CHAPTER 25

TOXIC EFFECTS OF RADIATION AND RADIOACTIVE MATERIALS

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INTRODUCTION

Among all the branches of toxicology, ionizing radiation provides the most quantitative estimates of health detriments for humans. Five large studies provide data on the health effects of radiation on people. These effects include those due to external x-rays and gamma-ray radiation and internal alpha radioactivity. The studies encompass radium exposures, including those sustained by radium dial painters, atom bomb survivors, patients irradiated with x-rays for ankylosing spondylitis, children irradiated with x-rays for tinea capitis (ringworm), and uranium miners exposed to radon and its short-lived daughter products. The only health effect subsequent to radiation exposure seen with statistical significance to date is cancer. The various types and the quantitative risks are described in subsequent sections.

All the studies provide a consistent picture of the risk of exposure to ionizing radiation. There are sufficient details in the studies of atom bomb, occupational, and medical exposures to estimate the risk from lifelong low-level environmental exposure. Natural background radiation is substantial, and only within the past two decades has the extent of the radiation insult to the global population from natural radiation and radioactivity been appreciated.

BASIC RADIATION CONCEPTS

The four main types of radiation are due to alpha particles, electrons (negatively charged beta particles or positively charged positrons), gamma rays, and x-rays. An atom can decay to a prod-

uct element through the loss of a heavy (mass = 4) charged (+2) alpha particle (He^{+2}) that consists of two protons and two neutrons. An atom can decay by loss of a negatively or positively charged electron (beta particle or positron). Gamma radiation results when the nucleus releases excess energy, usually after an alpha, beta, or positron transition. X-rays occur whenever an inner-shell orbital electron is removed and rearrangement of the atomic electrons results, with the release of the element's characteristic x-ray energy.

There are several excellent textbooks describing the details of radiologic physics (Evans, 1955; Andrews, 1974; Turner, 1986, Cember 1996).

Energy

Alpha particles and beta rays (or positrons) have kinetic energy as a result of their motion. The energy is equal to

$$E = 1/2\ mV^2 \qquad (1)$$

where m = mass of the particle
V = velocity of the particle

Alpha particles have a low velocity compared with the speed of light, and calculations of alpha particle energy do not require any corrections for relativity. Most beta particles (or positrons) have high velocity, and the basic expression must be corrected for their increased relativistic mass (the rest mass of the electron is 0.511 MeV). The total energy is equal to

$$E = 0.511/(1 - v^2/c^2) + 0.511 \qquad (2)$$

where v = velocity of the beta particle
c = speed of light

Gamma rays and x-rays are pure electromagnetic radiation with energy equal to

$$E = hv \qquad (3)$$

where h = Planck's constant (6.626×10^{-34} J s)
v = frequency of radiation

The conventional energy units for ionizing radiation are the electron volt (eV) or multiples of this basic unit, kiloelectron volts (keV) and million electron volts (MeV). The conversion to the international system of units, the Système Internationale (SI), is currently taking place in many countries, and the more fundamental energy unit of the Joule (J) is slowly replacing the older unit. The relationship is

$$1\ \text{eV} = 1.6 \times 10^{-19}\ \text{J}$$

Authoritative tables of nuclear data such as those of Browne and Firestone (1986) contain the older but more widely accepted units of MeV for energy.

Alpha Particles

Alpha particles are helium nuclei (consisting of two protons and two neutrons), with a charge of +2, that are ejected from the nucleus of an atom. When an alpha particle loses energy, slows to the velocity of a gas atom, and acquires two electrons from the vast sea of free electrons present in most media, it becomes part of the normal background helium in the environment. All helium in nature is the result of alpha particle decay. The formula for alpha decay is

$$\begin{matrix} A & A-4 \\ X \rightarrow & Y + He^{2+} + \text{gamma} + Q_\alpha \\ Z & Z-2 \end{matrix}$$

where Z = atomic number
A = atomic weight

The energy available in this decay is Q_α and is equal to the mass difference of the parent and the two products. The energy is shared among the particles and the gamma ray if one is present.

An example of alpha decay is given by the natural radionuclide radium (^{226}Ra):

$$\begin{matrix} 226 & 222 \\ Ra \rightarrow & Rn + \text{alpha (5.2 MeV)} \\ 86 & 84 \end{matrix}$$

The energy of alpha particles for most emitters lies in the range of 4 to 8 MeV. More energetic alpha particles exist but are seen only in very short-lived emitters such as those formed by reactions occurring in particle accelerators. These particles are not considered in this chapter.

Although there may be several alpha particles with very similar energy emitted by a particular element such as radium, each particular alpha particle is monoenergetic, i.e., no continuous spectrum of energies exists, only discrete energies.

Beta Particles, Positrons, and Electron Capture

Beta particle decay occurs when a neutron in the nucleus of an element is effectively transformed into a proton and an electron. Subsequent ejection of the electron occurs, and the maximum energy of the beta particle equals the mass difference between the parent and the product nuclei. A gamma ray may also be present to share the energy, Q_β:

$$\begin{matrix} A & A \\ X \rightarrow & Y + \text{beta} + Q_\beta \\ Z & Z+1 \end{matrix}$$

An example of beta decay is given by the natural radionuclide lead (^{210}Pb):

$$\begin{matrix} 210 & 210 \\ Pb \rightarrow & Bi + \text{beta (0.015 MeV)} + \text{gamma (0.046 MeV)} \\ 82 & 83 \end{matrix}$$

Unlike alpha particles in alpha decay, in which each alpha particle is monoenergetic, beta particles are emitted with a continuous spectrum of energy from zero to the maximum energy available for the transition. The reason for this is that the total available energy is shared in each decay or transition by two particles: the beta particle and an antineutrino. The total energy released in each

transition is constant, but the observed beta particles then appear as a spectrum. The residual energy is carried away by the antineutrino, which is a particle with essentially zero mass and charge that cannot be observed without extraordinarily complex instrumentation. The beta particle, by contrast, is readily observed with conventional nuclear counting equipment.

Positron emission is similar to beta particle emission but results from the effective nucleon transformation of a proton to a neutron plus a positively charged electron. The atomic number decreases rather than increases, as it does in beta decay.

An example of positron decay is given by the natural radionuclide copper (^{64}Cu), which decays by beta emission 41 percent of the time, positron emission 19 percent of the time, and electron capture 40 percent of the time:

$$^{64}_{29}\text{Cu} \rightarrow {}^{64}_{28}\text{Ni} + \text{positron (0.66 MeV)} \qquad 19 \text{ percent}$$

$$^{64}_{29}\text{Cu} \rightarrow {}^{64}_{30}\text{Zn} + \text{beta (0.57 MeV)} \qquad 41 \text{ percent}$$

$$^{64}_{29}\text{Cu} \rightarrow {}^{64}_{28}\text{Ni} \text{ electron capture} \qquad 40 \text{ percent}$$

The energy of the positron appears as a continuous spectrum, similar to that in beta decay, where the total energy available for decay is again shared between the positron and a neutrino. In the case of positron emission, the maximum energy of the emitted particle is the mass difference of the parent and product nuclide minus the energy needed to create two electron masses (1.02 MeV), whereas the maximum energy of the beta particle is the mass difference itself. This happens because in beta decay, the increase in the number of orbital electrons resulting from the increase in atomic number of the product nucleus cancels the mass of the electron lost in emitting the beta particle. This does not happen in positron decay, and there is an orbital electron lost as a result of the decrease in atomic number of the product and the loss of the electron mass in positron emission.

Electron capture competes with positron decay, and the resulting product nucleus is the same nuclide. In electron capture, an orbiting electron is acquired by the nucleus, and the transformation of a proton plus the electron to form a neutron takes place. In some cases the energy available is released as a gamma-ray photon, but this is not necessary, and a monoenergetic neutrino may be emitted. If the 1.02 MeV required for positron decay is not available, positron decay is not kinetically possible and electron capture is the only mode observed.

Gamma-Ray (Photon) Emission

Gamma-ray emission is not a primary process except in rare instances, but it occurs in combination with alpha, beta, or positron emission or electron capture. Whenever the ejected particle does not utilize all the available energy for decay, the nucleus contains the excess energy and is in an excited state. The excess energy is released as photon or gamma-ray emission coincident with the ejection of the particle.

One of the rare instances of pure gamma-ray emission is technetium 99m (^{99m}Tc), which has a 6.0-h half-life and is widely used in diagnostic medicine for various organ scans. Its decay product, ^{99}Tc, has a very long half life (2.13×10^5 years), and as all ^{99}Tc is ultimately released to the environment, a background of this nuclide is emerging.

$$^{99m}_{43}\text{Tc} \rightarrow {}^{99}_{43}\text{Tc} + \text{gamma (0.14 MeV)}$$

In many cases, the photon will not actually be emitted by the nucleus but the excess excitation energy will be transferred to an orbital electron. This electron is then ejected as a monoenergetic particle with energy equal to that of the photon minus the binding energy of the orbital electron. This process is known as internal conversion. In tables of nuclear data such as those of Browne and Firestone (1986), the ratio of the conversion process to the photon is given as e/ν. For example, the e/ν ratio for ^{99m}Tc is 0.11, and therefore the photon is emitted 90 percent of the time and the conversion electron is emitted 10 percent of the time.

INTERACTION OF RADIATION WITH MATTER

Ionizing radiation, by definition, loses energy when passing through matter by producing ion pairs (an electron and a positively charged atom residue). A fraction of the energy loss raises atomic electrons to an excited state. The average energy needed to produce an ion pair is given the notation W and is numerically equal to 33.85 eV. This energy is roughly two times the ionization potential of most gases or other elements because it includes the energy lost in the excitation process. It is not clear what role the excitation plays, for example, in damage to targets in the cellular DNA. Ionization, by contrast, can break bonds in DNA, causing strand breaks and easily understood damage.

All particles and rays interact through their charge or field with atomic or free electrons in the medium through which they are passing. There is no interaction with the atomic nucleus except at energies above about 8 MeV, which is required for interactions that break apart the nucleus (spallation). Very high energy cosmic-ray particles, for example, produce ^{3}H, ^{7}Be, ^{14}C, and ^{22}Na in the upper atmosphere by spallation of atmospheric oxygen and nitrogen.

Alpha and beta particles and gamma rays lose energy by ionization and excitation in somewhat different ways, as described in the following sections.

Alpha Particles

The alpha particle is a heavy charged particle with a mass that is 7300 times that of the electrons with which it interacts. A massive particle interacting with a small particle has the interesting property that it can give a maximum velocity during energy transfer to the small particle of only two times the initial velocity of the heavy particle. In terms of the maximum energy that can be transferred per interaction, this is

$$E_{(\text{maximum electron})} = 4/7300\ E_{(\text{alpha particle})} \qquad (4)$$

Although alpha particles can lose perhaps 10 to 20 percent of their energy in traveling 10 μm in tissue (1 cm in air), each interaction can impart only the small energy, given in the maximum, in Eq. (4). Thus, alpha particles are characterized by a high energy loss per unit path length and a high ionization density along the track length. This is called a *high linear-energy-transfer* (LET) or *high-LET* particle.

Hans Bethe (1953) derived an exact expression for the energy loss in matter, dE/dx or stopping power, with later modifications added by Bloch and others. For alpha energies between 0.2 and 10 MeV, the Bethe-Bloch expression can be simplified to

$$dE/dx = 3.8 \times 10^{-25}\ C\ NZ/E \ln\{548\ E/I\}\ \text{MeV} / \mu\text{m}^{-1} \quad (5)$$

where N = number of atoms cm^{-3} in medium
Z = atomic number of medium
I = ionization potential of medium
E = energy of alpha particle
C = charge correction for alpha particles with energy below 1.6 MeV

A simple rule of thumb derived by Bloch may be used to estimate the ionization potential of a compound or element,

$$I = 10(Z) \quad (6)$$

or the Bragg additivity rule (Attix et al., 1968) may be used for compounds when the individual values of ionization potential for the elements are available. A tabulation of values of ionization potential is given in ICRU 37 (ICRU, 1984), and the stopping power in all elements has been calculated in ICRU 49 (ICRU, 1993) and by Ziegler (Ziegler, 1977).

When alpha particles are near the end of their range, the charge is not constant at +2 but can be +1 or even zero as the particle acquires or loses electrons. A correction factor, C, is needed for energies between 0.2 and 1.5 MeV to account for this effect. Whaling (1958) published values for the correction factor by which Eq. (4) should be multiplied. These factors vary from 0.24 at 0.2 MeV, 0.75 at 0.6 MeV, 0.875 at 1.0 MeV, up to 1.0 at 1.6 MeV.

For the case of tissue, Eq. (5) reduces to

$$dE/dx_{\text{tissue}} = [0.126C/E] \ln\{7.99\ E\}\ \text{MeV} / \mu\text{m}^{-1} \quad (7)$$

Example 1 Find the energy loss (stopping power) of a 0.6 and a 5-MeV alpha particle in tissue.

$$\begin{aligned} dE/dx &= 0.126\ (0.75)/0.6 \ln (7.99 \times 0.6) \\ &= 0.25\ \text{MeV} / \mu\text{m}^{-1} \\ &= 0.126\ (1.0)/5.0 \ln (7.99 \times 5.0) \\ &= 0.093\ \text{MeV} / \mu\text{m}^{-1} \end{aligned}$$

The significance of this energy loss is seen in that it requires 33.85 eV to produce an ion pair; therefore, a 5-MeV alpha particle can produce $(0.25 \times 10^6\ \text{eV}/\mu\text{m}^{-1})$ / (33.86 ev / ion pair) = 7400 ion pairs in 1 μm, or enough damage to cause a double-strand break.

Beta Particles

The equations for beta particle energy loss in matter cannot be simplified, as in the case of alpha particles, because of three factors:

1. Even at low energies of a few tenths of an MeV, beta particles are traveling near the speed of light and relativistic effects (mass increase) must be considered.
2. Electrons are interacting with particles of the same mass in the medium (free or orbital electrons), so large energy losses per collision are possible.
3. Radiative or bremsstrahlung energy loss occurs when electrons or positrons are slowing down in matter. Such a loss also occurs with alpha particles, but the magnitude of this energy loss is negligible.

Including the effects of these three factors, the energy loss for electrons and positrons has been well quantitated. Tabulations of energy loss in various media have been prepared with the ionization energy loss and the radiative loss detailed. Tables of energy loss for electrons in tissue and many other substance as a function of electron energy can be found in ICRU 37 (1984).

Gamma Rays

Photons do not have a mass or charge, as do alpha and beta particles. The interaction between a photon and matter therefore is controlled not by the electrostatic Coulomb fields but by interaction of the electric and magnetic field of the photon with the electron in the medium. There are three modes of interaction with the medium.

