6 FIELD MEASUREMENT METHODS AND INSTRUMENTATION

6.1 Introduction

Measurement is used in MARSSIM to mean 1) the act of using a detector to determine the level or quantity of radioactivity on a surface or in a sample of material removed from a media being evaluated, or 2) the quantity obtained by the act of measuring. Three methods are available for collecting radiation data while performing a survey—direct measurements, scanning, and sampling. This chapter discusses scanning and direct measurement methods and instrumentation. The collection and analysis of media samples are presented in Chapter 7. Information on the operation and use of individual field and laboratory instruments is provided in Appendix H. Quality assurance and quality control (QA/QC) are discussed in Chapter 9.

Total surface activities, removable surface activities, and radionuclide concentrations in various environmental media (*e.g.*, soil, water, air) are the radiological parameters typically determined using field measurements and laboratory analyses. Certain radionuclides or radionuclide mixtures may necessitate the measurement of alpha, beta, and gamma radiations. In addition to assessing each survey unit as a whole, any small areas of elevated activity should be identified and their extent and activities determined. Due to numerous detector requirements, no single instrument (detector and readout combination) is generally capable of adequately measuring all of the parameters required to satisfy the release criterion or meet all the objectives of a survey.

Selecting instrumentation requires evaluation of both site and radionuclide specific parameters and conditions. Instruments should be stable and reliable under the environmental and physical conditions where they are used, and their physical characteristics (size and weight) should be compatible with the intended application. The instrument and measurement method should be able to detect the type of radiation of interest, and should, in relation to the survey or analytical technique, be capable of measuring levels that are less than the derived concentration guideline level (DCGL). Numerous commercial firms offer a wide variety of instruments appropriate for the radiation measurements described in this manual. These firms can provide thorough information regarding capabilities, operating characteristics, limitations, *etc.*, for specific equipment.

If the field instruments and measurement methods cannot detect radiation levels below the DCGLs, laboratory methods discussed in Chapter 7 are typically used. A discussion of detection limits and detection levels for some typical instruments is presented in Section 6.7. There are certain radionuclides that will be essentially impossible to measure at the DCGLs *in situ* using current state-of-the-art instrumentation and techniques because of the types, energies, and abundances of their radiations. Examples of such radionuclides include very low energy, pure beta emitters such as ^{3}H and ^{63}Ni and low-energy photon emitters such as ^{55}Fe and ^{125}I. Pure alpha emitters dispersed in soil or covered with some absorbing layer may not be detectable because alpha radiation will not penetrate through the media or covering to reach the detector. A

common example of such a condition would be ^{230}Th surface contamination, covered by paint, dust, oil, or moisture. NRC report NUREG-1507 (NRC 1997a) provides information on the extent to which these surface conditions may affect detection sensitivity. In circumstances such as these, the survey design will usually rely on sampling and laboratory analysis to measure residual activity levels.

6.2 Data Quality Objectives

The third step of the Data Quality Objectives (DQO) Process involves identifying the data needs for a survey. One decision that can be made at this step is the selection of direct measurements for performing a survey or deciding that sampling methods followed by laboratory analysis are necessary.

6.2.1 Identifying Data Needs

The decision maker and the survey planning team need to identify the data needs for the survey being performed, including the:

- type of measurements to be performed (Chapter 5)
- radionuclide(s) of interest (Section 4.3)
- number of direct measurements to be performed (Section 5.5.2)
- area of survey coverage for surface scans based on survey unit classification (Section 5.5.3)
- type and frequency of field QC measurements to be performed (Section 4.9)
- measurement locations and frequencies (Section 5.5.2)
- standard operating procedures (SOPs) to be followed or developed (Chapter 6)
- analytical bias and precision (*e.g.*, quantitative or qualitative) (Appendix N, Section N.6)
- target detection limits for each radionuclide of interest (Section 6.4)
- cost of the methods being evaluated (cost per measurement as well as total cost) (Appendix H)
- necessary turnaround time
- specific background for the radionuclide(s) of interest (Section 4.5)
- derived concentration guideline level (DCGL) for each radionuclide of interest (Section 4.3)
- measurement documentation requirements
- measurement tracking requirements

Some of this information will be supplied by subsequent steps in the DQO process, and several iterations of the process may be needed to identify all of the data needs. Consulting with a health physicist or radiochemist may be necessary to properly evaluate the information before deciding

between direct measurements or sampling methods to perform the survey. Many surveys will involve a combination of direct measurements and sampling methods, along with scanning techniques, to demonstrate compliance with the release criterion.

6.2.2 Data Quality Indicators

The data quality indicators identified as DQOs in Section 2.3.1 and described in Appendix N should be considered when selecting a measurement method (*i.e.*, scanning, direct measurement, sampling) or a measurement system (*e.g.*, survey instrument, human operator, and procedure for performing measurements). In some instances, the data quality indicator requirements will help in the selection of a measurement system. In other cases, the requirements of the measurement system will assist in the selection of appropriate levels for the data quality indicators.

6.2.2.1 Precision

Precision is a measure of agreement among replicate measurements of the same property, under prescribed similar conditions (ASQC 1995). Precision is determined quantitatively based on the results of replicate measurements (equations are provided in EPA 1990). The number of replicate analyses needed to determine a specified level of precision for a project is discussed in Section 4.9. Determining precision by replicating measurements with results at or near the detection limit of the measurement system is not recommended because the measurement uncertainty is usually greater than the desired level of precision. The types of replicate measurements applied to scanning and direct measurements are limited by the relatively uncomplicated measurement system (*i.e.*, the uncertainties associated with sample collection and preparation are eliminated). However, the uncertainties associated with applying a single calibration factor to a wide variety of site conditions mean these measurements are very useful for assessing data quality.

- Replicates to Measure Operator Precision. For scanning and direct measurements, replicates to measure operator precision provide an estimate of precision for the operator and the Standard Operating Procedure (SOP) or protocol used to perform the measurement. Replicates to measure operator precision are measurements performed using the same instrument at the same location, but with a different operator. Replicates to measure operator precision are usually non-blind or single-blind measurements.

