while 8 and 12 children in the above-mentioned age groups were diagnosed with leukemia. Compared with children born at a distance greater than 15 km from the plant, the RRs (95% CIs) for leukemia in the 0-4 and 0-15 age groups were 1.20 (0.60-2.41) and 1.05 (0.60-1.86), respectively.

Results presented little evidence for an association between residence at birth or diagnosis near nuclear power plants and risk of leukemia or other childhood cancers. Potential confounders that were considered included background ionizing radiation, electromagnetic radiation from power lines and other sources, carcinogens related to traffic, pesticide exposure, socioeconomic status, and proxies of population mixing and exposure to childhood infection (average number of children per household in the community and degree of urbanization) (Law, 2008). Although no data on radiation releases from the nuclear plants were available, additional analysis was performed where main dispersal directions of airborne emissions were accounted in the model. Results were consistent with the main results. Among the limitations of this study were the small sample size, particularly of 0-4-year-olds living close to the nuclear power plants, and lack of coverage of the earlier time periods when higher dose exposures may have occurred.

A.4.1.10 Israel

In Israel a study of the population near the Dimona nuclear plant (Sofer et al., 1991) examined new leukemia cases among those under 25 years of age who lived within 45 km of the station. The authors concluded that there was no excess incidence near the power plant.

A.4.1.11 Japan

A study by Yoshimoto et al. (2004) that covered the period 1972-1997 in 20 municipalities in Japan, containing 16 nuclear power plants showed no evidence of increased risk compared to control municipalities among the young residents. However, rates of mortality due to leukemia for the population overall were higher among those populations living in proximity to nuclear power plants in Japan.

A.4.2 Atomic Bombing Survivor Studies

The atomic bombs that exploded over the city of Hiroshima and three days later over Nagasaki, Japan, in August 1945 exposed the people of each city to whole-body doses of penetrating ionizing radiation. The number of deaths before the end of 1945 were estimated to be between 90,000 and 120,000 in Hiroshima (population at the time was 330,000) and between 60,000 and 80,000 in Nagasaki (with a population of about 250,000) and were attributed to traumatic blast injuries, burns, bone marrow depletion, and other physical consequences associated with the exposure. The information available on atomic bombing survivors and their children is highly relevant to the radiation protection policy of the general public (National Research Council, 2005; NCRP, 2009; UNSCEAR, 2006a,b).

The Radiation Effects Research Foundation and its predecessor, the Atomic Bomb Casualty Commission, track the mortality and cancer incidence—among other health effects—of the survivors of the bombings. The LSS cohort consists primarily of about 94,000 survivors of the atomic bombings of Hiroshima and Nagasaki. The cohort includes both a large proportion of survivors who were within 2.5 km of the hypocenters at the time of the bombings and a similar sized sample of survivors who were between 3 and 10 km from the hypocenters and whose radiation doses were almost negligible. Periodic analyses of the LSS mortality data have resulted in a series of reports; the fourteenth report (Ozasa et al., 2012), which covers the period 1950-2003 and includes an additional 6 years of follow-up since the last report of the series (Preston et al., 2003), was recently published.

Although the follow-up of the atomic bombing survivors is often perceived as a high-dose study (exposures 0.5-3 Sv range), about 86 percent of the survivors with estimated doses (i.e., 74,000 persons presenting 11,000 cancer cases) had colon doses under 0.2 Sv (Preston et al., 2007). Demographically, the population is large, and individuals were unselected with respect to sociodemographic or health-related status at the time of the bombings, but in order to be included they must have survived for at least 5 years after the bombings. All ages and both genders of individuals exposed to a wide range of radiation exposure levels are included, permitting a

dose-response analysis. Importantly, estimates of these individual doses are reasonably precise. Additionally, the population has a high rate of mortality and cancer-incidence follow-up. These strengths of the LSS study provide a high-quality, informative epidemiologic study. However, the radiation exposures were acute, received in a matter of seconds, and the population was exposed to a small amount of neutrons and not just gamma rays. Moreover, the fact that the population had to live in a war-torn country where there was malnutrition, poor sanitary conditions, and other severe difficulties makes generalizability of the findings to other populations an issue (Ozasa et al., 2012).

Subcohorts of LSS include the in utero cohort where persons born to mothers pregnant at the time of the bombing and controls are being followed, and the F1 cohort, where children of the exposed and unexposed parents are being followed for disease occurrence. While radiation doses were not directly measured at the time of the bombings, information on survivor locations and shielding were obtained in the early years, which combined with extensive physics calculations of the radiation source and transport have been used to retrospectively estimate the doses received by individual survivors (Cullings et al., 2006).

By the late 1940s, there were suggestions of an increased risk of leukemia among the atomic bombing survivors; the earliest evidence of an increased leukemia was reported in 1952 (Folley et al., 1952). The latest published LSS mortality data for leukemia are through 2000 and a 46 percent excess (93 excess deaths) are attributable to radiation exposure among the survivors to >0.005 Gy (Preston et al., 2004; Richardson et al., 2009). A clear dose-response relationship exists, with 90 percent of the leukemia deaths among those exposed to doses >1 Gy being excess deaths. Separate analyses also indicated strong dose responses for most subtypes of leukemia except chronic lymphocytic leukemia (Preston et al., 1994).

Because the atomic bombing survivors received whole-body exposure from penetrating radiation, a large number of organ sites were affected. An analysis by Preston et al. (2007) on solid cancer incidence in atomic bombing survivors for the period 1958-1998 showed that an excess of 11 percent of solid cancers are attributed to exposures >0.005 Gy (mean 0.23 Gy). The attributable proportion increases with increasing dose and reaches 48 percent among those who received at least 1 Gy. In ranking the sites based on excess cancers observed because of the exposure, the highest relative excess was found for bladder, female breast, and lung cancers, followed by cancers of the central nervous system, ovary, thyroid, colon, and esophagus (Preston et al., 2007). Overall, estimates for solid cancers were 50 percent higher among women, but if female cancers are excluded from the analysis, the estimates by gender are more comparable. Examination of the excess absolute risks (EARs) shows that the number of excess radiation-

related cancers occurring among males per 10,000 persons per year per Gy is about the same as among females. Excess risks are highly dependent on age at exposure and attained age. The excess relative risk (ERR) for persons exposed to the bombs at a younger age is higher than those exposed to the bombs when they were older, but it declines over time with increasing attained age (or time since exposure). However, the number of excess cancers occurring among 10,000 persons per year per Gy increases with attained age and indicates that radiation risk persists throughout the remaining lifetime. Both the in utero and early childhood groups exhibited statistically significant dose-related increases in incidence rates of solid cancer. At present, not only is there no evidence to support the hypothesis that in utero exposure confers greater adult-cancer risk than childhood exposure, but the risk might be lower (Preston et al., 2008).

Of particular pertinence to this document are the considerations related to risks among the low-dose part of the study population. In the most recent update of cancer incidence there was a statistically significant dose response within the range 0-150 mSv (Preston et al., 2007), suggesting there is dose-related risk even at relatively low dose levels. For cancer mortality, statistically significant upward curvature has been seen, but this is associated primarily with a sublinear degree of risk in the dose range of about 300-800 mSv and not sublinearity at low doses. However, other uncertainties need to be kept in mind in evaluating the low-dose data. First is the fact that some were exposed to residual radiation from neutron activation of soil elements which may have affected those who entered the high-exposure areas in the first few days after the bombings (e.g., in search of missing relatives). Certain areas also received "black rain," fallout which sometimes may have contained a degree of radioactive elements. There is very little information about who among the atomic bombing survivors may have received such exposures. In addition, the risk estimates may be affected by sociodemographic factors such as rural and urban differences and by selection effects having to do with the hardiness of the survivors of acute radiation effects. (However, the selection effects would more likely apply to high- and moderate-dose survivors than to low-dose survivors.) Because of these uncertainties, plus the other issues of generalizing to protracted exposures and to Western populations, corroborating evidence is needed from other studies to increase certainty in projecting low-dose risks.

A.4.3 Studies of Accidental Releases to Populations

A.4.3.1 Chernobyl

The Chernobyl nuclear power station accident in 1986 in northern Ukraine resulted in the largest accidental release of radionuclides (princi-

pally ^{131}I and ^{137}Cs) into the environment in history. Although there was a wide geographic dispersion of radionuclides, the accident had the greatest impact in Belarus, Ukraine, and the Russian Federation. A number of epidemiologic studies have investigated the impact of the Chernobyl accident and cancer risk, and most of the studies have been ecologic, where information on dose and health outcomes is available only at the population level. The radiation effects from the Chernobyl accident are comprehensively summarized in a recent report (UNSCEAR, 2008b). The most notable health consequence of the accident has been the large increase in thyroid cancer among those exposed as children or teenagers. The latency period for thyroid cancer was estimated to be 4-5 years after exposure (Ivanov et al., 2006; Kazakov et al., 1992).The increase in incidence of thyroid cancer was first observed in the early 1990s in Belarus. It is estimated that the thyroids of several thousand children received ^{131}I doses of at least 2 Gy. By 1995, the incidence of childhood thyroid cancer had increased to 4 per 100,000 per year compared to less than 0.05 cases per 100,000 per year prior to the accident (Stsjazhko et al., 1995). For the three most affected countries combined, the increase in incidence rate translated to 5,000 excess thyroid cancer cases in the first 16 years following the accident (Cardis et al., 2005a). A recent study—an update of an earlier report (Tronko et al., 2006)—evaluated the dose-response relationship for incident thyroid cancers using measurement-based individual ^{131}I thyroid dose estimates taken within 2 months after the accident. The 12,000 individuals who were part of the prospective cohort study were <18 years of age at the time of the accident and resided in three contaminated regions of Ukraine. Results suggested that thyroid cancers attributable to ^{131}I exposure continued to occur two decades after the exposure; the estimated ERR for incident thyroid cancer per gray was 1.91 (95% CI = 0.43-6.34) (Brenner et al., 2011). There is some indication that iodine deficiency at the time of exposure to ^{131}I may have increased the risk of developing thyroid cancer; conversely, prolonged iodine dietary supplementation may be protective for the disease (Cardis et al., 2005a).

Data on solid cancers other than thyroid among residents of the affected areas are limited. Among residents of the contaminated region of Kaluga in Russia, no indication of increased incidence or mortality of solid cancers was observed (Ivanov et al., 1997a). Exposure to ionizing radiation is a known risk factor for breast cancer. Pukkala et al. (2006) conducted an ecologic study to describe the trends in breast cancer incidence in Belarus and Ukraine. Despite the evident trends of increased breast cancer incidence due to improvements of diagnosis and registration, the authors showed that during the period 1997-2001, there was a twofold increase in risk in the highly contaminated (average accumulative dose 40 mSv or more) compared to the least contaminated areas.

Whether there is leukemia excess following the accident is much less clear, although several ecologic studies have examined the association between leukemia risk and exposure to radiation from Chernobyl in childhood. For example, the International Program on the Health Effects of the Chernobyl Accident pilot projects study aimed to examine leukemia and lymphoma incidence among populations residing in selected radioactively contaminated areas of the Ukraine, Russia, and Belarus during 1980-1992. Incidence was estimated before and after the Chernobyl accident and a statistically significant increase was observed following the accident (WHO, 1996). However, application of better screening systems and diagnostic procedures could account for the reported increase in incidence. The European Childhood Leukemia-Lymphoma Incidence Study examined trends in leukemia based on cancer registration data from 23 countries among children aged 0-14 years (Parkin et al., 1996). No significant associations with exposure to radiation from Chernobyl were identified. Other studies have not provided consistent evidence for an association (Ivanov et al., 1993, 1996; Noshchenko et al., 2001; Prisyazhiuk et al., 1991) but are limited by dependence on historical and current registration data of varying quality and lack of reliable dosimetry.

A case-control study was conducted to estimate the radiation-induced acute leukemia risk among those aged 0-20 at the time of the Chernobyl accident in Ukraine. Individual estimations of accumulated absorbed radiation dose to the bone marrow were assessed. The period of investigation was 1987-1997. Ninety-eight verified cases were compared to 151 randomly selected controls, matched for age, gender, and administrative region. The mean value of the estimated accumulated equivalent dose to the bone marrow was 4.5 mSv and the maximum was 101 mSv. Analysis showed that males whose estimated radiation exposure was higher than 10 mSv had a threefold higher risk of developing leukemia compared to those exposed to 1.9 mSv or less (Noshchenko et al., 2002). Many of the youngest subjects of the above-mentioned study were also participants of a larger multinational population-based case-control study of acute leukemia diagnosed among children who were in utero or less than 6 years of age at the time of the accident. Confirmed cases of leukemia diagnosed between 1986 and 2000 in Belarus, Russia, and Ukraine were included and compared to the same age, gender, and residence controls. The major findings of the study were that the median radiation doses received by the participants were low (<10 mGy), and there was an overall significant increase of leukemia risk with increasing dose, an association that was most evident in Ukraine, apparent in Belarus, and not evident in Russia (Parkin et al., 1996).

A.4.3.2 The Techa River Study

The Techa River cohort of an unselected population of men and women of all ages provides a unique opportunity to evaluate long-term human health risks from low-dose radiation exposures. Between 1949 and 1956, radioactive materials were released into the Techa River as a result of technological processes at the Mayak complex that produced plutonium for the Soviet nuclear weapons program. At the time of the Mayak releases, there were about 30,000 people living in 41 rural villages downstream on the river. This population received both external exposure primarily due to gamma exposure due to proximity to sediments and shoreline, and internal low-dose-rate radiation exposures, the more significant included drinking of water from the river (Degteva et al., 2000; Krestinina et al., 2005, 2007). Enhanced dose reconstruction efforts for individuals of the Techa River cohort were performed. Dosimetry information derived from annual village mean dose estimates that allowed for dose rate in air at the river bank and in residence areas, representative behavior patterns, intake of radionuclides with river water and food, and other factors (Degteva et al., 2000). Results provided clear evidence for radiation-associated increases in cancer mortality risks of the cohort. More specifically, the excess relative risk per gray for deaths from leukemia was 4.2 (95% CI = 1.2, 13). It was estimated that 2.5 percent of the solid cancer deaths and 63 percent of the leukemia deaths were associated with the radiation exposure (Krestinina et al., 2005). Studies on incidence of solid cancers (Krestinina et al., 2007) and leukemia (Ostroumova et al., 2006) in the cohort confirmed the association. More specifically, analysis of 83 leukemia cases diagnosed within the period 1950-1997 and 415 matched controls showed that the ORs per gray of total, external, and internal doses were 4.6, 7.2, and 5.4, respectively.

A.4.4 Studies of Nuclear Workers

Extrapolating results from databases such as that of the LSS to residential settings is problematic due to major differences in magnitude of dose and exposure periods (high-dose acute exposures versus low-dose protracted or fractionated exposures), study group demographics, and health of exposed populations. Studies of cancer risk assessment among workers in the nuclear industry could provide more relevant estimates of the effects of protracted, low-level ionizing radiation exposure. The great advantage of this approach is the availability of well-standardized and generally computerized individual whole-body dosimetry records that provide reliable information as the basis for epidemiologic estimates of radiation-induced cancer risk. The major limitation, however, is the "healthy worker effect," a concern in occupational epidemiology when health risk factors associated with workers (such as intended selection of more healthy persons for

employment, work-related medical care, higher socioeconomic status) are compared to those of the general population from which the workers are drawn.

The "healthy worker effect" reflects that an individual must be relatively healthy to be employable in a workforce; therefore, both disease and mortality rates are usually lower among workers than in the general population. Moreover, within the workforce studies, healthier workers are more likely to stay employed for longer periods of time than less healthy workers. This may give rise to a healthier occupational cohort (Li and Sung, 1999). There are several comprehensive reviews of the biases related to the comparison of workers and general population that includes selection bias, information bias, and confounding (Li and Sung, 1999; Pearce et al., 2007). An example of the latter is that some health-related behaviors such as smoking are not permitted during the hours of work, and certain personal traits such as obesity may be thought unfit for particular labor forces by industry (Wilcosky and Wing, 1987). Therefore, in view of the deficiency of background risk factors, the possibility of differential effects of ionizing radiation cannot be excluded. Although direct comparisons between the workforce and the general population in relation to the effects of ionizing radiation may be somewhat deceptive, examining the variation of the health outcome across a gradient of increasing exposure within the nuclear industry is very informative. It is worth noting that the healthy worker effect has often been found to be smaller for cancer than for other disease categories.

Workers in the nuclear plants are at potential risk of exposure to ionizing radiation both externally from radioactivity in the working environment and internally from radionuclides which enter the body by inhalation, ingestion, or through accidents that result in percutaneous wounds. The exposures may accumulate over a lifetime to doses of the order of 100 mGy. The possible carcinogenic effects of exposure to external sources of radiation among nuclear workers have been the subject of numerous investigations over the past 20 years. Estimates from these analyses are of limited precision because the sample sizes are small and the follow-up time not sufficiently long (Shore, 1990, 2009). Among white male employees of the Oak Ridge National Laboratory, leukemia mortality rates were 60 percent higher than national rates; however, there was no evidence of a dose-response relationship (Wing et al., 1991). Mortality data among 5,413 workers at the Rocky Flats plutonium weapons facility, although with limited precision, suggested an elevated risk for esophageal, stomach, colon, and prostate cancers among individuals with plutonium body burdens of 2 nCi or greater. No excess risk was reported for cancers of the bone, liver, and lung, the cancer sites most likely to be associated with plutonium exposure (Wilkinson et al., 1987). Combined analyses of mortality workers at

the Hanford Site, Oak Ridge National Laboratory, and Rocky Flats nuclear weapons plants provided no evidence of an association between radiation exposure and mortality from all cancers or from leukemia (Gilbert et al., 1989). The exception was multiple myeloma, which was found to exhibit a statistically significant correlation with radiation exposure. However, the observed association could be due to chance alone.

More recently, Schubauer-Berigan et al. (2007) combined the data from five nuclear facilities in the United States to evaluate leukemia mortality risk from ionizing radiation using a nested case-control study design. The authors reported an adjusted ERR per 10 mSv of 1.44 percent (95% CI = <–1.03% - 7.59%). In both reports, the results suggest that risks among nuclear workers are comparable to those observed in populations exposed acutely to high doses. An analysis of observed versus expected mortality of more than 29,000 nuclear workers in France, employed between 1950 and 1994 at two nuclear installations, showed a strong healthy worker effect with an observed 40 percent lower mortality rate among workers than expected from national mortality statistics (Telle-Lamberton et al., 2007). Of the 21 cancer sites examined, a statistically significant excess was observed only for skin melanoma. A significant dose-effect relationship was observed for leukemia after exclusion of chronic lymphoid leukemia (CLL). A larger study of 75,000 employees of the United Kingdom atomic energy authority, the atomic weapons establishment, and the Sellafield plant of British nuclear fuels demonstrated an approximately 20 percent lower all-cause mortality and 4 percent lower cancer associated mortality among workers compared to national rates. A positive association was observed for leukemia risk and exposure to radiation and weaker associations for melanoma and other skin cancers (Carpenter et al., 1994).

A.4.4.1 The Three-Country Study and the 15-Country Study of Nuclear Workers

The three-country study was coordinated by the International Agency for Research on Cancer (IARC) of the World Health Organization (WHO). In the analysis, Cardis and colleagues (1995) found a statistically significant correlation between mortality from leukemia (excluding CLL) and the cumulative individual dose of external radiation. The ERR coefficient was 2.18 (90% CI = 0.13, 5.7) per sievert. Cardis et al. (2005b) extended the IARC study to include countries with nuclear programs such as France and Japan to produce what is probably the largest study to date of cancer in the nuclear workforce. The investigation assessed mortality among workers in 155 nuclear facilities in 15 countries and was conducted to improve the precision of direct estimates of cancer risk following protracted low

doses of ionizing radiation and to advance the scientific basis for radiation protection standards. Analysis included more than 400,000 nuclear workers monitored individually for external radiation and followed up for an average of 12.7 years. The number of workers included in the study is approximately four times greater than in the three-country study. However, as discussed in a recent review, the increase in statistical power is not as great as the number of workers in the cohort may imply, primarily because of the inclusion of workers with low average doses and short periods of follow-up (Wakeford, 2005). About 10 percent of the cohort of workers received external doses exceeding 50 mGy, while 0.1 percent received doses exceeding 500 mGy. Additional problems of the 15-country study include the fact that the results are driven by the contribution of only one country, Canada (Ashmore et al., 2010). The Canadian data are being reexamined for their quality and validity of results. Areas of uncertainty in the 15-country study related to dosimetry, analytical methods, smoking data, and others have been described (Boice, 2010).