The Photoelectric Effect The photon interaction with an orbital electron in the medium is complete, and the full energy of the photon is given to the electron.

The Compton Effect Part of the photon energy is transferred to an electron, and the photon scatters (usually at a small angle from its original path) (Evans, 1955) with reduced energy. The governing expressions are

$$\begin{aligned} E' &= E\ 0.511/(1 + 1/a - \cos\ \theta) \\ T &= E\ a(1 - \cos\ \theta)/[1 + a(1 - \cos\ \theta)] \end{aligned} \quad (8)$$

where E, E' = initial and scattered photon energy in MeV
T = kinetic energy of electron in MeV
$a = E/0.511$
θ = angle of photon scatter from its original path

Pair Production Pair production occurs whenever the photon energy is greater than the rest mass of two electrons, 2(0.511 MeV) = 1.02 MeV. The electromagnetic energy of the photon can be converted directly to an electron-positron pair, with any excess energy above 1.02 MeV appearing as kinetic energy given to these particles.

The loss of photons and energy loss from a photon beam as it passes through matter are described by two coefficients. The attenuation coefficient determines the fractional loss of photons per unit distance (usually in normalized units of g/cm^2, which is the linear distance times the density of the medium). The mass energy absorption coefficient determines the fractional energy deposition per unit distance traveled. The loss of photons from the beam is given by

$$I/I_0 = \exp(-\mu/\rho\ d) \quad (9)$$

where I = intensity of photon beam (numbers of photons)
I_0 = beam intensity
μ/ρ = attenuation coefficient in medium for energy considered (in $m^2\ kg^{-2}$)
d = thickness of medium in superficial density units kg m^{-2} (thickness in m times density in kg m^{-3})

Superficial density is convenient in that it normalizes energy absorption in different media. For example, air and tissue have approximately the same energy absorption per kg m^{-2}, whereas in linear dimension, the energy absorption, say, per meter, is vastly different. The energy actually deposited in the medium per unit distance is calculated using the mass energy absorption coefficient as opposed to the overall attenuation coefficient and the energy loss is given by

$$\Delta E = (\mu_{en}/\rho)E_0 \tag{10}$$

where ΔE = energy loss in medium per unit distance (in MeV $m^2\ kg^{-1}$)
μ_{en}/ρ = mass energy absorption coefficient ($m^2\ kg^{-2}$)
E_0 = initial photon energy

The values for μ_{en}/ρ as a function of gamma-ray energy are shown in Table 25-1 for air and muscle. Energy loss can then be expressed per unit linear distance by multiplying by the density of the medium (kg m^{-3}).

Table 25-1
Mass Energy Absorption Coefficients for Air and Water

PHOTON ENERGY, MeV	AIR, μ_{en}/ρ($m^2\ kg^{-1}$)	MUSCLE, STRIATE (ICRU), μ_{en}/ρ($m^2\ kg^{-1}$)
0.01	0.46	0.49
0.015	0.13	0.14
0.02	0.052	0.055
0.03	0.015	0.016
0.04	0.0067	0.0070
0.05	0.0040	0.0043
0.06	0.0030	0.0032
0.08	0.0024	0.0026
0.10	0.0023	0.0025
0.15	0.0025	0.0027
0.20	0.0027	0.0029
0.30	0.0029	0.0032
0.40	0.0029	0.0032
0.50	0.0030	0.0033
0.60	0.0030	0.0033
0.80	0.0029	0.0032
1.00	0.0028	0.0031
1.50	0.0025	0.0028
2.00	0.0023	0.0026
3.00	0.0021	0.0023

SOURCE: Hubbell, 1982, with permission.

ABSORBED DOSE

Dose and Dose Rate

Absorbed dose is defined as the mean energy, e, imparted by ionizing radiation to matter of mass m (ICRU, 1993):

$$D = e/m \tag{11}$$

where D = absorbed dose
e = mean energy deposited in mass
m = mass

The unit for absorbed dose is the gray (Gy), which is equal to 1 J kg^{-1}. The older unit of dose is the rad, which is equal to 100 erg g^{-1}, a value numerically equal to 100 times the dose in gray. The conversion between the two units is 100 rad = 1 Gy.

For uncharged particles (gamma rays and neutrons), kerma (kinetic energy released in matter) is sometimes used. It is the sum of the initial kinetic energies of all the charged ionizing particles liberated in unit mass. The units of kerma are the same as those for dose.

Exposure often is confused with absorbed dose. Exposure is defined only in air for gamma rays or photons and is the charge of the ions of one sign when all electrons liberated by photons are completely stopped in air of mass m:

$$X = Q/m \tag{12}$$

where X = exposure
Q = total charge of one sign
m = mass of air

The unit of exposure is coulombs per kilogram of air. The older unit of exposure is the roentgen, which is equal to 2.58×10^{-4} C kg^{-1} of air.

Exposure and dose are used interchangeably in some publications, even though this is not correct. The reason is that the older numerical values of dose in rad and exposure in roentgen are similar. Although they are similar numerically, they are fundamentally different in that exposure is ionization (only in air) and dose is absorbed energy in any specified medium:

$$1 \text{ roentgen} = 0.87 \text{ rad (in air)}$$

The SI units are not numerically similar:

$$1 \text{ C kg}^{-1} = 33.85 \text{ Gy}$$

Dose Rate

Dose rate is the dose expressed per unit time interval. The dose rate delivered to the thyroid by ^{99m}Tc for a nuclear medicine scan, for example, diminishes with time because of the 6.0-h half-life of the nuclide. The total dose is a more pertinent quantity in this case because it can be related directly to risk and compared with the benefit of the thyroid scan. The total dose over all time is expressed by

$$D = D_0 \times \{T_{eff})/\ln 2$$

where D_0 = dose rate at time zero
T_{eff} = effective half-life = $\{T_r + T_b) / \{T_r \times T_b\}$
T_r = radiologic half-life
T_b = biological half-life

In general, substances in the body are removed through biological processes as well as by radioactive decay; therefore the effective half-life is shorter than the radiologic half-life.

The dose rate from natural body ^{40}K in all living cells, by contrast, is relatively constant throughout life and is usually expressed as the annual dose rate.

Equivalent Dose

Ionizing radiation creates ion pairs in a substance such as air or tissue in relatively dense or sparse distribution depending upon the particle. Alpha particles with large mass produce relatively intense ionization tracks per unit distance relative to beta particles, and beta particles produce more dense ionization than gamma-rays. The ability to produce more or less ionization per unit path in a medium is quantitated by the LET. The linear energy transfer in a substance such as water is readily calculated.

The calculated LET from alpha and beta particles is therefore much greater than it is for gamma rays. In considering the health or cellular effects of each particle or ray, it is necessary to normalize the various types of radiation. For a particular biological endpoint, such as cell death in an experiment with mouse fibroblasts, it is common to calculate a relative biological effectiveness (RBE). This is defined as the ratio of gamma ray dose that yields the same endpoint to the dose from the radiation under study, for example, cell death.

Although giving the same dose to an organ from alpha particles as opposed to gamma rays would result in greater effects from the alpha particles, such refinement in the normalization of endpoints (cancer) in the human is not possible with the available data. An attempt to normalize human health effects from different types of radiation, i.e., to calculate an "equivalent" dose is made through the values for LET of the various types of radiation in water. The ratio of the LET for gamma to the radiation in question is defined as a *radiation weighting factor,* w_r (formerly Q), and the normalized (or weighted) dose is called the *equivalent dose*. The unit for the equivalent dose is the sievert (Sv), and the older unit is the rem:

$$H = Dw_r \tag{13}$$

where H = equivalent dose in sievert (older unit rem)
D = dose in gray (older unit rad)
w_r = radiation weighting factor

Table 25-2 gives the values of LET for different particles or rays and is reproduced from the National Council on Radiation Protection (NCRP, 1993) and the International Commission on Radiation Protection (ICRP, 1990).

Example 2 Find the equivalent dose (in sievert) for a dose to lung from an internal emitter of 0.01-Gy alpha particles and 0.01 Gy from external gamma-ray radiation.

Alpha, $H = 0.01\ (20) = 0.20$ Sv
Gamma, $H = 0.01\ (1) = 0.01$ Sv

Effective Dose and Cancer Risk

The term *effective dose* (ED) (formerly *effective dose equivalent*) was introduced formally by the ICRP in 1977 to allow addition or direct comparison of the cancer and genetic risk from different partial-body or whole-body doses. A partial-body gamma-ray dose to the lung, for example, is thought to give 0.0064 cancers over a lifetime per sievert, whereas a whole-body dose of 1 Sv would result in 0.056 total cancers and early genetic effects over the same lifetime interval. Both values are derived from the human A-bomb follow-up data. The ratio 0.0064/0.056 was defined as a tissue weighting factor, w_t, for lung and is numerically equal to 0.12.

The effective dose, H_E, is defined as a doubly weighted dose, weighted for radiation type and the tissue at risk.

$$H_E = w_t \times (Dw_r) \tag{14}$$

Table 25-2
Recommended Values of W_r for Various Types of Radiation

TYPE OF RADIATION	APPROXIMATE W_r
X-rays, gamma rays, beta particles, and electrons	1
Thermal neutrons	5
Neutrons (other than thermal ≫100 kev to 2 Mev), protons, alpha particles, charged particles of unknown energy	20

SOURCE: NCRP, 1993, and ICRP, 1990.

Table 25-3
Recommended Values of the Weighting Factors, w_t, for Calculating Effective Dose

TISSUE OR ORGAN	TISSUE WEIGHTING FACTOR, w_t
Gonads	0.20
Bone marrow (red)	0.12
Colon	0.12
Lung	0.12
Stomach	0.12
Bladder	0.05
Breast	0.05
Liver	0.05
Esophagus	0.05
Thyroid	0.05
Skin	0.01
Bone surface	0.01
Remainder	0.05*†

NOTE: The values have been developed from a reference population of equal numbers of both sexes and a wide range of ages. In the definition of effective dose, they apply to workers, to the whole population, and to either sex.

*For purposes of calculation, the remainder is composed of the following additional tissues and organs: adrenals, brain, upper large intestine, small intestine, kidney, muscle, pancreas, spleen, thymus, and uterus. The list includes organs that are likely to be selectively irradiated. Some organs in the list are known to be susceptible to cancer induction. If other tissues and organs subsequently become identified as having a significant risk of induced cancer, they will then be included either with a specific w_t or in this additional list constituting the remainder. The latter also may include other tissues or organs selectively irradiated.

†In exceptional cases in which a single one of the remainder tissues or organs receives an equivalent dose in excess of the highest dose in any of the 12 organs for which a weighting factor is specified, a weighting factor of 0.025 should be applied to that tissue or organ and a weighting factor of 0.025 should be applied to the average dose in the rest of the remainder as defined above.

SOURCES: NCRP, 1987, and ICRP, 1990.

This concept is useful in the case of occupational exposure, because H_E values from different sources to different organs can be summed to yield a direct estimate of total cancer and genetic risk.

Table 25-3 is taken from ICRP (1990) and gives the values of w_t for various organs.

The occupational guideline for H_E is 20 mSv per annum (NCRP, 1987; ICRP, 1990). This requires that the sum of all H_E be less than or equal to this value:

$$H_E = \sum w_t H \leq 20 \text{ mSv} \tag{15}$$

In 1990, the ICRP revised its 1977 estimates of risk and adopted and published Publication 60. This document includes new estimates of risk for both fatal and nonfatal cancer and new guidelines for the exposure of workers to external and internal radiation. The risk estimates are based largely on the analysis of Japanese A-bomb survivors. The occupational guidelines for radiation protection developed from the 1990 document are 100 mSv in 5 years (average, 20 mSv per year) with a limit of 50 mSv in any single year. This is compared with the 1977 limit of 50 mSv per year.

The ICRP document (ICRP, 1990) is a response to the increase in lifetime cancer risk from ionizing radiation observed in A-bomb survivors. Mental retardation for those exposed in utero is a finding in the A-bomb survivor cohort and is now included in the risk estimates.

The overall risk per unit exposure for adult workers and the risk for the whole population given in the ICRP (1990) are shown in Table 25-4. The risk of fatal cancer is adopted as 0.04 per sievert [4 percent per sievert (100 rem)] for adult workers and 0.05 per sievert [5 percent per sievert (100 rem)], for the whole adult population.

The NCRP (1993) chose to limit exposure to 20 mSv per year with a lifetime limit of (age $\times$ 10 mSv). The ICRP had been criticized for excluding the effects of nonfatal cancer in previous documents. An attempt to correct this omission was made in ICRP 60 (1990). An attempt was made to calculate the total detriment and is given the notation *aggregated detriment*. The aggregated detriment is the product of four factors: the probability of attributable fatal cancer, the weighted probability of nonfatal cancer, the weighted probability of severe hereditary effects, and the relative length of life lost. The nominal probability of fatal cancer per sievert, F, and the aggregated detriment are shown in Table 25-4. The computation of the aggregated detriment proceeds as follows. A cancer lethality fraction, K (the fraction of total cancer that is lethal), is used as a weighting factor for nonfatal cancers. The total number of cancers (fatal plus nonfatal) Sv^{-1} will be F/K. The total number of nonfatal cancers is $(1 - K)F/K$. The total weighted detriment is then

$$F + K(1 - K)F/K = F(2 - K)$$

The aggregated detriment is then the product of $F(2 - K)$ times the relative length of life lost, $1/l_{av}$, for a particular cancer. The average length of life lost, l_{av}, is 15 years per cancer. The aggregated detriment is tabulated as 7.3×10^{-2} Sv^{-1} for the whole adult population and $\times 10^{-2}$ 5.6 Sv^{-1} for the working population.