- Replicates to Measure Instrument Precision. For scanning and direct measurements, replicates to measure instrument precision provide an estimate of precision for the type of instrument, the calibration, and the SOP or protocol used to perform the measurement. Replicates to measure instrument precision are measurements performed by the same operator at the same location, but with a different instrument. Replicates to measure instrument precision are usually non-blind or single-blind measurements.

For many surveys a combination of instrument and operator replicates are used to provide an estimate of overall precision for both scanning and direct measurements. Replicates of direct measurements can be compared with one another similar to the analytical results for samples. Results for scanning replicates may be obtained by stopping and recording instrument readings at specific intervals during the scanning survey (effectively performing direct measurements at specified locations). An alternative method for estimating the precision of scanning is to evaluate the effectiveness of the scanning survey for identifying areas of elevated activity. The results of scanning are usually locations that are identified for further investigation. A comparison of the areas identified by the replicate scanning surveys can be performed either quantitatively (using statistical methods) or qualitatively (using professional judgment). Because there is a necessity to evaluate whether the same number of locations were identified by both replicates as well as if the identified locations are the same, there is difficulty in developing precision as a DQO that can be evaluated.

6.2.2.2 Bias

Bias is the systematic or persistent distortion of a measurement process that causes error in one direction (EPA 1997a). Bias is determined quantitatively based on the measurement of materials with a known concentration. There are several types of materials with known concentrations that may be used to determine bias for scans and direct measurements.

- Reference Material. Reference material is a material or substance one or more of whose property values are sufficiently homogeneous and well established to be used for the calibration of an apparatus, the assessment of a measurement method, or for assigning values to materials (ISO 1993). A certified reference material is reference material for which each certified property value is accompanied by an uncertainty at a stated level of confidence. Radioactive reference materials may be available for certain radionuclides in soil (*e.g.*, uranium in soil), but reference building materials may not be available. Because reference materials are prepared and homogenized as part of the certification process, they are rarely available as double-blind samples. When appropriate reference materials are available (*i.e.*, proper matrix, proper radionuclide, proper concentration range) they are recommended for use in determining the overall bias for a measurement system. For scanning and direct measurements a known amount of reference material is sealed in a known geometry. This known material is measured in the field using a specified protocol (*e.g.*, specified measurement time at a specified distance from the reference material) to evaluate the performance of the instrument only.

- Performance Evaluation (PE) Samples. PE samples are used to evaluate the bias of the instrument and detect any error in the instrument calibration. These samples are usually prepared by a third party, using a quantity of analyte(s) which is known to the preparer but unknown to the operator, and always undergo certification analysis. The analyte(s)

used to prepare the PE sample is the same as the analyte(s) of interest (EPA 1991g). PE samples are recommended for use in determining bias for a measurement system when appropriate reference materials are not available. PE samples are equivalent to matrix spikes prepared by a third party that undergo certification analysis and can be non-blind or single-blind when used to measure bias for scanning and direct measurements.

- Matrix Spike Samples. Matrix spike samples are environmental samples that are spiked in the laboratory with a known concentration of a target analyte(s) to verify percent recoveries. They are primarily used to check sample matrix interferences but can also be used in the field to monitor instrument performance (EPA 1991g). Matrix Spike samples are often replicated to monitor a method's performance and evaluate bias and precision (when four or more pairs are analyzed). These replicates are often collectively referred to as a matrix spike/matrix spike duplicate (MS/MSD).

- Calibration Checks. Calibration checks are measurements performed to verify instrument performance each time an instrument is used (see Section 6.5.4). These checks may be qualitative or quantitative. Operators use qualitative checks to determine if an instrument is operating properly and can be used to perform measurements. Quantitative calibration checks require a specified protocol to measure a calibration source with a known instrument response, and the results are documented to provide a record of instrument precision and bias. The results of quantitative calibration checks are typically recorded on a control chart (see Section 6.2.2.7). Note that the calibration check source does not need to be traceable for qualitative or quantitative calibration checks as long as the instrument response has been adequately established (see Section 6.5.4). Because calibration checks are non-blind measurements they are only recommended when other types of QC measurements are not available.

Quality control measurements can also be used to estimate bias caused by contamination.

- Background Measurement. A background measurement is a measurement performed upgradient of the area of potential contamination (either onsite or offsite) where there is little or no chance of migration of the contaminants of concern (EPA 1991g). Background measurements are performed in the background reference area (Section 4.5), determine the natural composition and variability of the material of interest (especially important in areas with high concentrations of naturally occurring radionuclides), and are considered "clean." They provide a basis for comparison of contaminant concentration levels with measurements performed in the survey unit when the statistical tests described in Chapter 8 are performed.

- Measurement Blanks. Measurement blanks are samples prepared in the laboratory using certified clean sand or soil and brought to the field to monitor contamination for scanning and direct measurements. A measurement blank is used to evaluate contamination error associated with the instrument used to perform measurements in the field. Measurement blanks are recommended for determining bias resulting from contamination of instruments used for scanning and direct measurements.

6.2.2.3 Representativeness

Representativeness is a measure of the degree to which data accurately and precisely represent a characteristic of a population parameter at a sampling point (ASQC 1995) or measurement location. Representativeness is a qualitative term that is reflected in the survey design through the selection of a measurement method (*e.g.*, direct measurement or sampling).

Sample collection and analysis is typically less representative of true radionuclide concentrations at a specific measurement location than performing a direct measurement. This is caused by the additional steps required in collecting and analyzing samples, such as sample collection, field sample preparation, laboratory sample preparation, and radiochemical analysis. However, direct measurement techniques with acceptable detection limits are not always available. The location of the direct measurement is determined in Section 5.5.2.5, where random and systematic survey designs are selected based on survey unit classification. The coverage for a survey unit using scanning techniques is discussed in Section 5.5.3 and is also based primarily on survey unit classification. Because scanning locations are often selected based on professional judgment for survey units with less than 100% coverage, representativeness of these locations may be a concern. For both scanning and direct measurements the measurement locations and method for performing the measurements should be compared to the modeling assumptions used to develop the DCGLs.