Thirty-one cancer types were examined in the 15-country study. A significant association was seen between radiation dose and all-cause mortality (ERR = 0.42 per Sv, 90% CI = 0.07, 0.79); 18,993 deaths were attributed to mortality from all-cancer types (ERR/Sv = 0.97, 90% CI = 0.28, 1.77; 5,233 deaths). Lung cancer was the only cancer to show a statistically significant rise in the risk estimate; however, the association should be interpreted with caution as data on individual smoking characteristics were missing from the analysis. A borderline significant association was found for multiple myeloma. Stratified analysis by duration of employment had a large effect on the ERR/Sv, reflecting a strong healthy worker survivor effect in these cohorts.

A.4.4.2 The British National Registry of Radiation Workers

Perhaps the most precise estimates to date of mortality and cancer risks following occupational radiation exposure come from the third analysis of the British National Registry of Radiation Workers (Muirhead et al., 2009). Two earlier analyses that only looked at mortality data found a strong healthy worker effect and some evidence of an increasing trend in cancer risk (particularly leukemia) with increasing external dose; however, the confidence intervals for the observed trends were wide (Kendall et al., 1992; Muirhead et al., 1999). The third analysis of the series looked at an enlarged cohort of 175,000 workers, adding 9 years of follow-up (87,000 of these workers also were in the 15-country study described above). Due to the higher dose distribution and the larger number of cancers, this study had a greater statistical power than the 15-country study.

Within the cohort, there was evidence of an increasing trend in cancer

mortality with increasing external radiation dose. The trend with dose in the risk of all cancers other than leukemia was maintained when lung cancer was excluded from the analysis, supporting that the trend is not an artifact due to smoking. The cancer risk estimates obtained were consistent with values used to set radiation protection standards.

A.4.4.3 Emergency Chernobyl Workers

Cancer incidence (as opposed to mortality) data among nuclear workers is less available. An analysis has been published of solid cancer incidence rates during an 11-year follow-up (1991-2001) of emergency and cleanup workers after the Chernobyl accident in Russia. These persons worked in the 30-km zone in 1986-1987 and received on average higher doses than those involved in recovery operations in 1988-1990 and have been subject to annual medical checkups (Ivanov et al., 2004). Two control groups were selected for comparison: an "external control" representing age-adjusted incidence rates in Russia and an "internal control" representing emergency workers who were not exposed. The SIR and its 95% CI are similar to that obtained from the Russian population. The values of excess relative risk per unit dose (ERR/Gy) was estimated to be 0.33 (95% CI = –0.39, 1.22) for the follow-up period 1991-2001 and 0.19 (95% CI = –0.66, 1.27) for 1996-2001 compared to the internal control. The authors translate their findings as positive yet statistically insignificant excess of radiogenic solid cancers in the cohort of emergency workers (Ivanov et al., 2004).

Chernobyl recovery operation workers also have theoretically a high risk of developing cancer as a consequence of radioactivity from the accident. However, a number of investigations conducted among recovery workers have not found associations between leukemia incidence and exposure (Ivanov et al., 1997b, 2004). Risk factor analysis among 55 cases of leukemia among Chernobyl emergency workers reported between 1986 and 1995 showed that the risk of developing leukemia was not associated with radiation dose, effective exposure dose rate, or duration of stay in the zone (Konogorov et al., 2000).

A.4.4.4 The Mayak Workers Study

A cohort of about 25,000 Russian nuclear workers who worked at the Mayak plutonium production complex in the period 1948-1972 provides a great opportunity to evaluate cancer risks from exposure to plutonium. These workers were exposed to chronic low-dose-rate external gamma radiation as well as internal (inhaled) plutonium at levels much higher than workers in other countries. For example, for the nearly 11,000 monitored workers hired before 1959, the mean cumulative external dose was 1.2 Gy,

more than an order of magnitude higher than any of the nuclear cohorts described. Leukemia death rates increased significantly with increasing gamma-ray dose (Shilnikova et al., 2003). Excess cancers of the lung, liver, and bone, the organs that receive the largest doses of plutonium, have been described (Gilbert et al., 2000; Koshurnikova et al., 2000). Recent analysis with improved plutonium and external dose estimates verified the increase (Sokolnikov et al., 2008).

A.4.5 Studies of Medical Exposures to Radiation

Diagnostic and therapeutic radiation has been used in medicine for over a century. The continuing improvements in diagnostic imaging and radiotherapy as well as the aging of the population have led to greater use of medical radiation (Ron, 2003). Epidemiologic studies of persons exposed to radiation for medical reasons have provided unique opportunities in understanding the risks associated with fractionated radiation exposure. Additionally, medical records often contain information on a patient's personal past medical history as well as on demographic data and information on personal habits such as smoking, alcohol drinking, and medications. On the negative side, because of their possible underlying disease, patients may have different sensitivity to the radiogenic effects compared to a somewhat healthy population. Other concurrent treatments can affect radiation risks and it can prove difficult to untangle the impact of those different factors. Also, because patients come back for follow-up, other diseases are more likely to be detected and reported, leading to overrepresentation of diseases on this group compared to the general population (Ron, 2002).

A recent report from the NCRP entitled "Ionizing Radiation Exposure of the Population of the United States" indicated that in 2006, people in the United States were exposed to more than seven times as much ionizing radiation from medical diagnostic procedures than in 1980; the increase is fueled largely by the use of CT scans (NCRP, 2009). In 2006, over 67 million scans were performed, 4 to 7 million in children, and many patients receive multiple scans.

Diagnostic exposures are typically characterized by fairly low doses to individual patients (effective doses are typically in the range 0.1-10 mSv), sufficient to provide the required medical information. Because doses are typically low, their effects are difficult to study unless multiple examinations are performed. For example, an excess risk of breast cancer has been reported among women with tuberculosis who had multiple chest fluoroscopies (Delarue et al., 1975; Miller et al.,1989), women treated for benign breast disease (Mattsson et al., 1993), as well as among scoliosis patients who had frequent diagnostic x-rays during their late childhood and adoles-

cence (Doody et al., 2000). The potential risk attributed to mammography screening programs and understanding the balance between the number of breast cancer deaths induced and breast cancer deaths prevented continues to be an issue of debate especially when extended to women under the age of 50 (de Gelder et al., 2011; Hellquist et al., 2011). Exposure to diagnostic radiography in utero has been associated with increased risk of childhood cancer, particularly leukemia (Linet et al., 2009; Rajaraman et al., 2011; Wakeford, 2008).

In contrast to diagnostic radiation doses, therapeutic doses are much higher and precisely delivered to the targeted area such as the tumor (doses can be as high as 40 Gy or more) (Gilbert, 2009; UNSCEAR, 2008a) aiming to produce cell killing. Physicians need to consider the risks of the treatment against the potential benefits. Overall more than 100 studies of patients receiving diagnostic or therapeutic radiation have evaluated the potential risks and have been comprehensively reviewed elsewhere (Gilbert, 2009; NRC, 2005). Briefly, an association between leukemia and medical radiation exposure was first identified in a study of ankylosing spondylitis patients more than 50 years ago. Since then, leukemia has been linked with many medically exposed persons primarily adults (UNSCEAR, 2008a).

A.4.6 Exposure of the Offspring

Radiation could increase cancer risk of the offspring through parental preconception exposures that potentially cause germline mutations, or by in utero exposure of the fetus to radiation, which may cause somatic mutations.

A.4.6.1 Parental Preconception Exposure

Heritable mutations are particularly concerning, especially among women, as their oocytes are fixed at birth. A study in Sweden investigated, among other outcomes, risk of childhood malignancies in the offspring of women exposed to therapeutic radiation for treatment of skin hemangioma, when 18 months or less (Kallen et al., 1998). The mean ovarian dose was 6 cGy and the maximum was 8.6 Gy. No increase in childhood malignancies was detected. Similar results were obtained from a collaborative study from five countries: Denmark, Finland, Iceland, Norway, and Sweden, which included cancer survivors diagnosed when they were less than 20 years old (Sankila et al., 1998). Results from maternal or paternal radiation exposure from medical diagnostic procedures before conception were not associated with childhood cancer in some (Patton et al., 2004) but were in other studies (Graham et al., 1966; Shu et al., 1994a,b). Com-

prehensive studies of the children of cancer survivors exposed to high-dose radiotherapy and chemotherapy provide no evidence for heritable diseases (Signorello et al., 2012; Winther et al., 2012).

In Section A.4.1 we discussed the rejection of the hypothesis—known as the Gardner hypothesis, named after the investigator (Gardner et al., 1990)—that nuclear radiation exposure during work may have an effect on a father's germ cells, producing genetic changes in sperm that may be leukemogenic in the offspring (Draper et al., 1997; Kinlen et al., 1993; McLaughlin et al., 1993b; Pobel and Viel, 1997). Even in the offspring of male atomic bombing survivors in Hiroshima and Nagasaki, no increase in childhood cancer risk was observed (Izumi et al., 2003; Schull and Neel, 1959). A study examined the childhood cancer in the offspring of radiologic technologists in the United States, born in 1921-1984. Testis or ovary doses were estimated by undertaking a comprehensive dose reconstruction using work history data, badge dose data, and literature doses. No convincing evidence of an increased risk of childhood cancer in the offspring of radiologic technologists in association with parental occupational radiation exposure either preconception or in utero was found.

A.4.6.2 In Utero Exposure

A historic study, now known as the Oxford Survey of Childhood Cancers, was the first large study of in utero exposure to low doses of ionizing radiation (1-10 cGy) from diagnostic radiography and risk of childhood cancer. The study examined more than 15,000 case-control pairs and showed an approximately 50 percent increase in the frequency of childhood cancer among the exposed (Stewart et al., 1956). A consistent association has been found in many case-control studies; however, it is not universally accepted that the relationship is causal and not the effect of bias or confounding. Many people think that the observed association is the result of recall bias; mothers of the children who died of the disease would be more motivated to recall in detail the number of medical examinations they undertook during pregnancy, compared to the mothers of healthy children. It was not until later that a study in the United States that relied on hospital records rather than on mother's memory reported similar findings (MacMahon, 1962) that the results were taken seriously. Others believed that the relationship is due to confounding with some aspect of pregnancy that had given rise to the need for radiographic examinations itself. However, the theory was rejected when reanalysis of published data from the Oxford Childhood Cancer Survey showed that the frequency of leukemia and of solid cancers in childhood is greater following antenatal x-radiography, not only in singleton births but also in twins. The radiography rate for singletons and twins differed and was 10 and 55 percent, respectively, as

mothers of twins are x-rayed to determine fetal position before delivery, and not necessarily because of any illness or condition. A similar excess of leukemia and of solid cancers in the x-rayed with such different rates of radiography was strong evidence for irradiation as the cause (Mole, 1974). In support of a causal relationship is the demonstrated increase in risk with the increase in number of x-ray films used during the examination (Bithell and Stewart, 1975); the reduction in risk over time with reduction in fetal dose (Bithell and Stiller, 1988); and animal experiments that show the fetus to be susceptible to the induction of cancer by radiation. Based on the review of the evidence, it was concluded that "radiation doses of the order of 10 mGy received by the fetus in utero produce a consequent increase in the risk of childhood cancer. The excess absolute risk coefficient at this level of exposure is approximately 6% per gray" (Doll and Wakeford, 1997). Under the assumption that the relationship between in utero exposure to medical imaging and cancer is causal, the medical profession has in large part replaced x-rays by ultrasounds.

A reason for doubt of a causal relationship between cancer risk in childhood following prenatal exposure to ionizing radiation is the lack of evidence of a corresponding increased risk in cohort studies, most notably the atomic bombing survivors. Observations of those exposed in utero following the atomic bombings have been published since 1970. Possibly due to the small number of observed cancers, a dose-related increase in cancer mortality before age 15 could not be demonstrated (Jablon and Kato, 1970; Kato, 1971). More specifically, during the period 1950-1984, among atomic bombing survivors exposed in utero, there were only 18 cancer cases; 5 of them were in the "zero-dose" group. Two of these subjects developed childhood cancer and all the others developed cancer in adulthood. At present, there is no evidence to support the hypothesis that in utero exposure confers greater adult-cancer risk than childhood exposure (Preston et al., 2008).

An additional reason for doubt of a causal relationship is the unusual homogeneity of the relative risk of all childhood cancers in the Oxford Survey of Childhood Cancers. Regardless of the type of malignancy (i.e., childhood brain cancer, leukemia, neuroblastoma, Wilms tumor), the relative risks were consistent to a 40 to 50 percent increase in risk (Boice and Miller, 1999). Furthermore, in questioning the biological plausibility of increased cancer risk in childhood following prenatal exposure to ionizing radiation is whether embryonic tumors such as Wilms tumor and neuroblastoma could be induced by exposures that occurred primarily just before birth during pelvimetry in the measurement of the birth canal. These issues are sufficiently important to raise doubts as to the causal nature of the association and the ICRP in their most recent review concluded that the evidence for solid tumors, and in particular childhood brain cancer, was not strong (ICRP, 2003).

A.4.7 Noncancer Diseases and Radiation

The atomic bombing survivor studies and specifically the Adult Health Study is the principle source for information on diseases other than cancer related to radiation exposure. This is particularly true as there are no population-based disease incidence registries other than cancer.

A.4.7.1 Cardiovascular Diseases

The issue of radiation-induced cardiac damage has been demonstrated in studies of breast cancer and Hodgkin's lymphoma patients that received high-dose therapeutic radiation (>30-40 Gy) (Adams et al., 2003; Senkus-Konefka and Jassem, 2007). These patients have a life-long increased risk of fatal cardiovascular events. Data from the Japanese survivors demonstrated for the first time that subtherapeutic doses (<5 Gy) can also be associated with cardiovascular disease (Preston et al., 2003; Shimizu et al., 1992). A recent report indicated an excess relative risk of 14 percent per Sv (95% CI = 6%-23%) with an essentially linear dose response (Shimizu et al., 2010). However, there was substantial uncertainty in the amount of cardiovascular disease risk at doses under 0.5 Sv. Outside the atomic bombing studies, there is mixed epidemiologic evidence to support the notion that exposure to low doses of ionizing radiation increases risk of cardiovascular diseases (Little et al., 2008b, 2010; McGale and Darby; 2005; UNSCEAR, 2006b).

A.4.7.2 Cataracts

Posterior subcapsular or cortical cataracts are characteristic of radiation exposure. Cataracts were observed in survivors that received high doses of radiation within 3-4 years after the bombings in Hiroshima and Nagasaki (Cogan et al., 1949). More recent studies have shown an excess of opacities and cataracts at lower doses to the lens, both in the atomic bombing study (Nakashima et al., 2006; Neriishi et al., 2007) and in Chernobyl cleanup workers who received protracted radiation exposures (Worgul et al., 2007). Those studies suggest there may be a threshold for opacity effects at approximately 0.5 Sv.

A.4.7.3 Thyroid Diseases and Hyperparathyroidism

Nonmalignant thyroid diseases have been examined among those exposed as children or young adults as a result of fallout from the Chernobyl nuclear power plant accident in Ukraine (Zablotska et al., 2002). A signifi-

cant but small association between ^{131}I thyroid dose estimates and prevalent subclinical hypothyroidism with an excess estimated odds ratio per Gray of 0.10 (95% CI = 0.03-0.21) was observed in this cohort.

Together with thyroid cancer, the Hanford Thyroid Disease study examined risks associated with nonmalignant thyroid diseases such as benign thyroid nodules, thyroid nodules, autoimmune thyroiditis, and hypothyroidism. The study provided no evidence of an increase in any of the outcomes measured (Davis et al., 2004).

A study evaluated the prevalence of thyroid diseases and their radiation dose responses in atomic bombing survivors, some 55 years after the bombings. A significant linear radiation dose response for thyroid nodules (malignant and benign) was observed with an excess relative risk of 2.01 per Gray (Imaizumi et al., 2006). The prevalence of hyperparathyroidism was found to increase with an estimated excess relative risk of 3.1 at 1 Gy in the atomic bombing study (Fujiwara et al., 1992) and an excess relative risk of 1.1 at 1 Gy in a follow-up of those with medical irradiation in Chicago (Schneider et al., 1995); however, it was not clear whether there is an effect at low doses.

A.4.7.4 Neurological Effects

High doses of radiation to those with prenatal exposure to the atomic bombing were shown to increase the risk of mental retardation and decrements in intelligence (IQ) more generally (ICRP, 2003; Otake et al., 1996), but were limited to those exposed between 8 and 25 weeks of gestation. A review of the data by the ICRP concluded that there were dose thresholds for these effects of 300 mSv or greater for mental retardation and 100 mSv or greater for IQ (ICRP, 2003). Other related effects seen among those exposed during 8-25 weeks of gestation were diminished school performance and increased episodes of neurological seizures (Dunn et al., 1990; ICRP, 2003).

A.4.7.5 Life-Span Shortening

Life-span shortening provides an index that integrates a variety of possible adverse effects of ionizing radiation and has been seen in animal-model studies at high doses of several sieverts. A study of atomic bombing survivors indicated small amounts of life-span shortening at doses below 1 Sv, but proportionately more at higher doses. About 70 percent of the life-span shortening was due to excess cancer risk (Cologne and Preston, 2000).

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B

Biographical Sketches of Committee And Staff

Burris, John E., Chair

John E. Burris, Ph.D., became president of the Burroughs Wellcome Fund in July 2008. He is the former president of Beloit College. Prior to his appointment at Beloit in 2000, Dr. Burris served for 8 years as director and CEO of the Marine Biological Laboratory in Woods Hole, Massachusetts. From 1984 to 1992 he served as the executive director of the Commission on Life Sciences at the National Research Council/National Academies. He received an A.B. in biology from Harvard University in 1971, attended the University of Wisconsin-Madison in an M.D.-Ph.D. program, and received a Ph.D. in marine biology from the Scripps Institution of Oceanography at the University of California, San Diego, in 1976. A professor of biology at the Pennsylvania State University from 1976 to 1985, he held an adjunct appointment there until coming to Beloit. His research interests are in the areas of marine and terrestrial plant physiology and ecology. He has served as president of the American Institute of Biological Sciences and is or has been a member of a number of distinguished scientific boards and advisory committees including the Grass Foundation; the Stazione Zoologica "Anton Dohrn" in Naples, Italy; the American Association for the Advancement of Science; the Radiation Effects Research Foundation in Hiroshima, Japan; and the Morgridge Institute for Research. He has also served as a consultant to the National Conference of Catholic Bishops' Committee on Science and Human Values.

MEMBERS

Bailar, John C.

John C. Bailar III, MD, Ph.D. (statistics), is professor emeritus at the University of Chicago and founding chair of the Department of Health Studies there. For many years his professional interests centered on the causes and prevention of disease. More recently he has focused on improving quality and performance in science generally. He was at the U.S. National Cancer Institute (1956-1980), Harvard University (1980-1988), and McGill University (1988-1995) before he went to Chicago. At present he is scholar in residence at the National Academies. He was a MacArthur Fellow (1990-1995). He has published widely in the statistics and epidemiology literature, including, recently, the health effects of air pollution. Bailar has served on more than 30 committees at the U.S. National Academies, and as chair or co-chair of 12 of them.

Beck, Harold L.

Mr. Beck is an expert in radiation dose reconstruction. A physicist for the U.S. Department of Energy (DOE)/Atomic Energy Commission for over 36 years, he retired in 1999 as the Director of the Environmental Science Division of the DOE Environmental Measurements Laboratory (EML) in New York City and is presently a private consultant conducting various dose reconstructions in cooperation with scientists at the National Cancer Institute and Vanderbilt University. During his tenure at EML, he also served as director of the EML Instrumentation Division and as acting deputy director of the Laboratory. Mr. Beck has authored well over 100 publications on radiation physics, radiation measurement, dose reconstruction, environmental radiation, and radiation dosimetry. His efforts in the development of the scientific approach to reconstructing fallout doses to the U.S. population from above-ground nuclear weapons testing in Nevada earned him the DOE Meritorious Service Award in 1988, the second highest award in the department. Mr. Beck served as scientific vice president for radiation measurements and dosimetry of the National Council on Radiation Protection and Measurements (NCRP) from 1996 to 2003 and in 2004 was elected to distinguished emeritus membership in NCRP. From 2004 to 2006, he served as a member of the National Research Council's (NRC's) Board on Radiation Effects Research, Nuclear and Radiation Studies Board. He currently serves as a member of the Veterans (federal advisory) Board on Dose Reconstruction and the U.S. Scientific Review Group, Department of Energy Russian Health Studies Program. He has served as an expert member or chair on a number of NCRP and NRC scientific studies related to radiation dosimetry.

