11 Internal Radiation Safety

INTERNAL RADIATION

Internal radiation exposure occurs when radionuclides from environmental contamination enter the body. The consequences of this internal contamination may range from innocuous to very serious, depending on the quantity of the contaminating radionuclide and the dose that it delivers. In many instances, the level of contamination is known only after a lengthy investigation, which may include bioassay measurements. Accordingly, internal radiation safety is concerned mainly with preventing or minimizing the intake of radionuclides into the body and the deposition of radioactivity on the body. This is accomplished by a program designed to keep environmental contamination within acceptable limits and as low as reasonably achievable. This last point, keeping environmental contamination levels as low as reasonably achievable, is especially important in the context of internal radiation safety.

External radiation exposure is due to radiation originating in sources outside the body; there is no physical contact with the radiation source, and exposure ceases when one leaves the radiation area or when the source is removed. Since external radiation may be measured with relative ease and accuracy, the potential or actual hazards may be estimated with a good deal of confidence. In the case of internal contamination, however, the radiation dose cannot be directly measured; it can only be calculated. As a consequence of the fact that radioactivity is deposited on or within the body, irradiation of the contaminated person continues even after the person leaves the area where the contamination occurred. It must be emphasized, however, that the fact that an internally deposited radioisotope continues to irradiate as long as it is in the body is explicitly considered in calculating the dose from an internally deposited radionuclide or when calculating the annual limits on intake (ALIs) of the various radionuclides. In the context of potential harm, the radiation dose from an internally deposited radionuclide is no different from the same dose absorbed from external radiation. Therefore, it must be emphasized that, *dose for dose, the consequences of internal radiation are the same as those from external radiation; a milligray is a milligray and a millirem is a millirem, regardless of whether it was delivered from an internally deposited radionuclide or from external radiation.*

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PRINCIPLES OF CONTROL

Radioactive substances, like other noxious agents, may gain entry into the body through three pathways:

1. Inhalation—by breathing radioactive gases and aerosols.
2. Ingestion—by drinking contaminated water, eating contaminated food, or tactilely transferring radioactivity to the mouth.
3. Absorption—through the intact skin or through wounds.

Safety measures to counter internal radiation are designed to either block the portals of entry into the body or to interrupt the transmission of radioactivity from the source to the worker. This can be achieved either at the source, by enclosing and confining it, or at the worker, through the use of protective clothing and respiratory protective devices. Additionally, work practices and schedules should be designed so as to minimize contamination and exposure to contaminated environments. It should be noted that these control measures are the same as those employed by the industrial hygienist in the protection of workers from the effects of nonradioactive noxious substances. However, the degree of control required for radiological safety almost always greatly exceeds the requirements for chemical safety. This point is made clear by the figures in Table 11-1, which compare the maximum allowable atmospheric concentrations of several nonradioactive noxious substances to the maximum concentrations recommended by the International Commission on Radiological Protection (ICRP) for radioactive forms of the same element.

Control of the Source: Confinement

The simplest type of confinement and enclosure may be accomplished by limiting the handling of radioactive materials to well-defined, separated areas within a laboratory and by the use of subordinate isolating units such as trays. For low-level work, where there is no likelihood of releasing a gas, vapor, or aerosol to the atmosphere in a quantity exceeding 1 ALI, this may be sufficient. If the possibility exists of releasing to the atmosphere—either as gas, vapor, or aerosol—amounts of radioactivity between 1 and 10 times the ALI, the usual practice is to use a ventilated hood. The purpose of the ventilated hood is to dilute and to sweep out the released radioactivity with the air that flows through the hood. In order to accomplish this purpose, it thus is essential to have a sufficient amount of air flowing through the

TABLE 11-1   Concentration Limits of Several Substances Based on Chemical and Radiological Toxicity

	CONCENTRATION LIMITS (mg/m^3)		
	Nonradioactive	Radioactive	
Beryllium	0.002	^{7}Be	1.7×10^{-8}
Mercury	0.01	^{203}Hg	5×10^{-9}
Lead	0.05	^{210}Pb	1×10^{-9}
Arsenic	0.01	^{74}As	3×10^{-9}
Cadmium	0.01	^{115}Cd	4×10^{-10}
Zinc	5	^{65}Zn	1.2×10^{-8}

hood at all times. Constant airflow velocity may be maintained by using a bypass that opens as the face of the hood is closed. Openings along the bottom of the front-face frame facilitate the flow of air when the face is closed. Figure 11-1 shows a typical radiochemistry fume hood. The face velocity must be great enough to prevent contaminated air from flowing out of the face into the laboratory but not great enough to produce turbulence around the edges, which would allow the contaminated air from the hood to spill out into the laboratory. It has been found that velocities of 125–275 ft/min (0.6–1.4 m/s) are required. To minimize the possibility of contaminating the working environment with the exhaust from the hood, all ductwork must be kept under negative pressure. Any leakage in the ductwork will then be *into* the duct. This is most easily accomplished by locating the exhaust fan at the discharge end of the exhaust line, as shown in Figure 11-1.

For purposes of environmental control and hazard evaluation, aerosols are defined as airborne particles and are classified according to their size and manner of production:

Fumes—Solid particles resulting from a change of state. For example, lead monoxide (PbO) fumes are produced when lead is vaporized, whereupon the molecules are oxidized and then condensed to form solid particles. The particle sizes are less than 0.1 μm.

Dusts—Solid particles resulting from pulverizing large chunks of matter or re-suspending previously pulverized matter. Particle sizes range from ∼0.1 to 30 μm. Particles larger than 30 μm are not considered to be inhalable.

Inertials—Particles that are about 50 μm or greater in size.

Smokes—Products of combustion. Sizes range from submicron to several millimeters.

Mists—Liquid particles of any size.
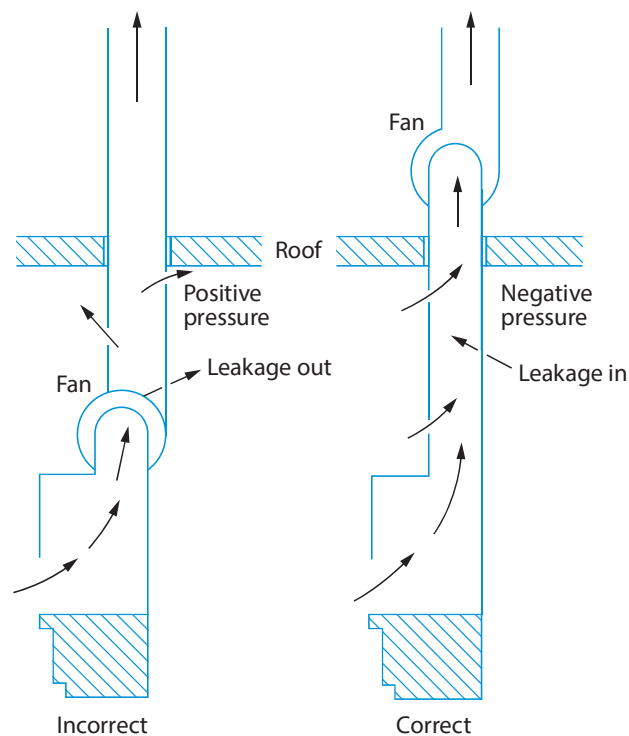
Figure 11-1. Effect of fan location on direction of leakage in the ductwork. The fan should be close to the discharge end, thereby creating a negative pressure in the ductwork and causing ductwork leakage to be into the duct.

If a hood is designed to remove only gases and vapors, an air velocity in the ductwork of about 2000 ft/min ($\sim$10 m/s) is sufficient. For fumes, the recommended duct velocity is 2500 ft/min (12–13 m/s). Since larger particulate matter tends to settle out, the air transport velocity must be on the order of 3500–4500 ft/min (18–22 m/s) if particle fallout is to be minimized. If the exhaust from the hood is of such a nature that it may create a radioactive pollution problem, the effluent from the hood must be decontaminated by an appropriate air-cleaning device. For this purpose, if the pollutant is an aerosol, a rough filter followed by a fire-resistant, high-efficiency filter is commonly employed. As used in this context, a high-efficiency filter is one that removes at least 99.997% of 0.3-μm-diameter homogeneous particles of dioctylphthalate (DOP). For a filter to be called HEPA in the United States, the filter must meet the criteria in "Specification for HEPA Filters Used by DOE Contractors (DOE-STD-3020)" or the equivalent EPA standard.

A filter loaded with radioactive dust can easily become a source of contamination if adequate precautions to prevent the dust from falling off the filter during the changing operation are not taken. A simple way to minimize dispersal of loose dust when removing the filter is to spray the filter faces with a bonding agent before removing the filter, thereby trapping the radioactive dust in the filter. For this purpose, access ports upstream and downstream of the filter should be provided in the ductwork.

If the nature of an operation involving radioactivity is such that it must be completely enclosed—that is, if the operation is potentially capable of contaminating the working environment with more than 10 times the recommended contamination limits or when the large quantities of air required by a hood are not available—then a glove box (Fig. 11-2) is used. It should be reemphasized here that, while the main function of a fume hood is to dilute and remove atmospheric contaminants, the main function of a glove box is to isolate the contaminant from the environment by confining it to the enclosed volume. Accordingly, the airflow through the glove box may be very small—on the order of 0.01–0.02 m³/s (25–50 ft³/min). Air is usually admitted into the glove box through a high-efficiency fiberglass filter (to prevent discharge of radioactive dust into the room in case of an accidental positive pressure inside the glove box) and is exhausted through a series of fire-resistant, rough, and high-efficiency filters. Airborne particles small enough to be carried by this flow of air are thus transferred out of the glove box into the filter; larger particles fall out inside the glove box and remain there until cleaned out. A negative pressure of at least 13 mm (0.5 in.) water inside the glove box assures that any air leakage will be into the box. Despite the negative pressure, however, it may be assumed that a small fraction, about 10^{-8}, of the activity inside the glove box will leak out during the course of normal use of the glove box. The laboratory should be prepared to handle such contamination, and the health physicist should be prepared to account for this activity in the design and operation of the surveillance program. For maximum safety, transfer of materials and apparatus into or out of the glove box is always done through an air lock. The viewing panel may be heat-resistant safety plate glass. Glove boxes are unshielded when used for handling radioisotopes that do not create high radiation levels. For radioisotopes that do create high levels of radiation, shielding must be added. When handling a high-energy, beta-emitting radioisotope, it may be necessary to use extra-thick gloves.

When we wish to contain substances that have a high degree of toxicity, we must have a highly effective containment. However, no hood provides absolutely perfect containment; some very small fraction of the noxious material that is emitted in the hood will escape to

Figure 11-2. Glove box for operations with low-intensity radioactive materials that might accidentally become dispersed into the environment if not handled in an enclosed volume. In use, long rubber gloves fit over the port flanges; material transferred into or out of the glove box through the air lock at the right of the glove box. (Used with permission from Innovative Technology, Inc.)

the environment. The effectiveness of a hood is determined by a number of factors, including its physical construction, face velocity, air currents in the laboratory, and so on. Several different methods are used to determine the effectiveness of a hood. All depends on releasing a vapor or a gas in the hood and then measuring the concentration of the test gas in the breathing zone directly in front of the hood.[1] In a commonly used method, a test gas or vapor, such as SF_6, is released in the hood, and its concentration in the exhaust air is measured. At the same time, a breathing-zone measurement is made. The hood protection factor, HPF, is defined as

$$\text{HPF} = \frac{\text{exhaust concentration}}{\text{breathing-zone concentration}}.$$

(11.1)

[1]ANSI/ASHRAE Standard 110-2016. Methods of Testing Performance of Laboratory Fume Hoods.

EXAMPLE 11-1

SF_6 is introduced into a chemical fume hood and is found to have a concentration of 1000 ppm in the hood's exhaust. A breathing zone sample taken and shows a breathing-zone concentration of 1 ppm. Calculate the value of HPF.

Solution

With the aid of Eq. (11.1), we find the HPF to be 1000. Hoods designed for radioisotope use have HPF values on the order of 10,000 or more.

Environmental Control

Environmental control of hazards from radioactive contamination begins with the proper design of the buildings, rooms, or physical facilities in which radioisotopes will be used; it continues with the proper design of the procedures and processes in which radioactivity will be employed. Since a finite probability exists that an accidental breakdown of a mechanical device or a human failure will occur despite the best efforts to prevent such a breakdown, the course of action to be taken in the event of an emergency must be known before the emergency occurs.

In the design of the physical facilities, attention must be paid to the decontaminability of working surfaces, floors, and walls; plumbing and means for monitoring or storing radioactive waste, both liquid and solid; means for incinerating radioactive waste; isotope storage facilities; change rooms and showers; and ventilation and the direction of airflow. Airflow should be directed from office to corridor to area where radioisotopes are handled to exhaust through an air-cleaning system that will assure radiological safety outside the building. Strict control—including monitoring of all persons, materials, and equipment leaving the radiation area—must be maintained over the area where radioisotopes are being used or stored in order to prevent the spread of contamination outside the radiation area. The degree to which each of these control measures is implemented depends, of course, on the types and amounts of isotopes handled and on the consequences of an accidental release of radioactivity to the environment. Radionuclides that are not in use should be stored in a locked cabinet or area to prevent loss of control when the legitimate user is not in the laboratory or in the workplace.

In order to maintain a radiologically safe environment and to prevent internal radiation hazards, good housekeeping and good ventilation must be practiced. In regard to ventilation requirements, several important facts should be emphasized. The first is that fine particles under the influence of gravity do not, for practical purposes, move independently of the air in which they are suspended. Such particles behave effectively as if they were weightless and can be assumed to remain suspended indefinitely in the air. *Control of airborne dust particles, thus, is reduced to a matter of airflow control.*

Control of the Worker: Protective Clothing

Radiation safety philosophy advocates the restriction of radiation exposure to levels as far below the recommended maximums as is reasonably achievable. Since it is extremely difficult to maintain absolute radiological asepsis when working with unsealed sources and the possibility of an accidental spill or release to the environment of radioactivity always exists, it is customary to require radioisotope workers to wear protective clothing in order to prevent contamination of the skin. Workers in nuclear power plants may be simultaneously exposed to multiple hazards, such as heat, noise, electric shock, physical trauma, and chemicals in addition to the radiation environment. They, therefore, may require special protective equipment and clothing, such as heat-resistant garments with built-in cooling systems to prevent heat stress, ear plugs prevent hearing loss, and rubber gloves to prevent electric shock. Clothing worn to prevent skin contamination (anti-C clothing), which may include laboratory coats, coveralls, caps, gloves, and shoes or shoe covers, must be restricted to the contaminated area. Protective clothing is always assumed to be contaminated and therefore must be removed when the worker leaves the contaminated area. To be most effective, the protective clothing should be designed so that the worker can remove it easily and without transferring contamination from the clothing to his or her skin or to the environment. To this end, the worker should be instructed in the proper sequence of removal of the protective clothing before stepping out of the contaminated area into a clean area. Workers should always be monitored before leaving the contaminated area.