6.2.2.4 Comparability

Comparability is a qualitative term that expresses the confidence that two data sets can contribute to a common analysis and interpolation. Generally, comparability is provided by using the same measurement system for all analyses of a specific radionuclide. Comparability is usually not an issue except in cases where historical data has been collected and is being compared to current analytical results, or when multiple laboratories are used to provide results as part of a single survey design.

6.2.2.5 Completeness

Completeness is a measure of the amount of valid data obtained from the measurement system. This is expressed as a percentage of the number of valid measurements that should have been

collected. Completeness is of greater concern for laboratory analyses than for direct measurements because the consequences of incomplete data often require the collection of additional data. Completeness is a concern for scanning only if the scanning results are invalidated for some reason. Direct measurements and scans can usually be repeated fairly easily while the personnel performing the measurements are still in the field. For this reason MARSSIM strongly recommends that scanning and direct measurement results be evaluated as soon as possible. Direct measurements performed on a systematic grid to locate areas of elevated activity are also a concern for completeness. If one direct measurement result is not valid, the entire survey design for locating areas of elevated activity may be invalidated.

6.2.2.6 Other Data Quality Indicators

Several additional data quality indicators that influence the final status survey design are identified as DQOs in Section 2.3.1. Many of these (*e.g.*, selection and classification of survey units, decision error rates, variability in the contaminant concentration, lower bound of the gray region) are used to determine the number of measurements and are discussed in detail in Section 5.5.2. The method detection limit is directly related to the selection of a measurement method and a specific measurement system.

Scanning and direct measurement techniques should be capable of measuring levels below the established DCGLs— detection limits of 10-50% of the DCGL should be the target (see Section 6.7). Cost, time, best available technology, or other constraints may create situations where the above stated sensitivities are deemed impractical. Under these circumstances, higher detection sensitivities may be acceptable. Although service providers and instrument manufacturers will state detection limits, these sensitivities are usually based on ideal or optimistic situations and may not be achievable under site-specific measurement conditions. Detection limits are subject to variation from measurement to measurement, instrument to instrument, operator to operator, and procedure to procedure. This variation depends on geometry, background, instrument calibration, abundance of the radiations being measured, counting time, operator training, operator experience, self-absorption in the medium being measured, and interferences from radionuclides or other materials present in the medium. The detection limit that is achievable in practice should not exceed the DCGL.

6.2.2.7 Using Control Charts to Provide Control of Field Measurement Systems

Control charts are commonly used in radioanalytical laboratories to monitor the performance of laboratory instruments. Control charts are also useful for monitoring the performance of field instruments and can be used to help control field measurement systems.

A control chart is a graphical plot of measurement results with respect to time or sequence of measurement, together with limits within in which the measurement values are expected to lie

when the system is in a state of statistical control (DOE 1995). Calibration check results are typically plotted on control charts for field measurements. However, control charts may be developed for any measurements where the expected performance is established and documented. A separate set of control charts for monitoring each type of measurement (*e.g.*, calibration check, background, measurement of PE samples) should be developed for each instrument.

The control chart is constructed by preparing a graph showing the arithmetic mean and the control limits as horizontal lines. The recommended control limits are two standard deviations above and below the mean, and three standard deviations above and below the mean. The measurement results in the appropriate units are shown on the y-axis and time or sequence is plotted using the x-axis. Detailed guidance on the development and use of control charts is available in *Quality Assurance of Chemical Measurements* (Taylor 1987) and *Statistical Methods for Quality Improvement* (Kume 1985).

As the quality control or other measurements are performed, the results are entered on the control chart. If the results are outside the control limits or show a particular trend or tendency, then the process is not in control. The control chart documents the performance of the measurement system during the time period of interest.

Quality control measurements for field instruments may be difficult or expensive to obtain for some surveys. In these cases control charts documenting instrument performance may represent the only determination of precision and bias for the survey. Because control charts are non-blind measurements they are generally not appropriate for estimating precision and bias. However, the control chart documents the performance of the field instruments. Provided the checks for precision and bias fall within the control limits, the results obtained using that instrument should be acceptable for the survey.

6.3 Selecting a Service Provider to Perform Field Data Collection Activities

One of the first steps in designing a survey is to select a service provider to perform field data collection activities. MARSSIM recommends that this selection take place early in the planning process so that the service provider can provide information during survey planning and participate in the design of the survey. Service providers may include in-house experts in field measurements and sample collection, health physics companies, or environmental engineering firms among others.

When the service provider is not part of the organization responsible for the site, these services are obtained using some form of procurement mechanism. Examples of procurement mechanisms include purchase orders or contracts. A graded approach should be used in determining the appropriate method for procuring services.

Potential service providers should be evaluated to determine their ability to perform the necessary analyses. For large or complex sites, this evaluation may take the form of a pre-award audit. The results of this audit provide a written record of the decision to use a specific service provider. For less complex sites or facilities, a review of the potential service provider's qualifications is sufficient for the evaluation.

There are six criteria that should be reviewed during this evaluation:

- Does the service provider possess the validated Standard Operating Procedures (SOPs), appropriate instrumentation, and trained personnel necessary to perform the field data collection activities? Field data collection activities (e.g., scanning surveys, direct measurements, and sample collection) are defined by the data needs identified by the DQO process.
- Is the service provider experienced in performing the same or similar data collection activities?
- Does the service provider have satisfactory performance evaluation or technical review results? The service provider should be able to provide a summary of QA audits and QC measurement results to demonstrate proficiency. Equipment calibrations should be performed using National Institute of Standards and Technology (NIST) traceable reference radionuclide standards whenever possible.
- Is there an adequate capacity to perform all field data collection activities within the desired timeframe? This criterion considers the number of trained personnel and quantity of calibrated equipment available to perform the specified tasks.
- Does the service provider conduct an internal quality control review of all generated data that is independent of the data generators?
- Are there adequate protocols for method performance documentation, sample tracking and security (if necessary), and documentation of results?