Bouville, Andre

Andre Bouville obtained his Ph.D. in physics at the University Paul-Sabatier in Toulouse in 1970. He was scientific secretary of the United Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) from 1970 to 1972 and remained associated with that committee as a consultant until 2000. From 1972 to 1984, Dr. Bouville was employed in France by the Institute of Radiation Protection and Nuclear Safety, where he contributed to a number of environmental and dosimetric studies related to nuclear facilities. He joined the National Cancer Institute in 1984, where, first as an expert and then as a senior radiation physicist, he has been involved mainly in the estimation of radiation doses resulting from radioactive fallout from atmospheric nuclear weapons tests and from the Chernobyl accident. He was head of the Radiation Dosimetry Unit of the Radiation Epidemiology Branch until his retirement at the end of 2010.

Corso, Phaedra S.

Phaedra S. Corso, Ph.D., MPA, is associate professor and head of the Department of Health Policy and Management in the College of Public Health at the University of Georgia (UGA). Prior to joining the UGA faculty in 2006, Dr. Corso worked for 15 years at the Centers for Disease Control and Prevention as an economic and policy analyst, most recently in the area of injury and violence prevention. Her research focuses on the practical application of economic evaluation for setting public health policy and assessing health-related quality of life in vulnerable populations. Dr. Corso has co-edited two editions of a primer on how to conduct economic evaluations in public health settings, a book on the incidence and economic costs of injury, and has produced numerous peer-reviewed articles on economic evaluation applied to prevention interventions. She holds a master's degree in public administration from UGA (1991) and a doctoral degree in health policy from Harvard University (2000).

Culligan, Patricia J.

Patricia J. Culligan, Ph.D., is professor of civil engineering and engineering mechanics at Columbia University and the vice dean of academic affairs for Columbia Engineering. Her research focuses on applying geoengineering principles to understand and control the migration of contaminants from waste disposal sites. She studies the behavior of miscible contaminants, nonaqueous phase liquids and colloids in soil and fractured rock and the effectiveness of in situ remediation strategies for the cleanup of waste sites. She also has interest and experience in the design of land-based disposal sites for waste materials. Dr. Culligan has received numerous awards, in-

cluding MIT's Arthur C. Smith Award for Undergraduate Service (1999), the National Science Foundation Career Award (1999), and Columbia University's Presidential Award for Outstanding Teaching (2007). She is also the author or coauthor of more than 80 journal articles, book chapters, and refereed conference papers. Dr. Culligan has a Ph.D. in civil engineering from Cambridge University, England. She currently serves on the Nuclear and Radiation Studies Board at the National Academies.

DeLuca, Paul M., Jr.

Paul M. DeLuca, Jr., Ph.D., received a bachelor of science degree in physics and math in 1966 and a doctorate in nuclear physics from the University of Notre Dame in 1971. That same year he joined the University of Wisconsin–Madison as a research associate, and in 1975 he was appointed to the faculty of the Department of Radiology. Following the creation of the Department of Medical Physics in 1981, he served as chair from 1987 through 1998 and holds an appointment as professor in the Departments of Medical Physics, Radiology, Human Oncology, Engineering Physics and Physics. In 1999, DeLuca assumed a role in the University of Wisconsin School of Medicine and Public Health as associate dean for research and graduate studies, and his administrative role was expanded in 2001 with his appointment as vice dean. In that role, he was closely involved with the development of the Wisconsin Institutes of Medical Research. He began serving as provost and vice chancellor for academic affairs in July 2009. His research interests have concentrated on fast neutron dosimetry including production of intense sources of fast neutrons, determination of elemental neutron kerma factors and application of microdosimetry to radiation dosimetry. DeLuca is an internationally recognized expert in high energy particle radiation effects on humans. He is a member of the International Commission on Radiation Units and Measurements and currently serves as vice chairman. He is also a member and chair of the Nonproliferation and International Security Division Review Committee at Los Alamos. Other national and international associations and professional society affiliations include the American Association of Physicists in Medicine, the American Physical Society, the Health Physics Society, the NCRP, the Council on Ionizing Radiation Measurements and Standards, and the Institute of Physics.

Guilmette, Raymond A.

Raymond L. Guilmette, Ph.D., received a B.S. in nuclear engineering from Rensselaer Polytechnic Institute and an M.S. in environmental health sciences and a Ph.D. in radiological health from New York University. For almost 40 years, he has been studying the metabolism, biokinetics, dosim-

etry, and biological effects of internally deposited radionuclides, developing methods for removing radionuclides from the body (decorporation), and studying the mechanisms of deposition, clearance, and retention of inhaled materials. Most of this research was performed at the Lovelace Respiratory Research Institute (LRRI; formerly the Inhalation Toxicology Research Institute), where he worked for 23 years. From 2000 through 2007, he was team leader for internal dosimetry at the Los Alamos National Laboratory, assessing radiation doses for workers who were exposed to radionuclides associated with the nuclear weapons industry. In 2007, he returned to LRRI as director of the Center for Countermeasures Against Radiation, where he is evaluating the efficacy of chemical compounds designed to decorporate radionuclides as well as drugs designed to ameliorate the effects of acute radiation syndrome from large external radiation doses. He is a past president of the Health Physics Society, received its Distinguished Scientific Achievement Award in 2002, and has given several honorary lectures (Newell Stannard Memorial Lecture, 2006; G. William Morgan Lecture, HPS, 2009; inaugural Patricia W. Durbin Memorial Lecture, Lawrence Berkeley National Laboratory, 2010). He is a member of scientific committees of the International Commission on Radiological Protection, the NCRP (also a board member), and the International Agency for Research on Cancer.

Hornberger, George M.

George M. Hornberger, Ph.D., is distinguished university professor at Vanderbilt University, where he is the director of the Vanderbilt Institute for Energy and the Environment. He has a shared appointment as the Craig E. Philip Professor of Engineering and as Professor of Earth and Environmental Sciences there. He previously was a professor at the University of Virginia for many years, where he held the Ernest H. Ern Chair of Environmental Sciences. He also has been a visiting scholar at the Australian National University, Lancaster University, Stanford University, the U.S. Geological Survey (USGS), the University of Colorado, and the University of California at Berkeley. His research is aimed at understanding complex water-energy-climate interrelationships and how hydrological processes affect the transport of dissolved and suspended constituents through catchments and aquifers. He is an ISI "Highly Cited Researcher" in environmental sciences and engineering, a recognition given to the top 250 individual researchers in each of 21 subject categories. Hornberger is a fellow of the American Geophysical Union (AGU), the Geological Society of America, and the Association for Women in Science. He was president of the Hydrology Section of AGU from 2006 to 2008. He has been a member of the Nuclear Waste Technical Review Board (a presidential appointment) since April 2004. He has served on numerous boards and committees of the

National Academies, including as chair of the Commission on Geosciences, Environment, and Resources (1996-2000) and chair of the Board on Earth Sciences and Resources (2003-2009). Professor Hornberger won the Robert E. Horton Award (Hydrology Section) from the AGU in 1993. In 1995, he received the John Wesley Powell Award from the USGS. In 1999, he was presented with the Excellence in Geophysical Education Award by the AGU and in 2007 he was selected Virginia Outstanding Scientist. Professor Hornberger was elected to the U.S. National Academy of Engineering in 1996.

Karagas, Margaret

Margaret Karagas, Ph.D., is professor of community and family medicine in the Department of Epidemiology at Dartmouth Medical School. She received her Ph.D. from the University of Washington. Professor Karagas' research includes several epidemiological studies focusing on the etiologic mechanisms and prevention of human cancers and other adverse health outcomes. Among these are investigations to determine the incidence rates of basal cell and squamous cell skin cancer and to assess the extent of any increases in rates over the past 20 years. Another aspect of this research is a population-based case-control study of these malignancies that is designed to quantify the risks associated with tanning lamps, ingestion of arsenic-containing drinking water, immunosuppressive therapy, and other factors. The research has been extended to study the effects of arsenic on bladder cancer and to conduct chemical analyses of household drinking water supplies. Her work also includes studies of melanoma among women and collaborative investigations of markers of individual susceptibility and biological response to environmental agents.

Kasperson, Roger E.

Roger E. Kasperson received his Ph.D. from the University of Chicago in 1966. Before joining the Clark University faculty he taught at the University of Connecticut and Michigan State University. He has written widely on issues connected with risk analysis, risk communication, global environmental change, risk and ethics, and environmental policy. Dr. Kasperson is a member of the National Academy of Sciences and the American Academy of Arts and Sciences. He has been honored by the Association of American Geographers for his hazards research, and he is a recipient of the 2006 Distinguished Achievement Award of the Society for Risk Analysis. He has been a consultant or advisor to numerous public and private agencies on energy and environmental issues and has served on various committees of the NRC and the Council of the Society for Risk Analysis. From 1992 to 1996 he chaired the International Geographical Union Commission on

Critical Situations/Regions in Environmental Change. He was vice president for academic affairs at Clark University from 1993 to 1996, and in 1999 he was elected director of the Stockholm Environment Institute, a post he held through 2004. He now serves on the Board on Environmental Sciences and Toxicology of the NRC and is on the executive steering committee of the START Programme of the IGBH. He is research professor and distinguished scientist at Clark University.

Klaunig, James E.

James E. Klaunig, Ph.D., is the Robert B. Forney Professor and director of toxicology in the Department of Pharmacology and Toxicology as well as the founding director of the Center for Environmental Health and associate director of the Cancer Center at Indiana University. He also serves as the program director of the Molecular and Environmental Carcinogenesis Program for the Indiana University Cancer Center. Dr. Klaunig's research interests are dedicated to understanding the mechanisms of chemically induced carcinogenesis with emphasis on the epigenetic (nongenotoxic) modes of action. This has involved studies into the role of oxidative stress/oxidative damage, Kupffer cell activation, modulation of cell-to-cell communication, cell growth and apoptosis in this process, and understanding the multistage nature of the cancer process. Dr. Klaunig also served the state of Indiana as the director of toxicology and the state toxicologist from 1991 to 2003. Dr. Klaunig is board certified in toxicology and a fellow in the Academy of Toxicological Sciences. He has published over 180 peer-reviewed manuscripts and book chapters in toxicology, carcinogenesis, and risk assessment and has mentored over 40 M.S., Ph.D., and postdoctoral fellows in toxicology and chemical carcinogenesis. He has served as an associate editor of *Toxicological Sciences* and is currently the editor-in-chief of *Toxicologic Pathology*. He received a B.S. in biology from Ursinus College and a Ph.D. in experimental pathology from the University of Maryland.

Mousseau, Timothy

Timothy Mousseau, Ph.D., received his doctoral degree in 1988 from McGill University and completed a Natural Sciences and Engineering Research Council of Canada postdoctoral fellowship in population biology at the University of California, Davis, before joining the faculty of the Department of Biological Sciences at the University of South Carolina in 1991. He is currently an associate vice president for research and graduate education and dean of the graduate school. Professor Mousseau's experience includes having served as a program officer at the National Science Foundation, on the editorial board for several journals, and on the USGS and a variety

of international grant foundation advisory panels. He has published over 100 scholarly articles and has edited two books. He is currently co-editor-in-chief of a new annual review series, The Year in Evolutionary Biology, published by the New York Academy of Sciences. He was elected a fellow of the American Association for the Advancement of Science in 2008. His primary areas of research interest include the genetic basis of adaptation in natural populations. Since 1999, Professor Mousseau and his collaborators have explored the ecological consequences of low-dose radiation in populations of plants, animals, and people inhabiting the Chernobyl region of Ukraine and Belarus. Dr. Mousseau's current research is aimed at elucidating the causes of variation among different species in their apparent sensitivity to radionuclides.

Murphy, Sharon B.

Sharon B. Murphy, M.D., joined the Institute of Medicine as a scholar-in-residence in October 2008. Previously, she was the inaugural director of the Greehey Children's Cancer Research Institute and professor of pediatrics at the University of Texas Health Science Center at San Antonio from 2002 to 2008. She earned her B.S. from the University of Wisconsin (1965) and her medical degree, cum laude, from Harvard Medical School (1969). She completed postdoctoral training in pediatrics at the University of Colorado (1969-1971) and in pediatric hematology and oncology at the University of Pennsylvania (1971-1973). A pediatric oncologist and clinical cancer researcher, Dr. Murphy has devoted the past 35 years to improving cure rates for childhood cancer, particularly childhood lymphomas and leukemias. She was chair of the Pediatric Oncology Group from 1993 to 2001. She has been recognized for her achievements by the Association of Community Cancer Centers (2001), the Distinguished Service Award for Scientific Leadership from the American Society of Clinical Oncology (2005), the Distinguished Career Award from the American Society of Pediatric Hematology and Oncology (2009), and the Pediatric Oncology Award from the American Society of Clinical Oncology (2010).

Shore, Roy E.

Roy E. Shore, Ph.D., DrPH, received his degrees from Syracuse University (Ph.D.) and Columbia University (DrPH in epidemiology). At New York University (NYU) School of Medicine he was a professor, director of the Epidemiology Program in the Department of Environmental Medicine, and an associate director of the NYU Cancer Center. He is currently vice chairman and chief of research at the Radiation Effects Research Foundation (RERF) in Hiroshima-Nagasaki, which conducts health studies of the Japa-

nese atomic bomb survivors. Dr. Shore has authored or co-authored over 100 publications pertaining to radiation epidemiology and risk assessment. He has served on a number of radiation committees for the NRC/National Academies and the NCRP, and he was a long-time member of Committee 1 of the International Commission on Radiological Protection pertaining to radiation biology and risk assessment. He has also served as an expert consultant to UNSCEAR and the World Health Organization (WHO).

Stram, Daniel O.

Daniel O. Stram, Ph.D., is professor in the Department of Preventive Medicine at the Keck School of Medicine of the University of Southern California. He received his Ph.D. in statistics from Temple University in 1983 and served as a postdoctoral fellow in the Biostatistics Department of the Harvard School of Public Health from 1984 to 1986. From 1986 to 1989 he was a research associate at RERF in Hiroshima, Japan. Dr. Stram's main areas of research are in the statistical problems that arise in the design, analysis, and interpretation of epidemiological studies of cancer and other diseases. His work on radiation epidemiology studies includes (1) helping to characterize the statistical nature of errors in dose estimates for the atomic bomb survivor study, (2) developing a multilevel variance components model for the dosimetry used in the Colorado Plateau uranium miners cohort for the purpose of better understanding dose and dose rate effects in those data, (3) characterizing study power and sample size issues in epidemiologic studies in which a complex dosimetry system is used to estimate radiation dose. Besides the field of radiation epidemiology, his past and current research has focused on statistical issues relevant to clinical trials of treatment for pediatric cancer, nutritional epidemiology studies, and to studies of the genetics of complex diseases. He is an elected fellow of the American Statistical Association and has authored or co-authored over 200 peer-reviewed articles.

Tirmache, Margot

Margot Tirmache, Ph.D., is director of scientific assessment at the Institute of Radiation Protection and Nuclear Safety (IRSN). She was the chief of the laboratory of epidemiology at IRSN for the period 1999-2008 and an epidemiologist in the same laboratory since 1980. She has a scientific background (Ph.D. equivalent) in biology and genetics, completed by specific diploma at the Medical University of Paris (Paris XI), related to epidemiology and oncology. During the period 1975-1979 she worked at the Institute of Cancer in Villejuif (IGR) in charge of the French coordination of a case-control study initiated by the National Cancer Institute, aiming to a joint

American-European analysis of lung cancer risk and tobacco consumption in different countries. She started in the radiation epidemiology field in 1980 and was in charge of the first cohort study in this field in France (uranium miners cohort). She conducted and coordinated several epidemiologic studies in relation to low chronic radiation exposure of various types: alpha exposure (radon decay exposure), external exposure (occupational cohorts), post-Chernobyl studies, and studies in the Urals. She also coordinated several multinational European contracts in the field of radiation epidemiology. She is a member of the French delegation at UNSCEAR, contributing to recently published reports on radon and on Chernobyl effects. She is also member of Committee 1 of the International Commission on Radiological Protection, where she is presently in charge of a working group that is analyzing cancer risk linked to alpha emitters (radon decay, uranium, plutonium). She is also an expert of the WHO.

Waller, Lance

Lance Waller, Ph.D., is the Rollins Professor and chair of the Department of Biostatistics and Bioinformatics at Emory University. His interests involve statistical analysis of spatially referenced data. Examples include tests of spatial clustering of disease cases, for example around a hazardous waste site; small area estimation; hierarchical models with spatially structured random effects; and spatial point process models. Recent applications include spatiotemporal mapping of disease rates, statistical methods for assessing environmental justice, the analysis of spatial trends in Lyme disease incidence and reporting, spatial modelling of the spread of raccoon rabies, and point process analysis of sea turtle nesting locations in Florida. He is interested in both the statistical methodology and the environmental and epidemiologic models involved in the analysis of this type of data. He teaches courses in spatial biostatistics, applied linear models, and Geographic Information Systems in public health. Waller has authored or coauthored more than 100 articles and one book. He has served the National Academies as a member of the Committee to Assess Potential Health Effects from Exposures to PAVE PAWS Low-level Phased-Array Radiofrequency Energy, the Committee on Review of Existing and Potential Standoff Explosives Detection Techniques, and the Committee on the Utility of Proximity-Based Herbicide Exposure Assessment in Epidemiologic Studies of Vietnam Veterans. He received his Ph.D. in operations research from Cornell University in 1992.

Woloschak, Gayle E.

Gayle E. Woloschak, Ph.D., is professor of radiation oncology at the Feinberg School of Medicine at Northwestern University. She received her bachelor's degree in biological sciences summa cum laude from Youngstown State University in Ohio and her Ph.D. in microbiology from the Medical College of Ohio in 1980. Afterward, she served as a postdoctoral research fellow in the Department of Immunology and Department of Cell Biology. In previous scientific positions she has worked at the Mayo Clinic and Argonne National Laboratory. Gayle Woloschak's laboratory is pursuing several areas of genetic research. Her projects include understanding the molecular basis of motor neuron disease in a mouse model and in humans. This project involves uncovering genes that cause motor neuron disease in a mouse model and also in humans. Her laboratory has several candidate genes that are being analyzed using a variety of different chip-based and protein-interaction approaches. Another project involves understanding the molecular basis of normal tissue responses to ionizing radiation and radiation-sensitivity syndromes. This project involves identifying differences in molecular responses of normal tissues to the effects of ionizing radiation. The hope is to identify genes that can be used to distinguish people who are more or less likely to have particular late effects following radiation exposure. Her laboratory is an investigator on a related project with Dr. Jeri Logemann to identify people at risk for swallowing problems following head and neck cancer radiotherapy.

Wong, Jeffrey J.

Jeffrey J. Wong, Ph.D., is chief scientist for the California Department of Toxic Substances Control (DTSC) at the California Environmental Protection Agency in Sacramento, California. For more than 20 years, he has managed DTSC's efforts in environmental measurements, biological and exposure monitoring, toxicology and risk assessment, and pollution prevention approaches and technologies; he is currently leading efforts focused on nanotechnologies, other emerging contaminants, and green chemistry. Prior to his work in the DTSC, Dr. Wong was involved in forensic investigations for the Department of Justice and pesticide toxicity evaluation for the Department of Food and Agriculture. Dr. Wong has served on panels for the National Academies, the U.S. Environmental Protection Agency, and DOE. He was appointed by President Clinton to serve on the Nuclear Waste Technical Review Board. Dr. Wong earned his Ph.D. at the University of California, Davis.

STAFF

Crowley, Kevin D.

Kevin D. Crowley is senior board director of the Nuclear and Radiation Studies Board (NRSB) at the National Research Council–National Academy of Sciences in Washington, DC. He is responsible for managing the NRSB's work on nuclear safety and security, radioactive-waste management and environmental cleanup, and radiation health effects. He is also the principal investigator for a long-standing cooperative agreement between the National Academy of Sciences and the U.S. Department of Energy to provide scientific support for the Radiation Effects Research Foundation in Hiroshima, Japan. Dr. Crowley's professional interests and activities focus on safety, security, and technical efficacy of nuclear and radiation-based technologies. He has directed over 20 National Research Council studies on these and other topics, including *Safety and Security of Commercial Spent Nuclear Fuel Storage* (2004, 2006); *Going the Distance? The Safe Transport of Spent Nuclear Fuel and High-Level Radioactive Waste in the United States* (2006); *Medical Isotope Production without Highly Enriched Uranium* (2009); *America's Energy Future: Technology and Transformation* (2009); and *Analysis of Cancer Risks in Populations near Nuclear Facilities*. Before joining the National Academies staff in 1993, Dr. Crowley held teaching/research positions at Miami University of Ohio, the University of Oklahoma, and the U.S. Geological Survey. He holds M.A. and Ph.D. degrees, both in geology, from Princeton University.