Protective clothing, by its very nature, must become contaminated; its main function is to intercept radioactivity that would otherwise contaminate the worker's skin or the clothing worn outside the radioactivity area. The degree of allowable contamination on the protective clothing varies with the type of work that the wearer does. Protective clothing requirements vary by installation; an example of requirements can be found in Table 11-2. Protective clothing can be disposable or laundered and reused.

Laundering contaminated protective equipment is a complex operation. It requires knowledge of the mechanisms of cleaning and of the effects of the cleaning agents on the composition and construction of the protective clothing. For most isotope laboratories, the simplest method for dealing with contaminated protective clothing is to lease the protective clothing from a commercial supplier and to return the contaminated clothing to the supplier. For those installations that do their own laundry, ordinary laundering procedures, using sodium hexa-meta-phosphate or sodium ethylene-diamine-tetra-acetic acid (Na-EDTA) added to the wash water, may facilitate the removal of the contaminants. After laundering, the protective clothing should be monitored to ascertain that it has, in fact, been decontaminated to some previously determined limit. If a piece of protective clothing is unusually or very severely contaminated, it may be simpler to dispose of the item as low-level radioactive waste (LLW) rather than to try to decontaminate it. Unless the wash water meets regulatory requirements for discharge into the sanitary sewer system (such as 10 CFR 20, Appendix B, Table 3), it must be treated as LLW.

TABLE 11-2   Guideline for Protective Clothing Use in Nuclear Facilities

RADIATION TYPE	REMOVABLE (dpm/100 cm^2)	FIXED + REMOVABLE (dpm/100 cm^2)
Beta/Gamma	>1000	>5000
Alpha	>20	>300

Control of the Worker: Respiratory Protection

When a worker is likely to be exposed to airborne radioactivity, respiratory protection must be considered. According to as low as reasonably achievable (ALARA) requirements, the sum of the internal and external doses must be minimized. Wearing a respirator decreases a worker's efficiency by approximately 20–25%.[2] Thus, if exposure to airborne radioactivity occurs simultaneously with external radiation exposure, an ALARA-based decision must be made regarding the use of a respiratory protective device, as explained in Chapter 8.

Medical Assessment

It must be strongly emphasized that a worker must be medically approved for respirator use before being allowed to put on a respirator or being fitted for one. Wearing a respirator effectively increases the volume of the upper respiratory tract, thereby decreasing the volume of air that reaches the deep respiratory tract (where gas exchange occurs). To compensate for this decreased air supply, the body's homeostatic mechanisms increase the respiratory rate and the rate of blood flow through the lungs. These rate increases lead to increased demands on the heart muscle. If the worker's cardiovascular and respiratory systems are in good health, then these increased cardiac demands are safely met. However, if the worker has an impaired cardiovascular system, the heart may not be capable of meeting this increased demand and a heart attack may ensue. For this reason, a worker who may have to wear a respirator on the job *must* be tested and approved for respirator use by a *qualified* physician before he or she is allowed to use a respirator.

Respiratory Protective Devices

The exact type of respiratory protective device that may be required depends on the nature of the airborne contaminant. *Respiratory protective devices may be used only for those hazards for which they are designed.* Half-mask or full-mask facepieces must not leak and must fit properly. Accordingly, the wearer of a respirator must be fit-tested before a respirator is assigned. Respiratory protective devices for radiological protection may be classified into several major categories, as shown in Table 11-3.

Air-purifying respirators remove the contaminant, either by use of a filter for aerosols or by chemical cartridges that remove gases. Because of the specific action of the chemical agents on the contaminant, different canisters must be used for different gases. For this reason, gas masks are not usually recommended for use against radioactive gases.

Supplied air respiratory protective devices may be used against either or both radioactive gases or radioactive aerosols. In this category of protective devices, we have two subcategories: (1) airline hoods, which utilize uncontaminated air under positive (with respect to the atmosphere) pressure supplied from a remote source, and (2) self-contained breathing apparatus (SCBA), in which breathing air is supplied either from a bottle carried by the user or from a canister containing oxygen-generating chemicals. The advantage of the supplied air device is that the pressure in the breathing zone is higher than atmospheric pressure. As a consequence, leakage is from the inside to outside. When using a supplied air device, it is imperative that the time limitation on the supply air is known. It is also imperative that the breathing air coupling *must be* incompatible with couplings for all other gases or nonrespirable laboratory or plant air, in order to prevent such gases from being inadvertently

[2]Johnson AT. Respirator masks protect health but impact performance: a review. *J Biol Eng.* 2016, February 9;10:4.

TABLE 11-3   Protection Factors for Respirators[a]

DESCRIPTION[c]	MODES[d]	PROTECTION FACTORS[b]	
		Particles Only	Particles, Gases, and Vapors
1. Air-purifying respirators			
Facepiece, half-mask	NP	10	
Facepiece, full	NP	100	
Facepiece, half	PP	50	
Facepiece, full & hood	PP	1000	
2. Atmosphere-supplying respirators			
a. Air-line respirator			
Facepiece, half	D		10
Facepiece, half	CF, PD		50
Facepiece, full	D		100
Facepiece, full	CF, PD		1000
Hood	CF		1000
b. Self-contained breathing			
apparatus (SCBA)			
Facepiece, full	D, DR		100
Facepiece, full	PD, RD		10,000
3. Combination respirators: Any combination of air-purifying and atmosphere-supplying respirators	Protection factor for type and mode of operation as listed above		

[a]For use in the selection of respiratory protective devices that are to be used only where the contaminants have been identified and the concentrations (or possible concentrations) are known. Note that iodine and tritium can be absorbed through intact skin.
[b]The protection factor is a measure of the degree of protection afforded by the respirator, defined as the ratio of the concentration of airborne radioactive materials outside the respiratory protective equipment to that inside the equipment (usually inside the facepiece) under conditions of use. It is applied to the ambient airborne concentration to estimate the concentrations inhaled by the wearer according to the following formula:

$$\text{Concentration inhaled} = \frac{\text{Ambient airborne concentration}}{\text{Protection factor}}$$

[c]Only for shaven faces where nothing interferes with the seal of tight-fitting facepieces against the skin. (Hoods and suits are excepted.)
[d]The mode symbols are defined as follows: CF, continuous flow; D, demand; NP, negative pressure (i.e., negative phase during inhalation); PD, pressure demand (i.e., always positive pressure); RD, demand, recirculating (closed circuit); RP, pressure demand, recirculating (closed circuit).

Source: Abstracted from NRC Regulations: 10 CFR Part 20, Appendix A, 2016.

supplied to the airline respirator. Color coding of couplings and outlets for this purpose is *not sufficient*, because a color-blind person would not recognize the color code.

As shown in Example 8.12, use of a respirator in an area where there is both external radiation and airborne radioactivity may actually increase the worker's total effective dose. In instances of both radiation and airborne radioactivity, the health physicist will recommend,

on the basis of total effective dose minimization, whether the worker should wear a respiratory protective device. A worker may insist on wearing a respirator, even if this results in an increased total effective dose, in the mistaken belief that a sievert from internal exposure is more serious than a sievert from external exposure.

SURFACE CONTAMINATION LIMITS

Contamination of personnel and/or equipment may occur either from normal operations or as a result of the breakdown of protective measures. An exact quantitative definition of contamination that would be applicable in all situations cannot be given. Generally, contamination means the presence of undesirable radioactivity—undesirable either in the context of health or for technical reasons, such as increased background, interference with tracer studies, etc. In this discussion, only the health aspects of contamination are considered.

Surface contamination falls into two categories: fixed and loose. In the case of fixed contamination, the radioactivity cannot be transmitted to personnel, and the hazard, consequently, is that of external radiation. For fixed contamination, therefore, the degree of acceptable contamination is directly related to the external radiation dose rate. Setting a maximum limit for fixed surface contamination thus becomes a relatively simple matter. The hazard from loose surface contamination arises mainly from the possibility of tactile transmission of the radioactive contaminant to the mouth or to the skin or of resuspending the contaminant and then inhaling it. It follows that the degree of hazard from surface contamination is strongly dependent on the degree to which the contaminant is fixed to the surface.

Dealing with loose surface contamination limits is not as straightforward as dealing with contamination of air and water. In the case of air and water contamination, safety standards can be easily set—at least in theory—on the basis of recommended dose limits. Using these criteria, we can calculate maximum annual intake of a radionuclide that would lead to the recommended dose limit. From the calculated intake limit, we go one step further from the basic radiation safety criteria and compute the maximum concentrations in air and water which, if continuously inhaled or ingested, would result in the ALI. For the case of surface contamination, we go one more step away from the basic criteria; we try to estimate the surface contamination that, if it were to be dispersed into the environment, would result in concentrations that might lead to an excessive body burden. Thus, specification of limits for loose surface contamination is three steps removed from the basic safety requirements.

From the foregoing discussion, it is clear that limits for surface contamination cannot be fixed in the same sense as limits for the concentration of radionuclides in air and water. Nevertheless, it is useful to compute a number that may serve as a guide in the evaluation of the hazard to workers from surface contamination and to assist the health physicist in deciding whether or not to require the use of special protective measures for workers in contaminated areas.

On the basis of per-unit quantity of radioactivity, inhalation is considered the most serious route of exposure. Surface contamination, therefore, is usually limited by the inhalation hazard that may arise from resuspension of the contaminant. The quantitative relationship between the concentration of loose surface contamination and consequent atmospheric concentration above the contaminated surface due to stirring up the surface is called the *resuspension factor*, f_r, and is defined by

$$f_r = \frac{\text{atmospheric concentration (Bq/m}^3)}{\text{surface concentration (Bq/m}^2)}. \tag{11.2}$$

Experimental investigation of the resuspension of loose surface contamination shows that the resuspension factor varies from about 10^{-4} to 10^{-8} m^{-1}, depending on the conditions under which the studies were conducted. A value of 10^{-6} m^{-1} is reasonable for the purpose of estimating the hazard from surface contamination.

EXAMPLE 11-2

Estimate the maximum surface contamination of "insoluble" strontium 90 (^{90}Sr) dust that may be allowed before taking special safety measures to protect personnel against a contamination hazard.

Solution

The derived atmospheric concentration of ^{90}Sr recommended in ICRP 30 is 60 Bq/m^3 (2×10^{-9} μCi/cm^3). Using a value of 10^{-6} m^{-1} for the resuspension factor in Eq. (11.2), we have

$$10^{-6}\,\text{m}^{-1} = \frac{60\ \text{Bq/m}^3}{\text{surface concentration}}.$$

Therefore, surface concentration $= 60 \times 10^6$ Bq/m^2 = 60 MBq/m^2 (2×10^{-3} μCi/cm^2).

A figure for loose surface contamination calculated by the method of Example 11.2 is intended only as a guide. In any particular case, the health physicist may, at his or her discretion and depending on the nature of the operation, the degree of ventilation, and other relevant factors, such as the area of contamination and the volume of the workroom, generally use more stringent surface-contamination limits for the use of personal protective clothing. On the basis of the ALARA principle, administrative limits for surface contamination are much more restrictive than those calculated using the resuspension factor. Various laboratories and nuclear installations have set their own limits for contamination of surfaces, personnel, equipment, and protective clothing. Tables 11-4 and 11-5 are given to illustrate some of the contamination standards maintained by several large users of radioisotopes.

WASTE MANAGEMENT

Proper collection and management of radioactive waste is an integral part of contamination control and internal (as well as external) radiation protection. In one sense, we cannot "dispose" of radioactive waste. All other types (nonradioactive) of hazardous wastes can be treated, either chemically, physically, or biologically, in order to reduce their toxicity. In the case of radioactive wastes, however, nothing can be done to decrease their radioactivity, and hence their inherent toxic properties. The only means of ultimate disposal is through time—to allow the radioactivity to decay. However, the wastes can be treated and stored in a manner that essentially eliminates their potential threat to the biosphere. Solid and liquid

TABLE 11-4 Recommended Action Levels for Removable Surface Contamination in Manufacturing Plants

TYPE OF SURFACE	TYPE OF RADIOACTIVE MATERIAL			
	ALPHA EMITTERS		Beta- or X-ray Emitters (μCi/cm^2)	Low-Risk Beta- or X-Ray Emitters (μCi/cm^2)
	High Toxicity (μCi/cm^2)	Lower Toxicity (μCi/cm^2)		
1. Unrestricted areas	10^{-7}	10^{-7}	10^{-6}	10^{-6}
2. Restricted areas	10^{-4}	10^{-3}	10^{-3}	10^{-2}
3. Personal clothing worn outside restricted areas	10^{-7}	10^{-7}	10^{-6}	10^{-6}
4. Protective clothing worn only in restricted areas	10^{-5}	10^{-5}	10^{-4}	10^{-4}

Note on Skin Contamination: Skin contamination should always be kept ALARA. Exposed areas of the body of persons working with unsealed radioactive materials should always be monitored and should be washed when any contamination is detected. It is important, however, that contaminated skin should not be so treated or scrubbed that the chance of intake of radioactivity into the body is increased.

High-toxicity alpha emitters include Am-243, Am-241, Np-237, Ac-227, Th-230, Pu-242, Pu-238, Pu-240, Pu-239, Th-228, and Cf-252. Lower toxicity alpha emitters include those having permissible concentrations in air greater than that for Ra-226 (s) in 10 CFR Part 20, Appendix B, Table 1, Column 1. Beta- or X-ray emitter values are applicable for all beta- or X-ray emitters other than those considered low risk. Low-risk nuclides include those whose beta energies are less than 0.2 MeV, whose gamma- or X-ray emission is less than 0.1 R/h at 1 m/Ci, and whose permissible concentration in air in 10 CFR Part 20, Appendix B, Table 1, is greater than 10^{-6} μCi/mL.

Contamination limits for unrestricted (noncontamination-controlled) areas in this table are considered to be compatible in level of safety with those for release of facilities and equipment for unrestricted use, as given in Regulatory Guide 1.86, "Termination of Operating Licenses for Nuclear Reactors," and in "Guidelines for Decontamination of Facilities and Equipment Prior to Release for Unrestricted Use or Termination of Licenses for Byproduct, Source, or Special Nuclear Material," which is available from the Division of Fuel Cycle and Material Safety, Office of Nuclear Material Safety and Safeguards, U.S. Nuclear Regulatory Commission, Washington, DC, 20555.