Potential service providers should have an active and fully documented quality system in place.[1] This system should enable compliance with the objectives determined by the DQO process in Section 2.3 and Appendix D (see EPA 1994c). The elements of a quality management system are discussed in Section 9.1 (ASQC 1995, EPA 1994f).

[1] The quality management system is typically documented in one or more documents such as a Quality Management Plan (QMP) or Quality Assurance Manual (QAM). A description of quality systems is included in Section 9.1.

6.4 Measurement Methods

Measurement methods used to generate field data can be classified into two categories commonly known as scanning surveys and direct measurements. The decision to use a measurement method as part of the survey design is determined by the survey objectives and the survey unit classification. Scanning is performed to identify areas of elevated activity that may not be detected by other measurement methods. Direct measurements are analogous to collecting and analyzing samples to determine the average activity in a survey unit. Section 5.5.3 discusses combining scans and direct measurements in an integrated survey design.

6.4.1 Direct Measurements

To conduct direct measurements of alpha, beta, and photon surface activity, instruments and techniques providing the required detection sensitivity are selected. The type of instrument and method of performing the direct measurement are selected as dictated by the type of potential contamination present, the measurement sensitivity requirements, and the objectives of the radiological survey. Direct measurements are taken by placing the instrument at the appropriate distance[2] above the surface, taking a discrete measurement for a pre-determined time interval (*e.g.*, 10 s, 60 s, *etc.*), and recording the reading. A one minute integrated count technique is a practical field survey procedure for most equipment and provides detection sensitivities that are below most DCGLs. However, longer or shorter integrating times may be warranted (see Section 6.7.1 for information dealing with the calculation of direct measurement detection sensitivities).

Direct measurements may be collected at random locations in the survey unit. Alternatively, direct measurements may be collected at systematic locations and supplement scanning surveys for the identification of small areas of elevated activity (see Section 5.5.2.5). Direct measurements may also be collected at locations identified by scanning surveys as part of an investigation to determine the source of the elevated instrument response. Professional judgment may also be used to identify location for direct measurements to further define the areal extent of contamination and to determine maximum radiation levels within an area, although these types of direct measurements are usually associated with preliminary surveys (*i.e.*, scoping, characterization, remedial action support). All direct measurement locations and results should be documented.

[2] Measurements at several distances may be needed. Near-surface or surface measurements provide the best indication of the size of the contaminated region and are useful for model implementation. Gamma measurements at 1 m provide a good estimate of potential direct external exposure.

If the equipment and methodology used for scanning is capable of providing data of the same quality required for direct measurement (*e.g.*, detection limit, location of measurements, ability to record and document results), then scanning may be used in place of direct measurements. Results should be documented for at least the number of locations required for the statistical tests. In addition, some direct measurement systems may be able to provide scanning data, provided they meet the objectives of the scanning survey.

The following sections briefly describe methods used to perform direct measurements in the field. The instruments used to perform these measurements are described in more detail in Section 6.5.3 and Appendix H.

6.4.1.1 Direct Measurements for Photon Emitting Radionuclides

There are a wide variety of instruments available for measuring photons in the field (see Appendix H) but all of them are used in essentially the same way. The detector is set up at a specified distance from the surface being measured and data are collected for a specified period of time. The distance from the surface to the detector is generally determined by the calibration of the instrument because photons do not interact appreciably with air. When measuring x-rays or low-energy gamma rays, the detector is often placed closer to the surface to increase the counting efficiency. The time required to perform a direct measurement may vary from very short (*e.g.*, 10 seconds) to very long (*e.g.*, several days or weeks) depending on the type of detector and the required detection limit. In general, the lower the required detection limit the longer the time required to perform the measurement. A collimator may be used in areas where activity from adjacent or nearby areas might interfere with the direct measurement. The collimator (usually lead, tungsten, or steel) shields the detector from extraneous photons but allows activity from a specified area of the surface to reach the detector.

Example:

The portable germanium detector, or *in situ* gamma spectrometer, can be used to estimate gamma-emitting radionuclide concentrations in the field. As with the laboratory-based germanium detector with multichannel analyzer, *in situ* gamma spectrometry can discriminate among various radionuclides on the basis of characteristic gamma and x-ray energies to provide a nuclide-specific measurement. A calibrated detector measures the fluence rate of primary photons at specific energies that are characteristic of a particular radionuclide (NRC 1995b). This fluence rate can then be converted to units of concentration. Under certain conditions the fluence rate may be converted directly to dose or risk for a direct comparison to the release criterion rather than to the $DCGL_W$. Although this conversion is generally made, the fluence rate should be considered the fundamental parameter for assessing the level of radiation at a specific location because it is a directly measurable physical quantity.

For outdoor measurements, where the contaminant is believed to be distributed within the surface soil, it may be appropriate to assume a uniform depth profile when converting the fluence rate to a concentration. At sites where the soil is plowed or overturned regularly, this assumption is quite realistic because of the effects of homogenization. At sites where the activity was initially deposited on the surface and has gradually penetrated deeper over time, the actual depth profile will have a higher activity at the surface and gradually diminish with depth. In this case, the assumption of a uniform depth profile will estimate a higher radionuclide concentration relative to the average concentration over that depth. In cases where there is an inverted depth profile (*i.e.*, low concentration at the surface that increase with depth), the assumption of a uniform depth profile will underestimate the average radionuclide concentration over that depth. For this reason, MARSSIM recommends that soil cores be collected to determine the actual depth profile for the site. These soil cores may be collected during the characterization or remedial action support survey to establish a depth profile for planning a final status survey. The cores may also be collected during the final status survey to verify the assumptions used to develop the fluence-to-concentration correction.

For indoor measurements, uncollimated *in situ* measurements can provide useful information on the low-level average activity across an entire room. The position of the measurement within the room is not critical if the radionuclide of interest is not present in the building materials. A measurement of peak count rate can be converted to fluence rate, which can in turn be related to the average surface activity. The absence of a discernible peak would mean that residual activity could not exceed a certain average level. However, this method will not easily locate small areas of elevated activity. For situations where the activity is not uniformly distributed on the surface, a series of collimated measurements using a systematic grid allows the operator to identify general areas of elevated contamination.