Kosti, Ourania (Rania)

Rania Kosti joined the staff of the Nuclear and Radiation Studies Board in January 2011. Prior to her current appointment, Rania was a post-doctoral fellow at the Lombardi Comprehensive Cancer Center at Georgetown University Hospital in Washington, DC, where she conducted research on biomarker development for early cancer detection using case-control epidemiologic study designs. She focused primarily on prostate, breast, and liver cancers and trying to identify those individuals who are at high risk of developing malignancies. She contributed on hypotheses generation, study design, data analysis and management of clinical databases and biospecimen repositories. Dr Kosti also trained at the National Cancer Institute (NCI) (2005-2007) in the Cancer and Developmental Biology Laboratory; the same period she volunteered in NCI's Division of Cancer Epidemiology and Genetics. Rania received a BSc. in biochemistry from the University of Surrey, UK, an MSc in molecular medicine from the University College London and a Ph.D in molecular endocrinology from St Bartholomew's Hospital in London, UK.

C

Presentations and Visits

Washington, DC, February 24, 2011

- The U.S. Nuclear Regulatory Commission's request to the National Academy of Sciences to Perform the Study, "Analysis of Cancer Risks in Populations Near Nuclear Facilities—Phase 1 Feasibility Study," Brian Sheron, Director, Office of Nuclear Regulatory Research; Terry Brock, Senior Program Manager, Office of Nuclear Regulatory Research

Chicago, IL, April 18, 2011

- U.S. NRC's program for keeping nuclear power plant offsite doses as low as reasonably achievable (ALARA), Steven Schaffer, senior health physicist, Office of Nuclear Regulatory Research; Richard Conatser, health physicist, Office of Nuclear Regulatory Research
- Radiological Environmental Monitoring Program at Exelon Nuclear, Willie Harris, director, Radiation Protection, Exelon nuclear; Ronald Chrzanowski, Corporate Chemistry Manager, Exelon Nuclear
- Health concerns and data around the Illinois nuclear power plants, Joseph Sauer, M.D., practicing physician, Indiana
- The North American Association of Central Cancer Registries (NAACCR), Betsy Kohler, executive director, NAACCR
- Childhood cancer: current knowledge and challenges in studying risk factors, Julie Ross, professor and director of the Division of Pediatric Epidemiology & Clinical Research, University of Minnesota

- Low-dose environmental radiation and cancer risk: Study design and methods considerations, Martha Linet, chief and senior investigator, Radiation Epidemiology Branch, National Cancer Institute

Atlanta, GA, May 23, 2011

- Uranium Recovery Regulations and Operations, Elise Striz, Office of Federal and State Materials and Environmental Management Programs, U.S. Nuclear Regulatory Commission (presentation prepared by: Randolph Von Till, Office of Federal and State Materials and Environmental Management Programs, U.S. Nuclear Regulatory Commission)
- Fuel-Cycle Facilities, John Pelchat, Region II, U.S. Nuclear Regulatory Commission; Gregory Chapman, project manager, Nuclear Regulatory Commission
- ATSDR's approach to site assessment and epidemiologic considerations for multisite studies, Steve Dearwent, Epidemiologist, Agency for Toxic Substances and Disease Registry (ATSDR), Department of Health and Human Services, Centers for Disease Control and Prevention
- Dose reconstruction in the epidemiologic study of the possible effect of ionizing radiation deriving from the operation of Spanish nuclear power plants and fuel-cycle facilities, Lucila Ramos, Deputy Director for Environmental Radiation Protection, Nuclear Safety Council (CSN), Spain
- Exposure to ionizing radiations arising from the operation of nuclear installations and its possible relationship with cancer mortality in Spain, Gonzalo López-Abente, National Center for Epidemiology, Carlos III Institute of Health, Spain
- Cancer risks near nuclear facilities: The importance of research design and explicit study hypotheses (round table discussion), Steve Wing, Associate Professor, Department of Epidemiology, University of North Carolina, Chapel Hill
- Challenges for the historical dose reconstruction of U.S. nuclear power plants (round table discussion), John Till, President, Risk Assessment Corporation
- Modeling for Environmental Radiation Dose Reconstruction, Bruce Napier, Staff Scientist, Energy and Environment Division, Pacific Northwest National Laboratory
- Designing large-scale case-control studies, Dana Flanders, Professor, Department of Environmental Health Epidemiology, Rollins School of Public Health, Emory University
- Overview of the National Program of Cancer Registries (NPCR),

Christie Eheman, Chief, Cancer Surveillance Branch, Centers for Disease Control and Prevention
- Overview of the Surveillance, Epidemiology and End Results (SEER) registry, Kevin Ward, Georgia Center for Cancer Statistics, Rollins School of Public Health, Emory University (on behalf of Brenda Edwards, Associate Director, Surveillance Research Program, National Cancer Institute)
- The Georgia Cancer Registry—A state's perspective, Kevin Ward, Georgia Center for Cancer Statistics, Rollins School of Public Health, Emory University
- The Georgia State's response to public concerns (round table discussion), Franklin Sanchez, Program Consultant, Environmental Health Branch, Georgia Department of Community Health, Chrissy McNamara, Epidemiologist, Georgia Comprehensive Cancer Registry

Irvine, CA, July 21, 2011

- Childhood cancer and nuclear power plants in Switzerland: National cohort study, Matthias Egger, Director, Institute of Social and Preventive Medicine, University of Bern, Switzerland
- Technical considerations for NAS Proposed Study of Cancer Risks in Populations Living Near Nuclear Facilities, Antone Brooks, Washington State University Tri-cities (retired professor); Helen Grogan, Cascade Scientific, Inc; David Hoel, Medical University of South Carolina; Phung Tran, Electric Power Research Institute; Bill Wendland, CN Associates
- Protocol for an analysis of cancer risk in populations living near nuclear-power facilities, 2009, Donna Cragle, Vice President and Director, Occupational Exposure and Worker Health, Oak Ridge Institute for Science and Education
- States' environmental monitoring at nuclear power plants, Alice Rogers, Chair, Conference of Radiation Control Program Directors (Texas Department of State Health Services)

Washington, DC, October 20, 2011

- Studies of health effects near Massachusetts nuclear power stations, Richard Clapp, Professor Emeritus, Boston University School of Public Health and Adjunct Professor, University of Massachusetts, Lowell
- Nuclear Regulatory Commission and stakeholder interactions, Scott Burnell, Public Affairs Officer, Office of Public Affairs, U.S.

Nuclear Regulatory Commission; Lance J Rakovan, Senior Communications Specialist, Office of the Executive Director for Operations, U.S. Nuclear Regulatory Commission

- Radiation risk communications: Challenges and opportunities, Tony Nesky, U.S. Environmental Protection Agency, Radiation Protection Division
- Next steps for the Analysis of Cancer Risk in Populations Near Nuclear Facilities Study, Terry Brock, Senior Program Manager, Office of Nuclear Regulatory Research, U.S. Nuclear Regulatory Commission

SITE VISITS

- April 20, 2011: Visit to Dresden Generating Station (Grundy County, Illinois)
- July 19, 2011: Visit to San Onofre Nuclear Generating Station (San Diego County, California)
- October 13, 2011: Visit to Nuclear Fuel Services Erwin nuclear fuel plant (Erwin, Tennessee)

D

Origin of Radioactivity in Nuclear Plants

Nuclear power reactors[1] are fueled with uranium that is slightly enriched in the isotope uranium-235.[2] This isotope is capable of sustaining a controlled nuclear chain reaction that is necessary for production of electrical energy. The chain reaction results in the production of neutrons that induce radioactivity in the fuel, cooling water, and structural components of the reactor.

Radioactivity is induced primarily through processes involving the capture of neutrons by uranium atoms in the fuel. *Fission* occurs when the nucleus of a uranium-235 atom (and less commonly a uranium-238 atom) captures a neutron, becomes unstable, and splits into two and (infrequently) three[3] lighter nuclei; these nuclei are referred to as *fission products*. Uranium fission produces a bimodal mass distribution of fission products shown in Figure D.1. The most common fission products have mass numbers around 90 and 137 (for example, strontium-90 and cesium-137).

The fission products produced in a nuclear power reactor span the periodic table. They include:

- Noble gases, for example, krypton-85 and xenon-133.
- Halogens, for example, iodide-131.

[1]The terms *nuclear power reactors* and *nuclear power plants* refer to reactors that are used on a commercial basis to produce electricity. Such reactors typically generate on the order of 1000 megawatts of electrical power and 3000 megawatts of thermal power.

[2]Natural uranium contains about 99.3 percent uranium-238 and 0.7 percent uranium-235. The fuel used in power reactors is typically enriched in uranium-235 to levels of 3-5 percent.

[3]Referred to as *ternary fission*.

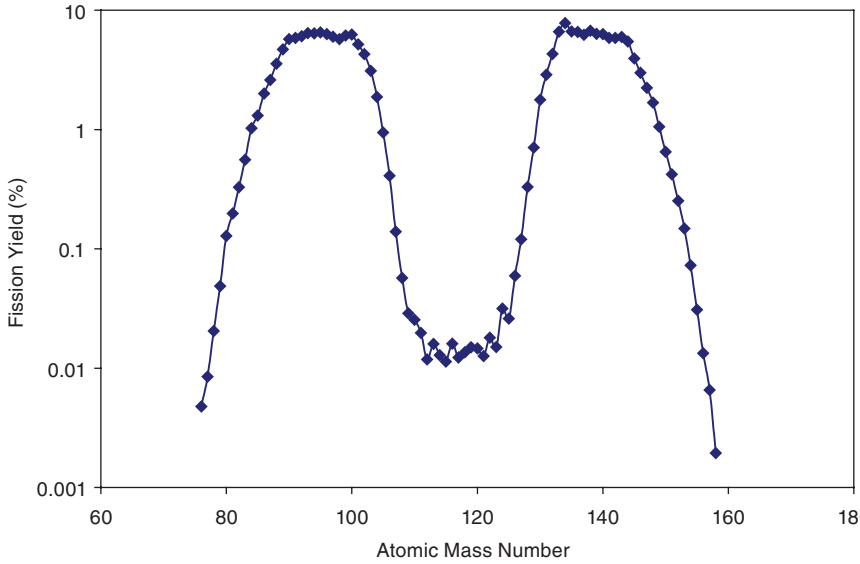

FIGURE D.1 Mass distributions resulting from fission of uranium-235 by thermal neutrons. SOURCE: Data from Joint Evaluated Fission and Fusion File, Incident-neutron data, http://www-nds.iaea.org/exfor/endf00.htm, October 2, 2006; see http://www-nds.iaea.org/sgnucdat/c1.htm.

- Alkali metals, for example, cesium-137.
- Alkaline earth metals, for example, strontium-90.
- Less commonly, hydrogen-3, more commonly referred to as *tritium* (T), from ternary fission of uranium atoms.

Neutron capture can also induce radioactivity through the *transmutation* of one chemical element into another. The transmutation process results in the emission of nuclear particles (e.g., protons) and radiation from the nucleus. Some transmutation reactions and products of significance in power reactors include the following:

- Production of nitrogen-16 through the capture of a neutron by the nucleus of an oxygen atom: oxygen-16 + neutron → nitrogen-16 + proton (abbreviated as $^{16}O(n,p)^{16}N$). Nitrogen-16 has a short (7-second) half-life and is primarily a hazard to workers at nuclear plants.
- Production of carbon-14 through the capture of neutrons by the nuclei of nitrogen, oxygen, or carbon atoms: $^{14}N(n,p)^{14}C$; $^{13}C(n,\gamma)^{14}C$; $^{17}O(n,\alpha)^{14}C$.

- Production of tritium (T) by the capture of a neutron by the nucleus of a boron atom: $^{10}B(n,2\alpha)T$. This is an important reaction in pressurized-water reactors, which use boron in cooling water to control reactivity.
- Production of tritium through capture of a neutron by a deuterium atom that is naturally present in the cooling water of a reactor.

Neutron capture can also induce radioactivity through *activation*. The capture of a neutron excites the nucleus, which quickly decays to a less energetic state through the emission of radiation. Some activation reactions and products of significance in power reactors include the following:

- Production of cobalt-60 from cobalt-59 through the reaction $^{59}Co(n, \gamma)^{60}Co$.
- Production of iron-55 from iron-54 through the reaction $^{54}Fe(n, \gamma)^{55}Fe$.

Cobalt-60 and iron-55 are common activation products in the structural components of reactors.

The isotopes produced by these neutron capture processes are almost always radioactive. Their decay involves the emission of alpha, beta, and gamma radiation, to produce both radioactive and nonradioactive *decay products*. A decay reaction of particular importance in nuclear power reactors is the following:

$$^{238}_{92}\mathrm{U} + {}^{1}_{0}\mathrm{n} \rightarrow {}^{239}_{92}\mathrm{U} \xrightarrow{\beta^-} {}^{239}_{93}\mathrm{Np} \xrightarrow{\beta^-} {}^{239}_{94}\mathrm{PU}$$

This reaction produces plutonium-239 by uranium-238 neutron capture followed by two beta decays.

The particles and other radiation emitted during neutron capture can interact with atoms in the fuel, coolant, and reactor structures to produce additional radioactivity. For example, the interaction of energetic electrons with materials in the reactor results in the emission of photons known as *bremsstrahlung*. This radiation appears as a faint blue glow when electrons interact with cooling water in the reactor and spent fuel pools.

E

Origin of Radioactivity in Fuel-Cycle Facilities

Fuel-cycle facilities are involved in the extraction and processing of uranium to produce fuel for nuclear reactors. Consequently, the most important radioactive effluent releases from these facilities involve uranium and its decay products (Table E.1).

Uranium and its decay products are present in equilibrium at mining and milling facilities (Figure E.1). The uranium decay products are removed during the milling process[1] and disposed of onsite as *mill tailings* (Figure E.2), which are potential sources of radioactive particulate and radon gas effluent releases from these facilities.

Other radioactive isotopes are sometimes present in effluent releases from enrichment and fuel fabrication facilities, usually at trace levels. These include cesium-137, technetium-99, as well as a number of actinide isotopes, most notably uranium-236, neptunium-237, and plutonium-239/240. These isotopes are produced by fission and neutron-capture reactions (these reactions are described in Appendix D). Their presence in an effluent release indicates that the facility has processed uranium that was previously irradiated in a nuclear reactor.[2]

[1]However, the decay products "grow back" into the uranium with time, especially those decay products near the top of the uranium decay chains, which have short half-lives (see Figure E.2).

[2]For example, recycled uranium (i.e., uranium obtained from reprocessing spent nuclear fuel) was enriched at the Paducah Gaseous Diffusion Plant between 1953 and 1975. This plant is still reporting releases of radioactive effluents from this recycled uranium.

TABLE E.1 Typical Effluent Releases from Fuel-Cycle Facilities

Facility Type	Typical Radioactive Effluents
Mining (in situ leaching)	Uranium, radon, and progeny
Milling	Uranium, radon, and progeny
Conversion	Uranium, radium-226, thorium-230
Enrichment	Natural uranium, uranium-235, thorium-230, technetium-99, neptunium-237, plutonium-239, 240
Fuel Fabrication	Uranium-234, 235, 236, 238

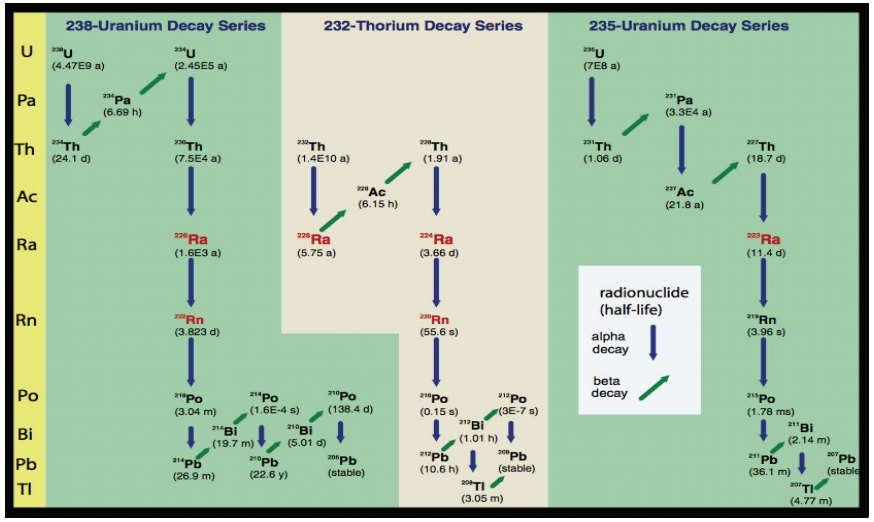

FIGURE E.1 Schematic illustration of the uranium-235, thorium-232, and uranium-238 decay chains showing decay modes (i.e., alpha or beta decay), half-lives, and progeny. SOURCE: U.S. Geological Survey, http://gulfsci.usgs.gov/tampabay/data/2_biogeochem/images/decaychain.gif.

FIGURE E.2 Aerial view of the White Mesa Uranium Mill near Blanding, Utah. The mill facilities can be seen in the upper right quadrant of the photo. The filled and active mill tailings ponds cells occupy most of the remainder of the photo. SOURCE: Elise A. Striz (USNRC) presentation at the Atlanta committee meeting.

F

Regulation of Effluent Releases

Effluent from nuclear facilities is permitted under regulations promulgated by the U.S. Nuclear Regulatory Commission, though it is controlled, monitored, and reported to authorities. These following requirements are intended to keep public exposures from radioactive effluent releases at levels that are as low as reasonably achievable (ALARA).

Title 10, Part 20 of the Code of Federal Regulations (10 CFR 20, Standards for Protection Against Radiation) establishes public dose limits for radioactive releases from nuclear plants. Specifically, Subpart D (Radiation Dose Limits for Individual Members of the Public) requires that nuclear plant licensees conduct operations so that:

- The total effective dose equivalent[1] to individual members of the public does not exceed 0.1 rem (1 mSv) in a year; and
- The dose in any unrestricted area[2] from external sources does not exceed 0.002 rem (0.02 mSv) in any one hour.

However, a licensee may apply for authorization to operate up to an annual dose limit of 0.5 rem (5 mSv) for an individual member of the public if there is a demonstrated need for the elevated exposures. However, there

[1]Total effective dose equivalent (TEDE) expresses the dose received by an individual in terms of a uniform whole-body dose, even though that actual dose may have been received by a particular organ or part of the body. The use of TEDE allows for comparisons of exposure risks for different kinds and levels of exposures.

[2]Unrestricted area is defined in 10 CFR 20.1003 as "an area, access to which is neither limited nor controlled by the licensee."

are additional requirements specified in 10 CFR 20.1301 that must be met by the licensee to obtain authorization for a higher dose limit.

To show compliance with these dose limits, licensees are required to survey radiation levels in unrestricted and controlled areas, as well as in the effluents released in these areas. The licensee must demonstrate that the total effective dose equivalent to the individual likely to receive the highest dose from the plant does not exceed the annual dose limit noted above; this demonstration can be made either by measurement or calculation. Alternatively, the licensee can demonstrate that the annual average concentrations of radioactive material released in airborne and liquid effluents at the boundary of the unrestricted area do not exceed radionuclide-specific values provided in the regulations,[3] and also that an individual continuously present in an unrestricted area would receive a dose not to exceed 0.002 rem (0.02 mSv) in an hour and 0.05 rem (0.5 mSv) in a year.

There are additional regulations on the control of effluent releases for nuclear power plants in 10 CFR 50. Part 50.34a (*Design objectives for equipment to control releases of radioactive material in effluents*) requires applicants for nuclear plant construction permits to estimate future releases for:

(i) The quantity of each of the principal radionuclides expected to be released annually to unrestricted areas in liquid effluents produced during normal reactor operations; and
(ii) The quantity of each of the principal radionuclides of the gases, halides, and particulates expected to be released annually to unrestricted areas in gaseous effluents produced during normal reactor operations.

Additionally, 10 CFR 50.36(a)(2) requires licensees to submit annual reports specifying the principal radionuclides released in liquid and gaseous effluents.

Part 50.36a (*Technical specifications on effluents from nuclear power reactors*) requires licensees to establish and follow procedures for the control of effluents. This Part also establishes an expectation that "the licensee will exert its best efforts to keep levels of radioactive material in effluents as low as is reasonably achievable."[4]

The release requirements for radioactive effluents are based on the calculated doses to members of the public from the effluents, and not on the total volume or type of radioactive material discharged. Thus, licensees have the discretion to control effluent releases in a manner that allows for

[3]These values are provided in Table 2 of Appendix B in 10 CFR 20.
[4]Appendix I in 10 CFR 50 establishes the numerical objectives for ALARA.

plant specific discharge streams, as well as the local setting of the plant. Compliance with 10 CFR 50.36a and Appendix I of 10 CFR 50 is established in a Licensee's radiological effluent release technical specifications, as based on dose calculations to a hypothetical maximally exposed member of the public living near the nuclear power plant.