Averaging is acceptable over inanimate areas of up to 300 cm^2 or, for floors, walls, and ceiling, 100 cm^2. These limits are allowed only in those restricted areas where appropriate protective clothing is worn.

Source: Reprinted with permission from King SH, Granlund RW. Organization and management of a radiation safety office. In: Miller KL, ed. *Handbook of Management of Radiation Protection Programs*. 2nd ed. Boca Raton, FL: CRC Press; 1992.

wastes are treated to minimize their volume, and liquid wastes are converted into solids by such means as vitrification, or by incorporating the liquid either into concrete or asphalt, or into an insoluble plastic. The treated solid waste is packaged in containers according to the class of the waste and buried either in shallow engineered trenches in seismologically and hydrologically stable soil that are then covered with soil, or in deep seismologically and hydrologically stable geologic formations.

Radioactive wastes, which include materials of widely differing types and activities, can originate from any industrial, medical, scientific, university, decommissioning, or agricultural activity in which radioisotopes are used or produced. For regulatory purposes, waste is considered to be radioactive if it contains radionuclides at concentrations or activities greater than those specified by a regulatory authority. For example, the U.S. Nuclear Regulatory Commission (U.S. NRC) regulations state that ^{3}H and ^{14}C in animal tissues and in liquid scintillation media in concentrations not greater than 0.05 μCi (1850 Bq) per gram may be disposed of as if it were not radioactive. It must be emphasized that this definition of radioactive waste is for regulatory purposes only. Waste materials whose activity, quantity, or concentration does not exceed this regulatory lower limit are radioactive from a physical point of view. However, because of their low levels of activity, they are not considered to be hazardous.

TABLE 11-5   Surface Contamination Limits Used by the U.S. Department of Energy

NUCLIDE[a]	REMOVABLE (dpm/100 cm^2)[b,c]	TOTAL FIXED PLUS REMOVABLE (dpm/100 cm^2)
U-natural, U-235, U-238, and associated decay products	1000 alpha	5000 alpha
Transuranics, Ra-226, Ra-228, Th-230, Th-228, Pa-231, Ac-227, I-125, I-129	20	500
Th-natural, Th-232, Sr-90, Ra-223, Ra-224, U-232, I-126, I-131, I-133	200	1000
Beta–gamma emitters (nuclides with decay modes other than alpha emission or spontaneous fission) except Sr-90 and others noted above. Includes mixed fission products containing Sr-90	1000 beta-gamma	5000 beta-gamma
Tritium organic compounds, surfaces contaminated by HT, HTO, and metal tritide aerosols[d]	10,000	—

[a]The values in this table apply to radioactive contamination deposited on but not incorporated into the interior of the contaminated item. Where contamination by both alpha- and beta–gamma-emitting nuclides exists, the limits established for the alpha- and beta–gamma-emitting nuclides apply independently.

[b]The amount of removable radioactive material per 100 cm^2 of surface area should be determined by swiping the area with dry filter or soft absorbent paper while applying moderate pressure and then assessing the amount of radioactive material on the swipe with an appropriate instrument of known efficiency. For objects with a surface area less than 100 cm^2, the entire surface should be swiped, and the activity per unit area should be based on the actual surface area. Except for transuranics, Ra-228, Ac-227, Th-228, Th-230, Pa-231, and alpha emitters, it is not necessary to use swiping techniques to measure removable contamination levels if direct scan surveys indicate that the total residual contamination levels are below the values for removable contamination.

[c]The levels may be averaged over 1 m^2 provided the maximum activity in any area of 100 cm^2 is less than three times the values in Table 11-5.

[d]Tritium contamination including special tritium compounds (STCs) may diffuse into the volume or matrix of materials. Evaluation of surface contamination shall consider the extent to which such contamination may migrate to the surface in order to ensure the surface contamination value provided in this appendix is not exceeded. Once this contamination migrates to the surface, it may be removable, not fixed; therefore, a "Total" value does not apply. In certain cases, a "Total" value of 10,000 dpm/100 cm^2 may be applicable either to metals of the types from which insoluble STCs are formed, that have been exposed to tritium, or to bulk materials to which insoluble STC particles are fixed to a surface.

Source: Adapted from *Radiological Control Manual*. DOE-STD-1098. Washington, DC: U.S. Department of Energy; 2008.

Waste Classification

For purposes of management and treatment, radioactive wastes may be classified according to their activity level and their half-lives. For example, the International Atomic Energy Agency (IAEA) defines high-level radioactive waste (HLW) as: "Waste with levels of activity concentration high enough to generate significant quantities of heat by the radioactive decay process or waste with large amounts of long lived radionuclides that need to be considered in the design of a disposal facility for such waste."[3] Generally, HLW is the waste from reprocessing of spent nuclear fuel or any other waste whose activity is comparable to fuel-reprocessing waste. Low- and intermediate-level wastes are defined by the IAEA as wastes whose radionuclide content and thermal power levels are below those of HLW. In the United States, the classification of radioactive wastes is similar to that of the IAEA. HLW includes reprocessing wastes from the nuclear weapons program and spent nuclear

[3] IAEA General Safety Guide GSG-1. *Classification of Radioactive Waste*. International Atomic Energy Agency, Vienna, 2009.

fuel from commercial nuclear power plants. When it is removed from the nuclear reactor, spent nuclear fuel contains about 1% ^{235}U, about 95% ^{238}U, and about 5% fission products plus transuranic elements, including plutonium isotopes. Although uranium and plutonium are potentially useful, it was decided for sociopolitical reasons not to reprocess spent nuclear reactor fuel in the United States. Accordingly, the Nuclear Waste Policy Act was passed in 1982. This law mandated that HLW and spent nuclear fuel are to be stored, under conditions that would ensure safety for thousands of years, in a geologic repository. Following the IAEA classification system, all radioactive waste in the United States that is not classified as HLW is called *low-level radioactive waste* (LLW). LLW is further subclassified, in 10 CFR 61, on the basis of the specific radioisotopes, their concentration, and their half-lives into class A, class B, and class C, with class A being the lowest level of potential hazard and class C having the highest degree of potential hazard. Class A and class B waste will decay away within 100 years. Wastes whose activity exceeds that of the class C limit are called greater-than-class-C (GTCC) wastes, and are treated like HLW.

Prior to 1980, LLW was put into containers, such as steel drums, after reduction in volume to the minimum practical size and, in the case of liquids, it was immobilized in order to minimize leakage. The drums were then placed into shallow trenches and covered with dirt. Six low-level burial sites were in operation in the United States. In the 1970s, three of these sites were closed because of leakage of radioactivity into the groundwater. As a consequence, and to lessen the burden on the remaining burial sites, the U.S. Congress passed the Low-Level Radioactive Waste Policy Act of 1980. This act made each state responsible for the disposal of waste generated within its borders. However, the states were authorized to form compacts for the establishment and operation of regional facilities for the disposal of LLW generated within the compact states. Currently (2016), the following sites in the United States accept LLW for burial: Barnwell, South Carolina; Hanford, Washington; Andrews, Texas; and Clive, Utah.

When we deal with waste disposal, we mean that there is no expectation of ever recovering it. However, the waste disposal site is not abandoned but is kept under surveillance and governmental control for an appropriate period of time. The long-term surveillance and stringent attention to long-term safety are in accordance with the IAEA's principle: "Radioactive waste shall be managed in such a way that will not impose undue burdens on future generations." The objective of shallow land burial is to confine the radioactivity and prevent it from reaching the biosphere for a long enough time that the radioactivity does not represent an unacceptable risk. The fitness of an LLW disposal site is therefore determined mainly by its hydrogeological characteristics as they relate to the prevention of migration of radioactivity outside of the site or the migration of radioactivity into the groundwater. Any activity that does migrate beyond the limits of the site should be of such low level that it will do no harm to humans or the environment.

Because of the wide range of activity in radioactive waste, ranging from very large amounts from the nuclear fuel cycle (Table 11-6) to the very small amounts from scientific laboratories that use tracer quantities of radioisotopes, several basically different methods are used in the management of radioactive waste. For very large amounts of radioactivity, the general principle is to concentrate and confine the waste, whereas for very small amounts of radioactivity, the waste may be diluted and dispersed. For radionuclides with very short half-lives, the radioactive waste may be stored until the activity is essentially gone. The exact manner of waste management depends on scientific and engineering criteria and on sociopolitical considerations. Included in the first set of criteria are the activity level and

TABLE 11-6   Radioactive Wastes from the Fuel Cycle

	TYPES OF WASTES AND PRINCIPAL CONSTITUENTS	REPRESENTATIVE NUCLIDES	APPROXIMATE RADIOACTIVITY LEVEL	
			Ci/m^3	Bq/m^3
Spent fuel	Fuel withdrawn from a reactor after irradiation	Fission products, ^{85}Kr, ^{133}Xe, ^{131}I, ^{129}I, ^{3}H, and Pu, Am, U	1×10^5 to 2×10^6	3.7×10^{15} to 7.4×10^{16}
High-level waste	Typically waste from reprocessing spent nuclear fuel	Fission products, ^{85}Kr, ^{133}Xe, ^{131}I, ^{129}I, ^{3}H, and Pu, Am, U, Pu	1×10^4 to 2×10^5	3.7×10^{14} to 7.4×10^{15}
Transuranic waste	Anthropogenic elements greater than atomic number 92	Np, Pu, Am, Cm, others	4×10^1 to 1×10^2	1.5×10^{12} to 3.7×10^{12}
Low-level waste	Defined in U.S. regulations as Class A, B, C	^{3}H, ^{14}C, ^{60}Co, ^{137}Cs, other fission products	3×10^3 to 7×10^3	1.1×10^{14} to 2.6×10^{14}
Uranium mill tailings	By products from uranium ore milling processes	Rn, Ra, Th	1×10^{-4} to 1×10^{-3}	3.7×10^6 to 3.7×10^7
Phosphogypsum waste	By product of fertilizer and construction product production	Ra, U, Th, K	6×10^{-6} to 5×10^{-5}	2.2×10^5 to 1.9×10^6
U.S. soils	—	U, Th, K, Ra, Rn	3×10^{-6} to 4×10^{-5}	1.1×10^5 to 1.5×10^6

Adapted from Andrews, A. *Radioactive Waste Streams: Waste Classification for Disposal, RL32163,* Congressional Research Service (CRC) Report for Congress, 2006.

half-life of the waste, the physical quantity of the waste, the nonradioactive matrix in which the radioactivity is dispersed, and whether the waste is in solid, liquid, or gaseous form. The second set of criteria includes consideration of the NIMBY ("not in my back yard") syndrome and the degree of public acceptance of scientifically based public policy decisions.

Mixed Waste

Waste that contains "hazardous waste," as defined by the U.S. Environmental Protection Agency (EPA) regulations in 40 CFR 261, subpart D, as well as radioactive waste is called *mixed waste.* Hazardous waste components are those that possess any one or more of the following characteristics:

- ignitability,
- corrosivity,
- reactivity, and
- toxicity.

A mixed waste encountered by health physicists in research laboratories is scintillation fluid such as benzene, toluene, or xylene containing $\geq$ 0.05 μCi (1850 Bq) of ^{3}H or ^{14}C per gram of liquid. If the activity concentration of an organic liquid scintillation fluid is <0.05 μCi ^{3}H or ^{14}C, it is considered nonradioactive, but still is considered to be a hazardous material on a chemical basis, and must be disposed of accordingly.

High-Level Liquid Wastes

High-level liquid wastes originate mainly as highly acidic solutions from the chemical processing of burned-up fuel, at specific activities of the order of 10^{12} Bq/L (some hundreds of curies per gallon). In the early years of the nuclear industry, these liquids were reduced in volume by evaporation and then stored in underground tanks. The problem of storing high-level liquid waste was complicated by the fact that the rate of heat production due to radioactive transformation is high. If we assume the mean decay energy to be 1 MeV per transformation, then 10^{13} Bq (270 Ci) generates 1 W of power as heat. Provision must therefore be made to remove this decay heat. Storage tanks for HLW were designed for strength, corrosion resistance, heat removal, and monitoring for leakage. Typical tanks, on the order of 3800 m^3 (10^6 gal), are steel-lined reinforced concrete, with an outer steel shell to serve as a backup in case of a leak and an integrated monitoring system for leak detection (Fig. 11-3).

Although storage in tanks was considered practical for short-term containment, it proved not to be adequate for long-term retention, on the order of many centuries. The potential hazards from high-level nuclear waste are twofold. The greatest threat is from indiscriminate release of the fission product waste to the biosphere. If this were to happen, successive bioconcentration of radionuclides by plants and animals in the food web could lead to unacceptably high levels of radioactivity in our food supply, which, in turn, could lead to an unacceptably high internal dose. The second threat, of smaller magnitude than the first, is the external radiation from the radioactive waste. After extensive research directed toward
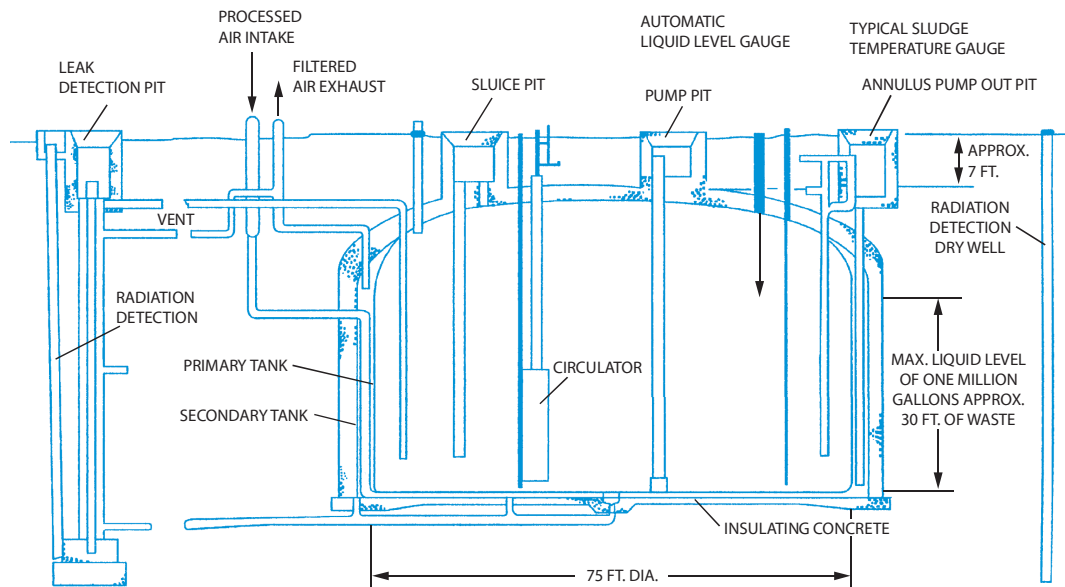
Figure 11-3. Tank for storing high-level liquid radioactive waste. (Reproduced from Roecker JH. *Radioactive Waste Management at Hanford*, 4th rev. Hanford, WA: US Department of Energy; 1979.)

new treatment methods that would remove these threats, it was found that the internal exposure pathway can be blocked by immobilizing the radioactivity in such a manner as to make it unavailable to the biosphere. This is accomplished by the process of vitrification, whereby the radioactive atoms are chemically incorporated into the chemical structure of glass beads, and become part of the glass itself. When this happens, the only way that the radionuclides can get into the biosphere is by dissolution of the glass, which would require time periods measured in geological terms rather than in historical time units. Thus, even if the glass were to escape from its containment into the environment, the radioactivity would remain locked in the glass and would not be available for uptake by flora or fauna. If an animal were to swallow one of these glass beads, the bead would pass through its gastrointestinal (GI) tract and be eliminated. Of course, the GI tract would be irradiated during passage of the bead, but the animal would not absorb any radioactivity from the bead and irradiation of the GI tract would cease when the bead passes out of it. Experience with thorium and obsidian, a naturally occurring volcanic glass, in the Morro do Ferro (Mountain of Iron) in Brazil, proves that glass is stable over geologic time periods. Having thus assured that the HLW will not enter into the biosphere, the second task is to isolate the radioactive beads so that they do not irradiate people or other living creatures. This can be accomplished by putting the glass beads into a suitable container, and then isolating the container deep underground, in a geologically acceptable formation such as a salt dome, or in the deep tunnels that were used for underground nuclear bomb tests, and thus are already radioactive and useless for most other purposes. By these methods, which effectively cut off exposure pathways, inherently toxic HLW can be rendered nonhazardous.