In assessing radiation risk from low-dose, low-dose-rate, low-LET radiation using risk coefficients derived from high-dose, high-dose-rate exposures, a dose rate–reduction factor (DREF) must be applied. NCRP (1980) and the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) (1988, 1993) have shown that the human data cover a range for the DREF of 2 to 10. That is, the original risk coefficients derived from the high-

Table 25-4
Nominal Probability Coefficients for Individual Tissues and Organs

	Probability of Fatal Cancer 10^{-2} *Sv*$^{-1}$		*Aggregated Detriment* 10^{-2} *Sv*$^{-1}$	
TISSUE OR ORGAN	WHOLE POPULATION	WORKERS	WHOLE POPULATION	WORKERS
Bladder	0.30	0.24	0.29	0.24
Bone marrow	0.50	0.40	1.04	0.83
Bone surface	0.05	0.04	0.07	0.06
Breast	0.20	0.16	0.36	0.29
Colon	0.85	0.68	1.03	0.82
Liver	0.15	0.12	0.16	0.13
Lung	0.85	0.68	0.80	0.64
Esophagus	0.30	0.24	0.24	0.19
Ovary	0.10	0.08	0.15	0.12
Skin	0.02	0.02	0.04	0.03
Stomach	1.10	0.88	1.00	0.80
Thyroid	0.08	0.06	0.15	0.12
Remainder	0.50	0.40	0.59	0.47
Total	5.00	4.00	5.92	4.74
	Probability of Severe Hereditary Disorders			
Gonads	1.00	0.6	1.33	0.80
Grand total (rounded)			7.3	5.6

NOTE: The values relate to a population of equal numbers of both sexes and a wide range of ages.
SOURCE: ICRP, 1990.

dose data are divided by the DREF factor to obtain the best estimate of effects at typical low-dose exposures. ICRP has used 2.5 as the adopted DREF; however, in ICRP (1990) a DREF of 2.0 was used, and this is incorporated in the nominal probability coefficients in Table 25-4. Table 25-4 is used universally to assess the effects of occupational exposure.

The overall objective of both NCRP and ICRP dose limitation recommendations is to control the lifetime risk to maximally exposed individuals. ICRP (1990) limits the lifetime occupational effective dose to (20 mSv × 50 years) = 1000 mSv. In 1993, NCRP (1993) reduced the U.S. Recommendation for Lifetime Exposure to (age × 10 mSv) or approximately 700 mSv, with an average annual limit of 20 mSv and a maximum annual limit of 50 mSv.

The dose to members of the public is also considered by ICRP and NCRP. For continuous exposure to human-made sources, i.e., other than medical or natural, it is recommended that the annual effective dose not exceed 1 mSv (100 mrem), and for infrequent exposures the annual effective dose should not exceed 5 mSv (500 mrem).

Committed Equivalent Dose

A problem arises with internal emitters in that once they are ingested, there is an irreversible dose that is committed because of the biokinetics of the particular element. The absorbed dose depends on the biological and physical half-times of the element in the body. For this reason, the concepts of committed equivalent dose and committed effective dose were derived to accommodate the potential for the dose to be delivered over long periods after incorporation in the body. The committed dose is taken over a 50-year interval after exposure and is equal to

$$H_{t,50} = \int_{t_0}^{t_0+50} H_t \, dt \qquad (16)$$

where $H_{t,50}$ = 50-year dose to tissue T for a single intake at time t_0
H_t = equivalent dose rate in organ or tissue T at time t

The NCRP (1987, 1993) recognizes that for radionuclides with half-lives ranging up to about 3 months, the committed equivalent dose is equal to the annual dose for the year of intake. For longer-lived nuclides, the committed equivalent dose will be greater than the annual equivalent dose and must be calculated on an individual basis. ICRP Publication 30 (ICRP, 1978) provides the details of this calculation for all nuclides.

Negligible Individual Risk Level (Negligible Dose)

The current radiobiologic principle commonly accepted is that of linear, nonthreshold cancer induction from ionizing radiation. Thus, regardless of the magnitude of the dose, a numerical cancer risk can be calculated. For this reason, the NCRP proposed the negligible individual risk level (NIRL) and defined it as "a level of annual excess risk of fatal health effects attributable to irradiation below which further effort to reduce radiation exposure to the individual is unwarranted."

The NCRP emphasized that the NIRL is not to be confused with an acceptable risk level, a level of significance, or a limit.

The NCRP recommended an annual effective equivalent dose limit for continuous exposure of members of the public of 1 mSv (0.1 rem). This value is in addition to that received from natural background radiation (about 2 mSv). In this context, the NIRL was taken to be 0.01 mSv (1 mrem). In NCRP (1993) the notation used currently is negligible individual dose (NID).

MECHANISMS OF DNA DAMAGE AND MUTAGENESIS

Energy Deposition in the Cell Nucleus

DNA is a double-helical macromolecule consisting of four repeating units: the purine bases adenine (A) and guanine (G) and the pyrimidine bases thymine (T) and cytosine (C). The bases are arranged in two linear arrays (or strands) held together by hydrogen bonds centrally and linked externally by covalent bonds to sugar-phosphate residues (the DNA "backbone"). The adenine base pairs naturally with thymine (A:T base pair), while guanine pairs with cytosine (G:C base pair), so that one DNA strand has the complementary sequence of the other. The sequence of the bases defines the genetic code; each gene has a unique sequence, but certain common sequences exist in control and structural DNA elements. Damage to DNA may affect any one of its components, but it is the loss or alteration of base sequence that has genetic consequences (UNSCEAR, 2000).

Ionizing radiation loses energy and slows down by forming ion pairs (a positively charged atom and an electron). Different ionization densities result from gamma rays, beta particles, and alpha particles. Their track structure is broadly characterized as from sparsely ionizing, (or low-LET), to densely ionizing (high-LET) radiation. Each track of low-LET radiation, resulting from x-rays or gamma rays, consists of a few ionizations across an average-sized cell nucleus (e.g., an electron set in motion by a gamma ray crossing an 8-μm-diameter nucleus gives an average of about 70 ionizations, equivalent to about 5 mGy (500 mrad) absorbed dose. Individual tracks vary widely about this value because of the stochastic nature of energy deposition, i.e., variability of ion pars per μm and path length through the nucleus. A high-LET alpha particle produces many thousands of ionizations and gives a relatively high dose to the cell. For example, a 4-MeV alpha-particle track yields on average, about 30,000 ionizations (3 Gy, 300 rad) in an average-sized cell nucleus. However, within the nucleus even low-LET gamma radiation will give some microregions of relatively dense ionization over the dimensions of DNA structures due to the low-energy electrons set in motion (UNSCEAR, 2000).

Direct and Indirect Ionization

Radiation tracks may deposit energy directly in DNA (direct effect) or may ionize other molecules closely associated with DNA, hydrogen or oxygen, to form free radicals that can damage DNA (indirect effect). Within a cell, the indirect effect occurs over very short distances, of the order of a few nanometers. The diffusion distance of radicals is limited by their reactivity. Although it is difficult to measure accurately the different contributions made by the direct and indirect effects to DNA damage caused by low-LET ra-

diation, evidence from radical scavengers introduced into cells suggests that about 35 percent is exclusively direct and 65 percent has an indirect (scavengeable) component (Reuvers et al., 1973).

It has been argued that both direct and indirect effects cause similar early damage to DNA; this is because the ion radicals produced by direct ionization of DNA may react further to produce DNA radicals similar to those produced by water-radical attack on DNA (Ward, 1975).

DNA Damage

Ionization frequently disrupts chemical bonding in cellular molecules such as DNA. If the majority of ionizations occur as single isolated events (low-LET radiation), the disruptions are readily repaired by cellular enzymes. The average density of ionization by high-LET radiations is such that several ionizations may occur as the particle traverses a DNA double helix. Therefore, much of the damage from high-LET radiations, as well as a minority of the DNA damage from low-LET radiations, will derive from localized clusters of ionizations that can severely disrupt the DNA structure (Goodhead, 1992; Ward, 1994). While the extent of local clustering of ionizations in DNA from single tracks of low and high-LET radiations will overlap, high-LET radiation tracks are more efficient at inducing larger clusters and hence more complex damage. Also, high-LET radiations will induce some very large clusters of ionizations that do not occur with low-LET radiations; the resulting damage may be irreparable and may also have unique cellular consequences (Goodhead, 1994). When a cell is damaged by high-LET radiation, each track will give large numbers of ionizations, so that the cell will receive a relatively high dose, as noted in the calculation above, and there will be a greater probability of correlated damage within a single DNA molecule. As a consequence, the irradiation of a population of cells or a tissue with a "low dose" of high-LET radiation results in a few cells being hit with a relatively high dose (one track) rather than in each cell receiving a small dose. In contrast, low-LET radiation is more uniformly distributed over the cell population. At doses of low-LET radiation in excess of about 1 mGy (for an average-size cell nucleus of 8 μm in diameter), each cell nucleus is likely to be traversed by more than one sparsely ionizing track.

The interaction of ionizing radiation with DNA produces numerous types of damage; the chemical products of many of these have been identified and classified according to their structure. These products differ according to which chemical bond is attacked, which base is modified, and the extent of the damage within a given segment of DNA. Table 25-5 lists some of the main damage products that can be measured following low-LET irradiation of DNA, with a rough estimate of their abundance (UNSCEAR, 2000). Attempts have also been made to predict the frequencies of different damage types from the knowledge of radiation track structure, with certain assumptions about the minimum energy deposition (number of ionizations) required. Interactions can be classified according to the probability they will cause a single-strand DNA alteration (e.g., a break in the backbone or base alteration) or alterations in both strands in close proximity in one DNA molecule (e.g., a double-strand break), or a more complex type of DNA damage (e.g., a double-strand break with adjacent damage). Good agreement has been obtained between these predictions and direct measurements of single-strand breaks, but there is less agreement for other categories of damage. While complex forms of damage are difficult to quantify with current experimental techniques, the use of enzymes that cut DNA at sites of base damage suggests that irradiation of DNA in solution gives complex damage sites consisting mainly of closely spaced base damage (measured as oxidized bases of abasic sites); double-strand breaks were associated with only 20 percent of the complex damage sites (Sutherland et al., 2000). It is expected that the occurrence of more complex types of damage will increase with increasing LET, and that this category of damage will be less repairable than the simpler forms of damage. Theoretical simulations have predicted that about 30 percent of DNA double-strand breaks from low-LET radiation are complex because of additional breaks (Nikjoo et al., 1977) and that this proportion rises to more than 70 percent, and the degree of complexity increases, for high-LET particles (Goodhead and Nikjoo, 1997).

Table 25-5
Estimated Yields of DNA Damage in Mammalian Cells Caused by Low-LET Radiation Exposures

TYPE OF DAMAGE	YIELD (NUMBER OF DEFECTS PER CELL Gy^{-1})
Single-strand breaks	1000
Base damage*	500
Double-strand breaks	40
DNA protein cross-links	150

*Base excision enzyme-sensitive sites or antibody detection of thymine glycol.
SOURCE: UNSCEAR, 2000.

Some of the DNA damage caused by ionizing radiation is chemically similar to damage that occurs naturally in the cell. This "spontaneous" damage arises from the thermal instability of DNA as well as endogenous oxidative and enzymatic processes. Several metabolic pathways produce oxidative radicals within the cell, and these radicals can attack DNA to give both DNA base damage and breakage, mostly as isolated events. The more complex types of damage caused by radiation may not occur spontaneously, since localized concentrations of endogenous radicals are less likely to be generated in the immediate vicinity of DNA (UNSCEAR, 2000).

HUMAN STUDIES OF RADIATION TOXICITY

There have been five major studies of the health detriment resulting from exposure of humans to ionizing radiation. Other studies of large worker populations exposed to very low levels of radiation and environmental populations exposed to radon are ongoing, but they are not expected to provide new data on the risk estimates from ionizing radiation. These worker or environmental populations are studied to ensure that there is no inconsistency in the radiation risk data in extrapolating from the higher exposures. The basic studies on which the quantitative risk calculations are founded include radium exposures, A-bomb survivors, underground miners exposed to radon, patients irradiated with x-rays for ankylosing spondylitis, and children irradiated with x-rays for tinea capitis (ringworm).

Radium Exposures (^{226,228}Ra)

Radium was discovered in the early part of the twentieth century. Its unique properties suggested a potential for the healing arts. It

was incorporated into a wide variety of nostrums, medicines, and artifacts. The highest exposure occurred in the United States among radium dial painters who ingested from 10s to 1000s of micrograms (microcuries). These exposed groups, including patients, chemists, and dial painters, have been studied for over 60 years to determine the body retention of radium and the health effects of long-term body burdens.

The only late effect of ingestion of ^{226,228}Ra seen is osteogenic sarcoma. It is significant that no study has identified a statistically significant excess of leukemia after even massive doses of radium. This implies that the target cells for leukemia residing in bone marrow are outside the short range of the radium series alpha particles (70 μm). Several thousand people were exposed to radium salts either as part of the modish therapies using radium in the era from 1900 to 1930 or occupationally in the radium dial–painting industry around 1920. Radium therapy was accepted by the American Medical Association, and in around 1915 advertisements were common for radium treatment of rheumatism and as a general tonic and in the treatment of mental disorders. Solutions were available for drinking containing 2 μg/60 cm^3 as well as ampoules for intravenous injection containing 5 to 100 μg radium (Woodard, 1980). Luminous paint was developed before World War I, and in 1917 there were many plants in New England and New Jersey painting watch dials, clocks, and military instruments (Woodard, 1980).

The first large studies on osteogenic sarcoma in radium-exposed people were done by Martland (1931) and Aub and associates (1952), who found 30 cases of bone sarcoma; Evans and associates (1969) with 496 cases of sarcoma out of 1064 studied at the Massachusetts Institute of Technology; and Rowland et al. (1978), with 61 cases out of 1474 female dial painters (Woodard 1980).

Radium, once ingested, is somewhat similar to calcium in its metabolism and is incorporated on bone surfaces into the mineralized portion of bone. The long half-life of ^{226}Ra (1600 y) allows distribution throughout the mineral skeleton over life. The target cells for osteogenic sarcoma reside in marrow on endosteal surfaces at about 10 μm from the bone surface. At long times after exposure, target cells are beyond the range of alpha particles from radium not on bone surfaces.