Regulations promulgated by the U.S. Environmental Protection Agency place additional requirements on releases from all fuel-cycle facilities. The regulations in 40 CFR 190 (Environmental Radiation Protection Standards for Nuclear Power Operations), Subpart 10 (Standards for Normal Operations) place annual limits of 0.025 rem (0.25 mSv) to the whole body, 0.075 rem (0.75 mSv) to the thyroid, and 0.025 rem (0.25 mSv) to any other organ of any member of the public as the result of planned discharges of radioactive materials, excluding radon and its progeny, to the general environment from uranium fuel-cycle operations and of exposures to radiation from these operations.

G

Radiological Effluent Technical Specifications (RETS)

The U.S. Nuclear Regulatory Commission (USNRC) requires that operators of nuclear plants and fuel-cycle facilities monitor and report on releases of radioactive effluents. For nuclear plants, he monitoring and reporting system is specified in the Radiological Effluent Technical Specifications (RETS).

RETS requires the licensee to monitor effluent releases at every significant release point at the facility. Effluent monitoring consists of continuous measurements of some effluent streams; periodic measurement of radioactive particles trapped on filters, and measurement of samples from effluents released in batches. Detailed information about the RETS program for a given plant is contained in the licensee's Offsite Dose Calculational Manual (ODCM), which is part of an operator's application for a USNRC license. The USNRC also requires that the licensee participate in an Interlaboratory Comparison Program to ensure the accuracy and precision of the licensee's data and also to carry out computational checks, data validation activities, and audits by USNRC personnel.

Methods for estimating airborne and liquid effluent dispersions from nuclear plants are described in Regulatory Guides 1.111 (Methods for Estimating Atmospheric Transport and Dispersion of Gaseous Effluents in Routine Releases from Light-Water-Cooled Reactors) (USNRC, 1977a) and Regulatory Guide 1.113 (Estimating Aquatic Dispersion of Effluents from Accidental and Routine Reactor Releases for The Purpose of Implementing Appendix I) (USNRC, 1977b), whereas methods used to derive the radionuclide concentrations in foodstuffs from the air and water concentrations are described in Regulatory Guide 1.109 (Calculation of Annual Doses to Man

from Routine Releases of Reactor Effluents for the Purpose of Evaluating Compliance with 10 CFR Part 50, Appendix I) (USNRC, 1977c). Guidance to calculate the annual doses to humans from effluent releases from nuclear plants is also included in Regulatory Guide 1.109.

Regulatory Guide 4.16 (Monitoring and Reporting Radioactive Materials in Liquid and Gaseous Effluents from Nuclear Fuel-Cycle Facilities) indicates that estimates of exposures resulting from effluent releases from nuclear fuel-cycle facilities also should be calculated consistent with the applicable guidance in Regulatory Guide 1.109. Alternatively, nuclear facility licensees can use Guide 4.20 (Constraint on Releases of Airborne Radioactive Material to the Environment for Licensees Other than Power Reactors) for estimating exposures from airborne releases. Of course, the nuclides of interest for exposures from nuclear fuel-cycle facilities differ from those for nuclear plants (see Chapter 2). The use of U.S. Environmental Protection Agency-approved codes (e.g., COMPLY) is accepted by the USNRC and these codes are generally used by fuel-cycle facilities to demonstrate compliance with exposure limits. These codes are generally conservative and overestimate exposures. Since external exposures from fuel-cycle facilities are essentially negligible compared to internal exposures, current models available in the literature are entirely sufficient. Similarly, current models are also sufficient for direct radiation exposure from stored waste, tailings piles, and depleted-uranium canisters.

G.1 EFFLUENT MONITORING AT NUCLEAR PLANTS

Regulatory Guide 1.21 (Measuring, Evaluating, and Reporting Radioactive Material in Liquid and Gaseous Effluents and Solid Waste) provides regulatory guidance for sampling and analysis of effluents from USNRC-licensed nuclear plants. Guidance to plant licensees on sampling and analysis methods and frequencies are provided in NUREG-1301 for Pressurized Water Reactors and NUREG-1302 for Boiling Water Reactors. These documents contain guidance on:

- Effluent monitoring instrumentation: Locations of monitoring instrumentation with respect to plant effluent systems, minimum number of operable channels, and surveillance (inspection) requirements.
- Effluent monitoring: Sampling and analysis frequency, type of analysis, and detection limits.

Site-specific monitoring programs can deviate from the guidance in these NUREGs with appropriate justifications and approvals.

Regulatory Guide 1.21 recommends that licensees monitor all locations at the plant at which >1 percent of activity is discharged as:

- liquid effluent,
- noble gases into the atmosphere, or
- anything else into the atmosphere.

Title 10, Part 50 of the Code of Federal Regulations (10 CFR 50.36(a)(2)) requires licensees to report the principal radionuclides in effluent releases.

These locations are referred to as *significant release points* and include vents and stacks for airborne effluents and liquid waste discharge points for liquid effluents. Releases are assessed using a combination of sample analyses, radiation monitoring, and flow, tank level, and system pressure indications, as appropriate, to ensure that the amount of radioactive material is not underestimated.

Licensees are also required to monitor unplanned leaks and spills. If such leaks and spills result in offsite releases, then the magnitude of the releases must be estimated and reported to the USNRC along with the releases from routine operations. If the leak or spill occurs onsite, then a bounding analysis can be used to assess the potential offsite hazard.

Continuous effluent releases are typically monitored by measuring gross radioactivity with a continuously indicating radiation monitoring system such as a sodium iodide detector. These gross measurements can be used to activate alarms and terminate effluent releases if radioactivity levels exceed allowable limits. These continuous measurements are combined with analyses of physical samples (e.g., particulate materials trapped on filters or air samples) from the effluent stream to obtain quantitative estimates of the radionuclide concentrations in the effluent stream. Such samples are usually taken at specified frequencies, the value of which depends on the expected variability of radioactivity in the effluent stream.

Batch effluent releases are sampled prior to purging or venting. Certain radionuclides, referred to as "hard-to-detect" radionuclides (e.g., iron-55, strontium-89, and strontium-90), may be analyzed after the release takes place. "Continuously indicating" radiation monitoring equipment may be used during the release to verify the representativeness of the grab sample or to more fully characterize the release.

Table G.1 summarizes the guidance on sampling and analyzing airborne and liquid waste. The guidance specifies analyses type, minimum sampling frequencies, and lower limits of detection for each type of release. The guidance for pressurized-water reactors in NUREG-1301 are similar, but some of the specified sampling points are different owing to the different design of these plants. Table G.1 footnotes list the principal radionuclides that should be measured by the monitoring program.

TABLE G.1 Radioactive Airborne Waste Sampling and Analysis Program

	Release Type	Sampling Frequency	Minimum Analysis Frequency	Type of Activity Analysis	Lower Limit of Detection[a] (μCi/ml)
Airborne	Offgas treatment system	Monthly Grab sample	Monthly	Principal gamma emitters[b]	1×10^{-4}
	Containment purge or vent	Prior to each purge[c] Grab sample	Prior to each purge[c] Monthly	Principal gamma emitters[b] Tritium (oxide)	1×10^{-4} 1×10^{-6}
	Other airborne release points	Monthly[c,d] Grab sample	Monthly[c]	Principal gamma emitters[b] Tritium (oxide)	1×10^{-4} 1×10^{-6}
	All release types listed above	Continuous[e]	Weekly[f] Charcoal sample	Iodine-131	1×10^{-12}
		Continuous[e]	Weekly[f] Particulate sample	Principal gamma emitters[a]	1×10^{-11}
		Continuous[e]	Monthly Composite particulate analysis	Gross alpha	1×10^{-11}
		Continuous[e]	Quarterly Composite particulate sample	Strontium-89 Strontium-90	1×10^{-11}
		Continuous[e]	Noble gas monitor	Noble gases Gross beta or gamma	1×10^{-6}
Liquid	*Batch Waste*[g] *Release Tanks*	Each batch—completed prior to each release	Each batch—completed prior to each release	Principal gamma emitters[b]	5×10^{-7}
				I-131	1×10^{-6}
	a.	Each batch—completed prior to each release; at least one per 31 days	At least one per 31 days	Dissolved and entrained gases (gamma emitters)	1×10^{-5}

TABLE G.1 Continued

Release Type	Sampling Frequency	Minimum Analysis Frequency	Type of Activity Analysis	Lower Limit of Detection[a] (μCi/ml)
b.	Each batch—completed prior to each release	Composite[i]—at least one per 31 days	H-3	1×10^{-5}
			Gross alpha	1×10^{-7}
c.	Each batch—completed prior to each release	Composite—at least one per 92 days	Sr-89; Sr-90	5×10^{-8}
			Fe-55	1×10^{-6}
Continuous	Continuous[j]	Composite—at least one per 7 days	Principal gamma emitters	5×10^{-7}
			I-131	1×10^{-6}
a.	Grab sample—at least one per 31 days	At least one per 31 days	Dissolved and entrained gases (gamma emitters)	1×10^{-5}
b.	Continuous	Composite—at least one per 31 days	H-3	1×10^{-5}
			Gross alpha	1×10^{-7}
c.	Continuous	Composite—at least one per 92 days	Sr-89, Sr-90	5×10^{-8}
			Fe-55	1×10^{-6}

[a]The LLD is defined, for purposes of these controls, as the smallest concentration of radioactive material in a sample that will yield a net count, above system background, that will be detected with 95% probability with only 5% probability of falsely concluding that a blank observation represents a "real" signal.

[b]Includes Kr-87, Kr-88, Xe-133, Xe-133m, Xe-135, and Xe-138 in noble gas releases; Mn-54, Fe-59, Co-58, Co-60, Zn-65, Mo-99, I-131, Cs-134, Cs-137, Ce-141, and Ce-144 in iodine and particulate releases; other gamma peaks that are identifiable must also be analyzed and reported.

[c]Sampling and analysis shall also be performed following shutdown, startup, or a thermal power change exceeding 15 percent of rated thermal power within a 1-hour period.

[d]Tritium grab samples shall be taken at least once every 7 days from the ventilation exhaust from the spent fuel pool area whenever spent fuel is in the spent fuel pool.

continued

TABLE G.1 Continued

[e]Guidance concerning the sample flow rate. See Table 4.11-2 footnotes in NUREG-1302 for details.

[f]Detailed guidance concerning sampling. See Table 4.11-2 footnotes in NUREG-1302 for details.

[g]A batch release is the discharge of liquid wastes of a discrete volume. Prior to sampling for analyses, each batch shall be isolated, and then thoroughly mixed by a method described in the ODCM to assure representative sampling.

[h]The principal gamma emitters for which the Lower Limit Detection (LLD) control applies include the following radionuclides: Mn-54, Fe-59, Co-58, Co-60, Zn-65, Mo-99, Cs-134, Cs-137, and Ce-141. Ce-144 shall also be measured, but with an LLD of 5×10^{-6}. This list does not mean that only these nuclides are to be considered. Other gamma peaks that are identifiable, together with those of the above nuclides, shall also be analyzed and reported in the Semiannual Radioactive Effluent Release Report pursuant to Control 6.9.1.4 in the format outlined in Regulatory Guide 1.21, Appendix B, Revision 1, June 1974.

[i]A composite sample is one in which the quantity of liquid sampled is proportional to the quantity of liquid waste discharged and in which the method of sampling employed results in a specimen that is representative of the liquids released.

[j]A continuous release is the discharge of liquid wastes of a nondiscrete volume, e.g., from a volume of a system that has an input flow during the continuous release. To be representative of the quantities and concentrations of radioactive materials in liquid effluents, samples shall be collected continuously in proportion to the rate of flow of the effluent stream. Prior to analyses, all samples taken for the composite shall be thoroughly mixed in order for the composite sample to be representative of the effluent release.

SOURCE: NUREG-1302, Table 4.11-2.

G.2 EFFLUENT MONITORING AT FUEL-CYCLE FACILITIES

Requirements for monitoring effluent releases from front-end nuclear fuel-cycle facilities are contained in the following regulations:

- 10 CFR 40.65 (*Effluent Monitoring Reporting Requirements*) applies to "Part 40" fuel-cycle facilities. These include in situ leaching facilities, milling facilities, and uranium conversion and deconversion[1] facilities.
- 10 CFR 70.59 (*Effluent Monitoring Reporting Requirements*) applies to "Part 70" fuel-cycle facilities. These include nuclear fuel fabrication plants as well as laser enrichment and centrifuge enrichment plants.
- 10 CFR 76.35(g) (*Contents of an Application*) applies to "Part

[1]A new uranium deconversion and fluorine extraction processing facility is planned for construction near Hobbs, New Mexico. This facility will deconvert depleted uranium hexafluoride tails from the enrichment process into a uranium oxide waste product for eventual disposal and will recover fluorine for commercial resale.

76" fuel-cycle facilities. These are the Paducah and Portsmouth Gaseous Diffusion Plants. Because the plants are owned by the U.S. Department of Energy,[2] they are subject to the regulations promulgated by the U.S. Environmental Protection Agency in 40 CFR 61 (*National Emission Standards for Hazardous Air Pollutants*), Subpart H (*National Emission Standards for Emissions of Radionuclides Other Than Radon from Department of Energy Facilities*) and Subpart Q (National Emission Standard for Radon Emissions from Department of Energy Facilities).

G.2.1 Milling Facilities

Guidance specifically for milling facility effluent monitoring is provided in Regulatory Guide 4.14. This guide recommends that a program of soil, water, air, vegetation, food, and fish sampling and direct radiation monitoring be initiated at least 12 months prior to the construction of the milling facility. The guide also recommends that an operational monitoring program be conducted during construction and after the commencement of milling operations. The recommended operational monitoring program includes the following elements:

- Sampling and analysis for natural uranium, thorium-230, radium-226, and lead-210 particulates from facility stacks.
- Sampling and analysis for natural uranium, thorium-230, radium-226, and lead-210 particulates in air from three locations at or near the site boundaries in sectors that are expected to have the highest concentrations of airborne particulates; from one or more locations at the closest residence(s) or occupy-able structure(s); and from one control location.
- Sampling and analysis for radon gas at five or more locations that were used for air particulate sampling.
- Measurement of direct radiation at five or more locations that were used for air particulate sampling.

G.2.2 Other Fuel-Cycle Facilities

Guidance for monitoring programs at other front-end facilities (e.g., conversion, enrichment, fuel fabrication) is provided in Regulatory Guide 4.16. This guide recommends that licensees:

- Establish a sampling program that is sufficient to determine quanti-

[2]These U.S. government-owned plants are leased to USEC, a private corporation.

ties and average concentrations of radioactive material discharges from the facility.
- Use continuous monitoring methods for determining releases of airborne effluents from process systems that have particulate or airborne materials that can be easily dispersed.
- Use grab-sampling methods to confirm releases at points that are continuously monitored.

Guidance for uranium recovery monitoring programs can be found in Table 2 of Regulatory Guide 4.14. This guide recommends that licensees perform:

- Soil sampling and analysis at five or more locations that were used for air particulate sampling.
- Surface water and groundwater sampling and analysis.
- Periodic fish, food, and vegetation sampling and analysis, if available.
- Sediment sampling and analysis.

Requirements for conducting an effluent monitoring program at the U.S. Department of Energy-owned gaseous diffusion plants are provided in 40 CFR 61, Subpart H. This subpart requires radionuclide emission measurements to be made at all release points that have a potential to discharge radionuclides into the air in quantities that could cause an effective dose equivalent in excess of 0.1 mrem per year to any member of the public. Confirmatory measurements are required for other release points that have a potential to release radionuclides into the air. The subpart also contains specific requirements for measurement and analysis procedures using approved methods and for quality assurance.

REFERENCES

USNRC (U.S. Nuclear Regulatory Commission) (1977a). Regulatory Guide 1.111, Methods for Estimating Atmospheric Transport and Dispersion of Gaseous Effluents in Routine Releases from Light-Water-Cooled Reactors. Revision 1.

USNRC (1977b). Regulatory Guide 1.113. Estimating Aquatic Dispersion of Effluents from Accidental and Routine Reactor Releases for the Purpose of Implementing Appendix I.

USNRC (1977c). Regulatory Guide 1.109. Calculation of Doses to Man from Routine Releases of Reactor Effluents for the Purpose of Evaluating Compliance with 10 CFR Part 50, Appendix I. Revision 1. October 1977.

H

Radiological Environmental Monitoring Program (REMP)

Under federal regulations, all nuclear power plants have stringent environmental monitoring programs to ensure there are no negative effects from plant operations. The U.S. Nuclear Regulatory Commission (USNRC) requires licensees to begin these programs at nuclear plant sites at least 2 years before the plant starts operating. Because radiation is naturally present in the environment, the preoperational monitoring is designed to establish a baseline the company later will use to ensure that the plant's impact on the environment remains minimal. The USNRC requires nuclear plants to submit a report each year on the results of their monitoring programs.

The USNRC requires the operators of nuclear power plants to sample air at various locations in the vicinity of the plants to determine if releases are detectable in the environment off site. The environmental monitoring system is covered under the Radiological Environmental Monitoring Program (REMP): typically, measurements are made at five stations: three near the plant boundary in the direction of most likely wind transport; one in the vicinity of a community likely to have the greatest chance of exposure; and one at control location 15 to 30 km distant in the upwind direction of prevailing winds (NUREG 1301). Radioiodine is measured weekly and gross beta activity of particulates captured on filters is measured quarterly. Analyses to identify gamma-emitting radionuclides are done on composite samples weekly.

The results of a licensee's effluent release program, which provides estimates of the public health impact of the releases, and radiological environmental monitoring program must be reported annually to the USNRC. Both reports are available to the public via the USNRC website. Historical

reports are available electronically in the USNRC system from about 2000 to the present. Prior to that, reports are available only in microfiche.

For a waterborne exposure pathway a sampling and analysis program shown in Table H.1 is recommended.

The Radiological Effluent Technical Specifications (RETS) require that the licensee submit:

1. An annual radiological environmental monitoring report which is designed to assess the impact of radiological effluent releases into the environment; and
2. A Special Report within 30 days of discovery of the event if predetermined levels of radioactivity are exceeded.

The USNRC also requires that the licensee participate in an Interlaboratory Comparison Program to ensure the accuracy and precision of the licensee's data.

The REMP has allowed licensees significant flexibility to make changes to their programs without prior USNRC approval.[1] The historical trend has been to reduce the scope of the program as a result of continued nondetection of radioactivity.

[1]However, licensees must notify the USNRC of any changes, and the USNRC has regulatory recourse if the changes are not in accord with regulations.

TABLE H.1 Water Sampling and Analysis Recommendations

Sample	Number of Representative Samples and Sample Locations	Sampling and Collection Frequency	Type and Frequency of Analysis
Surface water	One sample upstream (Wa1), one sample downstream (Wa2)	Composite sample over 1-month period	Gamma isotopic analysis monthly; composite for tritium analysis quarterly
Groundwater	Samples from one or two sources (Wb1, Wb2) only if likely to be affected	Quarterly	Gamma isotopic and tritium analysis quarterly
Drinking water	One sample of each of on to three (Wc1–Wc3) of the nearest water supplies that could be affected by its discharge; one sample from a control location (Wc4)	composite sample over 2-week period when I-131 analysis is performed; monthly composite otherwise	1-131 analysis on each composite when the dose calculated for the consumption of the water is greater than 1 mrem per year. Composite for gross beta and gamma isotopic analyses monthly. Composite for tritium analysis quarterly.
Sediment from shoreline	One sample from downstream area with existing or potential recreational value (Wd1)	Semiannually	Gamma isotopic analysis semiannually

NOTES:
a. Gamma isotopic analysis means the Identification and quantification of gamma-emitting radionuclides that may be attributable to the effluents from the facility.
b. The "upstream sample" shall be taken at a distance beyond significant influence of the discharge. The "downstream" sample shall be taken in an area beyond but near the mixing zone. "Upstream" samples in an estuary must be taken far enough upstream to be beyond the plant influence. Saltwater shall be sampled only when the receiving water is utilized for recreational activities.
c. A composite sample is one in which the quantity (aliquot) of liquid sampled is proportional to the quantity of flowing liquid and in which the method of sampling employed results in a specimen that is representative of the liquid flow. In this program composite sample aliquots shall be collected at time intervals that are very short (e.g., hourly) relative to the compositing period (e.g., monthly) in order to ensure obtaining a representative sample.
d. Groundwater samples shall be taken when this source is tapped for drinking or irrigation purposes in areas where the hydraulic gradient or recharge properties are suitable for contamination.
SOURCE: Offsite Dose Calculation Manual Guidance: Standard Radiological Effluent Controls for PWRs, Generic Letter 89-01, Supplement No. 1, April 1991, U.S. NRC, NUREG-1301.