Intermediate- and Low-Level Liquid Wastes

In the past, a common method for the disposal of intermediate- and low-level waste was to discharge it into the sea. The tremendous volume of the ocean seemed to make it an ideal medium for the "dilute-and-disperse" technique for the disposal of low- and intermediate-level waste. Furthermore, since seawater already contains a significant amount of radioactivity, mostly in the form of ^{40}K (it is estimated that the total radioactivity content of the oceans is on the order of 2×10^{22} Bq, or about 500,000 MCi), the addition of relatively small quantities of activity in the form of low- and intermediate-level waste would seem to add very little to the total activity of the ocean. However, because of uncertainties regarding the diverse physical, chemical, and biological processes that govern the distribution of radioisotopes in the sea and the possible transmission of these radioisotopes through the food chain to humans, it is very difficult to specify quantitatively the amounts of radioactivity that may be discharged from any point into the sea. Although undersea investigation of former disposal sites has found no deleterious effects, by an international agreement reached in 1993, the disposal of radioactive wastes into the ocean ceased in 1994.

Disposal of low- and intermediate-level wastes directly into the ground had been practiced where the hydrogeologic factors, the ion-exchange properties of the soil, and the population density were favorable. This method of disposal was called the "delay-and-decay" method, because the slow movement of the radioactivity through the ground affords the radionuclides sufficient time to decay to insignificant levels. Characteristics favorable to ground disposal include a deep water table, good ion-exchange properties of the soil in order to extract and retain relatively large fractions of radionuclides from the liquid waste as it percolates through the ground, few bodies of surface water in order to maximize the time

of underground flow, a large volume of groundwater flow to maximize dilution if the radioactivity should reach the water table, and a very low population density in the area around the ground-disposal site. In the practice of ground disposal, a wood-lined pit, called a "crib," of appropriate capacity was built into the ground and was filled with gravel. The liquid waste was pumped into the crib, from which it slowly percolated down into the ground. Because of uncertainties in hydrogeologic processes and the fear of serious contamination of the groundwater, direct disposal into the ground was halted.

Chemical processes for the decontamination of low- and intermediate-level liquid wastes include the standard methods of waste-water treatment and ion-exchange methods. Hydroxide flocs, which are produced by adding alum or ferric salts to the liquid wastes and then increasing the pH until aluminum or ferric hydroxide is precipitated, are useful for removing cations other than those of the alkaline earths and alkali metals. This treatment is especially effective for removing alpha emitters; it is not very effective for removing ^{90}Sr. Removal of about 95% of ^{90}Sr may be effected with a calcium phosphate floc under highly basic conditions (pH $\sim$ 11.5). Radiostrontium can also be effectively removed by lime-soda softening of the water. The degree of removal of ^{90}Sr is proportional to the degree of softening, since ^{90}SrCO^3 is precipitated with CaCO$_3$. Under certain conditions, liquid wastes may be decontaminated by ion-exchange methods. However, since nonradioactive ions are also adsorbed on the ion exchanger, the effectiveness of this method depends on the relative concentrations of radioactive and nonradioactive ions. Better than 99% reduction in radioactivity can be achieved under optimum conditions.

Water may also be decontaminated by biological means. However, biological removal of radionuclides is less effective than chemical treatment. Its main use, consequently, is for those cases where organic matter must be destroyed, as in sewage treatment or where high concentrations of organic complexing agents make ordinary chemical treatment difficult.

For nonvolatile radioactivity, evaporation is an effective means for decontaminating water. However, evaporation is very energy intensive, since the heat of vaporization of water is 540 cal/g (2.26×10^6 J/kg). Because evaporation requires removal of the solvent or the suspending medium, and since this component of the liquid waste usually accounts for more than 95% of the total pretreated volume, evaporation is a relatively expensive method for the treatment of liquid waste. Evaporation is usually reserved for those cases where a very high degree of decontamination is required. By means of evaporation, decontamination factors on the order of 10^4–10^6 may be obtained at vapor mass velocities ranging from about 20 to 3000 kg/m^2/h. The separated radioactivity, now in a relatively small volume, is processed further for disposal.

After separating the bulk of the radioactivity from the suspending liquid, the decontaminated water may be discharged into the storm sewer if it meets the regulatory requirements for such discharge. Very low level wastes, such as those produced in a laboratory handling trace amounts of radioactivity, may also be candidates for discharge into the sewer. Such discharges cannot be done indiscriminately but are subject to strict regulatory control. One reasonable policy that may be adopted for discharge into the public sewer system is that the quantity released in a day, when diluted by the average daily quantity of flow into the sewer from the institution, must not exceed the regulatory limits, such as those published by the NRC in 10 CFR 20, Appendix B, Table 3.

The NRC and EPA have established three layers of radiation protection limits to prevent potential health threats to the public from exposure to radioactive liquid discharges (effluents) from nuclear power plant operations.

- *Layer 1: 3 mrems/yr ALARA objective—Appendix I to 10 CFR Part 50*
  The NRC requires that nuclear plant operators must keep radiation doses from gas and liquid effluents ALARA to people offsite. For liquid effluent releases, such as diluted tritium, the ALARA annual offsite dose objective is 3 mrems effective dose to the whole body and 10 mrems to any organ of a maximally exposed individual who lives in close proximity to the plant boundary.
- *Layer 2: 25 mrems/yr standard—10 CFR 20.1301(e)*
  In 1979, EPA developed a radiation dose standard of 25 mrems to the whole body, 75 mrems to the thyroid, and 25 mrems to any other organ of an individual member of the public. The NRC incorporated these EPA standards (40 CFR 190.10) into its regulations in 1981, and all nuclear power plants must now meet these requirements. These standards are specific to facilities that are involved in generating nuclear power (commonly called the "uranium fuel cycle"), including where nuclear fuel is milled, manufactured, and used in nuclear power reactors.
- *Layer 3: 100 mrems/yr limit—10 CFR 20.1301(a)(1)*
  The NRC's final layer of protection of public health and safety is a dose limit of 100 mrems/yr to individual members of the public. This limit applies to everyone, including academic, university, industrial, and medical facilities that use radioactive material.

Although discharge of liquid wastes at concentrations within the legally prescribed limits is allowable, it is not societally acceptable. A convenient way to immobilize the radioactivity in low-level liquid waste is to convert the liquid into concrete and then properly package it and dispose of it in a LLW repository. This may be accomplished by using the aqueous waste as the water with which the cement is mixed to make the concrete. The radioactive concrete is then poured into a suitable container, such as a 1- or 5-gal can or a larger steel drum if necessary, and sealed. If the waste is to be transported elsewhere for burial, it must be packaged and marked according to transport regulations, and the radiation dose rate must not exceed 2 mGy (200 mrems) per hour on the surface or 0.1 mGy (10 mrems) per hour at 1 m from the surface.

Airborne Wastes

Airborne radioactivity may be either gaseous or particulate. Gases may arise from neutron activation of cooling air in a reactor and from gaseous fission products as well as from radiochemical reactions in which a gaseous product is produced. Particles may be due to a large variety of processes, ranging from condensate droplets formed during the treatment of high-level liquid wastes to dusts from incinerators in which inflammable solids are burned. Hazards from airborne wastes are best controlled at the source of the waste by limiting the production of airborne wastes. If airborne wastes are produced, the air must be sufficiently decontaminated so that it may be safely diluted and discharged into the atmosphere. If the levels of the airborne radioactivity are sufficiently low, the waste may be diluted to concentrations within the regulatory limits and dispersed into the environment without further treatment.

Gases are usually difficult to remove. For small quantities of iodine and the noble gases, adsorption on a bed of activated charcoal that is located in the negative pressure side of the exhaust duct may be used. Most of the radioactive gaseous wastes of the nuclear power industry are very short-lived. Accordingly, these gases may be compressed and stored in tanks until they decay. Some of the methods used against radioactive gases are summarized in Table 11-7. In many instances, the most expedient method for dealing with radioactive gas is to dilute it with ambient atmospheric air. The diluted mixture is then discharged to the atmosphere from

TABLE 11-7   Treatment Methods for Radioactive Gas

TREATMENT	GAS	EFFICIENCY (%)	VELOCITY (fpm)	PRESSURE DROP (Inches of Water)	COMMENTS
Detention chamber	Noble gases	100	0	0	Use to hold up relatively small volumes
Wet Scrubbing (Spray tower)	Halogens, HF	70–99	50	0.1–1.0	Precleaning or final cleaning for iodine removal
Packed tower	Radioiodine	95–99	50–200	1–10	Heated Berl saddles coated with silver nitrate
Adsorbent beds (Sorption)	Iodine and noble gases	99.95	168	2.8	Activated charcoal or molecular sieves: may be used to decay xenon
					May be refrigerated
Liquifaction column	Noble gases	99.9	—	—	Used to recover small amounts
Stripping column		90–95	—	—	Pilot studies only
Cryogenic trapping (carbon catalyst and carbon pellets)	Xenon, Krypton	99.9	—	—	Liquid nitrogen used for refrigerant
					Gases recovered by desorption

Data from IAEA-TECDOC-1744, *Treatment of Radioactive Gaseous Waste*, IAEA, Vienna, 2014. Table reproduced with permission from Silverman L. Economic aspects of air and gas cleaning for nuclear energy processes. In: *Disposal of Radioactive Wastes*. Vol 1. Vienna, Austria: International Atomic Energy Agency (IAEA); 1960:147.

a high stack in accordance with the appropriate regulatory limits, and atmospheric dispersion further dilutes the radioactivity to still lower levels when the effluent reaches ground level.

Particulate matter may be removed from gases by a variety of different devices, listed in Table 11-8, whose operating principles may be based on gravitational, inertial, electrostatic, thermal, or sonic forces; on physicochemical effects; or on filtration or barrier effects. The collection efficiencies of the different devices vary over a wide range. In considering an air-cleaning device for radioactive dusts, it should be borne in mind that the collection efficiency given by the manufacturer of air-cleaning devices for nonradioactive dusts is usually based on mass collection. Since the mass of a particle is proportional to the cube of its diameter, a single 10-μm particle is equivalent to a thousand 1-μm particles. Table 11-1 shows that the maximum allowable concentration of nonradioactive particles is on the order of a million times or more greater than the allowable concentration for radioactive particles. Air-cleaning devices that are designed to remove much mass from the air, and are thus designated as high-efficiency collectors, may nevertheless be inadequate for respirable radioactive dusts. When this is the case, the final air-cleaning device is usually a high-efficiency filter that is designed for radioactive dusts. The performance of some high-efficiency filters is given in Table 11-9.

The extremely rigorous filtration requirements for radioactive dusts make it desirable to specify the performance of a filter in a more meaningful way than "collection efficiency." Rather than designate the effectiveness of filters by filtration efficiency, in which there appears to be only a small difference between 99.99% and 99.995% (the former passes twice as many particles as the latter, 10 per 100,000 vs. 5 per 100,000), we often used the *decontamination factor* as the figure of merit for a filter. The decontamination factor, df, for a filter whose efficiency is E percent is defined as

$$\text{df} = \frac{100}{100 - E}. \tag{11.3}$$

The filter whose efficiency is 99.99% and thus passes 10 particles per 100,000 has a decontamination factor of 10,000, while the filter of 99.995% efficiency, which passes 5 particles per 100,000, has a decontamination factor of 20,000.

After filtration, the remaining radioactive particles are discharged into the atmosphere for dispersion of the nonfilterable low levels of activity. If the particles are small (i.e., <1 μm), the particulate terminal settling velocity is very low and the particles may be considered as part of the gas in regard to their diffusion into the atmosphere and transport with the gases that issue forth from the exhaust stack.

Meteorological Considerations

When a contaminant is discharged from a chimney, it is assumed that the contaminant will be carried downwind while, at the same time, it diffuses laterally and vertically. The two main consequences of this dispersion in the atmosphere are dilution of the contaminant and its eventual return to the breathing zone at ground level. Of particular interest in evaluating the safety of discharge into the air is the relationships between the rate of discharge and the ground-level concentrations—both in the breathing zone and on the ground (as fallout)—of the discharged radioactivity. The ground-level distribution of the discharged radioactivity depends on a number of factors, including atmospheric stability, wind velocity, type of terrain, the nature of the boundary layer of air (the air layer immediately over the ground for a distance of several hundred feet), and the height of the chimney. It is thus very difficult to predict precisely the pattern of ground-level distribution, although reasonable estimates may be made from one of several different sets of atmospheric diffusion equations.