The loss of radium from the body by excretion was determined to follow a relatively simple power function (Norris et al., 1955):

$$R = 0.54\, t^{-0.52} \tag{17}$$

where R = total body retention
t = time in days

Other models to fit the data were developed as more information became available, the most recent being that of Marshall et al. (1972). The entire body of radium data and the various models are shown in Fig. 25-1. It can be seen that the Norris function fits the observed data well except at very long times after exposure. A simplified form of the more complex later model of Marshall and associates (1972) which fits the human data over all observed times is

$$R = 0.8t^{-0.5}\,(0.5e^{-\lambda t} + 0.5e^{-4\lambda t}) \tag{18}$$

where R = whole body retention
λ = rate of bone apposition or resorption = 0.0001 day^{-1}
t = time in days

For most purposes, the Norris formula is applicable. It can be seen from Fig. 25-1 for the Norris equation that even 1 year after exposure, only about 2 percent of the radium is retained in the body but that after 30 years, about 0.5 percent still remains. The risk of osteogenic bone cancer after radium exposure has been summa-

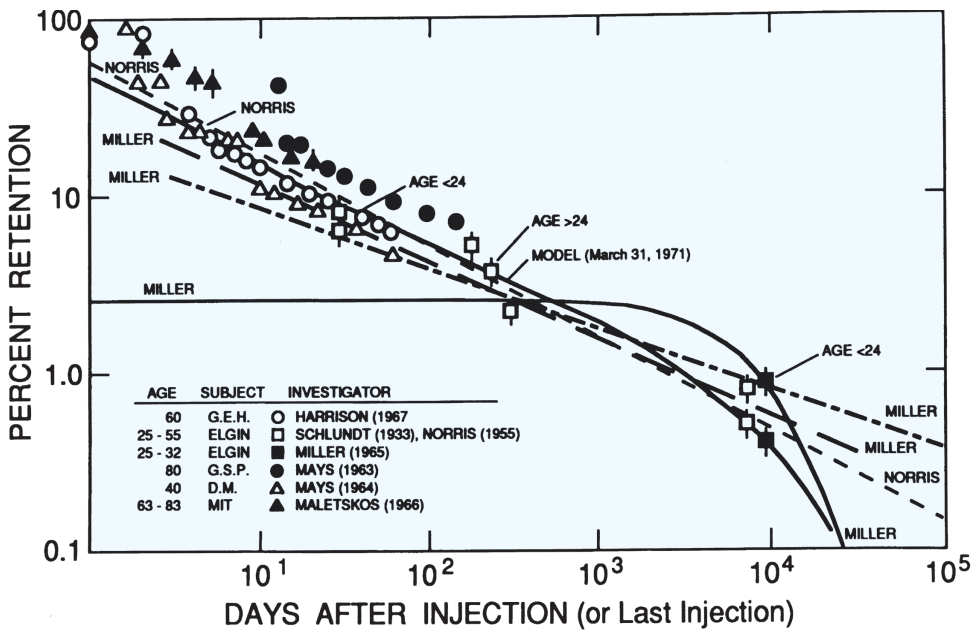

Figure 25-1. Whole-body radium retention in humans. Summary of all available data for adult humans. (From Marshall et al., 1972, with permission.)

rized in the National Academy of Sciences report BEIR IV (NAS, 1988).

Equations were proposed by Rowland and coworkers (1978) for the annual risk of sarcoma (including the natural risk), and expressed as a function of either radium intake or dose from ^{226,228}Ra. Risk per unit intake is

$$I = [0.7 \times 10^{-5} + (7 \times 10^{-8})\, D^2] \exp[-(1.1 \times 10^{-3})\, D] \quad (19)$$

where I = total bone sarcomas per person year at risk
D = total systemic intake of ^{226}Ra plus 2.5 times total systemic intake of ^{228}Ra, both in microcuries

Risk per unit dose is

$$I = [10^{-5} + (9.8 \times 10^{-6})D^2] \exp(-1.5 \times 10^{-2}\, D) \quad (20)$$

where I = total bone sarcomas per person year at risk
D = total mean skeletal dose in Gray from ^{226}Ra plus 1.5 times mean skeletal dose from ^{228}Ra

Raabe and associates (1980) modeled bone sarcoma risk in the human, dog, and mouse and determined that there is a practical threshold dose and dose rate (a dose low enough so that bone cancer will not appear within the human life span). The dose rate is 0.04 Gy per day or a total dose of 0.8 Gy to the skeleton. This practical threshold for bone cancer has useful implications in considering health effects from exposures to environmental radioactivity.

Radium Exposure (^{224}Ra)

In Europe, ^{224}Ra was used for more than 40 years in the treatment of tuberculosis and ankylosing spondylitis. The treatment of children was abandoned in the 1950s, but the ability to relieve debilitating pain from ankylosing spondylitis in adults has prolonged its use. ^{224}Ra is different from ^{226}Ra in that it has a short half-life (3.62 days) and the alpha dose is delivered completely while the radium is still on bone surfaces.

Spiess and Mays (1970) and Mays (1988) studied the health of 899 German patients given ^{224}Ra therapeutically. The calculated average mean skeletal dose was 30 Gy (range, 0.06 to 57.5 Gy) with injection time spans ranging from 1 to 45 months. There were two groups—juveniles and adults—and the bone sarcoma response was not significantly different for the two. There were 60 patients who developed bone sarcoma (Gossner 1999), 46 have been studied for histologic type. Further study of this group revealed other solid tumors with statistically significant excesses of male and female breast cancer, thyroid cancer and liver cancer (Nekolla et al., 1999).

In a second cohort, Wick and colleagues (1986, 1999) studied 1432 adult patients treated for ankylosing spondylitis with an average skeletal dose of 0.65 Gy. This study was originally started by Otto Hug and Fritz Schales and has been continued since their deaths. Four patients in this group have developed osteogenic sarcoma, and one in the control group.

Spiess and Mays (1973) found that the observed effectiveness of the ^{224}Ra in their cohort in producing bone sarcomas increased if the time span of the injections was long. Injections were given in 1, 10, or 50 weekly fractions. They developed an empiric expression to estimate the added risk from this protracted injection schedule:

$$I = \{0.003 + 0.014\,[1 - \exp(0.09m)]D \quad (21)$$

where I = cumulative incidence of bone sarcomas after most tumors have appeared (25 years)
m = span of injections in months
D = average skeletal dose in Gy

Chemelevsky and coworkers (1986) analyzed the Spiess data and developed an equation for the total cumulative sarcoma risk from ^{224}Ra:

$$R = (0.0085D + 0.0017D^2)\exp(-0.025D) \quad (22)$$

where R = cumulative risk of bone sarcoma
D = average skeletal dose in Gy

These two equations for risk predict 5.7 and 5.8 bone sarcomas in the second series of (spondylitis) patients, with 2 actually observed.

Chemelevsky and coworkers (1986) also showed that in the Spiess study, linearity (sarcoma response with dose) could be rejected. For example, equation 22 results in a lifetime risk of sarcoma of 0.02 Gy^{-1} at an average skeletal dose of 10 Gy but 0.01 Gy^{-1} at 1 Gy. Also, there was no difference in sarcoma response between juveniles and adults. These data are presented in Fig. 25-2. Again, no excess leukemia was found in either series of ^{224}Ra patients.

Atomic Bomb Survivors

On August 6, 1945, the U.S. military dropped an atomic bomb on the city of Hiroshima, Japan. Three days later a second bomb was dropped on Nagasaki which effectively ended World War II. The weapons were of two different types, the first being ^{235}U and the second a ^{239}Pu device.

Within 1 km of the explosions in both cities, a total of 64,000 people were killed by the blast and the thermal effects and as a result of the instantaneous gamma and neutron radiation released by the weapons. Others between 1 and 2 km from the hypocenter (the

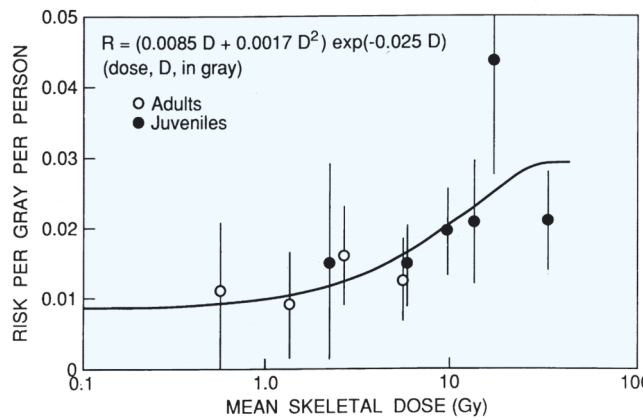

Figure 25-2. Lifetime risk per gray versus mean skeletal dose in ^{224}Ra-exposed subjects. (From Chemelevsky et al., 1986, with permission.)

point on earth directly below the detonation point in air) received radiation doses up to several gray.

Within a few years it was decided to follow the health of the people in both cities over their lifetime to determine quantitatively the effects of external ionizing radiation. The study of prospective mortality of A-bomb survivors was initiated by the Atomic Bomb Casualty Commission (ABCC) in 1950 and is ongoing by the Radiation Effects Research Foundation (RERF). The main study, called the Life Span Study (LSS), included 92,228 people within 10,000 m of the hypocenter and 26,850 people who were not in either city at the time of bombing (ATB). The most recent reports of the RERF (1987) are follow-up of the cancer mortality of a subcohort (DS86 subcohort) of 75,991 persons over the periods 1950–1985, and 1950–1990 (Pierce and Preston 1993).

In 1978, questions arose that the original dose estimates for persons in the LSS might be somewhat in error and that an effort should be made to improve the dose estimates. This study was published in a United States–Japan joint reassessment of dose called DS86—Dosimetry System 1986 (RERF, 1987).

Dose estimation by reconstruction of the event is always problematic, but direct computation of dose to about 18,500 persons in the LSS with detailed shielding information is complete. The remaining DS86 dose values for 57,000 individuals without detailed shielding information are also incorporated into the mortality study by various estimation techniques. Of the 75,991 persons in the DS86 subcohort, 16,207 were within 2000 m of the hypocenter, and these are the individuals who received a substantial exposure.

Previous reports of cancer risk estimates were based on the air dose (gamma ray plus neutron tissue kerma in air) adjusted for shielding by structures or terrain. The 1987 and 1988 reports also include DS86 organ dose estimates, and these are about 80 percent of the shielded kerma (Shimizu et al., 1988).

The dose from fallout at Hiroshima and Nagasaki has not been included in the health effects studies. Fallout was found in certain restricted localities in Nagasaki and Hiroshima. The absorbed dose from gamma rays at Nagasaki for persons continuously in the fallout area from 1 h on ranged from 0.12 to 0.24 Gy. The absorbed doses at Hiroshima ranged from 0.006 to 0.02 Gy. Because the region of fallout was quite limited, the total contribution of fallout to survivor dose was probably negligible in Hiroshima but may have been significant for a limited number of survivors in Nagasaki, where an exposure of one-fifth the maximum extends over some 1000 hectares. Estimates of internal dose from ingested ^{137}Cs yield about 0.0001 Gy integrated over 40 years (Harley, 1987; RERF, 1987).

Complete mortality data and the dose estimates are reported in RERF Technical Reports 5-88 (RERF, 1988) and were updated by Pierce and Preston (1993) and in UNSCEAR (2000). The projected lifetime cancer risks as of the follow-up through 1990 are reported in UNSCEAR (2000), and these data are summarized in Table 25-6.

No statistically significant excess cancer of the gallbladder, bone, pancreas, uterus, or prostate or of malignant lymphoma has been seen in the LSS to date.

It is of interest to consider the effect of smoking, as it is the most important factor in assessing lung cancer risk. The analysis performed by Shimuzu and associates (1988) examined the interaction of smoking and radiation in detail. The results showed no interaction indicating that smoking and the atom bomb radiation act independently rather than multiplicatively in lung cancer induction.

It is also possible to model the risk over the full life if a projection model is assumed. RERF has preferred a constant relative risk model (radiation mortality is a constant fraction of the baseline age-specific mortality per gray) for this purpose. There is evidence in the atom bomb mortality and in several other studies discussed later (ankylosing spondylitis patients, uranium miners) that the constant relative risk model is not appropriate but that the risk coefficient decreases with time subsequent to exposure. In most cases this also means that the absolute excess cancer risk (risk above that expected) declines with time. This is a biologically plau-

Table 25-6

Observed and Expected Cases of Cancer in A-Bomb Survivors with DS86 Dose Estimate and Projections of Lifetime Cancer Incidence from Radiation for a Dose of 0.1 Sv

				*Projection of Lifetime Cancer Incidence (%) for a Dose of 0.1 Sv**	
CANCER TYPE	OBSERVED CASES	EXPECTED CASES	MEAN DOSE (Sv)	MALES	FEMALES
Esophagus	84	77.4	0.23	0.04	0.02
Stomach	1307	1222	0.23	0.17	0.17
Colon	223	193.7	0.23	0.13	0.19
Liver	284	254.5	0.24	0.23	0.41
Lung	456	364.7	0.25	0.23	0.07
Breast	295	200	0.27	0.0	0.52
Thyroid	132	94.3	0.26	0.07	0.04
Urinary bladder	115	98.1	0.23	0.03	0.13
Other solid cancer				0.26	0.15
Solid cancer				1.16	1.70
Leukemia	141	67.4	0.25	0.05	0.05
Total				1.21	1.75

*Projection of cancer incidence based on the attained age model, UNSCEAR (2000), and a whole-body exposure of 0.1 Sv at age 30. Mortality projections for total cancer 0.67 and 0.99 percent per 0.1 Sv, for males and females respectively.

sible model suggesting the loss or repair of the damaged stem cell population.

The estimates of lifetime risk of cancer may increase somewhat with time, but given the present age of the population, the final values are unlikely to be significantly higher than the values in Table 25-6.

Tinea Capitis (Ringworm) Irradiation

During the period 1905–1960, x-ray epilation in the treatment of tinea capitis was performed regularly in children. The treatment was introduced by Sabouraud in 1904 and was standardized by Kienbock (1907) and Adamson (1909). Over the half century it was used, as many as 200,000 children worldwide may have been irradiated (Albert et al., 1986).

No follow-up studies of the long-term effects of irradiation were performed until Albert and Omran (1968) reported on 2200 children irradiated at the Skin and Cancer Unit of New York University Hospital during 1940–1959. Subsequent publications on this group have appeared at regular intervals (Shore et al., 1976, 1984, 1990).

Since the New York University (NYU) study, a follow-up of 11,000 children irradiated in Israel was performed (Ron and Modan, 1984; Ron et al. 1991).

The mean age of children irradiated in both the New York and Israeli studies was between 7 and 8 years. Dose reconstruction in the NYU series was performed using a head phantom containing the skull of a 7-year-old child covered with tissue-equivalent material (Schulz and Albert, 1963; Harley et al., 1976, 1983). The doses to organs in the head and neck for a typical Adamson-Kienbock five-field treatment of the scalp are shown in Table 25-7, and the dose to the skin is shown in Fig. 25-3.

In the NYU series there were 2 thyroid cancers with 1.4 expected cases. In the Israeli series there were 43 thyroid cancers with 10.7 expected cases. In the NYU series there are 83 skin lesions, predominantly basal cell carcinoma, with 24 expected cases, in 41 persons. Fairness of skin is an important factor in the appearance of skin cancer (Shore et al., 1984, 1990). Skin cancer was found only in whites even though 25 percent of the study population consisted of blacks. This and the fact that there appears to be a much lower dose response on the hair-covered scalp than on the face and neck (Harley et al., 1983) suggest that the promotional effects of UV radiation play an important role in skin cancer.

The dose estimate for the thyroid in the Israeli study is 0.09 Gy compared with 0.06 Gy in the NYU study.