The NRC draft report *Measurement Methods for Radiological Surveys in Support of New Decommissioning Criteria* (NRC 1995b) provides a detailed description of the theory and implementation of *in situ* gamma spectrometry. *In situ* spectrometry is provided as one example of a useful tool for performing direct measurements for particular scenarios, but interpretation of the instrument output in terms of radionuclide distributions is dependent on the assumptions used to calibrate the method site-specifically. The depth of treatment of this technique in this example is not meant to imply that *in situ* gamma spectrometry is preferred *a priori* over other appropriate measurement techniques described in this manual.

6.4.1.2 Direct Measurements for Alpha Emitting Radionuclides

Direct measurements for alpha-emitting radionuclides are generally performed by placing the detector on or near the surface to be measured. The limited range of alpha particles (*e.g.*, about 1 cm or 0.4 in. in air, less in denser material) means that these measurements are generally restricted to relatively smooth, impermeable surfaces such as concrete, metal, or drywall where the activity is present as surface contamination. In most cases, direct measurements of porous (*e.g.*, wood) and volumetric (*e.g.*, soil, water) material cannot meet the objectives of the survey. However, special instruments such as the long range alpha detector (see Appendix H) have been developed to measure the concentration of alpha emitting radionuclides in soil under certain conditions. Because the detector is used in close proximity to the potentially contaminated surface, contamination of the detector or damage to the detector caused by irregular surfaces need to be considered before performing direct measurements for alpha emitters.

6.4.1.3 Direct Measurements for Beta Emitting Radionuclides

Direct measurements for beta emitting radionuclides are generally performed by placing the detector on or near the surface to be measured, similar to measurements for alpha emitting radionuclides. These measurements are typically restricted to relatively smooth, impermeable surfaces where the activity is present as surface contamination. In most cases, direct measurements of porous (*e.g.*, wood) and volumetric (*e.g.*, soil, water) material cannot meet the objectives of the survey. However, special instruments such as large area gas-flow proportional counters (see Appendix H) and arrays of beta scintillators have been developed to measure the concentration of beta emitting radionuclides in soil under certain conditions. Similar to direct measurements for alpha emitting radionuclides, contamination of the detector and damage to the detector need to be considered before performing direct measurements for beta emitters.

6.4.2 Scanning Surveys

Scanning is the process by which the operator uses portable radiation detection instruments to detect the presence of radionuclides on a specific surface (*i.e.*, ground, wall, floor, equipment). The term scanning survey is used to describe the process of moving portable radiation detectors across a suspect surface with the intent of locating radionuclide contamination. Investigation levels for scanning surveys are determined during survey planning to identify areas of elevated activity. Scanning surveys are performed to locate radiation anomalies indicating residual gross activity that may require further investigation or action. These investigation levels may be based on the $DCGL_W$, the $DCGL_{EMC}$, or some other level as discussed in Section 5.5.2.6.

Small areas of elevated activity typically represent a small portion of the site or survey unit. Thus, random or systematic direct measurements or sampling on the commonly used grid spacing may have a low probability of identifying such small areas. Scanning surveys are often relatively quick and inexpensive to perform. For these reasons, scanning surveys are typically performed before direct measurements or sampling. This way time is not spent fully evaluating an area that may quickly prove to be contaminated above the investigation level during the scanning process. Scans are conducted which would be indicative of all radionuclides potentially present, based on the Historical Site Assessment, surfaces to be surveyed, and survey design objectives. Surrogate measurements may be utilized where appropriate (see Section 4.3.2). Documenting scanning results and observations from the field is very important. For example, a scan that identified relatively sharp increases in instrument response or identified the boundary of an area of increased instrument response should be documented. This information is useful when interpreting survey results.

The following sections briefly describe techniques used to perform scanning surveys for different types of radiation. The instruments used to perform these measurements are described in more detail in Section 6.5.3 and Appendix H.

6.4.2.1 Scanning for Photon Emitting Radionuclides

Sodium iodide survey meters (NaI(Tl) detectors) are normally used for scanning areas for gamma emitters because they are very sensitive to gamma radiation, easily portable and relatively inexpensive. The detector is held close to the ground surface (~6 cm or 2.5 in.) and moved in a serpentine (*i.e.*, snake like, "S" shaped) pattern while walking at a speed that allows the investigator to detect the desired investigation level. A scan rate of approximately 0.5 m/s is typically used for distributed gamma emitting contaminants in soil; however, this rate must be adjusted depending on the expected detector response and the desired investigation level. Discussion of scanning rates versus detection sensitivity for gamma emitters is provided in Section 6.7.2.1.

Sodium iodide survey meters are also used for scanning to detect areas with elevated areas of low-energy gamma and x-ray emitting radionuclides such as ^{241}Am and ^{239}Pu. Specially designed detectors, such as the FIDLER (field instrument for the detection of low energy radiation) probe with survey meter, are typically used to detect these types of radionuclides.

6.4.2.2 Scanning for Alpha Emitting Radionuclides

Alpha scintillation survey meters and thin window gas-flow proportional counters are typically used for performing alpha surveys. Alpha radiation has a very limited range and, therefore, instrumentation must be kept close to the surface—usually less than 1 cm (0.4 in.). For this reason, alpha scans are generally performed on relatively smooth, impermeable surfaces (*e.g.*,

concrete, metal, drywall) and not on porous material (*e.g.*, wood) or for volumetric contamination (*e.g.*, soil, water). In most cases, porous and volumetric contamination cannot be detected by scanning for alpha activity and meet the objectives of the survey because of high detection sensitivities. Under these circumstances, samples of the material are usually collected and analyzed as discussed in Chapter 7. Determining scan rates when surveying for alpha emitters is discussed in Section 6.7.2.2 and Appendix J.