TABLE 11-8   Basic Characteristics of Air-Cleaning Equipment

TYPE OF EQUIPMENT	PARTICLE SIZE RANGE, MASS MEDIAN (μm)	EFFICIENCY FOR SIZE IN COLUMN 2 (%)	VELOCITY (fpm)	PRESSURE LOSS (Inches of Water)	CURRENT APPLICATION IN U.S. ATOMIC ENERGY PROGRAMS
Simple settling chambers	>50	60–80	25–75	0.2–0.5	Rarely used except for chips and recovery operations
Cyclones, large diameter	>5	40–85	2000–3500 (entry)	0.5–2.5	Precleaners in mining, ore handling and machining operations
Cyclones, small diameter	>5	40–95	2500–3500 (entry)	2–4.5	Same as above
Mechanical centrifugal collectors	>5	20–85	2500–4000	—	Same as large cyclone application
Baffle chambers	>5	10–40	1000–1500	0.5–1.0	Incorporated in chip traps for metal turning
Spray washers	>5	20–40	200–500	0.1–0.2	Rarely used, occasionally as cooling for hot gases
Wet filters	Gases and 0.1–25 μm mists	90–99	100	1–5	Used in laboratory hoods and chemical separation operations
Packed towers	Gases and soluble particles >5	90	200–500	1–10	Gas absorption and precleaning for acid mists Roughing for incinerators
Cyclone scrubber	>5	40–85	2000–3500 (entry)	1–5	Pyrophoric materials in machining and casting operations, mining, and ore handling.
Inertial scrubbers, power-driven	8–10	90–95	—	3–5 HP/1000 cfm	Pyrophoric materials in machining and casting operations, mining and ore handling
Venturi scrubber	>1	99 for H_2SO_4 mist. SiO_2, oil smoke, etc. 60–70	12,000–24,000 at throat	6–30	Incorporated in air-cleaning train of incinerators

Viscous air conditioning filters	10–25	70–85	300–500	0.03–0.15	General ventilation air
Dry spun-glass filters	5	85–90	30–35	0.1–0.3	General ventilation air. Precleaning from chemical and metallurgical hoods
Packed beds of graded glass fibers 1–20 μm 40 in. deep	<1	99.90–99.99	20	10–30	Dissolver off-gas cleaning
High-efficiency cellulose-asbestos filters	<1	99.95–99.98	5 through media 250 at face	1.0–2.0	Final cleaning for hoods, glove boxes, reactor air, and incinerators
All-glass web filters	<1	99.95–99.99	5 through media 250 at face	1.0–2.0	Same as above
Conventional fabric filters	>1	90–99.9	3–5	5–7	Dust and fumes in feed materials production
Reverse-jet fabric	>1	90–99.9	15–50	2–5	Same as above
Single-stage electrostatic precipitator	<1	90–99	200–400	0.25–0.75	Final clean-up for chemical and metallurgical hoods. Uranium machining
Two-stage electrostatic precipitator	<1–5	85–99	200–400	0.25–0.50	Not widely used for decontamination

Reproduced with permission from Silverman L. Economic aspects of air and gas cleaning for nuclear energy processes. In: *Disposal of Radioactive Wastes*. Vol 1. Vienna, Austria: International Atomic Energy Agency (IAEA); 1960:139–179.

TABLE 11-9   Performance of High-Efficiency Filters (at Normal Air Temperatures and Standard-Density Air)

MEDIUM	TEST AEROSOL		AIR VELOCITY		RESISTANCE, (Inches of Water)	EFFICIENCY (%)	METHOD	REMARKS
	Name	Size in μm (homogeneous except[a])	fpm	cm/s				
CC-6	Methylene blue	—	4	2	0.8	99.9871	Discoloration	
	Dioctyl–phthalate (DOP)	0.3	5	2.5	0.67	99.85	Penetrometer	
Cellulose-asbestos paper	Atmospheric dust	0.5[a]	5	2.5	0.67	99.9+	Count	
	Duralumin	0.18	500	250	100	97.7	Count	Note excessive velocity causes greater generation of fine size
	Duralumin	0.18	2	1	0.28	99.7	Count	Reduced velocity improves performance
	Potassium permanganate ($KMnO_4$)	0.02	20	10	2.7	93.0	Count	Size for maximum penetration
AEC No. 1	DOP	0.3	5	2.6	0.7	99.78	Penetrometer	
Cellulose-asbestos	Duralumin	0.18	2	1	0.28	92.9	Count	
	Duralumin	0.18	40	20	5.6	99.6	Count	
	Atmospheric dust	0.58	5	2.5	0.7	99.98	Count	
	$KMnO_4$	0.01	4	2	0.56	91.0	Count	Size for maximum penetration
		0.02						

All-glass superfine fibers—Hurlbut-MSA1106B	DOP	0.3	5	2.5	1.05	99.999	Penetrometer	
	Atmospheric dust	0.5^a	5	2.5	1.05	99.9+	Count	
Sand	KMnO$_4$	0.015	20	10	4.4	93.0	Count	Size for maximum penetration
	Cell ventilation gases	1	3–5	1.5–2.5	4.5–5.5	99.5–99.8	Radioactivity	69 in. deep: graded size from $2\frac{1}{4}$–50 mesh
								Will not withstand high moisture conditions
Composite glass wool	Process off-gases	1	20	10	4.0	99.0		Composition given in reference
	Methylene blue	0.6 MMD	20	10	4.0	99.99	Gravimetric	Same
Compressed glass fibers	Atmospheric dust	0.5	5.25	2.6	0.69	99.997	Count	0.02 in. thick 50% 1.3 µ and 50% 3.0 µ fibers
Resin wool	Atmospheric dust	0.5	14	7	0.3	99.6	Discoloration	These filters are known to decrease in performance when exposed to ionizing radiation
Glass	Uranium oxide	0.12	2.3–7.8	1.2–3.9	0.30–1.23	95.5–99.5	Gravimetric	Special glass formulation developed by A.D. Little. Aluminum separators and furnace cement seals

^{a}Heterogeneous distribution.

Atmospheric stability depends on the temperature gradient of the air (Fig. 11-4). Meteorologists refer to the temperature gradient of the atmosphere as the *lapse rate*. A parcel of air that is rising expands as a result of the decreasing atmospheric pressure. If no heat is gained or lost by this parcel of air, the expansion will be adiabatic, and the temperature of the air parcel will drop. For dry air, this adiabatic cooling results in a temperature decrease of 1°C per 100 m (5.4°F per 1000 ft) of ascent; for average moist air, the lapse rate
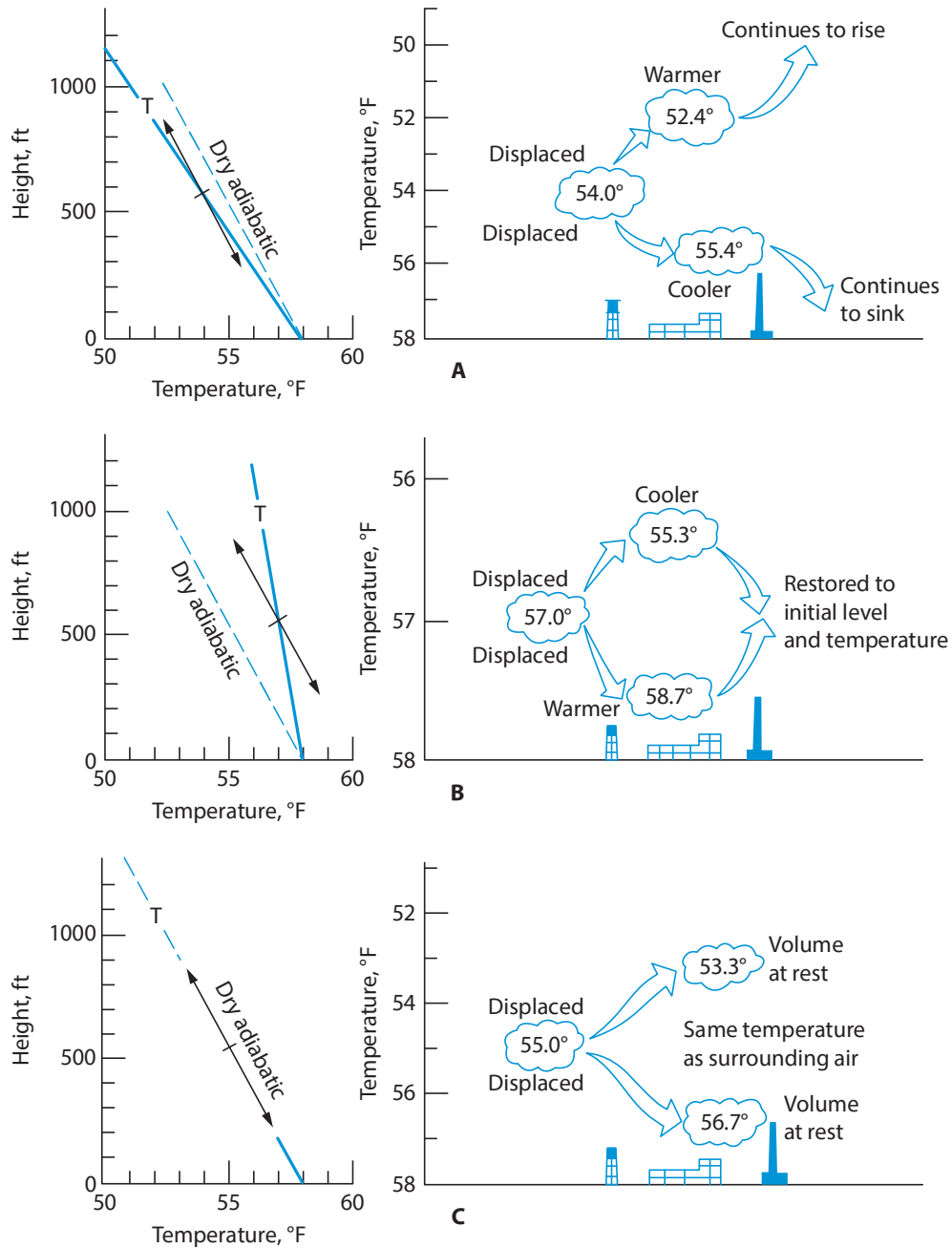
Figure 11-4. Effect of atmospheric temperature gradient—or lapse rate—on a displaced volume of air. (**A**) Unstable lapse rate. (**B**). Stable lapse rate. (**C**). Neutral lapse rate. (Reproduced from *Meteorology and Atomic Energy.* Washington, DC: US Atomic Energy Commission; 1955.)

is 0.65°C per 100 m (3.5°F per 1000 ft). If the temperature gradient of the atmosphere is less than adiabatic but still negative, we have a *stable lapse rate*. In this case, a rising parcel of air cools faster than the surrounding atmosphere. It, therefore, is denser than the air in which it is immersed and tends to sink. A sinking parcel of air is warmer than the surrounding air and thus is less dense, which results in a tendency to rise. A stable lapse rate, therefore, tends to restrict the width of the plume in the vertical direction, thereby decreasing the dilution effect of the atmosphere.

If the lapse rate is positive—that is, if the air temperature increases with increasing height—then the superstable condition known as an *inversion* occurs (since the temperature gradient is "inverted"). The rising effluent from the chimney becomes much denser than the surrounding air as it cools adiabatically and thus sinks. The overall effect of an inversion is to trap the effluent from the chimney and to prevent its ascent to higher altitudes.

A *superadiabatic lapse rate*, one in which the rate of decrease of temperature with increasing height is greater than 1°C per 100 m (5.4°F per 1000 ft), produces an unstable condition that helps to promote vertical dispersion of the contaminated effluent from the chimney. Under the conditions of such an unstable lapse rate, a rising parcel of air does not cool fast enough, because of its adiabatic expansion, and therefore it remains warmer and less dense than the surrounding air and thus continues to rise. By the same reasoning, a falling parcel of air continues to fall.

Dispersion of Gas from a Continuous Source

Although we often speak of atmospheric diffusion, the fact is that very little atmospheric dispersion of gases is due to diffusion. The effects of turbulence are usually so great that molecular diffusion is completely masked. For this reason, estimates of the dispersion of gases in the atmosphere are based on mathematical models that consider the meteorological state of the atmosphere rather than on classical diffusion theory. One of the more commonly used models for estimating the ground-level concentration of a gaseous effluent from a point source, such as a chimney, is the Gaussian plume, straight-line trajectory model (Fig. 11-5). In this model, the contaminant is assumed to be normally distributed around the central axis of the plume in both the vertical and the horizontal directions; it is also assumed that atmospheric stability and wind speed determine the atmospheric dispersion characteristics of the contaminant in the downwind direction. The gas dispersion model is described by the Pasquill–Gifford equation:

$$\chi(x,y) = \frac{Q}{\pi \sigma_y \sigma_z \mu} e^{\left[-\frac{1}{2}\left(\frac{y^2}{\sigma_y^2} + \frac{H^2}{\sigma_z^2} \right) \right]}, \tag{11.4}$$

where

$\chi(x, y)$ = ground level concentration in Bq (or Ci) per cubic meter at point (x, y),

x = downwind distance on plume center line, m,

y = crosswind distance, m,

Q = emission rate, Bq (or Ci) per second of the gas,

σ_y, σ_z = horizontal and vertical standard deviations of contaminant concentration in the plume, m,

μ = mean wind speed at level of plume center line, m/s, and

H = effective chimney height, m.
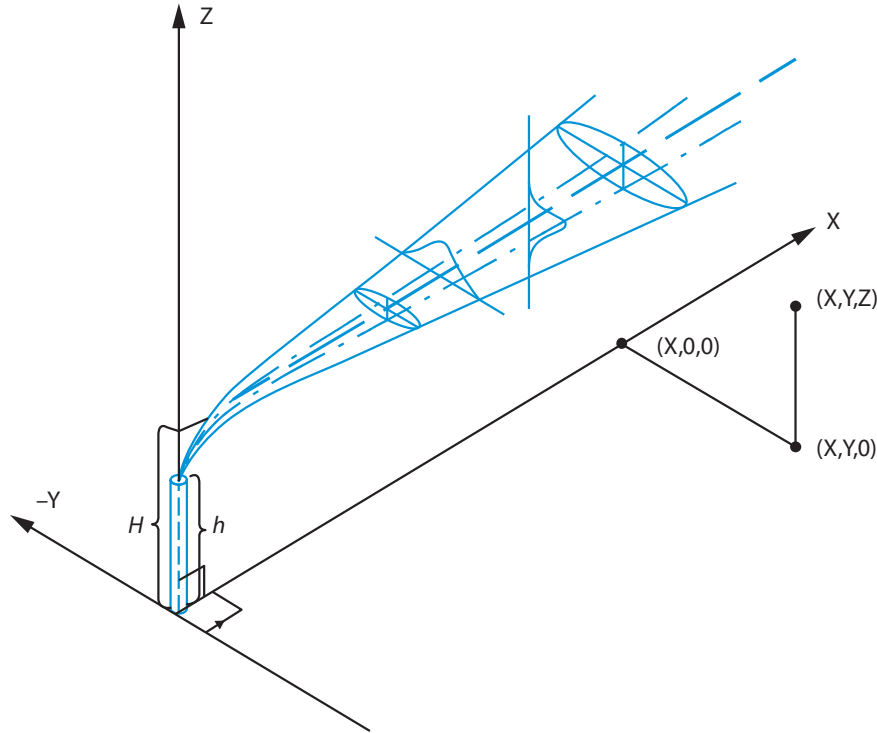
Figure 11-5. Gaussian plume dispersion model for a continuous point source.