A risk projection model was used to estimate the lifetime risk of basal cell carcinoma (BCC) for facial skin and for the hair-covered scalp after x-ray epilation in whites. The model used was a cumulative hazard plot which assumes that the BCC appearance rate in the exposed population remains constant over time (Harley et al., 1983). The result of this risk projection for BCC is shown in Table 25-8.

The small numbers of tumors other than skin cancers in the NYU study make it of dubious value in estimating the lifetime risk per Gy although an excess is appearing. The tinea capitis studies are prospective, and sound numerical values are forthcoming as these populations age. These are particularly important studies because children were the exposed group and because only partial body irradiation was involved. The temporal pattern of appearance of these tumors is also important. The dose was delivered over a short time interval (minutes at NYU and 5 days in Israel), and lifetime patterns will be indicative of the underlying carcinogenic mechanisms.

Skin and thyroid cancers are of importance in documenting health effects from ionizing radiation. However, both types of cancer are rarely fatal. NCRP (1985) reported that about 10 percent of thyroid cancer is lethal. It is estimated that the fatality rate of skin cancer is 1 percent (NCRP, 1990). The lifetime risk per gray derived by NCRP for total thyroid cancer incidence (0.003 for females and 0.0014 for males for external x-ray or gamma radiation for persons under 18 years of age) is about a factor of 10 lower than that reported by Ron and Modan (1984, 1991) in tinea capitis irradiations. However, the tinea irradiations were given to children with a mean age of about 7 years, also in the Israeli study there is apparently an increased sensitivity resulting from ethnicity.

The effect of ethnicity and sex is also suggested by NCRP (1985) for thyroid cancer. The incidence rates of spontaneous thyroid cancer for persons of Jewish origin in Europe and North America are three to four times that for other racial groups. There is an obvious susceptibility of women for thyroid cancer and adenomas in both the NYU and Israeli tinea capitis studies.

Table 25-7
Average Dose to Organs in the Head and Neck from Measurements Performed with a Phantom for a Child's Head

ORGAN	AVERAGE DOSE AT 25 cm TREATMENT DISTANCE, rad
Scalp	220–540
Brain	140
Eye	16
Internal ear	71
Cranial marrow	385
Pituitary	49
Parotid gland	39
Thyroid	6
Skin (eyelid)	16
Skin (nose)	11
Skin (midneck)	9

Chernobyl and Radioactive Iodine (^{131}I)–Induced Thyroid Cancer

The Chernobyl accident (April 26, 1986) was the result of efforts to conduct a test on the electrical control system, which allows power to be provided in the event of a station blackout. The details of the accident have been published in a report of the International Atomic Energy Agency (IAEA 1992), and UNSCEAR (2000). Basically, there was a rapid increase in the reactor power. Part of the fuel in the pressurized water reactor was vaporized, resulting in an explosion that blew the reactor core apart and destroyed much of the containment building. The estimates of the significant radionuclides for health effects released during the accident are 1800 PBq (4.9×10^7 Ci) ^{131}I and 85 PBq (2.3×10^6 Ci) ^{137}Cs, and 10 PBq (2.7×10^5 Ci) ^{90}Sr (UNSCEAR, 2000).

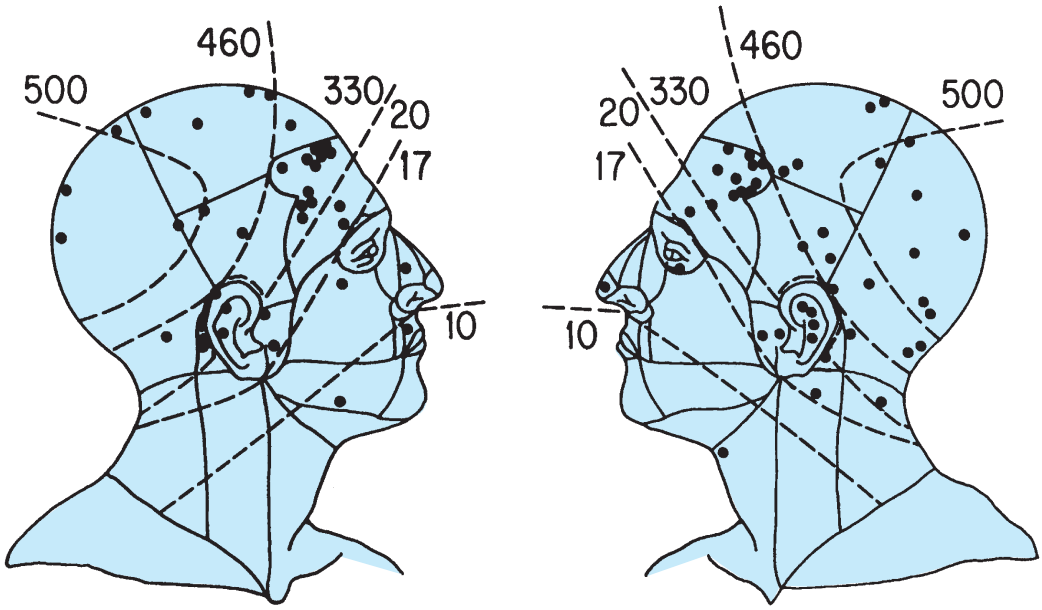

Figure 25-3. X-ray dose in rads for the Adamson-Kienbock five-field tinea capitis treatment and locations of basal cell lesions. (From Shore et al., 1984, with permission.)

The accident caused the deaths within days or weeks of 30 power plant employees and firemen (including 28 deaths that were due to radiation exposure). During 1986, 220,000 people were evacuated from areas surrounding the reactor, and, after 1986, of about 250,000 people were relocated from what were at that time three constituent republics of the Soviet Union: Belarus, the Russian Federation, and Ukraine. Large areas in these three republics were contaminated, and deposition of fission product radionuclides was measurable in all countries of the northern hemisphere. In addition, about 240,000 workers, termed "liquidators," were mobilized in 1986 and 1987 to take part in major mitigation activities at the reactor and within the 30-km zone surrounding the reactor. Residual mitigation activities continued until 1990. In all, about 600,000 persons received the special status of "liquidator."

The radiation exposures resulting from the Chernobyl accident were due initially to ^{131}I and short-lived radionuclides and subsequently to radiocesium (^{134}Cs and ^{137}Cs) from both external exposure and the consumption of foods contaminated with these radionuclides. UNSCEAR (1988) estimated that, outside the regions of Belarus, the Russian Federation, and Ukraine, thyroid doses averaged over large portions of European countries were 25 mGy for 1-year-old infants. However, the dose distribution was very heterogeneous, especially in countries near the reactor site. For example, in Poland, although the countrywide population-weighted average thyroid dose was estimated to be 8 mGy, the mean thyroid doses for the populations of particular districts ranged from 0.2 to 64 mGy. Individual dose values for about 5 percent of the children were 200 mGy. UNSCEAR (1988) estimates that effective dose averaged over large portions of European countries were 1 mSv or less in the first year after the accident and approximately two to five times the first-year dose over a full lifetime.

To date there is a significant increase in the number of thyroid cancers in the three territories. Approximately 1600 thyroid cancers have been identified in children under the age of 17 at the

Table 25-8
Estimated Lifetime Risk Estimates for Basal Cell Carcinoma (BCC) and Thyroid Cancer after X-Ray Irradiation for Tinea Capitis

	TOTAL INCIDENCE, RISK Gy^{-1}	MORTALITY, RISK Gy^{-1}
Skin malignancies (NYU study)		
BCC (facial skin)	0.32	
BCC (hair-covered scalp)	0.01	
Thyroid malignancies (Israeli study)		
Male	0.01	0.001
Female	0.04	0.004

time of the accident. The expected number of thyroid cancer cases is not known, but is evidently a small fraction of those observed. No solid tumors other than thyroid cancer have been identified resulting from the accident. Figure 25-4 shows the sequential increase in the number of thyroid cancers in children under the age of 14 at the time of the accident. It is well established that release of ^{131}I presents the major health effect in a nuclear accident. Many countries prepare for such accidents and have ready a large supply of potassium iodide (KI), which effectively blocks the thyroid uptake of radioactive iodine. Had this been available in time, it is questionable whether the large number of tumors seen would have occurred.

Medical Administration of ^{131}I Iodine-131 is given medically in three ways. Very large quantities, 3.7×10^9 Bq (100 mCi) or more are administered to ablate the thyroid in thyroid cancer, lesser quantities (about 10 mCi or 3.7×10^8 Bq) are given for hyperthyroidism, and the lowest quantity given (0.1 mCi or 3.7×10^6) is for diagnostic purposes (UNSCEAR, 1993). Individuals have also been exposed to ^{131}I as a result of nuclear weapons testing. Very few thyroid cancers have been found subsequent to these exposures, with the exception of the 243 Marshall Island inhabitants who received a large dose from a mixture of radionuclides (^{131}I, ^{132}I, ^{133}I, ^{134}I, and ^{135}I), tellurium, and gamma-ray radiation from the 1954 Bravo thermonuclear test (Conard, 1984) in the Pacific. The mean thyroid dose was estimated as 3 to 52 Gy in children and 1.6 to 12 Gy in adults. Over a 32-year follow-up period, 7 of 130 women and 2 of 113 men developed thyroid cancer.

Attempts have been made to relate external gamma-ray radiation and ^{131}I exposure. The NCRP (NCRP, 1985) estimated from human data that the effectiveness ratio of ^{131}I/gamma-ray radiation is between 0.1 and 1.0. In a more recent review of the human data, Shore (1992) found 8.3 observed excess cancers derived from all ^{131}I studies and 37 cases based on risk estimates from external exposure. The ratio 8.3/37 yields an estimate for the effectiveness ratio of 0.22. The protracted dose to the thyroid during the decay of ^{131}I may explain the difference; however, the nonuniform distribution of ^{131}I in the thyroid also may be a factor (Sinclair et al., 1956).

It is evident that ^{131}I can expose large populations after nuclear weapons testing or nuclear accidents. Generally, it is the ingestion pathway that is most significant. Iodine is ingested quickly either from surface deposition on edible plants or from pasture grass to the cow, to milk, and to the thyroid. A large body of data exists on the transfer coefficients, P_{24} (intake to the body per unit deposition), P_{45} (effective dose per unit intake), and P_{25} ($P_{25} = P_{24} \times P_{45}$) = effective dose per unit deposition). UNSCEAR (1993) reported the transfer coefficients for ^{131}I to be

$P_{24} = 0.07$ Bq per Bq m^{-2}
$P_{45} = 61$ nSv per Bq intake (effective dose for the thyroid)
$P_{25} = P_{24} \times P_{45} = 4.2$ nSv per Bq m^{-2} (effective dose for the thyroid)

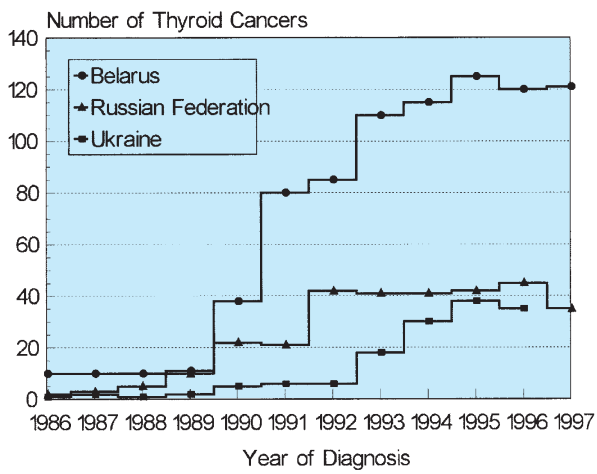

Figure 25-4. Thyroid cancer in children under 14 at the time of the Chernobyl accident. (From UNSCEAR, 2000.)

Ankylosing Spondylitis

About 14,000 persons, mostly men, were treated with x-rays for ankylosing spondylitis at 87 radiotherapy centers in Great Britain and Northern Ireland between 1935 and 1954. Court Brown and Doll (1957) were the first to report that these patients had a leukemia risk substantially in excess of that for the general population. Subsequent publications have developed the time pattern of appearance not only of leukemia but also of solid tumors (Court Brown and Doll, 1959, 1965; Smith and Doll, 1978, 1982; Smith, 1984; Darby et al., 1985, 1987; Weiss et al., 1994).

A group was selected consisting of 11,776 men and 2335 women all of whom had been treated with x-rays either once or twice. About half the total group received a second x-ray treatment or treatment with thorium. The reports on the ankylosing spondylitis patients attempt to consider health effects from only the first x-ray treatment. For this reason, an individual receiving a second treatment is included in their follow-up only until 18 months after the second course (a short enough time so that any malignancies in this interval cannot be ascribed to the second x-ray treatment).

The appearance of excess leukemia is now well documented, and solid tumors are also apparent in the population. The part of the body in the direct x-ray beam (spine) received the highest dose, but it is thought that other sites received substantial radiation from scatter or from the beam itself.

The importance of this study lies in the health effects of partial body exposure and in the temporal pattern of appearance of solid tumors in irradiated adults. Smith and Doll (1978, 1982), Darby et al. (1985, 1987), and Weiss et al. (1994, 1995), in the most recent follow-up publications concerning these patients, have shown that the excess risk for solid tumors diminishes with time since exposure, with maximum appearance 5 to 20 years after exposure. This has significant implications for risk projection modeling. Many projection models assume a constant rate of appearance either as an absolute number of tumors per person per unit exposure (constant absolute risk) or as a fraction of the baseline age-specific cancer mortality rate (constant relative risk). The emerging pattern is that constant risk models, either absolute or relative, are not correct for certain cancers, such as lung cancer. Thirty-five years after the first treatment, excess lung cancer had completely disappeared.

The dosimetry was redone in 1988 (Lewis et al., 1988), and although better estimates of dose are now available, it is still the dose that is most uncertain for the cohort. No details about the x-ray machines used to deliver the exposures, such as output, kilovoltage, and half-value layer, are reported.

The excess cancers and the estimate of lifetime cancer risk at three sites in the ankylosing spondylitis cohort are shown in

Table 25-9. For the purpose of calculating lifetime risks as of the time of follow-up, the number of persons used here as the individuals at risk is the number actually receiving only one x-ray treatment (6158). This assumes that those followed for 18 months after the second treatment do not contribute significantly to the malignancies.

The relatively low risk for leukemia (compared with atom bomb survivors) has been suggested to be due to cell sterilization at the high dose delivered. It is also possible that the low risk is due to partial irradiation of the skeletal red marrow. The volume of bone marrow irradiated in the spine, rib, and pelvis is much less than 50 percent of that in whole-body irradiation.

The deaths resulting from causes other than neoplasms in the total cohort are about 30 percent higher than expected. This higher total mortality is of significance in risk modeling as the premature deaths resulting from competing causes decrease the observed fractional cancer mortality. Thus, the lifetime risk in this population probably underestimates the risk when projecting the effects of exposure in a healthy population.