I

Radiation Dose Assessment

Under normal operating conditions, nuclear facilities release radioactive effluents in many physical and chemical forms (See Appendixes D and E). These effluents can travel through the environment in a number of physical pathways to expose individuals and populations surrounding the facilities. Individuals may be exposed to radiation from immersion in clouds of radioactive gases, inhalation of radioactive materials in the air, ingestion of radioactive materials from contaminated foods and liquids, and other less common pathways. Each pathway generates different patterns of whole-body and organ exposures, often with different time courses. For example:

- The immersion of an individual in a cloud of radioactive iodine generates an exposure pattern characteristic of external radiation—namely, absorbed doses delivered at various depths in tissues from penetrating radiation (e.g., gamma rays) as well as skin exposure due to finite-range charged particles (e.g., electrons from beta decay). These doses are relatively uniform with the exception of bone and red marrow doses, which can differ by as much as a factor of 2. These exposures persist only when the radioactive material is present.
- Alternatively, intakes of radioactive iodine by inhalation and ingestion can result in exposures of individual organs, most prominently the thyroid in the case of soluble forms of iodine. The organ doses can vary according to biokinetic properties of radioactive iodine.

As a result, organ-specific doses can vary significantly for different organs.

Organ absorbed doses for these many exposure pathways have been studied for decades and for most radionuclides. The absorbed dose to individual organs is well established and provided in a series of reports published by the International Commission on Radiological Protection (ICRP). ICRP recommendations address ingestion and inhalation scenarios.

U.S. Nuclear Regulatory Commission (USNRC) licensing activities for nuclear plants are based on the very simplistic dosimetry model reported in ICRP Publication 2 (1959). In this model, the concept of the *critical organ* is applied. The critical organ is defined as the organ, which can include the whole body, that is expected to receive the largest radiation dose. In contrast, current ICRP guidelines account for the exposure of all organs and tissues. Doses from intakes of radionuclides by individuals generally are much more accurately and comprehensively modeled under these guidelines.

Estimating the radiation exposure to individuals in the vicinity of a nuclear facility is a strong function of the type of facility, local conditions such as distances from effluent release points, and of course environmental conditions. Although there are wide variations in these conditions, estimating radiation exposures reduces to knowing effluent release patterns as a function of time, exposure pathways, and the quantity and type of radionuclide(s) released.

Some of this required information is quite complex. For example, to estimate radiation exposures from atmospheric release, one needs to know radionuclide quantities, concentrations, and release locations as a function of time, the local weather pattern also as a function of time, and any occupancy at appropriate locations surrounding the facility.

When the information discussed above is convolved with the aforementioned dosimetric models, individual and population absorbed doses can be estimated on an individual-by-individual basis. The reliability of these estimates will depend on the availability and quality of all the required input data.

I.1 EXTERNAL DOSES

External doses resulting from atmospheric releases of radioactive effluents consist of three components: (1) dose from airborne noble gases and fission (plus activation) products; (2) doses from radionuclides deposited on the ground or in water; and (3) dose due to direct exposure to radioactive material at the facility, including nitrogen-16 in turbine buildings (in boiling water reactor plants) and other radionuclides in stored wastes.

I.1.1 Dose from Airborne Noble Gases and Fission Products

Estimates of nuclide-specific ground-level activity concentrations in air at a particular direction and distance and annual and quarterly doses can be calculated as a function of distance and direction using accepted air dispersion methodologies that account for radioactive decay and plume depletion during transport, release height, and average annual (or longer) meteorology (wind speed, direction, atmospheric stability) as well as site-specific features such as terrain features. The organ dose resulting from immersion in air containing radioactive gases (sometimes referred to as a radioactive *plume*) at any location can then be calculated fairly accurately for each of the specific nuclides released and their specific gamma and beta emissions (Federal Guidance Report 12 [USEPA, 1993]).

The exposure rate from immersion in a plume of noble gases varies significantly with the composition of the gaseous cloud versus distance. The exposure rate from the various radioactive gases varies significantly due to large differences in the energies of their respective radiation emissions. As shown in Table I.1, the effective dose factors for short-lived emitters such as krypton-87 and 88 are significantly higher than that for longer-lived xenon-133, which comprises most of the airborne effluents from currently operating nuclear plants. However, because of the shorter half-lives of these radioisotopes, their relative contribution to doses to persons living farther downwind will be somewhat less than the relative effective dose factors shown in Table I.1.

I.1.2 Doses from Deposited Radionuclides

Calculations of external exposure and organ doses from particulate radioactive materials deposited on the ground are based on the same transport model used for estimating noble gas concentrations downwind and models for calculating dry and wet deposition and the dose rate per unit

TABLE I.1 Exposure Rate Dose Conversion Factors

Nuclide	Half-life	Effective Dose Factor (Sv Bq^{-1} s m^{-3})
Kr-87	76 min	4.0×10^{-14}
Kr-88	2.8 h	9.7×10^{-14}
Xe-133	5.2 d	1.3×10^{-15}
Xe-135	9.1 h	1.1×10^{-14}
Xe-135m	15 min	1.9×10^{-14}
Xe-138	14 min	5.5×10^{-14}

SOURCE: Effective dose factors from Federal Guidance Report 12.

activity concentration in soil of each nuclide. Recommended models for estimating doses from external radiation exposure are discussed in USNRC regulatory guides as well in guidance published by the U.S. Environmental Protection Agency (USEPA) and the National Council on Radiation Protection and Measurements (NCRP). Some of these models conservatively assume that the activity is deposited onto the surface of the ground (no ground roughness correction) and that no weathering occurs to reduce the integral exposure over time. Nevertheless, the estimated doses from nuclear plant effluents are a small fraction of those resulting from immersion in the plume of noble gases that are released from the plants, and they are almost always too low to be measured directly.

The exposure rate from radionuclides deposited onto the ground varies with the energy of the emissions. However, longer-lived nuclides can build up in the soil with time. Table I.2 shows conservative estimates of exposure in air per unit surface activity concentrations for selected radionuclides of importance in airborne effluents. The tabulated values are for a plane surface source. The exposure rates for a given activity concentration in the soil will decrease as the activity moves down into the soil profile over time as a result of rainfall and human activity. Because of the very low effluent rates and the diffusion of the airborne activity over a large area, only the longer-lived nuclides such as cesium-137 and cobolt-60 can potentially build up to activity levels high enough for the exposure rate to be distinguishable from even the temporal variations in terrestrial background levels at any site. Modern gamma-ray spectrometric techniques might allow the detection of very low levels of cobalt-60 in soil at close-in sites that might occur after many years of plant operation, but cesium-137 from the facility, even if present, would be undetectable because it is expected to be present in all soils from nuclear weapons testing fallout.

TABLE I.2 Exposure Rate per Unit Deposition Density

Nuclide	Half-life	Exposure Rate (μR/h per nCi/m^2)
I-131	8 d	0.0073
Cr-51	28 d	0.0006
Co-60	5 y	0.0432
Cs-134	2.1 y	0.0291
Cs-137	30 y	0.0107
Ba-140	12 d	0.0027

SOURCE: Beck (1980).

I.2 INTERNAL EXPOSURES

Calculation of radiation doses from internally deposited radionuclides is done by determining the spatial and temporal distribution of energy deposited in tissues and organs after intake. Generally, this requires knowledge of the distribution of sources and targets in space and time. The source is the radionuclide of concern, and the target is the biological entity considered most relevant for determining dose and risk. The choice of target can range from molecules and cells to organs and tissues to whole organisms. For radiation protection, the level of averaging of radiation dose has consistently been at the tissue or organ level.

Regardless of dosimetry system employed, the following information is needed:

- decay characteristics of the radionuclide,
- chemical and physical nature of the exposure material,
- intake route,
- solubility of the exposure material in vivo,
- tissue and organ distribution pattern in the body,
- retention times for the radionuclide in the various target tissues, and
- an appropriate anatomic or physiologic model of a human.

Taken together, this information allows both dose rate and dose patterns from intakes of radionuclides to be calculated.

For calculating internal doses resulting from the release of radionuclides from nuclear facilities, the USNRC continues to use dosimetry methods published by the International Commission on Radiological Protection in 1959 (commonly referred to as ICRP 2 methods) (ICRP, 1960). This is described in USNRC Regulatory Guide 1.109 (USNRC, 1977), which implements the guidance in Appendix I of 10 CFR Part 50. The ICRP 2 dosimetry model (ICRP, 1960) was developed primarily for providing radiation protection guidance for occupational environments, although recommendations for members of the public living in the neighborhood of controlled areas are also provided. However, the ICRP recommendations for the public did not take into account differences in dose limits between workers and members of the public, nor did they use different biokinetic models; thus, the differences in maximum permissible concentrations only reflect different exposure periods, that is, 40-hour weeks for workers versus 168-hour weeks for the public.

In general, the guidance protects workers by controlling the dose to the "critical organ," which is defined as that organ of the body that receives the highest dose or is the most radiosensitive organ receiving a significant dose from an intake of a given radionuclide." Through the use of the critical

organ and maximum permissible doses defined in ICRP 2, the risk to the individual is then controlled through the use of the Maximum Permissible Body Burden (q). This quantity is applied to specific exposure scenarios (e.g., chronic exposure for 168 hours per week for 50 years) and used with defined anatomic and physiologic parameters for ingestion and inhalation to yield Maximum Permissible Concentrations for a radionuclide in air (MPC_a) and water (MPC_w). Although the USNRC does not use the dose constraints proposed in ICRP 2, but rather those in 10 CFR 50 Appendix I, it still uses the ICRP 2 methodology for calculating doses to the maximally exposed member of the public.

The models used in ICRP 2 to define intakes from ingestion and inhalation exposure to radionuclides are very basic, reflecting the state of knowledge of the behavior of radionuclides at the time this methodology was issued. All physiologic parameters were provided for a Standard Man, and thus do not provide for individual variations in body size, intake, or metabolic rates. The Standard Man, which was defined at the Tripartite Conference in Chalk River (Warren et al.,1949), was designed to represent a typical or average adult who is exposed occupationally. Although the USNRC has modified the application of the Standard Man approach as applied to intake of radionuclides in effluents from nuclear plants, the essential features of Standard Man are described here for reference.

Water balance in Standard Man is defined in terms of food, fluids, and oxidation by-products intake and excretion rates, as shown Table I.3. Other physiologic parameters were also defined (Table I.4). These values allow the calculation of intakes from ingestion and inhalation in terms of the quantity of radionuclides in food, water, and air. In addition, a separate empirical model was defined for intakes of particulates by inhalation (Table I.5). Although it was recognized that the retention of particulate matter depends on many factors, such as the size, shape, and density of the particles, as well as their chemical form and whether the person is a nose or mouth breather, ICRP indicated that specific data were lacking, and therefore the distribution and fate of inhaled particles could adequately be described as in Table I.5.

Thus, there is no particle size dependence in this model, which strongly affects both total and regional deposition in the respiratory tract. Additionally, the fate of material, whether being cleared via feces as particles or absorbed to blood, was described simply in terms of whether the inhaled particles were relatively soluble or not. For the soluble compounds, the 25 percent deposited in lungs was assumed to translocate to blood within the first 24 hours after inhalation. For the insoluble particles, half of the 25 percent that deposited in the lung was assumed to be eliminated from lung and swallowed in the first 24 hours after inhalation; this meant that 62.5 percent of the materials deposited in the upper respiratory tract (URT)

TABLE I.3 Intake and Excretion of Standard Man (Water Balance)

Intake (cm^3/d)		Excretion (cm^3/d)	
Food	1000	Urine	1400
Fluids	1200	Sweat	600
Oxidation	300	From lungs	300
		Feces	200
TOTAL	2500		2500

SOURCE: Warren et al. (1949).

TABLE I.4 Other Physiologic Parameters for Standard Man

Vital capacity of lung (male)	3-4 L
Vital capacity of lung (female)	2-3 L
Air inhaled per 24-h day	2×10^7 cm^3 d^{-1}
Interchange area of lungs	50 m^2
Area of upper respiratory tract, trachea and bronchi	20 m^2
Total surface area of respiratory tract	70 m^2
Total water in body	4.3×10^4 g
Average lifespan of man	70 y

SOURCE: Warren et al. (1949).

TABLE I.5 Behavior of Inhaled Particulates in the Respiratory Tract of Standard Man

Distribution	Readily Soluble Compounds	Other Compounds (insoluble)
Exhaled	25	25
Deposited in URT and swallowed	50	50
Deposited in lungs (LRT)	25 (to blood)	25 (12.5% swallowed; 12.5% to blood)

SOURCE: Warren et al. (1949).

and lower respiratory tract (LRT) was removed by mucociliary clearance, swallowed, and subsequently would be excreted via feces. The remaining 12.5 percent of the amount deposited in LRT was absorbed to blood with a 120-day half-time.

To calculate the absorbed doses, the retention and fate of a radionuclide taken into the body by ingestion or inhalation had to be described for individual radionuclides once they reached the blood. To do this for most of the radionuclides, particularly those for which the bone and GI tract were

not the critical organs, a simple exponential model was assumed as default. This was expressed by the equation:

$$qf_2 = P(1 - e^{-\lambda t})/\lambda \quad (1)$$

where

qf_2 = amount of the radionuclide in the critical body organ (Ci)
f_2 = fraction of radionuclide in the critical organ to that in total body
λ = effective decay constant = 0.693/T
T = effective half-time $((T_r T_b)/(T_r + T_b))$ (days)
T_r = radiological half-time (days)
T_b = biological half-time (days)
T = period of exposure (for occupational exposure, t = 50 years)
P = rate of uptake of the radionuclide by the critical body organ (μCi d^{-1})

The quantities and their radionuclide-specific values needed to calculate the absorbed and rem doses were provided in Table 12 of ICRP 2 (1960) and included the reference organ for dose calculation; the physical, biological, and effective half-times; the fraction of ingested radionuclide that reached the blood (f_1); the critical organ fraction (f_2); and the fractions reaching the critical or reference organ from water (f_w) or air (f_a).

ICRP dosimetry models have been improved markedly since the release of ICRP 2, and the models used in ICRP 2 have been replaced by more current dosimetry models. These models have been designed to calculate age-dependent dose coefficients (dose per unit intake) for members of the public. These include doses from ingestion (ICRP, 1989, 1993) and inhalation (ICRP, 1995a,b), doses to the embryo and fetus from radionuclide intakes by the mother (ICRP, 2001), doses to infants from ingestion of mothers' milk (ICRP, 2004), a new respiratory tract dosimetry model (ICRP, 1994), and an alimentary tract dosimetry model (ICRP, 2006). Also contained in the above documents are radioelement-specific biokinetic models that describe the systemic tissue and organ uptake and retention of radionuclides once they have reached the blood. These systemic models are coupled with the appropriate intake model (ingestion, inhalation) and a dosimetric model that calculates the dose to all target organs and tissues per radionuclide decay to obtain exposure-specific dose coefficients.

I.2.1 ICRP Models to Support an Epidemiologic Study

The first internal dosimetry system was published in 1959 (ICRP, 1960) and has generally been replaced sequentially by the ICRP 30-based sys-

tem (ICRP, 1979), which was focused on occupational workers; the ICRP 56-based system (ICRP, 1989), which related to members of the public; and the current system outlined (but not described) in ICRP 103 (2007). Although the ICRP 2 system is still implemented by USNRC for performance compliance dosimetry of radioactive effluent releases from nuclear plants in some ways, it has been described previously. Rather, the more current ICRP (ICRP 56+) dosimetry system (ICRP, 1990, 1992, 1995a,b), which may be most applicable for calculating doses for an epidemiologic study, is described below.

Over the past 50 years, a substantial increase in knowledge about radionuclide metabolism and biokinetics in humans and experimental animal models has occurred and has provided a basis for the development of more realistic biokinetic models of radionuclide uptake and retention, particularly at the organ and tissue level. This plus better understanding of the disposition of inhaled and ingested radionuclides both in the deposition and systemic organs has further provided the basis for significant improvements in internal dosimetry and modeling.

The current generation of ICRP models for internal dosimetry of intakes of radionuclides by the public offers the following advantages:

1. More complete radionuclide physical decay schemes;
2. Improved physical anthropometric models, which allow more accurate calculation of absorbed fractions of radiation resulting from the distribution of radionuclides in various source organs;
3. Better description of organ-level biokinetics of radionuclides that reach the blood and circulation (systemic models);
4. More anatomically and physiologically accurate model of the respiratory tract together with improved description of deposition, retention, translocation, and excretion of inhaled radionuclides;
5. More anatomically and physiologically accurate model of the alimentary tract, which extends the number of tissues modeled and includes a better understanding of the biokinetics within the alimentary tract and relative radiosensitivities of the various target tissues within the alimentary tract;
6. Improved age-dependent modeling of radionuclide biokinetics in humans of different ages;
7. Addition of radionuclide biokinetic modeling of the uptake and retention of radionuclides in the embryo and fetus from intakes by the mother, both before and during pregnancy;
8. Inclusion of a milk pathway of intake for newborns who are nursed by mothers who have had intakes of radionuclides.

These improvements in modeling have necessarily come at the expense

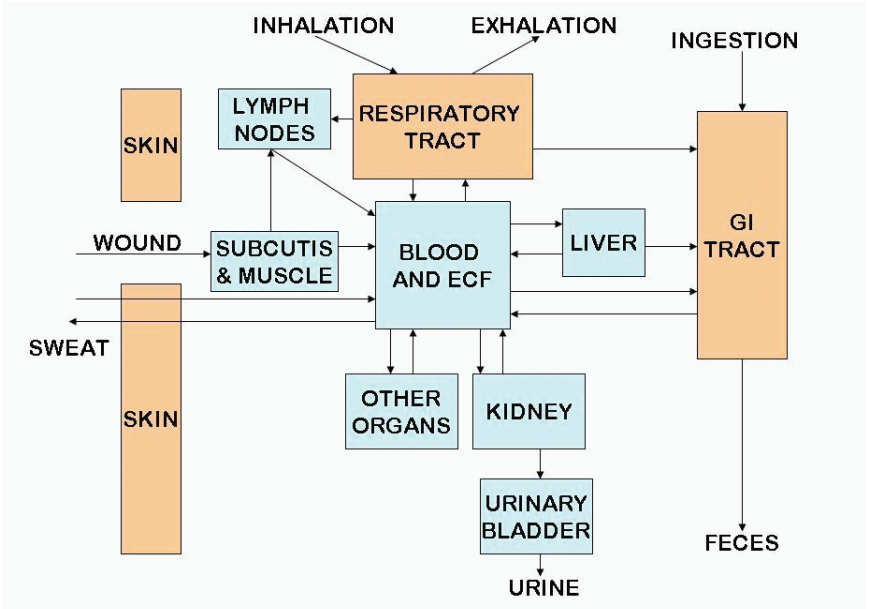

FIGURE I.1 Example compartmental model representation of a radionuclide biokinetic model SOURCE: Adapted from ICRP (1997).

of having much more complicated models, which require the use of computer software to calculate radiation doses.

Figure I.1 shows an example of the type of biokinetic models being used by ICRP in its current series of dosimetry models. Among the general features of the modeling approach are (1) allowance of intakes by ingestion, inhalation, wounds, and transcutaneous absorption across intact skin. (2) Compartments in brown are tissue sites of entry of radionuclide into the body. These may be described in more detail in other models, e.g., a respiratory tract model. (3) Compartments in blue are systemic deposition sites that communicate directly with blood. (4) Current models allow for recycling between compartment, which can be a more accurate representation of the flow of radionuclides between compartments. Different levels of subcompartments within a tissue compartment can also be used when multicomponent retention patterns are needed. For example, multiple compartments have been employed for the liver in the plutonium systemic biokinetic model of ICRP publication 67 (1992).

The complexity of a given set of biokinetic models depends on the tissues and organs that are the principal deposition sites for a given radionuclide, and are therefore usually at greater risk of receiving radiation

dose. When designing the models, a full range of radionuclides is considered. Additionally, the list of tissues and organs is also influenced by those considered to be at risk of biological effects from radiation. Since this list includes irradiation from both external and internal sources, essentially all tissues and organs of the body are considered. ICRP Publication 103 (2007) lists the following organs: red bone marrow, colon, lung, stomach, breast, gonads, urinary bladder, esophagus, liver, thyroid, bone surface, brain, salivary glands, skin, adrenals, extrathoracic region of the respiratory tract (head airways), gall bladder, heart, kidneys, lymph nodes, muscle, oral mucosa, pancreas, prostate, small intestine, spleen, thymus, and uterus/cervix.