If the effluent gas has a significant exit velocity or if it is at a high temperature, it will rise to a level higher than the chimney. The *effective chimney height, H,* therefore, is the sum of the actual chimney height plus a factor that accounts for the exit velocity and the temperature of the effluent gas:

$$H = h + d \left(\frac{v}{\mu}\right)^{1.4} \left(1 + \frac{\Delta T}{T}\right),$$ (11.5a)

where

$h =$ actual chimney height, m,
$d =$ chimney outlet diameter, m,
$y =$ exit velocity of gas, m/s,
$\mu =$ mean wind speed, m/s, at top of chimney,
$\Delta T =$ difference between ambient and effluent gas temperatures, and
$T =$ absolute temperature of effluent gas.

Wind velocity, both speed and direction, changes with increasing height. At the ground surface–air interface, the air speed is zero due to friction. As the height above the surface increases, the frictional effect decreases and the wind speed increases. Because the Earth is rotating, the frictional effect on the surface "drags" the boundary layer of air with it. With increasing height, the decreasing frictional effect results in less "dragging" of the air, and the wind direction changes. In the Northern Hemisphere, the change with respect to the surface direction is clockwise, and in the Southern Hemisphere, the change is in the

counterclockwise direction. ("Ground" level wind speeds are measured at a height of 10 m above ground.) For wind speeds ≥ 6 m/s (13 mph), the directional change is very small; for low wind speeds, the directional changes increase significantly with decreasing wind speed. Changes in wind speed depend on the atmospheric stability and on the surface of the terrain. Air motion is due to atmospheric pressure differences, due to the Coriolis force because of the Earth's rotation, and due to frictional forces. When all these forces are balanced, the resulting wind is called the *gradient* wind. The wind speed μ at a height h may be estimated with the following relationship, where μ_{10} is the "ground" level wind speed at a height of 10 m above the ground:

$$\frac{\mu}{\mu_{10}} = \left(\frac{h \text{ m}}{10 \text{ m}} \right)^{\alpha}, \tag{11.5b}$$

where α is a function of the atmospheric stability:

STABILITY	α
Very unstable	0.02
Neutral	0.14
Very stable	0.5

For example, if the ground level wind speed is 6 m/s in a neutral atmosphere, the wind speed at a height of 60 m is:

$$\mu(60) = 6 \frac{\text{m}}{\text{s}} \cdot \left(\frac{60}{10} \right)^{0.14} = 7.7 \frac{\text{m}}{\text{s}}.$$

Although SI units are shown in Eqs. (11.4) and (11.5), any consistent set of units may be used. The maximum ground-level concentration occurs on the plume center line, at the downwind distance where

$$\sigma_z = \frac{H}{\sqrt{2}}. \tag{11.6}$$

The spread of the plume at any downwind distance is determined by the atmospheric stability, wind speed, and the downwind distance. For purposes of calculating ground-level concentrations with the use of Eq. (11.4), Pasquill proposed the stability categories listed in Table 11-10. For each of the stability categories, the values of the standard deviations in the horizontal and vertical planes through the plume center line, σ_y and σ_z, as a function of downwind distance, are given in Figures 11-6 and 11-7.

The many uncertainties implicit in Eq. (11.4), such as type of terrain, fluctuations in meteorological conditions, etc., lead to a degree of imprecision in the calculated ground-level concentrations. The standard deviation of the calculated concentration is thought to be about a factor of 3. That is, 68 times out of 100, the true concentration can be expected to lie between $x/3$ and $3x$, while 96 times out of 100, the true concentration can be expected to lie between $x/6$ and $6x$.

TABLE 11-10 Pasquill's Categories of Atmospheric Stability

A: Extremely unstable conditions D: Neutral conditions[a]
B: Moderately unstable conditions E: Slightly stable conditions
C: Slightly unstable conditions F: Moderately stable conditions

SURFACE WIND SPEED (m/s)	DAYTIME INSOLATION			THIN OVERCAST or ≥4/8 CLOUDINESS[b]	≤3/8 CLOUDINESS
	Strong	Moderate	Slight		
<2	A	A–B	B		
2	A–B	B	C	E	F
4	B	B–C	C	D	E
6	C	C–D	D	D	D
>6	C	D	D	D	D

[a]Applicable to heavy overcast, day or night.
[b]The degree of cloudiness is defined as that fraction of the sky above the local apparent horizon which is covered by clouds. (Manual of Surface Observations [WBAN], Circular N [7th ed.], paragraph 1210, U.S. Government Printing Office, Washington, July 1960.)

Reproduced from Hilsmeier WF, Gifford FA, Jr. *Graphs for Estimating Atmospheric Dispersion*. Oak Ridge, TN: Oak Ridge National Laboratory; 1962. Report ORO-545.
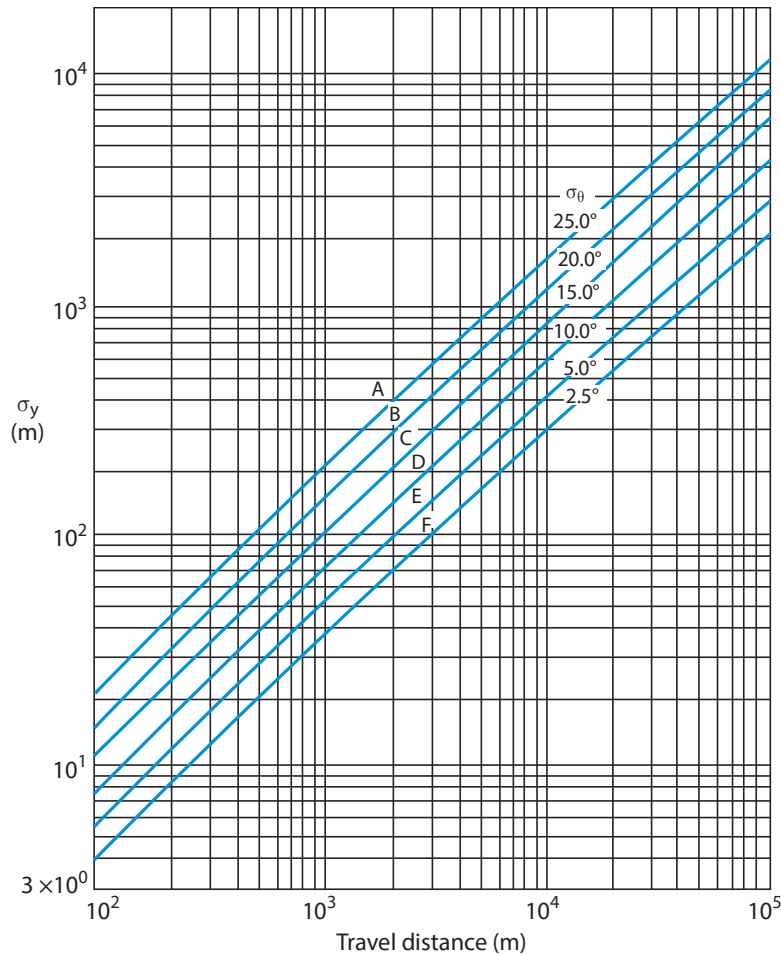
Figure 11-6. Horizontal diffusion standard deviation, σ_y, versus downwind distance from a point source for Pasquill's atmospheric stabilities. (Reproduced from Slade DH, ed. *Meteorology and Atomic Energy*. Washington, DC: US Atomic Energy Commission, Technical Information Division; 1968.)
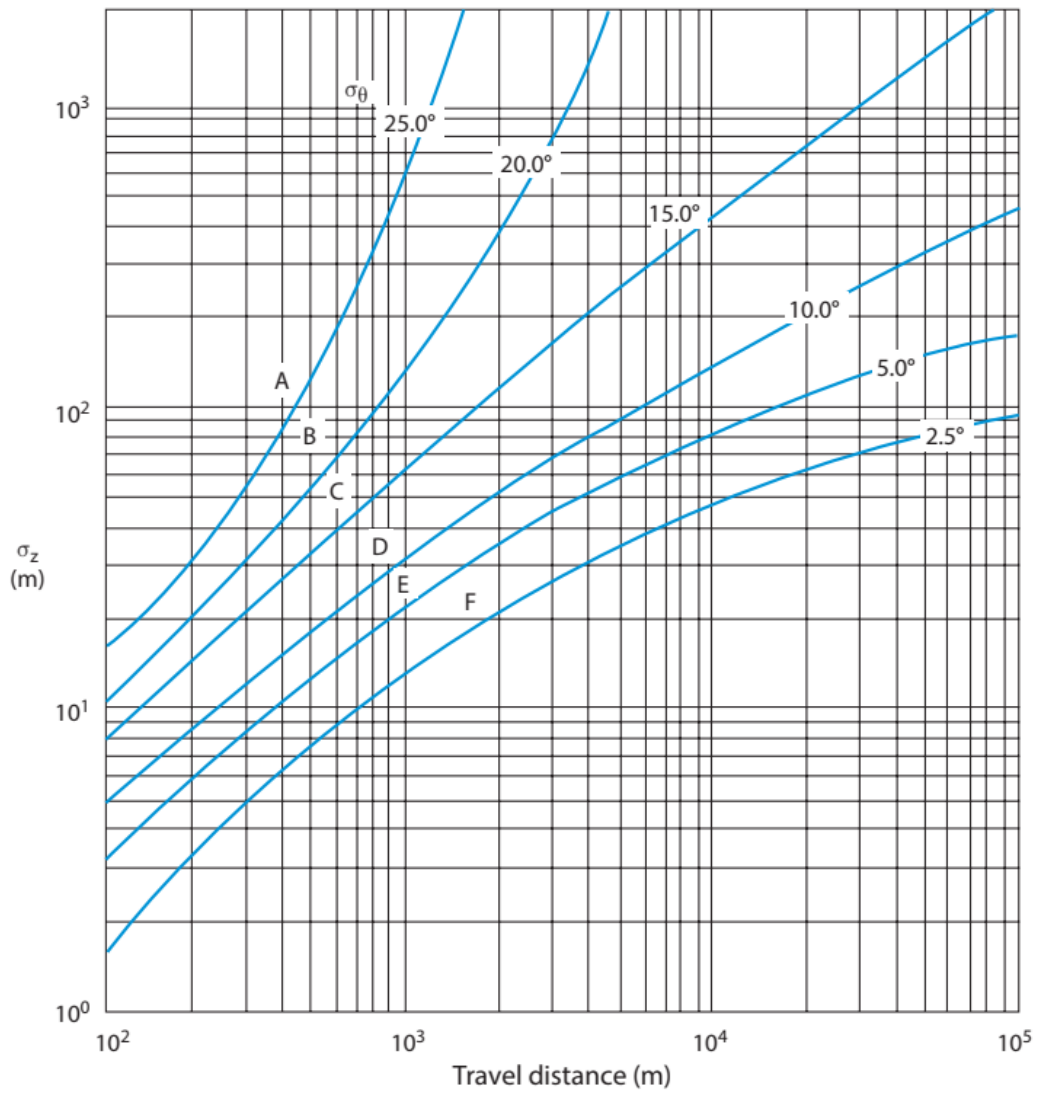
Figure 11-7. Vertical diffusion standard deviation, σ_z, versus downwind distance from a continuously emitting point source for Pasquill's atmospheric stability categories. (Reproduced from Slade DH, ed. *Meteorology and Atomic Energy*. Washington, DC: US Atomic Energy Commission, Technical Information Division; 1968.)

In this calculation, no allowance was made for the fact that ^{41}Ar has a half-life of 110 minutes and thus will undergo a significant decrease in activity during the transit times required for travel of the effluent to the point in question. Generally, if this travel time is significant relative to the half-life of the radioisotope, the emission rate, Q, in Eq (11.4), is multiplied by the decay factor, $e^{-\lambda t}$, where t is the transit time to the point in question.

Particles

Equation (11.4), which gives the ground-level concentration of a gas that is continuously emitted from a point source, is based on total reflection of the gas by the ground. If the pollutant in the plume were retained on the ground, however, as would be true in the case of particles, then the ground-level concentration would be only one-half of the value given by Eq. (11.4):

$$\chi(x, y) = \frac{Q}{2\pi\sigma_y\sigma_z\mu} e^{\left[-\frac{1}{2}\left(\frac{y^2}{\sigma_y^2} + \frac{H^2}{\sigma_z^2}\right)\right]}. \tag{11.7}$$

Furthermore, if a chimney emits particles, then the depletion of the radioactivity in the plume due to gravitational settling, impaction on surfaces protruding from the ground, and precipitation scavenging must be considered in estimating downwind concentrations. Gravitational settling is important for large particles, about 15 μm or larger; impaction and wet deposition are important mechanisms of plume depletion mainly for small particles. Generally, because of the great number of factors that determine depletion of particles, such as particle-size distribution, wetability, solubility, humidity, etc., we can only at best estimate depletion of particles from a plume. The effect of gravitational settling is to tilt the axis of the plume downward through an angle $\theta = \tan^{-1} v_t/\mu$ from the horizontal, where v_t is the terminal settling velocity and μ is the mean wind speed. The effective height of the plume centerline line (H') at any downwind distance x becomes

$$H' = H - x \tan\theta \tag{11.8}$$

or

$$H' = H - \frac{xv_t}{\mu}. \tag{11.9}$$

Equation (11.7) may be modified to estimate the concentration of particles at or near ground level by substituting H', from Eq. (11.9), for H in Eq. (11.7):

$$\chi(x, y) = \frac{Q}{2\pi\sigma_y\sigma_z\mu} e^{\left[-\frac{1}{2}\left(\frac{y^2}{\sigma_y^2} + \frac{\left(H - \frac{xv_t}{\mu}\right)^2}{\sigma_z^2}\right)\right]}. \tag{11.10}$$

The rate of ground deposition, ω, of particles at point (x, y) is found by multiplying the ground-level concentration (Eq. 11.10) by the deposition velocity v_g of the particulate matter:

$$\omega \frac{Bq/m^2}{s} = \chi \frac{Bq}{m^3} \cdot v_g \frac{m}{s}. \tag{11.11}$$

Deposition velocity is determined mainly by micrometeorological conditions near the surface and thus cannot be calculated with any reasonable degree of accuracy. Experimentally determined values range from about 0.1 cm/s to several centimeters per second; an average deposition velocity of about 0.01 m/s is commonly used as a default value.