Uranium Miners

Radon is ubiquitous on earth. It is found outdoors and in all dwellings as a result of the decay of the parent ^{226}Ra, which is present in all of earth's minerals.

Although the risk of developing lung cancer from radon exposure among underground miners is firmly documented and quantitative risk estimates are available, the current interest lies in whether this risk carries over into environmental situations. Radon levels in homes that are comparable to those in mines surely confer risks to the residents. The question remains: Can the risks in mines for exposures at higher concentrations over short time periods be used to model risks at lower environmental levels over a lifetime?

Radon Exposure in Underground Mines There are 11 large follow-up studies of underground miners exposed to high concentrations of radon and radon decay products, and the documentation of excess lung cancer is convincing (NCRP, 1984; NAS, 1988; NIH, 1994; NAS, 1998). The carcinogen in the case of radon is actually the alpha-emitting short-lived decay products of radon, ^{218}Po, and ^{214}Po. The decay scheme for the entire uranium series, including radon and the daughter species, is shown in Fig. 25-5. The decay

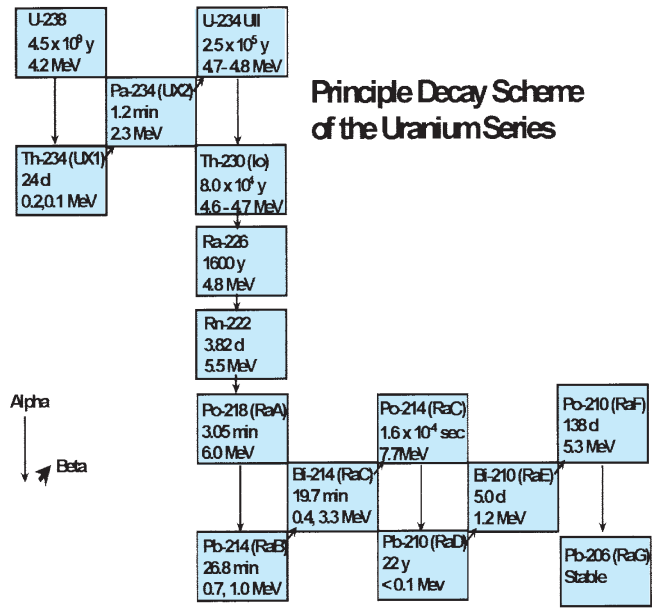

Figure 25-5. Uranium-238 decay series. (From NCRP, 1987.)

products or daughter products are solids and are deposited on the bronchial airways during inhalation and exhalation according to the laws of diffusion. As the airway lining (bronchial epithelium) is only 40 μm thick, the alpha particles emitted are able to reach and transfer a significant amount of energy to all the cells implicated in lung cancer induction. Although the daughters are the carcinogen, the term *radon* is here used interchangeably for *radon decay products,* because without the parent radon, the daughters could not exist longer than a few hours. The measurements in mines were usually of the daughter species rather than radon, and the term *working level* (WL) was adopted for occupational exposure. It indicated the total potential energy content in 1 L of air for complete decay of the short-lived daughters.* The exposure attributed to miners was developed in working-level months (WLMs), which is the

*One working level (WL) is any combination of short-lived daughters in 1 L of air that will result in 1.3×10^5 MeV of alpha energy when complete decay occurs. One working level is approximately equal to 7400 Bq m^{-3} (200 pCi/L) in a home and 11,000 Bq m^{-3} (300 pCi/L) in a mine.

Table 25-9
Excess Cancer in 6158 Ankylosing Spondylitis Patients Given a Single X-Ray Treatment as of the Last Follow-up

SITE	OBS	EXP	DOSE, Gy	LIFETIME, RISK Gy^{-1}
Leukemia	53	17	4.8	0.0011
Lung*	563	469	2.5	0.0027
Esophagus	74	38	5.6	0.0007

*Lung cancer appearing less than 5 years after exposure is not included as less than the minimum latency for tumor expression. The doses to the pulmonary lung and main bronchi were estimated as 1.8 and 6.8 Gy, respectively. The majority of lung cancer is bronchogenic, and the dose estimates for the main bronchi are probably most pertinent. Lifetime risk calculated as 30-year risk.

SOURCE: UNSCEAR, 2000.

numerical value of WL times the time exposed in multiples of a working month of 170 h (Holaday et al., 1957).

$$\text{WLM} = \text{WL (hours exposed}/170).$$

Estimating Lung Cancer Risk from Underground Miner Epidemiology The follow-up studies from 11 large underground mining cohorts in Australia, Canada, Czechoslovakia, France, Sweden, and the United States have all produced data that show that the excess lung cancer risk from exposure to radon is about 1 to 3 per 10,000 persons per WLM exposure (Radford and Renard, 1984; Hornung and Meinhardt, 1987; Sevc et al., 1988; Muller et al., 1989; Howe et al., 1986, 1987; Tirmarche et al., 1993; Woodward et al., 1991; NCRP 1984, NIH, 1994, NAS 1998). Expressed another way, radon exposure increases the normal age-specific lung cancer risk by about 0.5 percent for each WLM exposure. This way of expressing risk leads to the concept that many epidemiologists prefer—that the lung cancer risk is proportional to the normal baseline risk. This means, for example, that the lifetime excess lung cancer risk from radon is different for smokers and nonsmokers (NAS, 1988; NIH, 1994).

The actual data from the underground studies are not clear-cut with regard to the effect of smoking, and it is apparent from more recent analyses that radon exposure does not simply multiply the baseline risks of the population by a constant factor. This is considered in the discussion of risk, earlier in this chapter. The excess lung cancer risk in each of the exposure cohorts for the 11 major mining populations as of the date of the last published follow-up is summarized in Fig. 25-6. It can be seen in the figure that the range of risks for the same exposure varies by about a factor of 10 among the different studies. The differences are likely accounted for by errors in measuring and estimating total exposure. However, the Czech mine atmosphere contained arsenic as well as radon, and the arsenic contributed to the excess lung cancers observed. A maximum value of 50 percent lung cancer risk is the highest value ever observed in a mining population and was reported in mines in Saxony at the turn of the century (Muller, 1989). These mines are thought to have had about 100,000 Bq m^{-3} of radon. It is noteworthy that concentrations this high have been reported in a few homes in the United States. In Fig. 25-6, the lowest (well-documented) exposures were in the Ontario mines, and a mean exposure of 29 WLM has given an excess lung cancer risk of about 1 percent.

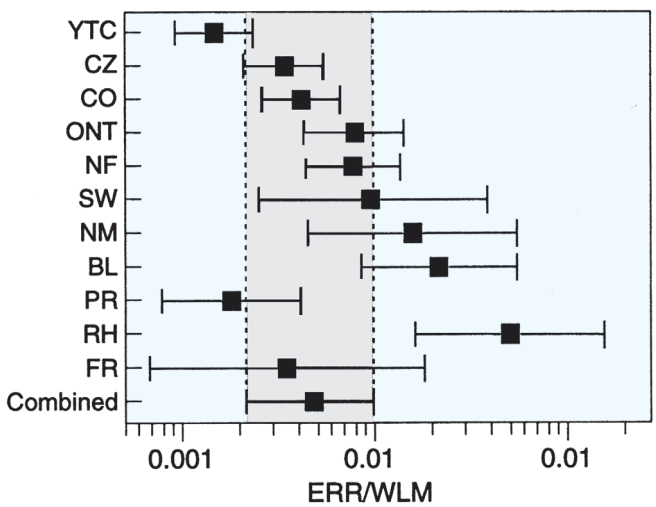

Figure 25-6. Excess relative risk (ERR) of lung cancer per working-level month (WLM) for each of the 11 cohorts studied.

RRs plotted per WLM exposure. YTC, Yunnan China Tin miners; CZ, Chechoslovakian uranium miners; CO, Colorado uranium miners; ONT, Ontario uranium miners; NF, Newfoundland fluorspar miners; SW, Swedish iron miners; NM, New Mexico uranium miners; BL, Beaverlodge, Canada, uranium miners; PR, Port Radium, Canada, uranium miners; RH, Radium Hill, Australia, uranium miners; FR, French uranium miners. (From NIH, 1994.)

To date, the most comprehensive epidemiologic analysis of underground miners exposed to high concentrations of ^{222}Rn is a joint analysis of 11 underground mining studies conducted by the National Cancer Institute (NIH, 1994), and updated by NAS (1998). The study encompasses the Chinese, Czechoslovakian, Colorado, Ontario, Newfoundland, Swedish, New Mexico, Beaverlodge, Port Radium, Radium Hill, and French mining data. Domestic studies do not prove useful in obtaining risk estimates for radon. This is because the effects of ^{222}Rn in the environment are obscured by typical low exposure rates and the large numbers of lung cancers caused by smoking. In mines, the average concentration can be thousands of Bq m^{-3} compared with less than 100 Bq m^{-3} in a typical domestic environment. The occupational exposure in mines is relatively short compared with that over a full life in homes. Although the time exposed in mines is shorter, the cumulative exposure in mines is generally many times that in homes. The joint analysis of the 11 cohorts (NIH, 1994) focused on 10 variables:

1. The estimation of excess relative risk per working level month (ERR/WLM) and the form of the exposure response
2. The variation of ERR/WLM with attained age
3. The variation of ERR/WLM with duration of exposure, considering total exposure as well as exposure rate
4. The variation of ERR/WLM with age at first exposure
5. The variation of ERR/WLM as a function of time after exposure ceased
6. The evaluation of an optimal exposure lag interval, that is, the interval before lung cancer death during which ^{222}Rn exposure has no effect
7. The consistency among the 11 cohorts
8. The joint effect of smoking and ^{222}Rn exposure
9. The role of exposure to other airborne contaminants in mines
10. The direct modeling of the relative risk of lung cancer with duration and rate of exposure

The relative risk of lung cancer for the Colorado uranium miners is shown in Fig. 25-7 and that for all 11 cohorts—showing only the data for exposures below 400 WLM—is shown in Fig. 25-8.

The pooled cohorts included 2620 lung cancer deaths among 60,570 exposed miners, accumulating 1.2 million person-years of observation. The excess relative risk of lung cancer is seen in Fig. 25-7 to be linearly related to the cumulative radon decay product exposure in units of WLM. The exception is at the highest exposures, where a clear reduction in excess relative risk is evident. This is often noted as an *inverse dose rate effect*. Confusion exists concerning the inverse dose rate effect in that the relative risk is not *higher at lower dose rates* but is *lower at higher dose rates,* in agreement with cell killing and removal of damaged cells from

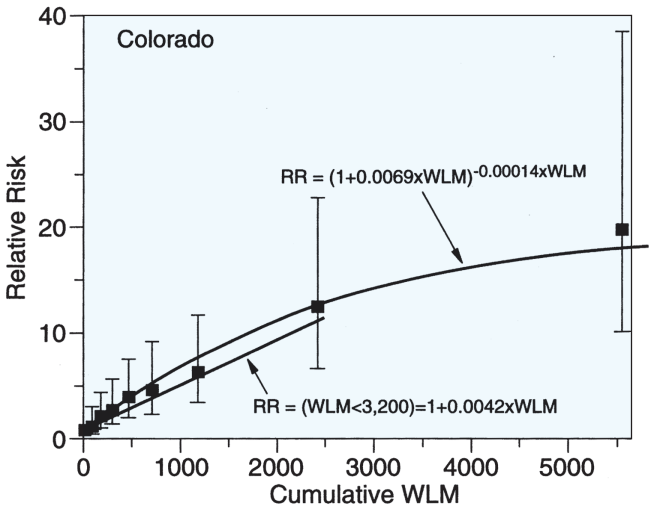

Figure 25-7. Excess relative risk per unit exposure for the Colorado mining cohort. (From NIH, 1994.)

the pool of cells that are potential sites of malignant transformation. This effect is seen in all studies of radiation damage at high dose.

Data on tobacco use were available in six of the cohorts. The ERR/WLM was not related to age at first exposure in this analysis. The joint effect of smoking and ^{222}Rn exposure did not show a clear pattern except that the risk was consistent with a relationship that was intermediate between additive and multiplicative.

Lung (Bronchial) Dose from Radon Exposure When radon gas decays to its solid decay products, some 8 to 15 percent of the ^{218}Po atoms do not attach to the normal aerosol particles. This ultrafine species (unattached fraction) is deposited with 100 percent efficiency on the upper bronchial airways. In mines the unattached fraction is low (4 to 5 percent) because of the normal aerosol loading. The rest of the decay products attach to the ambient aerosol of about 100-nm average diameter (George and Breslin, 1980) and only a few percent of this aerosol is deposited on these airways. Measurements in mines have mostly involved the short-lived radon daughters, as they are the easiest to measure rapidly. The alpha dose from radon gas itself is very low in comparison with that from the daughters, as the daughters deposit and accumulate on the airway surfaces. The upper airways of the bronchial tree are the region where almost all the lung cancers appear. This is true in general, not only for miners exposed to radon daughters but also for smokers.

The alpha dose from radon daughters therefore must be calculated in these airways, not in the pulmonary or gas-exchange regions. Although the dose to the pulmonary region should not be neglected, it is about 15 percent of that to the airways (Saccomanno et al., 1995). Several calculations regarding the absorbed alpha dose exist for radon daughters (NCRP, 1984; ICRP, 1987; Harley, 1987, 1989; Harley et al., 1996, NRC, 1991). The authors make different assumptions about the atmospheric and biological parameters that go into the dose calculation, and this can cause discrepancies among the models. The most significant variables are the particle size of the ambient aerosol, the assumed breathing rate, and the target cells considered.

Very small particles deposit more efficiently in the airways. Therefore if small particles, such as those from open burning flame (Tu and Knutson, 1988), contribute to the atmosphere, the dose delivered to the bronchial epithelium can be higher per unit WLM exposure than is the dose predicted from an average particle size. Conversely, a hygroscopic particle can increase in size in the humid environment of the bronchial airways, and deposition will be diminished. The particle size of the aerosol in mines is somewhat larger than that for environmental conditions (200 to perhaps 600 nm versus 100 nm) (George et al., 1975). Figure 25-9 shows the

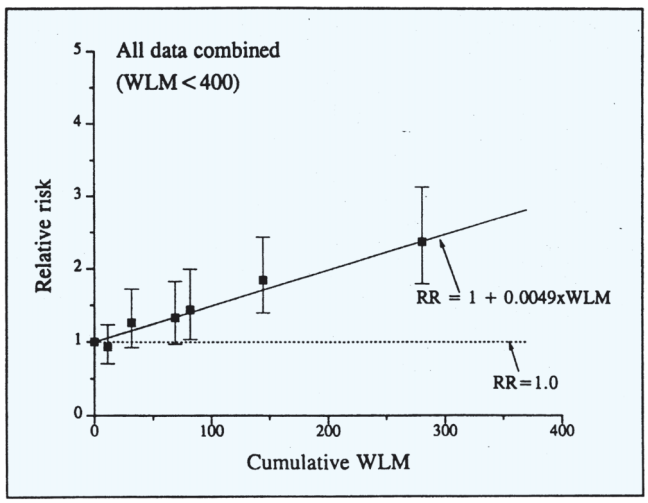

Figure 25-8. Relative risk (RR) of lung cancer by cumulative working-level month (WLM) and fitted linear excess RR model for each cohort and for all data combined (WLM < 400).