To calculate organ-specific absorbed doses, two biokinetic models are required. The first model is used to relate radionuclide concentration in air or solid media (food or water) to intake. This is done using the Human Respiratory Tract Model (HRTM) (ICRP, 1994) or the Human Alimentary Tract Model (HATM) (ICRP, 2006) for inhalation and ingestion, respectively. The second model, which is radioelement specific, is the systemic biokinetic model, which describes in detail the spatial and temporal distribution of a radionuclide once it has reached the blood. These models are coupled mathematically so that the number of disintegrations occurring in the various organs and tissues of interest can be calculated and used together with an appropriate anatomical model and physical dosimetry model to calculate the pattern of deposition of energy in the organs.

I.2.1.1 Human Respiratory Tract Model

The HRTM is actually a second-generation replacement of the simple respiratory tract model published in ICRP 2 (1960); it replaced the intermediate model published in ICRP Publication 30 (1979). The HRTM was developed by ICRP over a 10-year period and represented the state-of-the-art knowledge about the behavior of inhaled particles and gases in the human respiratory tract. In this model, the respiratory tract is subdivided into five anatomical compartments (Figure I.2), ranging from two extrathoracic regions (ET_1, ET_2), to bronchi, bronchioles, and the parenchymal region of the lung (AI). Regional deposition efficiencies were calculated for these anatomic compartments for particle sizes ranging from 0.001 μm through 100 μm. As part of the definition of these anatomic regions, different geometric constructs were created for each of the regions. The critical cell populations at risk to stochastic health effects were purported to exist within these geometrically prescribed subregions so that only energy deposited in these subregions is used to calculate the absorbed dose to that anatomic compartment. Additionally, each of the anatomic compartments has been risk-weighted by apportioning the radiation detriment to the dif-

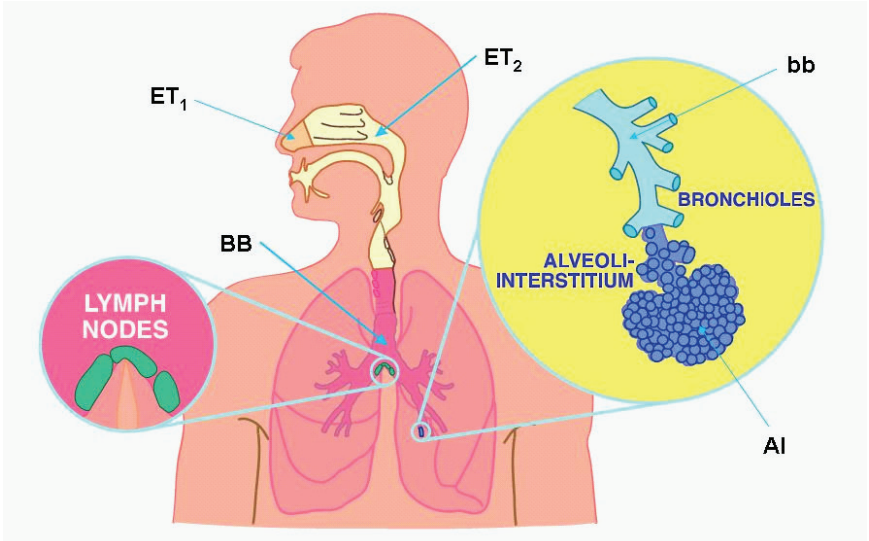

FIGURE I.2 HRTM anatomic model. SOURCE: ICRP (1994).

ferent compartments based on human and experimental data regarding the frequency of different types of respiratory cancer (ICRP, 1994).

The fate of deposited radionuclides in the respiratory tract[1] is modeled by considering clearance based on three pathways: (1) mucociliary clearance from both the head airways and the lung leading to swallowing of the cleared material, and subsequent excretion into feces or absorption through the GI tract to blood; (2) clearance of particles through the interstitium leading to uptake in the lymph nodes that drain the various regions of the respiratory tract; and (3) dissolution of the radionuclide on or near the airways of the respiratory tract followed by either local binding to tissue constituents (less likely and applicable to only a few radioelements, e.g., plutonium and americium) or absorption to blood (most likely). These processes are modeled mathematically as competing pathways and are dependent on the physical and chemical properties of the inhaled radionuclide.

The HRTM is an age-dependent dosimetry model whose morphometric and physiologic characteristics have been defined for reference ages of 3 months; 1, 5, 10, and 15 years; adult; and all for both genders. As such, age-dependent dose coefficients (dose per unit intake) have been published for members of the public in ICRP Publication 71 (1995). The model also

[1]It is important to note that not all inhaled material is deposited in the respiratory tract. About 40-50 percent of most inhaled material is exhaled without depositing anywhere.

has examined the role of personal factors such as smoking and respiratory disease on deposition and clearance of inhaled particles, both of which affect the dose coefficients.

Because of the complexity of the HRTM, several software programs have been developed and implement the model for use in dose assessment and bioassay interpretation (e.g., Bertelli et al., 2008; Jarvis and Birchall, 1994).

I.2.1.2 Human Alimentary Tract Model

The HATM (ICRP, 2006) is a biokinetic and dosimetric model of the human alimentary tract that replaces the previous GI tract model of ICRP Publication 30 (1979). This expanded model is applicable to all radionuclide intakes by children and adults. As such, it provides age-dependent parameter values for the dimensions of the alimentary tract as well as age- and gender-dependent transit rates. Although the default is for absorption of radionuclide to blood to occur in the small intestine, the model does allow for absorption to occur in other regions. The HATM also allows for local binding of radionuclides to the structures of the various regions of the alimentary tract, thus allowing for calculation of radiation dose to subcompartments of the HATM.

Figure I.3 illustrates the compartmental model for the HATM. It depicts the entire alimentary tract from oral cavity to rectosigmoid colon. Input occurs into the oral cavity via ingestion and clearance of inhaled deposited radionuclides from the respiratory tract into the esophagus (the HATM was designed to be consistent with the HRTM in terms of structure, clearance processes, and dosimetric modeling). The movement of contents through the alimentary tract is sequential, and the transit rates are modeled by first-order exponential processes. It was recognized that modeling transit in this way was a considerable simplification, but by indexing the emptying half-time to the reported mean transit times of material through a given segment, a reasonable estimate of the transit rate was obtained, which allowed dose calculation to be done in a straightforward way. The bulk of the material ends up in feces. It should be noted that the behavior of a given radionuclide in terms of absorption to blood versus excretion in the bulk material depends on its physical and chemical form.

Absorption of solutes, including radionuclides, to blood is allowed through the walls of all HAT organs, but the default is that absorption is limited through the small intestine. Deposition and retention of radionuclides in teeth, oral mucosa, and GI tract walls allows these tissues to be both sources for radionuclide retention and targets for calculating radiation absorbed doses, although these tissues are targets in any case. Typically, the geometry identified for dose calculations in the various tissues of the HAT

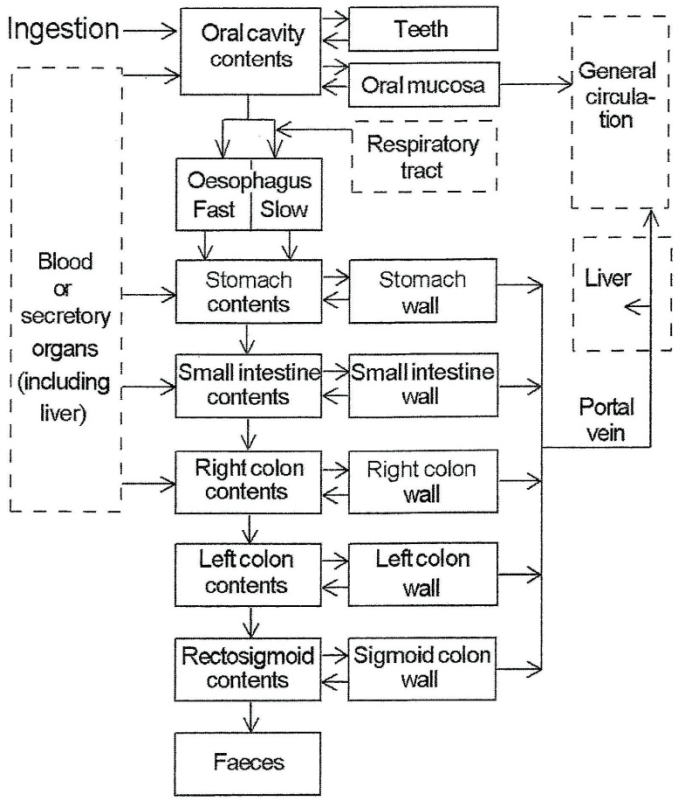

FIGURE I.3 Compartments of the HATM. SOURCE: ICRP (2006).

comprise the layers of epithelial cells contained in those tissues. This is due to the fact that most of the cancers linked to radiation in the alimentary tract are epithelial in origin.

Transit time parameter values have been provided for different types of ingested materials (solids, caloric and noncaloric liquids, and total diet) and for subjects having ages of 3 months, 1 year, 5-15 year, adult male, and adult female.

I.2.1.3 Systemic Biokinetic Models

The development of radioelement-specific systemic biokinetic models is ongoing within the committees and task groups of the ICRP. Presently the only relatively complete set of systemic models, i.e., for all radioele-

ments, is that contained in the ICRP 30 series of publications, which apply only to adult workers. From the structural point of view, these models are nonrecycling models whose physiologic relevance is often questionable, but they are useful for their intended purpose of designing radiation protection programs as well as interpretation of human bioassay data.

A smaller number of age-dependent, recycling biokinetic models have been published by ICRP, namely for the alkaline earths and lead, calcium, plutonium, neptunium, americium, and curium. Age-specific biokinetic data have been developed for the most common radionuclides (isotopes of hydrogen, carbon, zirconium, niobium, ruthenium, iodine, cesium, cerium, plutonium, americium, and neptunium) for a total of 31 radioelements (ICRP, 1989). Recycling models continue to be developed by ICRP for other radioelements, but these may not become available during the timeframe needed for this project. Nevertheless, ICRP in its publication 72 (1995a) added 60 other radioelements to its age-specific dose coefficient database by using the nonrecycling models of ICRP publication 30 together with age-specific organ masses.

ICRP publication 72 (ICRP, 1995a) provides age-specific dose coefficients that are needed for the purposes of epidemiologic study dosimetry. Although ICRP states "[b]ecause changes in biokinetics are considered with age and have not been considered fully, these additional dose coefficients should be used with care for assessing doses to infants and children," the dose coefficients nevertheless provide the best set of age-dependent dose coefficients available. Additionally equivalent doses have been provided in electronic form by ICRP on CD.

I.2.1.4 Comparison of USNRC and Recent ICRP Dose Coefficients

In Table I.6, the inhalation dose coefficients from USNRC Regulatory Guides are compared with those derived from recent ICRP publications (ICRP, 1995b) for radionuclides commonly encountered in effluent releases from nuclear power plants. It is clear that very large differences are observed between the two sets of data. USNRC dose coefficients are derived from USNRC Regulatory Guide 1.109 (1977), Table E-7. The tabulated values were converted to Sv/Bq from mrem/pCi by dividing the latter by 3700.

ICRP dose coefficients were calculated using the AIDE dose assessment software (Bertelli et al., 2008). All coefficients were calculated assuming an aerosol particle size of 1.5 μm AMAD, inhaled by a male worker. Solubility classes (F or M) are shown in the radionuclide column. The systemic models were derived either from ICRP 56 or ICRP 67.

TABLE I.6 Comparison of Inhalation Dose Coefficients (Committed Dose per Unit Uptake) Derived from USNRC and ICRP Dosimetric Approaches for Adults (Sv/Bq intake)

Radionuclide	Model	Bone	Liver	Lung	GI-LLI[a]
H-3	USNRC	—	4.27E-11	4.27E-11	4.27E-11
H-3 (F)	ICRP (56)	8.14E-12	8.14E-12	8.17E-12	8.61E-12
Co-60	USNRC	—	3.89E-10	2.02E-7	9.62E-9
Co-60 (M)	ICRP (67)	3.72E-9	8.11E-9	4.89E-8	5.56E-9
Sr-90	USNRC	3.35E-6	—	3.24E-7	2.43E-8
Sr-90 (F)	ICRP (56)	3.63E-7	6.58E-10	7.12E-10	1.21E-8
Ru-106	USNRC	2.28E-9	—	3.16E-7	3.08E-8
Ru-106 (M)	ICRP (30)	2.71E-9	2.82E-9	1.76E-7	2.55E-8
I-131	USNRC	8.54E-10	1.21E-9	4.03E-7 (Thyroid)	2.12E-10
I-131 (F)	ICRP (56)	5.95E-11	2.01E-11	1.76E-7 (Thyroid)	4.04E-11
Cs-137	USNRC	1.61E-8	2.10E-8	2.54E-9	2.85E-10
Cs-137 (F)	ICRP (56)	5.60E-9	5.52E-9	5.19E-9	6.76E-9

[a]GI-LLI, gastrointestinal tract–lower large intestine.

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J

Modeling Incidence and Mortality Data in an Ecologic Study

A starting point for ecologic modeling of cancer rate is Poisson regression for rates and counts. In classic Poisson regression, a count, N_i of some data item (e.g., a count of childhood leukemias) is modeled as a Poisson random variable, with a probability distribution function equal to:

$$\frac{\mu_i^N e^{-\mu}}{N_i!} \tag{1}$$

Here μ_i is the expected value of N_i (i.e., the number of cancer incident cases or deaths in a particular geographic unit expected from broad population rates, typically cross-classified by other variables such as age, gender, and race/ethnicity with i as the identifying index). In Poisson regression the mean, μ_i, is unknown but assumed to be a function of known covariates. For example, in generalized linear regression (McCullagh and Nelder, 1989) a model for the mean involves a covariate vector $X_i = (X_{i1},X_{i2},\ldots,X_{ip})^T$ observed for each i. These X_i may be either continuous variables, such as dose, or indicator variables, indicating levels taken by categorical variables. The generalized linear model for μ_i is of form:

$$g(\mu_i) = \alpha_1 X_{i1} + \alpha_2 X_{i2} + \ldots + \alpha_p X_{ip} = X_i^T \alpha \tag{2}$$

Here $\alpha = (\alpha_1,\alpha_2,\ldots,\alpha_p)^T$ and α_1 is the regression coefficient relating covariate value X_{i1} to the mean μ_i, α_2 relates X_{i2} to μ_i, etc. Here g is a link

function, for example when (as is often the case) g is the log function then the model is equivalent to:

$$\mu_i = \exp(\alpha_1 X_{i1} + \alpha_2 X_{i2} + \ldots + \alpha_p X_{ip}) \quad (3)$$

When N_i counts the number of events observed over a period of time, t_i (years), for a known number of individuals, k_i , then the person-years of observation, py_i, defined as $t_i k_i$ will be made a part of model as:

$$\mu_i = \exp(\alpha_1 X_2 + \alpha_2 X_{i2} + \ldots + \alpha_p X_{ip} + \log(py_i)) = py_i \exp(X_i^T \alpha) \quad (4)$$

so that the mean of the counts is proportional to the person-years of observation multiplied by the effect of covariates.

In the setting described here N_i would correspond to a single entry in a cross-tabulation of events (death due to or incidence of a particular cancer) by each geographical unit, and by gender, race, age, calendar time, and any other relevant variable known (from the cancer registry) about the cases. For each cell in the table the number of events and person-years at risk, py_i, are required to be calculated (see discussion below) in addition the variable of interest, dose D_i, and other covariates available for each geographical unit (i.e., indices of social economic status) are required for each table entry i.

A variation on model , known as the linear excess relative risk (ERR) model, is commonly used in radiation epidemiology. The linear ERR model incorporates dose in the model for μ_i as:

$$py_i \exp(X_i^T \alpha)(1 + \beta D_i) \quad (5)$$

Here $py_i \exp(X_i^T \alpha)$ is the background rate of disease (for unexposed cells), multiplied by person-years at risk, and the ERR parameter β is the excess relative risk associated with dose or dose surrogate D_i. Much more complex models can be considered and software for generalized Poisson regression is available (Epicure, Hirosoft Software, Seattle, Washington). The background rate of disease is allowed to vary depending on race, gender, age, and calendar time (to allow for disease rates to differ by age and for age-specific rates to vary by calendar year, for example). Covariates in ecologic models are not individual covariates, but instead are summaries obtained for each geographical unit, although these can also vary in time; for example, we may have information about some socioeconomic variables at the level of census tract and these variables may change with time over the period of interest. Such variables are incorporated by including (categories of) calendar time as a cross-classification variable.

J.1 DOSE AND DOSE SURROGATES

The presumed effect on risk of the dose or dose surrogate variable, D_i, in model is much simpler (involving only the ERR parameter, β) than the model for the background risk (involving many additional parameters α); however, D_i will also vary in time. For example, if D_i is cumulative dose from a particular nearby plant for representative individuals, then D_i for all census tracts near that plant would be zero until the start of operations of that plant and would accumulate in time during operation. Even treatment of much simpler dose surrogates (exposed or not exposed according to distance) should reflect startup times of each plant or facility.

Other factors may also need to be considered in the calculation of D_i; for example, if it is known that a population around a particular plant or facility has been highly mobile over the period of exposure then it would be desirable to incorporate that mobility into the calculation of D_i in order to approximate the average cumulative dose to the individuals in each census tract for each time period considered. If distance is to be used as a dose surrogate then time-weighted distance could also be considered.

J.2 PERSON-YEAR CALCULATIONS

Another key issue in Poisson modeling is to adequately approximate person-years of exposure to some hazard, py_i, as well as counting the number of events N_i. For each cell in the tabulation of events cross-classified by geographical unit, race, age, and calendar time, census data are required in order to determine the population size for each table entry, i.e., the whole population must be classified according to these same variables. Data from each decennial census must be interpolated to the out years. The accuracy of person-year approximations affect the modeling of N_i using Poisson regression and inaccuracies in estimation of person-years is one (among many) reasons to assume that the Poisson model may not adequately capture the variability of the observed counts N_i.

J.3 OVERDISPERSION

It is likely that observed counts N_i will depart from the Poisson regression distribution in a way that must be adequately accommodated when fitting the regression models such as (5). If a random variable is distributed according to the Poisson distribution then the variance of N_i is also equal to μ_i. However, there are good reasons why we expect that the actual variability of N_i will be greater than that predicted by Poisson distribution. For example, as mentioned above, for the out years at least, the population size and hence person-years will not be known exactly. Even more importantly, however, is that other known and unknown risk factors that influence dis-

ease occurrence are not being accounted for in the variables that are used in the ecologic regression. Even if those risk factors are completely independent of distance or dose from a plant or facility then they will still increase the dispersion of N_i while leaving the model for the mean unaffected. Ignoring overdispersion will lead to underestimation of standard errors of the estimates of the regression parameters, including those of most interest (i.e., β). The treatment of overdispersion in Poisson regression models has been considered by a number of authors (Liu and Pierce, 1993; McCullagh and Nelder, 1989; Moore, 1986). A simple and usually effective approach (McCullagh and Nelder, 1989) to solving this problem is to fit the means model using Poisson regression but then to estimate an overdispersion term σ^2 with $\sigma^2 > 1$ so that the variance of N_i is estimated to be equal to $\sigma^2\mu_i$. Inference about the significance of the parameters of interest (i.e., β) is performed after adjusting the usual standard error estimates (assuming the Poisson model). A method of moments approaches for fitting this and similar models is described by Moore (1986). More generally, the "sandwich estimator" of Zeger and Liang (1986) can be used to compute variances of the parameter estimates that adequately reflect the variability of the counts.

The overall approach described above relates observed disease rates to distance or other dose surrogates in a systemic way, i.e., addressing the question of whether or not disease risk appears to be associated with proximity to a nuclear facility, or to other dose surrogates, averaging over all the facilities. For some common cancers it will be possible to consider site-specific analyses, i.e., whether proximity to a specific facility or plant is associated with risk. Such analyses are subject to concerns about multiple comparisons (as described in the main text) but may also be particularly sensitive to the problem of overdispersion described above. If one uses an uncorrected test, i.e., a test based upon the assumption that the Poisson distribution holds exactly, then it is very likely that there will be some sites where for some cancers proximity is "significantly" associated with risk, but for which the inference differs greatly depending upon whether or not purely Poisson variation of counts is assumed. The estimation of overdispersion terms $\sigma^2 > 1$ (or providing other treatment of overdispersion as in a random effects analysis) is crucial in order to avoid overinterpretation of random fluctuation that simply are greater in magnitude (due to unmeasured characteristics affecting disease risk) than expected under the Poisson model. These problems appear in many different kinds of settings and have been described by a number of different authors (Efron, 1992). Modeling of both the mean (as in equation (5) of the appendix) and the variance of counts will be essential in ensuring that unrealistic inference from fitting these models is avoided; this is true both for the overall analysis of risk in relation to plant proximity and especially for site-specific analyses.