6.4.2.3 Scanning for Beta Emitting Radionuclides

Thin window gas-flow proportional counters are normally used when surveying for beta emitters, although solid scintillators designed for this purpose are also available. Typically, the beta detector is held less than 2 cm from the surface and moved at a rate such that the desired investigation level can be detected. Low-energy (<100 keV) beta emitters are subject to the same interferences and self-absorption problems found with alpha emitting radionuclides, and scans for these radionuclides are performed under similar circumstances. Determination of scan rates when surveying for beta emitters is discussed in Section 6.7.2.1.

6.5 Radiation Detection Instrumentation

Traditional radiation instruments consist of two components: 1) a radiation detector, and 2) electronic equipment to provide power to the detector and to display or record radiation events. This section identifies and very briefly describes the types of radiation detectors and associated display or recording equipment that are applicable to survey activities in support of environmental assessment or remedial action. Each survey usually requires performing direct field measurements using portable instrumentation and collection of samples for laboratory analysis. The selection and proper use of appropriate instruments for both direct measurements and laboratory analyses will likely be the most critical factors in assuring that the survey accurately determines the radiological status of a site and meets the survey objectives. Chapter 7 provides specific information on laboratory analysis of collected samples. Appendix H contains instrument specific information for various types of field survey and laboratory analysis equipment currently in use.

6.5.1 Radiation Detectors

The particular capabilities of a radiation detector will establish its potential applications in conducting a specific type of survey. Radiation detectors can be divided into four general classes based on the detector material or the application. These categories are: 1) gas-filled detectors, 2) scintillation detectors, 3) solid-state detectors, and 4) passive integrating detectors.

6.5.1.1 Gas-Filled Detectors

Radiation interacts with the fill gas, producing ion-pairs that are collected by charged electrodes. Commonly used gas-filled detectors are categorized as ionization, proportional, or Geiger-Mueller (GM), referring to the region of gas amplification in which they are operated. The fill gas varies, but the most common are: 1) air, 2) argon with a small amount of organic methane (usually 10% methane by mass, referred to as P-10 gas), and 3) argon or helium with a small amount of a halogen such as chlorine or bromine added as a quenching agent.

6.5.1.2 Scintillation Detectors

Radiation interacts with a solid or liquid medium causing electronic transitions to excited states in a luminescent material. The excited states decay rapidly, emitting photons that in turn are captured by a photomultiplier tube. The ensuing electrical signal is proportional to the scintillator light output, which, under the right conditions, is proportional to the energy loss that produced the scintillation. The most common scintillant materials are NaI(Tl), ZnS(Ag), Cd(Te), and CsI(Tl) which are used in traditional radiation survey instruments such as the NaI(Tl) detector used for gamma surveys and the ZnS(Ag) detector for alpha surveys.

6.5.1.3 Solid-State Detectors

Radiation interacting with a semiconductor material creates electron-hole pairs that are collected by a charged electrode. The design and operating conditions of a specific solid-state detector determines the types of radiations (alpha, beta, and/or gamma) that can be measured, the detection level of the measurements, and the ability of the detector to resolve the energies of the interacting radiations. The semiconductor materials currently being used are germanium and silicon which are available in both n and p types in various configurations.

Spectrometric techniques using these detectors provide a marked increase in sensitivity in many situations. When a particular radionuclide contributes only a fraction of the total particle fluence or photon fluence, or both, from all sources (natural or manmade background), gross measurements are inadequate and nuclide-specific measurements are necessary. Spectrometry provides the means to discriminate among various radionuclides on the basis of characteristic energies. *In-situ* gamma spectrometry is particularly effective in field measurements since the penetrating nature of the radiation allows one to "see" beyond immediate surface contamination. The availability of large, high efficiency germanium detectors permits measurement of low abundance gamma emitters such as ^{238}U as well as low energy emitters such as ^{241}Am and ^{239}Pu.

6.5.1.4 Passive Integrating Detectors

There is an additional class of instruments that consists of passive, integrating detectors and associated reading/analyzing instruments. The integrated ionization is read using a laboratory or hand-held reader. This class includes thermoluminescence dosimeters (TLDs) and electret ion chambers (EICs). Because these detectors are passive and can be exposed for relatively long periods of time, they can provide better sensitivity for measuring low activity levels such as free release limits or for continuing surveillance. The ability to read and present data onsite is a useful feature and such systems are comparable to direct reading instruments.

The scintillation materials in Section 6.5.1.2 are selected for their prompt fluorescence characteristics. In another class of inorganic crystals, called TLDs, the crystal material and impurities are chosen so that the free electrons and holes created following the absorption of energy from the radiation are trapped by impurities in the crystalline lattice thus locking the excitation energy in the crystal. Such materials are used as passive, integrating detectors. After removal from the exposure area, the TLDs are heated in a reader which measures the total amount of light produced when the energy is released. The total amount of light is proportional to the number of trapped, excited electrons, which in turn is proportional to the amount of energy absorbed from the radiation. The intensity of the light emitted from the thermoluminescent crystals is thus directly proportional to the radiation dose. TLDs come in a large number of materials, the most common of which are LiF, CaF_2:Mn, CaF_2:Dy, $CaSO_4$:Mn, $CaSO_4$:Dy, Al_2O_3:C.

The electret ion chamber consists of a very stable electret (a charged Teflon® disk) mounted inside a small chamber made of electrically charged plastic. The ions produced inside this air filled chamber are collected onto the electret, causing a reduction of its surface charge. The reduction in charge is a function of the total ionization during a specific monitoring period and the specific chamber volume. This change in voltage is measured with a surface potential voltmeter.

6.5.2 Display and Recording Equipment

Radiation detectors are connected to electronic devices to 1) provide a source of power for detector operation, and 2) enable measurement of the quantity and/or quality of the radiation interactions that are occurring in the detector. The quality of the radiation interaction refers to the amount of energy transferred to the detector. In many cases, radiation interacts with other material (*e.g.*, air) prior to interacting with the detector, or only partially interacts with the detector (*e.g.*, Compton scattering for photons). Because the energy recorded by the detector is affected, there is an increased probability of incorrectly identifying the radionuclide.