RRs are plotted at mean WLM for category. When the referent category for RRs is not zero exposure, a fitted exposure-response line is adjusted to pass through the mean of the referent category. For the China, Ontario, and Beaverlodge cohorts, the excess RR model was fitted with a free intercept. (From NIH, 1994.)

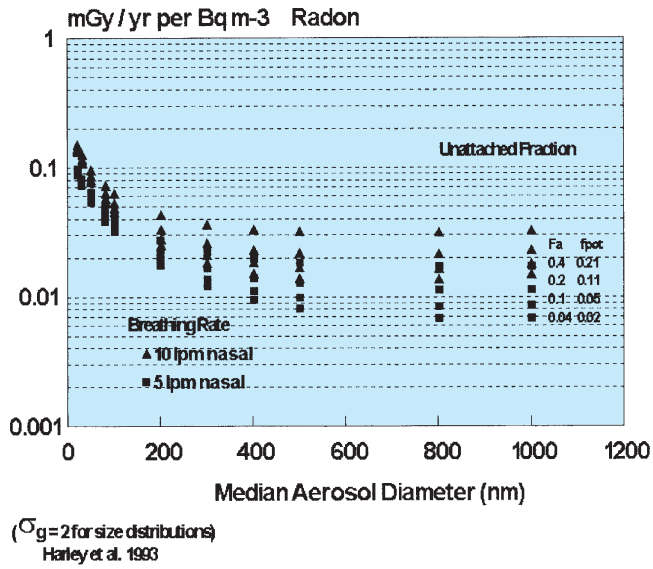

Figure 25-9. Radon decay product bronchial dose as a function of inhaled aerosol diameter, breathing rate, and unattached fraction. (Breathing rates from Ruzer et al., 1995; target cells from Robbins and Meyers, 1995, with permission.)

alpha dose per unit exposure as it is related to the variables (particle size, unattached fraction, breathing rate) known to affect dose.

As carcinogenesis is related to absorbed alpha dose, Fig. 25-9 shows that particle size is an important determinant of risk. The average dose per unit exposure in WLM for miners in Fig. 25-9 is about the same as that for average environmental conditions, assuming 100 nm aerosol in homes and 200 to 600 nm aerosol in mines.

Radon can deliver a greater or lesser carcinogenic potential by about a factor of 2 over the range of realistic indoor conditions (average particle size ranging from 80 to 300 nm). The allowable effective dose for continuous exposure of the population in the United States is 1 mSv/year (100 mrem/year) (NCRP, 1993). This limit would be delivered by exposure to 10 Bq m^{-3} of radon, or one-quarter the actual average measured indoor concentration in most countries where measurements have been made. Thus, the guidelines for exposure to radon cannot be set in the usual way from dosimetric considerations.

LIFETIME ENVIRONMENTAL LUNG CANCER RISK PROJECTIONS FOR RADON EXPOSURE

There are at present six sets of models based on the underground miner epidemiology that provide risk projection calculations for exposure to radon daughters in the home. The following sections describe each in detail.

National Council on Radiation Protection and Measurements

In 1984, the NRCP on Protection and Measurements (NCRP, 1984) developed a model to project the risk derived from miner studies to whole-life risk in the environment. It is a modified absolute risk model that reduces the risk subsequent to exposure with a half-life of 20 years. Risk is not accumulated until after age 40, the time when lung cancer normally appears in the population. There is no indication that early exposure produces any significant shift to younger ages, even for young miners exposed at significantly higher concentrations. This model was the first to incorporate a time since exposure reduction in risk.

National Academy of Sciences

The National Academy of Sciences report in 1988 (BEIR IV) developed a model based on examination of the raw data from five mining cohorts (NAS, 1988). The data indicated that the highest risk appears from 5 to 15 years after exposure. After 15 years, the risk is one-half that of the 5- to 15-year risk (per unit exposure), and this risk was assumed to persist to the end of life. Again, no significant risk appears before 40, the usual age for the appearance of lung cancer. The NAS model also included a correction for attained age (at age 65, the risk is 0.4 of that for ages 55 to 64). The BEIR IV committee assumed a relative risk model (risk is proportional to the normal age-specific lung cancer risk per unit radon exposure), but with risk dependent on time from exposure. This was the first modified relative risk model. This means that the risk for smokers and nonsmokers differs because of their different baseline lung cancer values. Although the miners' epidemiology did not support this strictly multiplicative relationship, the NAS chose the relative risk model as a conservative one. Its analysis supported the risk reduction subsequent to exposure by using a two-step risk reduction window.

NAS 1998 (BEIR VI) expanded on the models developed in NAS (1988) using 11 underground mining cohorts rather than 4. The models for the 11 cohorts were developed first by NIH (1994), see below, and updated for the NAS report.

International Commission on Radiation Protection

The IRCP (ICRP, 1987) developed two risk projection models: one was based on a constant relative risk and the other was a constant absolute risk model. Although neither risk model is correct because of the temporal reduction pattern of lung cancer subsequent to the cessation of exposure, the numerical values obtained for the lifetime risk of lung cancer from radon exposure are not significantly different from those in other models. Later follow-up of the Czechoslovakian underground uranium miners presented by Kunz and Sevc (1988) indicates that the excess lung cancer risk may actually be reduced to zero 35 years after exposure. If this factor were included in the NAS model (zero risk after 35 years), it would reduce those values by about a factor of 2. The risk values obtained from the various models are shown in Table 25-10. In 1993, ICRP simply adopted a lifetime lung cancer fatality coefficient of 3×10^{-4} per WLM.

NIH Joint Analysis of 11 Underground Mining Cohorts

The pooled analysis from the 11 underground mining cohorts was used to develop two models for full-life risk projection (NIH, 1994). The models are similar to the model used by the NAS (1988), utilizing time since exposure reduction and reduction with attained age. Three time windows for reduction of risk with time since exposure are used instead of two. Also, an additional parameter is incorporated; one model decreases the risk with increasing exposure rate, and the other decreases the risk with decreasing exposure duration. The lifetime domestic risk for lifetime exposure to unit concentration was not reported. However, the ratio of the relative risk of lung cancer for the BEIR IV and joint analysis model was given as 0.9 for continuous exposure to 4 $pCi l^{-1}$ (1 WLM per year) for the model incorporating exposure duration as a parameter. The joint analysis estimated that there are 15,000 lung cancer deaths in the United States attributable to ^{222}Rn: 10,000 in smokers and 5000 in those who have never smoked.

BEIR VI (NAS 1998) updated these two models and increased the calculated risk in the United States to 15,400 or 21,800 per year for ever smokers and never smokers for the two model values. The BEIR VI best annual estimate of deaths for ever smokers is stated to be 11,000 per year with 2100 or 2900 calculated deaths for never smokers, depending upon the model chosen.

ENVIRONMENTAL EPIDEMIOLOGY

The Environmental Studies

There are at least 24 published studies that attempt to define or detect the effect of radon exposure in the environment. Most are summarized by Borak and Johnson (1988), Neuberger (1989, 1992),

Table 25-10
Lung Cancer Risk for Continuous Whole-life Exposure to 4 pCi/L (150 Bq m^{-3} or 0.58 WLM per year at Indoor Conditions) as Predicted by Various Models of Domestic Exposure*

MODEL	LIFETIME RISK %	MODEL TYPE	COMMENT
NCRP (1984a)	0.50	Modified absolute risk. Two parameter model.	Risk decreases with time since exposure.
ICRP (1987)	0.90	Constant relative risk.	
ICRP (1987)	0.62	Constant additive risk.	
ICRP (1993)	0.56	Single-value risk per WLM.	Adopted lifetime risk per WLM exposure.
BEIR IV (NRC 1988)	1.1	Modified relative risk. Two-time windows. Two-parameter model.	Risk decreases with time since exposure.
NIH (1994)	1.8	Modified relative risk, three-time windows, age and exposure rate. Three parameter model.	Risk decreases with time since exposure and decreases with very high exposures.
BEIR VI (NRC 1998)	2.0	Modified relative risk. Three time windows, age and exposure rate. Three parameter model.	Risk decreases with time since exposure and decreases with very high exposures.
Meta-analysis of eight domestic case-control studies (Lubin and Boice, 1997)	0.7	Observed mortality.	Linear regression fit to data from eight domestic studies.

*Exposure assumes a home concentration of 148 Bq m^{-3} (4 pCi l^{-1} or 0.56 WLM), calculated with 40 percent decay product equilibrium, and actual exposure is 70 percent of the home exposure.
SOURCE: NAS, 1999.

and Samet and associates (1991). A study in the United States was performed in 1989 by the New Jersey Department of Health (NJDOH) (Schoenberg and Klotz, 1989; Schoenberg et al., 1990). This is a case-control study of women, 433 lung cancer cases and 402 controls with yearlong measurements of radon in the homes where the individuals lived for 10 or more years. This study devoted considerable effort to quality control concerning the exposure measurements. The results of this study are slightly positive, suggesting an association of radon and lung cancer even at concentrations of 80 Bq $m^{-3,}$ but the results are not statistically significant. A case-control study of 538 nonsmoking women (1183 controls) in Missouri with an average exposure of 70 Bq m^{23} also showed no statistically significant increase in lung cancer (Alavanja et al., 1994).

The largest case-control study to date concerning the effects of residential ^{222}Rn exposure was conducted nationwide in 109 municipalities in Sweden. It included all subjects 35 to 74 years old who had lived in one of the 109 municipalities at some time between January 1980 and December 31, 1984, and who had been living in Sweden on January 1, 1947. Fifty-six of the municipalities were known to have elevated ^{222}Rn concentrations on the basis of earlier measurements.

Thus, an attempt was made to study a large group of persons living in a known area of greater than average ^{222}Rn and to estimate their exposure over a large fraction of life (34 years). The primary aim of the study was to narrow the uncertainty in the estimation of lung cancer risk.

The environmental epidemiologic studies conducted before this study suffered from the small numbers of persons observed and relatively low ^{222}Rn exposures. For this reason, although the risk in underground miners was seen clearly, the outcome regarding the lung cancer risk from residential exposure has been ambiguous. All the existing domestic studies, including the measurement protocols, have been reviewed (Neuberger, 1992, 1994; Samet, 1989; Samet et al., 1991, 1989; Lubin et al., 1990).

Pershagen and coworkers (1994) included 586 women and 774 men with lung cancer diagnosed between 1980 and 1984. For a comparison control population, 1380 women and 1467 men were studied.

The ^{222}Rn concentration in 8992 homes was measured for 3 months during the heating season. The geometric and arithmetic mean concentrations were 1.6 and 2.9 pCi L^{21} (60 and 106 Bq m^{-3}). The cumulative exposure since 1947 was estimated for each subject by the addition of the products of concentration by the length of time the subject lived in each residence.

The data were reported in terms of the relative risk (RR) of lung cancer (ratio of observed to expected lung cancer) normalized to a relative risk of 1.0 for persons who never smoked and who had radon exposure below 50 Bq m^{-3}. The excess risk due to smoking could be seen easily. Smokers smoking less than or more than 10 cigarettes per day with a radon concentration of $\ll$50 Bq m^{-3} had RR of 6.2 (with a confidence interval from 4.2 to 9.2) and 12.6 (CI from 8.7 to 18.4), respectively.

The only statistically significant lung cancer excess resulting from ^{222}Rn was seen in those who smoked fewer than 10 cigarettes per day and had a time-weighted mean ^{222}Rn concentration $>$400 Bq m^{-3}. Their relative risk was 25.1 (CI 7.7 to 82.4). For those smoking more than 10 cigarettes per day, the relative risk com-

pared with those who had never smoked and had ^{222}Rn concentrations <50 Bq m^{-3} was 32.5 (CI 10.3 to 23.7). Although this relative risk appears higher than that for those smoking <10 cigarettes per day, the result is not statistically signficant. If the effect of ^{222}Rn alone is examined by comparing the risk only among smokers, that is, those with <50 Bq m^{-3} against smokers having >400 Bq m^{-3}, the relative risk due to ^{222}Rn alone is 3.7 (CI 1.1 to 11.7) for those smoking less than 10 cigarettes per day and 2.5 (CI 0.8 to 7.9) for those smoking more than 10 cigarettes per day (Pershagen et al., 1994). Because the confidence interval includes 1.0, it cannot be stated with statistical certainty that there was increased lung cancer caused by ^{222}Rn exposure although the point estimate RR = 2.5 suggests at least an upper bound of risk.

The analysis was done for the combined group of men and women (Pershagen et al., 1994). There were no details given concerning lung cancer and sex difference. However, the preliminary report (Pershagen et al., 1994) suggested that women may indeed have had less lung cancer than men for the same exposure conditions. Also of interest in the study of Pershagen and associates (1994) was the relative risk of lung cancer by histological type. In the >400 Bq m^{-3} group, only small cell carcinoma and adenocarcinoma had a statistically significant increased risk.

The pattern emerging from the domestic studies indicates that the lung cancer risk from ^{222}Rn exposure is difficult to determine with accuracy or precision. This is mostly due to the high background lung cancer mortality caused by smoking.

Among the 24 published domestic studies, 13 are ecologic and 11 are case-control (Neuberger 1989). Ecologic studies depend on relating the disease response of a population to some measure of a suspected causative agent. There usually are not enough data on all the variables involved in the disease to infer any reliable associations. Ecologic studies are the weakest type of epidemiologic exploration. Unless a biological marker for radon-induced lung cancer is found, it is unlikely that environmental epidemiology will be effective in assessing risk. The effects of radon in the environment are subtle compared with the overwhelming lung cancer mortality that results from smoking.