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K

Letter Template to State Cancer Registries

Re: Committee on Analysis of Cancer Risks in Populations Near Nuclear Facilities: Request for information on cancer incidence and mortality data availability and release

Dear Director:

I am writing to request information on your state's cancer incidence and mortality data availability and the release of criteria of those data. Information about the accessibility of data will be used in support of a study being carried out by the National Academy of Sciences (NAS) *Committee on Analysis of Cancer Risks in Populations near Nuclear Facilities*, at the request of the U.S. Nuclear Regulatory Commission. The Statement of Task related to the study is attached.

I would appreciate information on the following issues:

- Year from which complete cancer incidence and cancer mortality data for your State are available.
- Year from which registry records include address.
- Year from which census tract of reported cases is available.
- Year from which the data are available electronically.
- Assessments of the quality of the incidence and mortality data over time (i.e. completeness of ascertainment of cancer cases, completeness and accuracy of data variables requested).
- The registry's efforts to retrieve missing or incomplete information (e.g. missing age, or details about the site of cancer).
- Whether there is active follow-up of reported cancer cases through

surveillance programs or other means to ascertain survival, length of follow-up (e.g., 5 years, lifetime), and completeness of follow-up.
- Whether there is passive follow-up of reported cancer cases through linkage of cancer registry reports with death records to update vital status, and completeness.
- Whether year and place of birth of diseased/deceased individuals are available
- Procedures for request and release of cancer incidence and cancer mortality data at the address and census tract level, including associated costs and estimated time frames.
- References to any publications about registry operations or findings.
- Any other information you think might be relevant to a national study of cancer incidence and mortality among persons living near nuclear facilities.

I have also addressed the request for cancer mortality data information to *[Name]* from the Office of Vital Statistics. However, I understand that cancer registries can often facilitate access to the mortality data; therefore I hope that you will be able to provide me with the requested information.

Any information you provide to me by November 15, 2011 will be considered in support of the study. Please note that all written information that you provide will be included in the Public Access File for this study.

If you have questions about this request or would prefer to respond by telephone I would be happy to contact you. I can be reached by email (kcrowley@nas.edu) or telephone (202-334-3066).

Sincerely,

Kevin Crowley
Study Director

L

Letter Template to State Vital Statistics Offices

Re: Committee on Analysis of Cancer Risks in Populations near Nuclear Facilities: Request for information on availability and release of birth records and mechanisms of linkage with cancer registries and death records.

Dear Director:

I am writing to request information about your office's availability of birth records and the release criteria of those data for research purposes. Information about the accessibility of data will be used in support of a study being carried out by the National Academy of Sciences *Committee on Analysis of Cancer Risks in Populations near Nuclear Facilities*, at the request of the U.S. Nuclear Regulatory Commission. The Statement of Task related to the study is attached.

I would appreciate information on the following issues:

Census tract level:

- Year from which aggregated birth records data are available; release criteria for those data, and variables that can be released (e.g., age, race/ethnicity, gender, date of birth, number of siblings of index child).

Individual birth level:

- Year from which individual birth records are available, release criteria for those records, and variables that can be released (e.g., age, race/ethnicity, gender, date of birth, number of siblings, address and telephone number of parents at birth of the index child).

Policies and mechanisms:

- For review and approval of projects aiming to link birth, cancer incidence, and mortality data within the state and outside the state; matching criteria on which linkage is based.
- For contact of individuals identified via birth records, especially minors and/or their families.

We would also like to know about additional information that your office may collect for individual births, such as birth defects reported after the birth certificate is filed, and any other information you think might be relevant to a national study of pediatric cancer incidence within a birth cohort of children living near nuclear facilities.

Any information you provide to us by December 15, 2011 will be considered in support of the study. Please note that all written information that you provide will be included in the Public Access File for this study.

If you have questions about this request or would prefer to respond by phone I would be happy to contact you. I can be reached at email: kcrowley@nas.edu or phone: 202 334 3066.

Sincerely,

Kevin Crowley
Study Director

M

Letter Template to Departments of Public Health

Re: Committee on Analysis of Cancer Risks in Populations Near Nuclear Facilities: Request for Information on Public Concerns Related to Nuclear Power Plants and Other Nuclear Facilities

Dear Director:

I am writing to request information from your Department on reported public health concerns related to living near a nuclear power plant or other nuclear facilities. This information will be used in support of a study being carried out by the National Academy of Sciences (NAS) *Committee on Analysis of Cancer Risks in Populations near Nuclear Facilities* at the request of the U.S. Nuclear Regulatory Commission. The Statement of Task and background information related to the study are provided in the attachment.

I would appreciate information related to the Statement of Task and in particular on the following issues if they apply to your Department:

- Reports from members of the public on health concerns or suspected health effects related to nuclear power plants or nuclear fuel-cycle facilities in their communities.
- Reports from physicians or other health care providers concerning suspected disease clusters that could be due to the releases from these facilities.
- Assessments of cancer risks in association with nuclear facilities that were carried out by your Department.
- Other individual or organized activities that have been undertaken

by your Department in response to environmental monitoring or health surveillance programs.

- Interactions between your Department and communities around nuclear facilities to solicit feedback on potential health concerns.

Any information you provide to us by September 15, 2011, will be considered in support of the study. Please note that all written information that you provide will be included in the Public Access File for this study.

If you have questions about this request or would prefer to respond by phone I would be happy to contact you. I can be reached at email: kcrowley@nas.edu or phone: (202)-334-3066.

Sincerely,

Kevin Crowley
Study Director

N

Glossary

Alternative hypothesis: the hypothesis that observations are influenced by some nonrandom cause; contrast to null hypothesis.

Analytical study: a study designed to examine associations often concerned with measuring the effect of a risk factor; contrast to descriptive study.

Association: the relationship between two or more events, characteristics or other variables; does not necessarily imply cause and effect.

Absolute risk: in the context of a disease such as cancer is the observed or calculated probability that a person will develop a disease over a certain period of time, as contrasted with the relative risk.

Background radiation: ionizing radiation to which a person is exposed from natural sources, such as terrestrial radiation, cosmic radiation, and naturally occurring radionuclides deposited in the body.

Becquerel (Bq): the international (SI) special name for the unit of activity; one Bq is equal to one disintegration per second, or 2.7×10^{-11} curies (Ci).

Bias: tendency for an estimate to over- or underpredict an actual event due to a systematic error in an epidemiologic study.

Biological plausibility: the criterion that an observed association could be causal based on existing biological knowledge.

Biomarker: a substance or molecular/cellular event that is used as an indicator of a specific biologic state and which can link a specific environmental exposure to a health outcome.

Carcinogenesis: the process by which normal cells are transformed into cancer cells.

Case-control study: the epidemiologic observation of a group of persons with a disease of interest and a group of persons without the disease; cases and controls are compared for the frequency of the factor that is believed to be associated with the disease.

Causality: the relationship between an event (e.g., an exposure) and a second event (e.g., the disease) in which the second event is explained as a consequence of the first.

Census: the enumeration of an entire population that includes demographic information.

Census tract: a geographic area for which details on population structure are tabulated at a given census. Census tracts typically contain 1,200 to 8,000 people (with a target of 4,000 people).

Centroid: the geographic or population center for a geographic unit.

Classification of diseases: arrangement of diseases into categories based on shared characteristics such as body site that they occur, etiology, histology, and others.

Cluster: a grouping of health related events that are related in time, space, or both.

Cohort study: the epidemiologic observation of a group of persons with the exposure hypothesized to be associated with a disease of interest and a group of persons without the exposure; exposed and unexposed persons are often followed with time until the disease of interest develops and the frequency of disease occurrence by exposure is calculated.

Cold shutdown: a state of a nuclear reactor in which it is deemed subcritical and its coolant system is at atmospheric pressure and at a temperature less than 200 °F.

Confounder: a variable that is associated with both an exposure of interest and disease of interest and may result in statistically false cause or prevent detection of a cause-effect relationship between the exposure and outcome of interest.

Confidence intervals: the computed range with a particular confidence level, commonly set up at 95 percent, intended to give the assurance that if a statistical model is correct, the true value of the parameter (for example risk estimation) is within the range indicated; if the 95% CI range does not include 1, then the estimated risk is significantly different from that of a comparison group.

Correlation: a statistical measurement of the relationship between two variables. Correlation can be positive (as one variable goes up, the other variable goes up), or negative (as one variable goes up, the other variable goes down).

Covariate: a variable that is associated with the outcome of interest. For example, in a study of cancer risks, covariates of interest may be age, race/ethnicity, socioeconomic status, smoking status, and others.

Curie (Ci): a special name for the unit of radioactivity equal to 37 billion decays per second.

Decommission: removal of a nuclear facility from service.

Descriptive study: a study concerned with reporting the existing distribution of variables, e.g., cancer registry data analyses that often occur in ecologic studies; contrast to analytical study.

Distribution: the frequency of the values or categories of a measurement made on a population. For example, the age distribution of a population may be summarized as how many people in this population are 0-15, 16-25, 26-45 years of age, and so on.

Dose dose-rate effectiveness factor (DDREF): a factor by which the effect caused by a specific dose or dose rate of radiation changes at low doses or dose rates.

Ecologic fallacy: error in inference associated with ecologic studies due to extrapolating correlations observed at the group level to individuals; e.g., it has been shown that countries with high dietary fat intake have high incidence of breast cancer (the fallacy would be to infer from this observation alone that it is the individuals that have a high fat diet are those that develop breast cancer).

Ecologic study: a study in which the unit of analysis is a population or group (countries, states, counties, communities) and not individuals.

Effluent: solid, liquid, or gaseous release from a nuclear facility.

Epidemiology: the study of the distribution of diseases and other health-related conditions in populations and the application of this study to address health.

Excess risk: an estimate of the amount of risk due to the exposure of interest when the effects of other risk factors are removed. Can be relative or absolute risk.

Experimental study: a study in which the conditions are being directed by the investigator, e.g., a clinical trial in which patients are separated in two groups where some receive a new drug and some receive a placebo.

Follow-up: observation over a period of time of an individual or a population to retrieve new information and record changes in the health status.

Geocoding: the process of finding geographic coordinates (often expressed as latitude and longitude) from other geographic data such as address.

Gray (Gy): the international (SI) special name for the unit of absorbed dose; one Gy is equal to 1 joule per kilogram, or 100 rad.

Hazard: an act or phenomenon (e.g., ionizing radiation) that has the potential to produce harm or other undesirable consequences to humans or what they value (NCRP Report No. 139).

Half-life: the time required for half the atoms of a radioactive isotope to decay.

Healthy worker effect: the notion that an individual must be relatively healthy to be employable in a workforce; therefore, both disease and mortality rates are typically lower among workers than in the general population. Within the workforce studies, healthier workers are more likely to stay employed for longer periods of time compared to the relatively unhealthy workers which would have the shortest duration of employment.

Incidence: the number of persons that have developed a disease of interest in a specified population in a specific period of time.

Information bias: a flaw in estimating risk because of the difference in quality or accuracy of information collected for comparison groups.

Latency period: the lag time between exposure to a disease-causing agent and clinical recognition of disease. In terms of cancer due to exposure to radiation, the concept of *minimum* latency period is important and is often considered to be 2 years for leukemia and 10 years for solid cancers.

Lifetime risk: the risk to an individual that a given health effect or disease such as cancer will occur, without consideration of time elapsed since exposure.

Matching: the process during epidemiologic study design of making comparison groups similar to one or more extraneous factors so that the factor of interest is examined by eliminating the "noise" of other factors.

Misclassification: the erroneous attribution of a value into a category other than that it should be assigned.

Mortality rate: the number of deaths from all causes or a specific cause in a specified time period.

Multiple comparison: a problem in detection of a likely false positive association due to chance alone that arises when too many comparisons are made.

Multivariate analysis: a method used to study the effect of variation of many variables simultaneously.

Nested case-control study: a case-control study in which the study subjects are selected from a cohort study; presents a number of advantages over case-control studies, notably less inherited bias.

Null hypothesis: the hypothesis that one variable and another variable are not associated, e.g., a risk factor and a disease; in statistics, equivalent to *test hypothesis*, which the investigator will reject or accept based on available data; contrast to alternative hypothesis.

Observational study: a study in which the investigator does not have control of the conditions, but observes and reports information as nature takes its course.

Odds ratio (OR): the ratio of the odds of an event occurring in one group to the odds of the event occurring in a comparison group.

Population mixing hypothesis: proposes that childhood leukemia can be caused by a yet unidentified infectious agent transmitted due to the influx of people into rural areas where susceptible individuals are more prevalent than the average results in epidemics of this infection.

Prevalence: the number of people with a disease in a given population at a designated time; often used to describe incidence rate.

Prospective study: a cohort study that follows individuals that differ with respect to a factor of interest over time.

***P* (probability) value:** a measure of the compatibility of data with the null hypothesis; traditionally, $P < 0.05$ is considered sufficiently unlikely for the association to have occurred by chance and justifies the designation "statistically significant."

Radiation: the energy that comes from a source and travels through some matter or through space. Two types of radiation are commonly differentiated in the way they interact with matter: *ionizing* and nonionizing radiation. Ionizing radiation, which includes alpha particles, beta particles, gamma rays and x-rays, and neutrons, is considerably more energetic compared to nonionizing radiation such as that found in microwaves. In general, ionizing radiation is far more harmful to living organisms per unit of energy deposited than nonionizing radiation, since it has the potential to cause DNA damage and consequently cancer.

Radiation exposure: the absorption of ionizing radiation by an object; this absorption can impact health.

Radioactivity: the property or characteristic of an unstable atomic nucleus to spontaneously transform with the emission of energy in the form of radiation.

Relative risk (RR): in the context of a disease such as cancer is the probability of the disease occurring in an exposed group relative to the probability occurring in a nonexposed group.

Rad: special name for the unit of absorbed radiation dose; one rad is equivalent to 1/100 Gy.

Reference group: the group to which the population under study is compared.

Release: a discharge to the environment of radioactive materials, either during normal operations or due to an accident.

Rem: special name for the unit of radiation dose equivalent; the product of absorbed dose (measured in rads) and a weighting factor which accounts for biological damage caused by radiation (1 rem = 1/100 Sv).

Retrospective study: a study in which past exposures related to past or current disease is explored; can be case-control or cohort in design.

Risk assessment: An analysis of the potential adverse impacts of an event (e.g., releases of radioactive material from a nuclear facility) on the health or well-being of an individual or population. Risk assessment is a process by which information or experience concerning causes and effects under a set of circumstances is integrated with the extent of those circumstances to quantify or otherwise describe risk (NCRP Report No. 139).

Risk communication: an interactive process of exchange of information and opinion among individuals, groups, and institutions; often involves multiple messages about the nature of risk or expressing concerns, opinions, or reactions to risk messages or to legal and institutional arrangements for risk management.[1]

Risk management: The process by which results of risk assessments are integrated with other information (e.g., results of cost-benefit analysis, judgments about acceptable risk, and other societal concerns) (NCRP Report No. 139).

Sample size: the number of individuals selected from a population to be the subjects of an epidemiologic study.

Selection bias: a flaw in estimating real risk because of systematic differences in characteristics of those that participate in the study and those that do not.

Sievert (Sv): the international (SI) special name for the unit of dose equivalent radiation measured in J/kg, calculated by multiplying the absorbed dose (in Gy) with a weighting factor; 1 Sv = 100 rem.

[1]NRC (National Research Council) (1989). Improving Risk Communication. Washington, DC: National Academy Press.

Standardization: method for removing the effect of potential confounders such as age, gender, race from risk estimations.

Standardized incidence rate (SIR): the ratio of incident cases observed in the study group or population in a time period to the number of expected deaths if the study population has the mortality experience of the standard population.

Standardized mortality rate (SMR): the ratio of deaths observed in the study group or population in a time period to the number of expected deaths if the study population has the mortality experience of the standard population.

Standby mode: nuclear facilities available for operation but not currently operating.

Statistical power: the probability that a test will reject a null hypothesis when the hypothesis is actually false.

Statistical significance: refers to a result that is unlikely to be caused by chance; see "*P* (probability) value."

Stratification: the process of separating a sample into categories according to a specific criterion, e.g., age, gender, smoking status.

Susceptibility: the risk of becoming afflicted by something that can impact health.

Temporality: the issue associated with specific study designs (e.g., cross-sectional studies, case-control studies) that makes it difficult to understand if exposure or disease came first.

Uncertainty: Lack of sureness or confidence in predictions of models or results of measurements (NCRP Report No. 158).

O

Acronyms

ABCC	Atomic Bomb Casualty Commission
ACS	American Community Survey
ADAMS	Agencywide Documents Access and Management System
AHS	Adult Health Study
ALARA	As Low As (is) Reasonably Achievable
ALL	Acute Lymphocytic Leukemia
AML	Acute Myeloid Leukemia
AREER	Annual Radioactive Effluent Release Reports
BEIR	Biologic Effects of Ionizing Radiation
BRFSS	Behavioral Risk Factor Surveillance System
BWR	Boiling-Water Reactor
CANUPIS	Childhood Cancer and Nuclear Power Plants in Switzerland
CCRN	Childhood Cancer Research Network
CDC	Centers for Disease Control and Prevention
CFR	Code of Federal Regulations
CEDE	Committed Effective Dose Equivalent
CI	Confidence Interval
CLL	Chronic Lymphocytic Leukemia
CML	Chronic Myeloid Leukemia
CNS	Central Nervous System
COG	Children's Oncology Group
COMARE	Committee on Medical Aspects of Radiation in the Environment

CRDA	Cancer Registry Data Access
CT	Computed Tomography
DEP	Department of Environmental Protection
DDREF	Dose and Dose Rate Effectiveness Factor
DMV	Department of Motor Vehicles
ECLIS	European Childhood Leukemia-Lymphoma Incidence Study
ERR	Excess Relative Risk
GCCR	German Childhood Cancer Registry
GIS	Geographic Information System
GPI	Groundwater Protection Initiative
GU	Geographic Unit
HATM	Human Alimentary Tract Model
HIPAA	Health Insurance Portability and Accountability Act
HPIC	High-Pressure Ionization Chambers
HRTM	Human Respiratory Tract Model
IAEA	International Atomic Energy Agency
IARC	International Agency for Research on Cancer
ICD	International Classification of Diseases
ICRP	International Commission on Radiological Protection
ICRU	International Commission on Radiation Unites
IRB	Institutional Research Board
KiKK	*Kinderkrebs in der Umgebung von Kernkraftwerken*
LET	Linear Energy Transfer
LNT	Linear No-Threshold
LSS	Life Span Study
MCL	Maximum Contaminant Level
MDL	Minimum Detection Limit
MEI	Maximally Exposed Individual
MPC	Maximum Permissible Concentration
NAACCR	North American Association of Central Cancer Registries
NAS	National Academy of Sciences
NCCDPHP	National Center for Chronic Disease Prevention and Health Promotion
NCI	National Cancer Institute

NCHS	National Center for Health Statistics
NCRP	National Council on Radiation Protection and Measurements
NDI	National Death Index
NEI	Nuclear Energy Institute
NFS	Nuclear Fuel Services
NHAMCS	National Hospital Ambulatory Medical Care Survey
NHANES	National Health and Nutrition Examination Survey
NHCS	National Hospital Care Survey
NHDS	National Hospital Discharge Survey
NHIS	National Health Interview Survey
NIH	National Institutes of Health
NPCR	National Program of Cancer Registries
NPP	Nuclear Power Plant
ODCM	Offsite Dose Calculational Manual
OMB	Office of Management and Budget
OR	Odds Ratio
ORAU	Oak Ridge Associated Universities
PNL	Pacific Northwest Laboratory
PNNL	Pacific Northwest National Laboratory
PRA	Paperwork Reduction Act
PWR	Pressurized-Water Reactor
QA	Quality Assurance
QF	Quality Factor
RDD	Random-Digit Dialing
REMP	Radiological Environmental Monitoring Program
REF	Radiation Effectiveness Factors
RERF	Radiation Effects Research Foundation
RETS	Radiological Effluent Technical Specifications
RR	Relative Risk
SEER	Surveillance, Epidemiology and End Results
SIR	Standardized Incidence Rate
SMR	Standardized Mortality Rate
SSN	Social Security Number
TEDE	Total Effective Dose Equivalent
TLD	Thermoluminescent Dosimeter
TRI	Toxics Release Inventory

UNSCEAR	United Nations Scientific Committee on the Effects of Atomic Radiation
USCS	United States Cancer Statistics
USDOE	U.S. Department of Energy
USEPA	U.S. Environmental Protection Agency
USNRC	U.S. Nuclear Regulatory Commission
USPS	U.S. Postal Service
WECARE	Women's Environment, Cancer, and Radiation Epidemiology
WHO	World Health Organization
ZCTA	ZIP Code Tabulation Area