Solid Wastes

Except for "delay and decay" for short-lived radionuclides (generally less than 120 days half-life), not very much can be done to solid waste to reduce its radioactivity. The main treatment is volume reduction. For noncombustible materials, this can be done by filling a steel drum with the solid waste, covering the drum, and then compressing the drum and its contents into a solid mass with a powerful press. This compressed mass of radioactive waste can then be properly packaged and shipped to a LLW repository for disposal. Combustible waste can be burned. Incineration may either concentrate the activity by burning away the substrate in which the activity—if nonvolatile—is held, or it may disperse the activity with the effluent from the chimney if the activity is volatile or if the contaminated waste is transformed physically into fly ash. Generally, the chimney effluent passes through an air-cleaning device, such as an electrostatic precipitator, before being discharged to the atmosphere. In the instances where the radioactivity is concentrated in the bottom ash or removed from the effluent stream by the electrostatic precipitator, we have a case of volume reduction. The ashes still must be collected and packaged for disposal. (The collection of ashes from an incinerator in which radioactive waste had been burned should be done under the supervision of a health physicist. Appropriate respiratory protective equipment should be available if necessary.) If the activity goes up the stack to be diluted and dispersed in the atmosphere, the rate of incineration of the radioactive waste should be controlled in order to limit the activity discharged from the stack to acceptable levels.

In certain other instances, the radioactivity could be converted into a gas and then discharged to the atmosphere. An illustration of how this may be accomplished within limits prescribed by radiation safety regulations is shown in the example below.

EXAMPLE 11-5

Three hundred and seventy MBq (10 mCi) ^{14}C waste, in the form of 1 g $BaCO_3$, will be disposed of by interacting the $BaCO_3$ with HCl to change the chemical form of the carbon to $^{14}CO_2$ and then discharging the radioactive gas to the atmosphere. The chemical manipulations will be carried out in a fume hood whose face opening is 2 m wide and 0.8 m high

and whose face velocity is 0.5 m/s (100 ft/min). The $^{14}CO_2$ will be vented to the atmosphere through an exhaust stack from the hood. The chemical conversion from the carbonate to the gas will be accomplished by the addition of 1 N HCl. What is the maximum rate at which acid may be added to the $BaCO_3$ if the maximum effluent concentration of 11.1 kBq/m³ $(3 \times 10^{-7} \, \mu Ci/cm^3)$ is not to be exceeded at the discharge end of the exhaust stack?

Solution

The conversion of the carbonate to CO_2 proceeds according to the reaction

$$BaCO_3 + 2HCl \rightarrow BaCl_2 + CO_2 + H_2O.$$
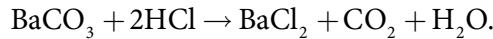
Since the formula weight of $BaCO_3$ is 197.4 (the additional weight due to the ^{14}C is very small, and may be neglected), 1 g $BaCO_3$ is

$$\frac{1 \, g}{197.4 \, \frac{g}{mol}} = 0.00506 \, mol.$$

To convert all the $BaCO_3$ to CO_2, $2 \times 0.00506 = 0.0101$ mol HCl is needed. Since 1-N HCl contains 1 mol acid per liter, the required amount of acid will be contained in 0.0101 L, or 10.1 mL HCl. According to the chemical equation, 1-mol $BaCO_3$ reacts with 2-mol acid to yield 1-mol CO_2. Therefore, if the reaction goes to completion, 0.00506-mol $^{14}CO_2$ will be produced. The gas will occupy a volume, under standard conditions of temperature and pressure, of

$$5.06 \times 10^{-3} \, mol \cdot 22.4 \, \frac{L}{mol} = 0.113 \, L.$$

The specific activity of the $^{14}CO_2$ produced in this reaction is

$$\frac{10 \, mCi \cdot 10^3 \, \frac{\mu Ci}{mCi}}{0.113 \, L} \cdot 10^{-3} \, \frac{L}{mL} = 88.2 \frac{\mu Ci}{mL} \left(3.3 \times 10^{12} \, \frac{Bq}{m^3}\right).$$

According to 10 CFR 20, the maximum permissible effluent concentration for airborne $^{14}CO_2$ is $3 \times 10^{-7} \, \mu Ci/mL$ (11.1×10^3 Bq/m³). The minimum volume of air in which the generated $^{14}CO_2$ must be mixed in order to meet the required concentration at the point of emission is

$$\frac{10 \, mCi \cdot 10^3 \, \frac{\mu Ci}{mCi}}{V \, mL} = \frac{3 \times 10^{-7} \, \mu Ci}{1 \, mL}$$

$$V = 3.33 \times 10^{10} \, mL = 3.33 \times 10^4 \, m^3.$$

The volume of air that flows out of the discharge stock is

$$Q = \text{face area} \cdot \text{velocity}$$

$$= 2 \, m \cdot 0.8 \, m \cdot 30 \frac{m}{min} = 48 \frac{m^3}{min} \quad (1700 \, cfm).$$

If the conversion of the $BaCO_3$ to CO_2 proceeds at a uniform rate of speed, it must take at least

$$\frac{3.33 \times 10^4 \ m^3}{48 \frac{m^3}{min}} = 694 \ min.$$

The 1-N HCl, therefore, must flow into the gas generator at a rate not exceeding

$$\frac{10.1 \ mL}{694 \ min} = 1.46 \times 10^{-2} \frac{mL}{min},$$

or about 69 min/mL.

ASSESSMENT OF HAZARD

A realistic assessment of hazard based on the dose from an internally deposited radioisotope requires more consideration than merely comparing an environmental concentration to legally prescribed limits. The derived air concentrations (DACs) are not maximum permissible concentrations. They are administrative quantities that are used in radiation protection practice for controlling exposure and for dose tracking as a means of demonstrating compliance with the prescribed dose limit (under U.S. regulations and ICRP26/30). Administratively (in U.S. NRC regulations), 1 DAC-hour of exposure to an airborne contaminant corresponds to a committed effective dose equivalent of 0.025 mSv (2.5 mrems). However, the ALIs and their corresponding DACs, with few exceptions, are based on generalized properties of classes of compounds, such as oxides and sulfides, rather than on the metabolic characteristics of specific chemical compounds. This fact, together with the successive rounding of the numbers to one significant figure in the calculational steps leading to the ALIs and DACs, may result in large discrepancies between the dose determined from the actual metabolic behavior and the nominal dose based on the administrative equivalence of 1 DAC-hour to 0.025 mSv (2.5 mrems). If the ALI and DAC are calculated from a physiologically based biokinetic model for a specific compound, as is the case for CO and CO_2, then we might expect reasonably close agreement between the nominal dose based on DAC-hours and the calculated dose. In this regard, the ICRP says that in cases where there are specific data showing that the behavior of any specific material differs significantly from that of the dosimetric model used, then changes should be made in the application of the model to make the model compatible with the specific data.

EXAMPLE 11-6

^{14}CO will be produced in a pilot study in which excess H_2SO_4 will react with H-^{14}COONa, whose specific activity is 47.5 MBq (1.28 mCi) per mmol, to produce ^{14}CO. The NIOSH recommended exposure level (REL) for CO gas, based on its chemical toxicity, is 35 ppm for occupational exposure. The DAC for occupational exposure to ^{14}CO is listed in 10 CFR 20, Appendix B, Table 1, as 7×10^{-4} µCi/mL (26 MBq/m³).

(a) Will the industrial hygiene control that limits CO to 35 ppm be sufficient to meet the regulatory requirements for radiological safety?

(b) The industrial hygienist, believing that control of the ^{14}CO according to the chemical REL is sufficient for the radiological hazard, allows a chemical engineer to be exposed to 35 ppm of the ^{14}CO for a period of 2 hours. What is the chemical engineer's dose commitment as a result of his exposure?

Solution

(a) The molar concentration of CO in the atmosphere corresponding to 35 ppm is

$$35 \text{ ppm} = \frac{35 \text{ mol CO}}{10^6 \text{ mol atmosphere}} = 3.5 \times 10^{-5} \frac{\text{mol CO}}{\text{mol atmosphere}}.$$

Since there is one carbon atom per molecule of sodium formate and also only one carbon atom per CO molecule, the specific activity of the tagged ^{14}CO will also be 47.5 MBq (1.28 mCi) per mmol. The radioactivity concentration corresponding to 35 ppm is therefore

$$\frac{3.5 \times 10^{-5} \dfrac{\text{mol CO}}{\text{mol atm}} \cdot 47.5 \times 10^9 \dfrac{\text{Bq}}{\text{mol CO}}}{22.4 \times 10^{-3} \dfrac{\text{m}^3}{\text{mol atm}}}$$

$$= 7.4 \times 10^7 \frac{\text{Bq}}{\text{m}^3} \left(2 \times 10^{-3} \frac{\mu\text{Ci}}{\text{mL}}\right).$$

Use of industrial hygiene criteria would, in this case, lead to an atmospheric concentration of ^{14}CO, that is

$$\frac{2 \times 10^{-3} \ \mu\text{Ci/mL}}{7 \times 10^{-4} \ \mu\text{Ci/mL}} = 2.9$$

times the DAC limit for continuous exposure.

(b) In order to calculate the absorbed dose, certain facts must be known about the physiological behavior of CO. When CO is inhaled, it diffuses across the capillary bed in the lungs and dissolved in the blood. It is then absorbed by the erythrocytes and combines with the hemoglobin to form carboxyhemoglobin. Since carboxyhemoglobin is incapable of transporting oxygen, the inhalation of CO leads to cellular anoxia, which in turn may lead to unconsciousness or death—depending on the amount of CO that is absorbed into the blood. Hemoglobin has an affinity for CO that is 210 times greater than that for oxygen. The maximum amount of an inhaled gas that can be absorbed, which is called the saturation value, depends on the partial pressure (P) of the gas in the atmosphere. The saturation value for CO, S_∞, as percent hemoglobin tied up as carboxyhemoglobin, is given by

$$S_\infty = \frac{210 \cdot P_{CO}}{(210 \cdot P_{CO}) + P_{O_2}} \cdot 100, \tag{11.12}$$

where P_{CO} is the percent CO in the air, and P_{O_2} is the percent oxygen in the alveolar air (P_{O_2} is usually equal to 15). One hundred percent saturation corresponds to 20 mL CO per 100 mL blood. Rate of absorption, in most cases, follows first-order kinetics, that is, the fractional approach to saturation per unit time remains constant. Thus, if 1% of the saturation value is

absorbed in 1 minute after a person begins inhaling the gas, 1% of the remaining 99% will be absorbed during the second minute, then 1% of the 98.01% left, and so on. Since saturation is approached asymptotically, we usually refer to the *saturation half time* to designate the rate of absorption of an inhaled gas. The numerical value for the saturation half time is independent of the atmospheric concentration of the gas (except for very high concentrations). For CO, the saturation half time is about 47 minutes.[4] The absorption of CO is analogous to the buildup of a radioactive daughter as it approaches secular equilibrium and is described by a similar equation:

$$S = S_\infty \left(1 - e^{-\frac{0.693}{T} \cdot t_i} \right), \tag{11.13}$$

where S_∞ is the saturation value corresponding to a particular atmospheric concentration of the CO; S is the percent of the hemoglobin bound with CO; T is the saturation half time; and t_i is the inhalation time.

For an atmospheric concentration of 35 ppm (35×10^{-6} parts CO per part air), which corresponds to 35×10^{-4}%, or 0.0035%, the hemoglobin saturation value is calculated from Eq. (11.12) to be

$$S_\infty = \frac{210 \cdot 0.0035}{(210 \cdot 0.0035) + 15} \cdot 100 = 4.7\%.$$

After the 2-hour exposure, the percentage of the worker's hemoglobin that is bound with CO is calculated from Eq. (11.13) to be

$$S = 4.7 \cdot \left(1 - e^{-\frac{0.693}{47 \text{ min}} \cdot 120 \text{ min}} \right) = 3.9\%.$$

The blood volume of the reference man is 7.7% of his weight, or 5.4 L for a 70-kg man; it therefore can hold 1/5 of 5.4 L, or 1080 mL CO or oxygen. Since 3.9% of this capacity is tied up with CO, the quantity of CO in the man's body is

$$0.039 \cdot 1080 \text{ mL} = 42 \text{ mL } {}^{14}\text{CO at STP} \quad (0°\text{C and 760 mm Hg}).$$

Since the specific activity of the ^{14}CO is 47.5×10^6 Bq (1.28 mCi) per mmol, the body burden following 2 hours of inhalation is

$$\frac{42 \text{ mL}}{22.4 \dfrac{\text{mL}}{\text{mmol}}} \cdot 47.5 \times 10^6 \frac{\text{Bq}}{\text{mmol}} = 8.9 \times 10^7 \text{ Bq} \quad (2.4 \text{ mCi}).$$

Assuming the blood, and hence the ^{14}C, to be uniformly distributed throughout the body of a 70-kg man, the dose rate due to this body burden of ^{14}C is calculated from Eq. (6.47):

$$\dot{D} = \frac{8.9 \times 10^7 \text{ Bq} \cdot 1 \dfrac{\text{tps}}{\text{Bq}} \cdot 5 \times 10^{-2} \dfrac{\text{MeV}}{\text{transf}} \cdot 1.6 \times 10^{-13} \dfrac{\text{J}}{\text{MeV}} \cdot 3.6 \times 10^3 \dfrac{\text{s}}{\text{h}}}{70 \text{ kg} \cdot 1 \dfrac{\text{J/kg}}{\text{Gy}}}$$

$$= 3.7 \times 10^{-5} \frac{\text{Gy}}{\text{h}} \quad \left(3.7 \frac{\text{mrads}}{\text{h}} \right).$$

[4]G. S. Zavorsky, J. Tesler, J. Rucker, L. Fedorko, J. Duffin, and J. A. Fisher. Rates of carbon monoxide elimination in males and females. *Physiological Reports*. 2(e12237), 2014.