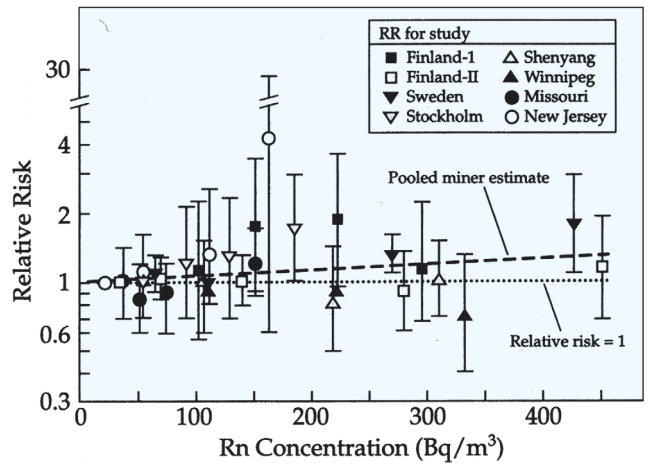

Figure 25-10. Meta-analysis of eight domestic radon case-control studies. $\sigma g = 2$ for size distributions. (From Harley et al., 1993, with permission.)

Meta-analysis of Environmental Epidemiology

In an attempt to combine the largest domestic studies to determine whether any risk from radon exposure in the home was apparent, Lubin and Boice (1997) performed a meta-analysis of 8 domestic case control studies. A meta-analysis combines the published information from several studies into one study without actually having the raw data available. The results are shown in Fig. 25-10. The data showed that essentially no study found statistically significant cancer deaths due to radon, but the authors state that the combined trend in the relative risk with increasing exposure was

Table 25-11
Equivalent Dose Rates to Various Tissues from Natural Radionuclides Contained in the Body

	Equivalent Dose Rate, mSv yr^{-1}			
RADIONUCLIDE	BRONCHIAL EPITHELIUM	SOFT TISSUE	BONE SURFACES	BONE MARROW
^{14}C	—	0.10	0.08	0.30
^{40}K	—	1.80	1.40	2.70
^{87}Rb	—	0.03	0.14	0.07
^{238}U-^{234}Th	—	0.046	0.03	0.004
^{230}Th	—	0.001	0.06	0.001
^{226}Ra	—	0.03	0.90	0.15
^{222}Rn	—	0.07	0.14	0.14
^{222}Rn daughters	24	—	—	—
^{210}Pb-^{210}Po	—	1.40	7.00	1.40
^{232}Th	—	0.001	0.02	0.004
^{228}Ra-^{224}Ra	—	0.0015	1.20	0.22
^{220}Rn	—	0.001	—	—
Total	24	3.50	11.00	5.00

Table 25-12
Estimated Total Effective Dose Rate for a Member of the Population in the United States and Canada from Various Sources of Background Radiation

	Total Effective Dose Rate, mSv yr^{-1}					
SOURCE	LUNG	GONADS	BONE SURFACE	BONE MARROW	OTHER TISSUES	TOTAL
w_t*	0.12	0.25	0.03	0.12	0.48	1.0
Cosmic	0.03	0.07	0.008	0.03	0.13	0.27
Cosmogenic	0.001	0.002	—	0.004	0.003	0.01
Terrestrial	0.03	0.07	0.008	0.03	0.14	0.28
Inhaled	2.0	—	—	—	—	2.0
In body	0.04	0.09	0.03	0.06	0.17	0.40
Total	2.1	0.23	0.05	0.12	0.44	3.0

*Tissue weighting factor–see Table 25.3
SOURCE: NCRP, 1987.

statistically significant, with an estimated RR of 1.14 (95 percent CI = 1.0 to 1.3) at an exposure of 150 Bq m^{-3} (4 pCil^{-1}).

What Is Known about Radon Exposure in the Home

Four concepts have emerged from the radon research so far:

1. The mining epidemiology indicates that short exposure to high levels of radon and daughters produces a clear excess of lung cancer.
2. Particle size can change the actual dose delivered by radon to bronchial tissue, with small particles giving a substantially higher dose per unit exposure. The use of open flames, electric motors, and the like indoors produces a higher dose per unit exposure.
3. Smokers are at higher risk from radon per unit exposure than are nonsmokers. The relative risk for nonsmokers is about three times that for smokers, but their age-specific lung cancer mortality is about ten times lower than that for smokers. Thus, the overall lifetime lung cancer risk is three times higher for smokers.
4. Urban areas almost universally have low radon, and apartment dwellers removed from the ground source have particularly low radon exposure at home.

The miners' data show clearly that there is a risk of lung cancer from exposure to high concentrations of radon delivered over short periods. Comparable exposures delivered over a lifetime in the home have not produced statistically significant increases in lung cancer mortality except among smokers in one large study in Sweden. The risk can still exist, but the confounding effects of other carcinogens, such as smoking and urbanization, make it impossible to extract the more subtle impact of radon in existing studies.

Table 25-13
Lifetime Effective Dose (in mSv from Birth to Age 85) from Natural Radionuclide Exposure

	LUNG	BONE MARROW	WHOLE BODY
Effective dose	180	10	260

SOURCE: NCRP, 1987.

NATURAL RADIOACTIVITY AND RADIATION BACKGROUND

The occupational, accidental, and wartime experiences detailed in the preceding sections have provided the bases for all the current radiation risk estimates. For many years, the radioisotopes deposited internally were compared with ^{226}Ra to evaluate the maximum permissible body burden for a particular emitter. The present limits for external and internal radiation are based on dose estimates that, in turn, can be related to cancer risks. One standard of comparison has always been the exposure from natural background, and this source is assessed here.

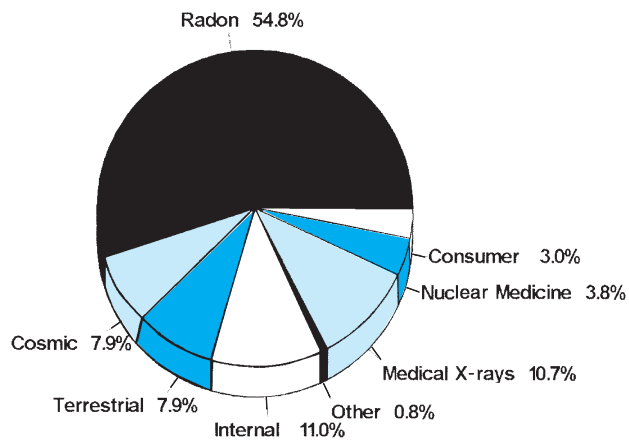

Figure 25-11. Contribution from natural background to effective dose of radiation in the U.S. population. Annual average effective dose, 3.6 mSv. (From NCRP, 1987.)

Background radiation from all sources is described in detail in NCRP report 94 (1987b), and some of the information is summarized here.

The risk estimates in the previous sections must be placed in context with the radiation dose received by all humans from natural background radiation. A substantial dose is received annually from cosmic radiation and from external terrestrial radiation present from uranium, thorium, and potassium in the earth's crust. Internal emitters are present in the body as a consequence of dietary consumption and inhalation. For example, potassium is a necessary element in the body and is under homeostatic control. Radioactive ^{40}K constitutes a constant fraction of all natural potassium. Potassium delivers the largest internal dose from the diet of 0.15 mSv per year. However, the data are scanty on the dietary intake of other radionuclides in the U.S. population. Given the usual distribution of intakes across a large population, it is probable that

Table 25-14

Estimates of Radionuclide Released and Collective Effective Dose from Human-Made Environmental Sources of Radiation

	Release (PBq)						*Collective Effective Dose* Person Sv*	
SOURCE	^{3}H	^{14}C	NOBLE GASES	^{90}Sr	^{131}I	^{137}Cs	LOCAL AND REGIONAL	GLOBAL
Atmospheric nuclear testing	240,000	220		604	650,000	910		2,230,000
Local								
Semipalatinsk							4600	
Nevada							500†	
Australia							700	
Pacific test site							160†	
Underground nuclear testing			50		15		200	
Nuclear weapons fabrication								
Early practice								
Hanford							8000‡	
Chelyabinsk							15,000§	
Later practice							1000	10,000
							30,000¶	
Nuclear power production								
Milling and mining							2700	
Reactor operation	140	1.1	3,200		0.04		3700	
Fuel reprocessing	57	0.3	1,200	6.9	0.004	40	4600	
Fuel cycle							300,000¶	100,000
Radioisotope production and use	2.6	1.0	52		6.0		2000	80,000
Accidents								
Three Mile Island			370		0.0006		40	
Chernobyl					630	70		600,000
Kyshtym				5.4		0.04	2500	
Windscale			1.2		0.7	0.02	2000	
Palomares							3	
Thule							0	
SNAP 9A								2100
Cosmos 954				0.003	0.2	0.003		20
Ciudad Juarez							150	
Mohammedia							80	
Goiania						0.05	60	
Total							380,000	23,100,000
Total collective effective dose (Person Sv)								23,500,000

*Truncated at 10,000 years.

†External dose only.

‡From release of ^{131}I to the atmosphere.

§From releases of radionuclides into the Techa River.

¶Long-term collective dose from release of ^{222}Rn from tailings.

SOURCE: UNSCEAR, 1993.

other emitters, notably ^{210}Pb, could deliver a significant dose to a fraction of the population.

The largest dose received by the population is from the inhaled short-lived daughters of radon. These are present in all atmospheres because radon is released rather efficiently from the ^{226}Ra in rock and soil. The short-lived daughters, ^{218}Po, ^{214}Pb, ^{214}Bi-^{214}Po, have an effective half-life of 30 min, but the 3.8-day parent radon supports their presence in the atmosphere. Figure 25-5 shows the entire uranium series decay.

Average outdoor concentrations in every state in the United States have been measured and summarized as 15 Bq m^{-3}, and indoors, as 40 Bq m^{-3} (NAS, 1999). A structure such as a house prevents the rapid upward distribution of radon into the atmosphere, and substantial levels can be built up indoors. The source of radon is the ground; therefore levels in living areas above the ground are generally one-third to one-fifth the concentrations measured in basements. An effective barrier across the soil-building interface also inhibits the entry of radon to buildings. Ventilation with outdoor air reduces indoor radon. For this reason, industrial buildings with more substantial foundations and higher ventilation rates tend to have lower radon concentrations than do single-family (or detached) houses. Apartments above ground level have radon concentrations about half the average of those in single-family dwellings.

It is of significance that an average radon concentration indoors of 40 Bq m^{-3} results in an equivalent dose to bronchial epithelium of 24 mSv/year or an effective dose of 2 mSv per year.

The equivalent doses for the major natural internal emitters are shown in Table 25-11. These are reproduced from NCRP (1987).

The annual effective dose equivalents for all the external and internal emitters from natural background are summarized in NCRP Report 94 (1987) and are shown in Table 25-12.

The lifetime dose from natural emitters is shown in Table 25-13, assuming an average exposure from birth to a full life of 85 years. It should be recognized that the actual dose accumulated by an individual depends on dietary habits, location (Denver, for example, at an altitude of 1.6 km, has double the average cosmic-ray exposure), and the dwelling. An apartment dweller would accumulate approximately half the dose from inhaled radon daughters as would a person living in a single-family dwelling. Table 25-13 is informative in considering the effects of radiation exposure from other than natural sources. For example, in assessing an occupational dose, which might add, say, 10 mSv effective dose equivalent, natural background would be a strong confounder. Any health detriment would have to be calculated rather than observed directly. No study would be able to detect an increase in health effects from 10 mSv above the average whole-life natural background of 260 mSv.

Figure 25-11 shows the average components of natural background in the United States, and Harley (2000) gives a detailed summary of background radiation and internal radioactivity.

LOCAL ENVIRONMENTAL RELEASES

Large- and small-scale accidents will undoubtedly occur that release radioactivity into the environment. The accident at the Windscale nuclear power reactor in 1957 was a local incident in Great Britain. The nearby population has been studied for over 30 years without the appearance of significant health effects.

The nuclear power accident at Three Mile Island caused enormous financial damage, but the containment vessel was not breached and virtually no radioactivity escaped.

The accident at the Chernobyl nuclear power plant was another such occasion, and in this case containment did not exist and some of the radioactivity was widespread over Europe. The United Nations Scientific Committee on the Effects of Atomic Radiation

Table 25-15

Lifetime Cancer Mortality per Gray from Five Major Epidemiologic Studies (in parentheses, risk per sievert for alpha emitters, $w_r = 20$)*

STUDY	ALL SITES	LEUKEMIA	LUNG	FEMALE BREAST	BONE	THYROID	SKIN
Atom bomb whole-body, gamma	0.05	0.005	0.0085	0.002	0.0005	0.0008	0.0002
Uranium miner bronchial epithelium, alpha			(0.04) 0.0020				
Ankylosing spondylitis, spincal x-ray		0.0011	0.0008 0.0028	0.0015			
Tinea capitis, head x-ray						0.0010§	0.0030‡
Radium ingestion, bone,* alpha (^{226}Ra)					0.004 (0.0002)		
Radium ingestion, bone,† alpha (^{224}Ra)					0.02 (0.0010)		

*The lifetime risk is calculated for an average skeletal dose of 10 Gy, assuming that the risk persists for 50 years and using Eq. (20). The risk is nonlinear and is about 0.01 Gy^{-1} at 100 Gy, for example.

†The lifetime risk is calculated for an average skeletal dose of 10 Gy using equation 22. The risk is nonlinear and is about 0.01 Gy^{-1} for a skeletal dose of 1 Gy.

‡The mortality for skin cancer is estimated as 1 percent of the incidence; see text.

§Thyroid mortality for males and females. Estimated as 10 percent of incidence.

(UNSCEAR, 1988, 1993) has summarized the committed dose from measurements made in the affected countries from various releases, and these are shown in Table 25-14. Table 25-14 includes environmental releases from most known sources.

Highly radioactive emissions from industrial sources that are "lost," such as the ^{60}Co in Goiania or Thailand, do harm to the few persons involved, often with a few deaths due to very high radiation exposure and dose.

The criticality accident on September 30, 1999, at a fuel reprocessing facility in Tokai-Mura, Japan, resulted in the death of two workers and caused neighbors to receive a small local dose of about 2 mSv (Komura et al 2000).

Local exposures and doses from accidents can only be anticipated to increase, as the use of radioactive materials industrially is widespread.

SUMMARY OF HUMAN CANCER RISKS FROM RADIATION

The details of the five major studies have been given in the preceding sections. The data are summarized in Table 25-15. This table shows the lifetime cancer risks that are significant. The risks are given in units of per gray (or per Sievert where appropriate for alpha emitters).

Within the table, leukemia and cancers of the lung and female breast are the most critical. Osteogenic sarcoma is seen in the radium exposures. There is no clear linear dose response for ^{224,226}Ra. This has been attributed to the existence of an apparent threshold. The cancer risk to individual organs from different study groups is in general agreement regardless of radiation type or whole- or partial-body exposure.

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