If inhalation had continued until the hemoglobin saturation value were attained, the body burden would have reached

$$\frac{4.7\% \text{ at saturation}}{3.9\% \text{ tied up in CO}} \cdot 8.9 \times 10^7 = 1.1 \times 10^8 \text{ Bq},$$

and the dose rate would have proportionately increased to $\dot{D}_\infty$, the maximum possible value under the conditions of exposure, to 4.6×10^{-5} Gy/h (4.6 mrads/h). In this case, the body burden, and hence the dose rate, varied with time, as shown in Figure 11-8. The instantaneous dose rate during the period of inhalation (period I in Fig. 11-8) is given by

$$\dot{D} = \dot{D}_\infty \left(1 - e^{-kt_i} \right), \tag{11.14}$$

where k is the carboxyhemoglobin dissociation rate constant, $0.693/T$, and t_i is the inhalation time. The total dose during the period of inhalation is
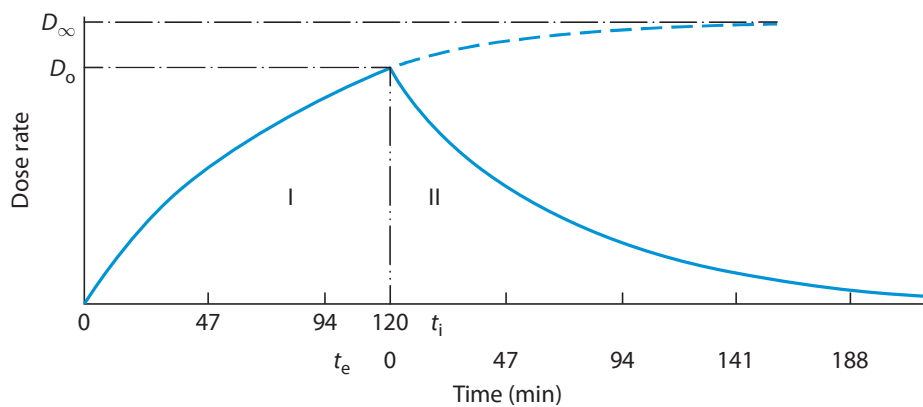
$$D_i = \dot{D}_\infty \int_0^{t_i} \left(1 - e^{-kt_i} \right) dt, \tag{11.15}$$

which, when integrated, yields

$$D = \dot{D}_\infty \left[t_i + \frac{1}{k} \left(e^{-kt_i} - 1 \right) \right]. \tag{11.16}$$

If we substitute the previously calculated values into Eq. (11.16) in order to calculate the absorbed dose during the 2-hour inhalation period, we have

$$D = 4.5 \times 10^{-5} \frac{\text{Gy}}{\text{h}} \left[2 \text{ h} + \frac{1}{\left(\frac{0.693}{47 \text{ min}} \cdot \frac{60 \text{ min}}{\text{h}} \right)} \left(e^{-\left[\frac{0.693}{(47 \text{ min})} \cdot \frac{60 \text{ min}}{\text{h}} \right] \cdot 2 \text{ h}} - 1 \right) \right]$$

$$D = 4.8 \times 10^{-5} \text{ Gy} \quad (4.8 \text{ mrads}).$$

Figure 11-8. Variation of dose rate with time after start of ^{14}CO inhalation. Region I under the curve represents the period of inhalation, and region II represents the period of exhalation. $\dot{D}_0$ is the dose rate at the end of the inhalation period, and $\dot{D}_\infty$ is the dose rate due to the saturation amount of radioactive CO.

The dose absorbed during the time period t_e, when the CO leaves the blood (area II in Fig. 11-8), is given by Eq. (6.58) as

$$D = \frac{\dot{D}_0}{\lambda_E}.$$

When 3.7×10^{-5} Gy/h (dose rate at the end of the 2-hour exposure to ^{14}CO) is substituted into Eq. (6.58) for the initial dose rate for the exhalation period in Figure 11-8, the dose during the CO elimination period is calculated as follows:

$$D = \frac{3.7 \times 10^{-5} \frac{Gy}{h}}{\left(\frac{0.693}{47 \text{ min}} \cdot \frac{60 \text{ min}}{h} \right)} = 4.2 \times 10^{-5} \text{ Gy} \quad (4.2 \text{ mrads}).$$

The dose commitment due to the 2-hour-long inhalation of the ^{14}CO is the sum of the doses during inhalation and elimination, which is equal to 9×10^{-5} Gy (9 mrads). Since the dose equivalent is numerically equal to the dose for beta radiation, the committed effective dose equivalent (CEDE) from this exposure is 0.09 mSv or 9 mrems. The nominal CEDE from a 2-hour exposure at 2.9 DACs at 0.025 mSv per DAC-hour (2.5 mrems per DAC-hour) is

$$CEDE = 2 \text{ h} \cdot 2.9 \text{ DAC} \cdot 0.025 \frac{mSv}{DAC\text{-}h} = 0.15 \text{ mSv} \ (15 \text{ mrems}).$$

This example shows that if available, biologically based biokinetic information rather than the assumption that 1 DAC-hour is equivalent to 2.5 mrems (0.025 mSv) should be used to estimate the dose from the intake of a radionuclide.

Maximum Credible Accident

The concept of a maximum credible accident is useful in advance planning for the purpose of minimizing radiation dose in the event of an accident or for designing safety limitations into an experiment. Consider the following example.

EXAMPLE 11-7

An engineer wishes to use tritiated water in an experimental study of a closed pressurized system. The system's capacity is 3-L water, which will be kept at a temperature of 150°C (302°F). The experiment will be done in a ventilated laboratory whose dimensions are 3 m × 3 m × 3 m. The maximum credible accident is one in which the system will rupture, and the entire 3 L of tritiated water will be sprayed into the room as steam. The laboratory ventilation rate is 7 m³/min (250 cfm). The laboratory has its own exhaust line and stack, so there is no possibility of spreading the tritium, in the event of an accident, to other laboratories in the building.

What is the maximum amount of tritium, as tritiated water, ^{3}HOH, that may be in the system, assuming a maximum credible accident, if the engineer is not to inhale more tritium than that which would deliver a dose of 30 mGy (3 rads) over a period of 13 weeks. In the event of such an accident, it is estimated that the engineer might remain in the laboratory for as long as 2 minutes.

For tritium:

(a) the critical organ is the total body, weight $= 70$ kg;
(b) the biological half-time is 12 days;
(c) the radiological half-life is 12.3 years;

    (i) the effective half-life
    (ii) the effective elimination rate

(d) pure beta emitter, average beta energy $= 0.006$ MeV; and
(e) all the inhaled tritium is assumed to be absorbed.

Solution

The initial dose rate, $\dot{D}_0$, that will result in a total dose of 30 mGy (3000 mrads) over a period of 13 weeks (91 days) is calculated from Eq. (6.57):

$$30 \text{ mGy} = \frac{\dot{D}_0}{0.0578 \text{ d}^{-1}}\left(1 - e^{-0.0578 \text{ d}^{-1} \cdot 91 \text{ d}}\right)$$

$$\dot{D}_0 = 1.74 \text{ mGy/d} \quad (174 \text{ mrads/d}).$$

The body burden, q Bq, that will deliver this initial dose rate is calculated from Eq. (6.47), and is found to be

$$1.74 \times 10^{-3} \text{ Gy/d}$$

$$= \frac{q \text{ Bq} \cdot 1\frac{\text{tps}}{\text{Bq}} \cdot 6 \times 10^{-3} \frac{\text{MeV}}{\text{transf}} \cdot 1.6 \times 10^{-13} \frac{\text{J}}{\text{MeV}} \cdot 8.64 \times 10^4 \frac{\text{s}}{\text{d}}}{70 \text{ kg} \cdot 1\frac{\text{J/kg}}{\text{Gy}}}$$

$$q = 1.47 \times 10^9 \text{ Bq} \quad (39.7 \text{ mCi}).$$

If the entire 3 L (assume that 3 L $= 3$ kg) of water were vaporized, the density of the steam would be

$$\rho(\text{steam}) = \frac{3 \text{ kg}}{27 \text{ m}^3} = 0.11 \frac{\text{kg}}{\text{m}^3}.$$

For the density of water vapor to be this high, the temperature must be approximately 56°C (132°F). This is an unreasonably high ambient temperature. If we assume an ambient temperature of 38°C (100°F), then the saturated water vapor density is 0.0463 kg/m³, and the amount of water in the air is

$$\frac{0.0463 \frac{\text{kg}}{\text{m}^3}}{0.11 \frac{\text{kg}}{\text{m}^3}} \cdot 3 \text{ kg} = 1.26 \text{ kg}.$$

The tritium activity in this amount of water must be restricted to that quantity that would lead to an inhalation of no more than 1.47×10^9 Bq (39.7 mCi) during 2 minutes of breathing air saturated with the tritiated water vapor at a temperature as high as 38°C.

Assume that the worker's breathing rate is 20 respirations per minute and that the tidal volume is 0.5 L. If there were no ventilation and the concentration of the tritium had remained constant, the atmospheric concentration C Bq/L that would lead to the maximum acceptable body burden is

$$C\,\frac{\mathrm{Bq}}{\mathrm{L}} \cdot \text{respiration rate,} \ \frac{\mathrm{L}}{\mathrm{min}} \cdot \text{exposure time, min} = q \ \mathrm{Bq}.$$

However, the atmospheric concentration does not remain constant; the ventilation system changes the air of the laboratory at a turnover rate k of

$$k = \frac{7\,\dfrac{\mathrm{m}^3}{\mathrm{min}}}{3\,\mathrm{m} \cdot 3\,\mathrm{m} \cdot 3\,\mathrm{m}} = 0.26 \ \mathrm{min}^{-1}.$$

The atmospheric concentration of tritium in the laboratory C at any time t after release, assuming instantaneous release and uniform concentration C_0, is given by

$$C = C_0 e^{-k\,\mathrm{min}^{-1} \cdot t\,\mathrm{min}}, \tag{11.17}$$

where k is the turnover rate of the air. The total amount of inhaled tritium, during any exposure at a mean respiration rate of RR, is

$$q = \mathrm{RR} \cdot C_0 \int_0^t e^{-kt} \ dt, \tag{11.18}$$

which yields, upon integration,

$$q = \mathrm{RR} \cdot \frac{C_0}{k} \left(1 - e^{-kt}\right). \tag{11.19}$$

In the case under consideration, the respiration rate is 0.5 L per inspiration × 20 inspirations per minute $= 10$ L min^{-1}. Substituting into Eq. (11.19) to solve for C_0, we have

$$1.47 \times 10^9 \ \mathrm{Bq} = 10\,\frac{\mathrm{L}}{\mathrm{min}} \cdot \frac{C_0\,\dfrac{\mathrm{Bq}}{\mathrm{L}}}{0.26\,\mathrm{min}^{-1}} \left(1 - e^{-0.26\,\mathrm{min}^{-1} \cdot 2\,\mathrm{min}}\right)$$

$$C_0 = 9.43 \times 10^7 \ \mathrm{Bq/L} \quad (2.55 \ \mathrm{mCi/L}).$$

Since the room volume is 27 m^3, or 27,000 L, and since only 1.26 kg of the 3 kg of water will be in the vapor state, the maximum amount of tritium that may be contained in the tritiated water is

$$\frac{3\,\mathrm{kg}}{1.26\,\mathrm{kg}} \cdot 9.43 \times 10^7\,\frac{\mathrm{Bq}}{\mathrm{L\ vapor}} \cdot 2.7 \times 10^4 \ \mathrm{L\ vapor} = 6.1 \times 10^{12} \ \mathrm{Bq} \quad (164 \ \mathrm{Ci}).$$

According to these calculations the use of no more than 6.1×10^{12} Bq (164 Ci) tritium in the 3 L of tritiated water would ensure against overexposure in the event of the maximum credible accident. If this is not enough activity for the purpose of the experiment, then additional precautions, such as enclosure of the process or increased ventilation, would have to be employed.

Radon

Radon gas is a naturally occurring radioactive gas that occurs in the uranium ($4n + 2$ series) and the thorium ($4n$ series) as described in Chapter 4. Since both these series are ubiquitous, radon gas is produced, to a lesser or greater degree in all soils, depending on the concentration of uranium or thorium. The gas diffuses out of the ground and into the air. Because of the relatively slow diffusion rate and the short half-lives of the two Rn isotopes, most of the radon that is exhaled from the ground originates in the top 30 cm (1 foot) of soil. Radon exhalation from the soil is on the order of several picocuries per square meter per second; the exact rate depends on the concentration of the uranium and thorium series precursors in the soil. The concrete foundation or slab on which a building rests acts as a barrier against the radon. However, radon gas can gain entry into the building by diffusing through the concrete or through cracks that may occur in the concrete. Entry rates through such concrete barriers are usually less than one-half the exhalation rate from the soil. Once in the house, air exchange by general ventilation and radioactive decay will lead to a radon and radon daughter concentration that depends on the rate of radon influx into the house, on the ventilation rate, and on the radon concentration in the outside air with which the house is ventilated. In the United States, the average outdoor Rn concentration is 0.4 pCi/L (15 Bq/m^3) and the mean indoor concentration is 0.67 pCi/L (24.3 Bq/m^3).[5]

In general ventilation, the concentration C to which the airborne contaminant is reduced is determined by the rate of generation G of the contaminant and the ventilation rate Q. The rate of change of the quantity of contaminant in the air is simply the difference between the rate of generation and the rate of removal of the contaminant. If we have a workplace whose volume is V, the rate of change of the quantity of airborne radioactivity is as follows:

Activity change $=$ generated activity $-$ ventilated activity $-$ decayed activity

$$V\,\mathrm{d}C = [G - (CQ) - (\lambda VC)]\mathrm{d}t \tag{11.20}$$

$$V\,\mathrm{d}C = [G - C(Q + \lambda V)]\mathrm{d}t. \tag{11.21}$$

Separating the variables gives

$$\int_{C_0}^{C} \frac{\mathrm{d}C}{G - C(Q + \lambda V)} = \int_{0}^{t} \frac{1}{V}\mathrm{d}t \tag{11.22}$$

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[5]Health Effects of Exposure to Radon: BEIR VI. National Research Council, National Academy of Sciences, Washington DC, 1999.

and integrating Eq. (11.22) gives

$$C(t) = \frac{G}{Q + \lambda V}\left[1 - e^{-\left(\frac{Q+\lambda V}{V}\right)t}\right] + C_0 e^{-\left(\frac{Q+\lambda V}{V}\right)t}. \tag{11.23}$$

As $t \to \infty$, $\exp\left[-\left(\frac{Q+\lambda V}{V}\right)t\right] \to 0$, and Eq. (11.23) approaches the steady-state concentration of the contaminant:

$$C(\text{steady state}) = \frac{G}{Q + \lambda V}. \tag{11.24}$$

If λV is much less than Q, then Eq. (11.24) reduces to

$$C(\text{steady state}) = \frac{G}{Q}. \tag{11.25}$$

The effective rate of generation of the contaminant is increased if the ventilating air also contains the contaminant. The rate of introduction of the contaminant in the ventilating air (outdoor radon in this case) into the ventilated space is the product of the concentration in the ventilating air, C_v, and the ventilation rate, Q. Thus, the steady-state concentration becomes

$$C(\text{steady state}) = \frac{G + (C_v Q)}{Q}. \tag{11.26}$$