The most common recording or display device used for portable radiation measurement systems is a ratemeter. This device provides a display on an analog meter representing the number of events occurring over some time period (*e.g.*, counts per minute). Digital ratemeters are also commercially available. The number of events can also be accumulated over a preset time period using a digital scaling device. The resulting information from a scaling device is the total number of events that occurred over a fixed period of time, where a ratemeter display varies with time and represents a short term average of the event rate. Determining the average level on a ratemeter will require judgment by the user, especially when a low frequency of events results in significant variations in the meter reading.

Pulse height analyzers are specialized electronic devices designed to measure and record the number of pulses or events that occur at different pulse height levels. These types of devices are used with detectors which produce output pulses that are proportional in height to the energy deposited within them by the interacting radiation. They can be used to record only those events occurring in a detector within a single band of energy or can simultaneously record the events in multiple energy ranges. In the former case, the equipment is known as a single-channel analyzer; the latter application is referred to as a multichannel analyzer.

6.5.3 Instrument Selection

Radiation survey parameters that might be needed for site release purposes include surface activities, exposure rates, and radionuclide concentrations in soil. To determine these parameters, field measurements and laboratory analyses may be necessary. For certain radionuclides or radionuclide mixtures, both alpha and beta radiations may have to be measured. In addition to assessing average radiological conditions, the survey objectives should address identifying small areas of elevated activity and determining the extent and level of residual radioactivity.

Additionally, the potential uses of radiation instruments can vary significantly depending on the specific design and operating criteria of a given detector type. For example, a NaI(Tl) scintillator can be designed to be very thin with a low atomic number entrance window (*e.g.*, beryllium) such that the effective detection capability for low energy photons is optimized. Conversely, the same scintillant material can be fabricated as a thick cylinder in order to optimize the detection probability for higher energy photons. On the recording end of a detection system, the output could be a ratemeter, scaler, or multichannel analyzer as described in Section 6.5.2. Operator variables such as training and level of experience with specific instruments should also be considered.

With so many variables, it is highly unlikely that any single instrument (detector and readout combination) will be capable of adequately measuring all of the radiological parameters necessary to demonstrate that criteria for release have been satisfied. It is usually necessary to select multiple instruments to perform the variety of measurements required.

Selection of instruments will require an evaluation of a number of situations and conditions. Instruments must be stable and reliable under the environmental and physical conditions where they will be used, and their physical characteristics (size and weight) should be compatible with the intended application. The instrument must be able to detect the type of radiation of interest, and the measurement system should be capable of measuring levels that are less than the DCGL (see Section 6.7).

For gamma radiation scanning, a scintillation detector/ratemeter combination is the usual instrument of choice. A large-area proportional detector with a ratemeter is recommended for scanning for alpha and beta radiations where surface conditions and locations permit; otherwise, an alpha scintillation or thin-window GM detector (for beta surveys) may be used.

For direct gamma measurements, a pressurized ionization chamber or *in-situ* gamma spectroscopy system is recommended. As an option, a NaI(Tl) scintillation detector may be used if cross-calibrated to a pressurized ion chamber or calibrated for the specific energy of interest. The same alpha and beta detectors identified above for scanning surveys are also recommended for use in direct measurements.

There are certain radionuclides that, because of the types, energies, and abundances of their radiations, will be essentially impossible to measure at the guideline levels, under field conditions, using state-of-the-art instrumentation and techniques. Examples of such radionuclides include very low energy pure beta emitters, such as ^{3}H and ^{63}Ni, and low energy photon emitters, such as ^{55}Fe and ^{125}I. Pure alpha emitters dispersed in soil or covered with some absorbing layer will not be detectable because the alpha radiation will not penetrate through the media or covering to reach the detector. A common example of such a condition would be ^{230}Th surface contamination covered by paint, dust, oil, or moisture. In such circumstances, sampling and laboratory analysis would be required to measure the residual activity levels unless surrogate radionuclides are present as discussed in Section 4.3.2.

The number of possible design and operating schemes for each of the different types of detectors is too large to discuss in detail within the context of this document. For a general overview, lists of common radiation detectors along with their usual applications during surveys are provided in Tables 6.1 through 6.3. Appendix H contains specific information for various types of field survey and laboratory analysis equipment currently in use. Continual development of new technologies will result in changes to these listings.

Table 6.1 Radiation Detectors with Applications to Alpha Surveys

Detector Type	Detector Description	Application	Remarks
Gas Proportional	<1 mg/cm^2 window; probe area 50 to 1000 cm^2 <0.1 mg/cm^2 window; probe area 10 to 20 cm^2 No window (internal proportional)	Surface scanning; surface contamination measurement Laboratory measurement of water, air, and smear samples Laboratory measurement of water, air, and smear samples	Requires a supply of appropriate fill gas
Air Proportional	<1 mg/cm^2 window; probe area ~50 cm^2	Useful in low humidity conditions	
Scintillation	ZnS(Ag) scintillator; probe area 50 to 100 cm^2 ZnS(Ag) scintillator; probe area 10 to 20 cm^2 Liquid scintillation cocktail containing sample	Surface contamination measurements, smears Laboratory measurement of water, air, and smear samples Laboratory analysis, spectrometry capabilities	
Solid State	Silicon surface barrier detector	Laboratory analysis by alpha spectrometry	
Passive, integrating electret ion chamber	<0.8 mg/cm^2 window, also window-less, window area 50-180 cm^2, chamber volume 50-1,000 ml	Contamination on surfaces, in pipes and in soils	Useable in high humidity and temperature

6.5.4 Instrument Calibration

Calibration refers to the determination and adjustment of the instrument response in a particular radiation field of known intensity. Proper calibration procedures are an essential requisite toward providing confidence in measurements made to demonstrate compliance with cleanup criteria. Certain factors, such as energy dependence and environmental conditions, require consideration in the calibration process, depending on the conditions of use of the instrument in the field. Routine calibration of radiation detection instruments refers to calibration for normal use under typical field conditions. Considerations for the use and calibration of instruments include:

Table 6.2 Radiation Detectors with Applications to Beta Surveys

Detector Type	Detector Description	Application	Remarks
Gas Proportional	<1 mg/cm^2 window; probe area 50 to 1,000 cm^2	Surface scanning; surface contamination measurement	Requires a supply of appropriate fill gas
	<0.1 mg/cm^2 window; probe area 10 to 20 cm^2	Laboratory measurement of water, air, smear, and other samples	
	No window (internal proportional)	Laboratory measurement of water, air, smear, and other samples	Can be used for measuring very low-energy betas
Ionization (non-pressurized)	1-7 mg/cm^2 window	Contamination measurements; skin dose rate estimates	
Geiger-Mueller	<2 mg/cm^2 window; probe area 10 to 100 cm^2	Surface scanning; contamination measurements; laboratory analyses	
	Various window thickness; few cm^2 probe face	Special scanning applications	
Scintillation	Liquid scintillation cocktail containing sample	Laboratory analysis; spectrometry capabilities	
	Plastic scintillator	Contamination measurements	
Passive, integrating electret ion chamber	7 mg/cm^2 window, also window-less, window area 50-180 cm^2, chamber volume 50-1,000 ml	Low energy beta including H-3 contamination on surfaces and in pipes	Useable in high humidity and temperature

- use of the instrument for radiation of the type for which the instrument is designed
- use of the instrument for radiation energies within the range of energies for which the instrument is designed
- use under environmental conditions for which the instrument is designed
- use under influencing factors, such as magnetic and electrostatic fields, for which the instrument is designed
- use of the instrument in an orientation such that geotropic effects are not a concern
- use of the instrument in a manner that will not subject the instrument to mechanical or thermal stress beyond that for which it is designed

Table 6.3 Radiation Detectors with Applications to Gamma Surveys

Detector Type	Detector Description	Application	Remarks
Gas Ionization	Pressurized ionization chamber; Non-pressurized ionization chamber	Exposure rate measurements	
Geiger-Mueller	Pancake (<2 mg/cm^2 window) or side window (~30 mg/cm^2)	Surface scanning; exposure rate correlation (side window in closed position)	Low relative sensitivity to gamma radiation
Scintillation	NaI(Tl) scintillator; up to 5 cm by 5 cm	Surface scanning; exposure rate correlation	High sensitivity; Cross calibrate with PIC (or equivalent) or for specific site gamma energy mixture for exposure rate measurements.
	NaI(Tl) scintillator; large volume and "well" configurations	Laboratory gamma spectrometry	
	CsI or NaI(Tl) scintillator; thin crystal	Scanning; low-energy gamma and x-rays	Detection of low-energy radiation
	Organic tissue equivalent (plastics)	Dose equivalent rate measurements	
Solid State	Germanium semi-conductor	Laboratory and field gamma spectrometry and spectroscopy	
Passive, integrating electret ion chamber	7 mg/cm^2 window, also window-less, window area 50-180 cm^2, chamber volume 50-1,000 ml		Useable in high humidity and temperature

Routine calibration commonly involves the use of one or more sources of a specific radiation type and energy, and of sufficient activity to provide adequate field intensities for calibration on all ranges of concern.

Actual field conditions under which the radiation detection instrument will be used may differ significantly from those present during routine calibration. Factors which may affect calibration validity include:

- the energies of radioactive sources used for routine calibration may differ significantly from those of radionuclides in the field
- the source-detector geometry (*e.g.*, point source or large area distributed source) used for routine calibration may be different than that found in the field
- the source-to-detector distance typically used for routine calibration may not always be achievable in the field
- the condition and composition of the surface being monitored (*e.g.*, sealed concrete, scabbled concrete, carbon steel, stainless steel, and wood) and the presence of overlaying material (*e.g.*, water, dust, oil, paint) may result in a decreased instrument response relative to that observed during routine calibration

If the actual field conditions differ significantly from the calibration assumptions, a special calibration for specific field conditions may be required. Such an extensive calibration need only be done once to determine the effects of the range of field conditions that may be encountered at the site. If responses under routine calibration conditions and proposed use conditions are significantly different, a correction factor or chart should be supplied with the instrument for use under the proposed conditions.

As a minimum, each measurement system (detector/readout combination) should be calibrated annually and response checked with a source following calibration (ANSI 1996). Instruments may require more frequent calibration if recommended by the manufacturer. Re-calibration of field instruments is also required if an instrument fails a performance check or if it has undergone repair or any modification that could affect its response.

The user may decide to perform calibrations following industry recognized procedures (ANSI 1996b, DOE Order 5484.1, NCRP 1978, NCRP 1985, NCRP 1991, ISO 1988, HPS 1994a, HPS 1994b), or the user can choose to obtain calibration by an outside service, such as a major instrument manufacturer or a health physics services organization.

Calibration sources should be traceable to the National Institute of Standards and Technology (NIST). Where NIST traceable standards are not available, standards obtained from an industry recognized organization (*e.g.*, the New Brunswick Laboratory for various uranium standards) may be used.

Calibration of instruments for measurement of surface contamination should be performed such that a direct instrument response can be accurately converted to the 4π (total) emission rate from the source. An accurate determination of activity from a measurement of count rate above a surface in most cases is an extremely complex task because of the need to determine appropriate chacteristics of the source including decay scheme, geometry, energy, scatter, and self-absorption. For the purpose of release of contaminated areas from radiological control, measurements must provide sufficient accuracy to ensure that cleanup standards have been

achieved. Inaccuracies in measurements should be controlled in a manner that minimizes the consequences of decision errors. The variables that affect instrument response should be understood well enough to ensure that the consequences of decision errors are minimized. Therefore, the calibration should account for the following factors (where necessary):

- Calibrations for point and large area source geometries may differ, and both may be necessary if areas of activity smaller than the probe area and regions of activity larger than the probe area are present.
- Calibration should either be performed with the radionuclide of concern, or with appropriate correction factors developed for the radionuclide(s) present based on calibrations with nuclides emitting radiations similar to the radionuclide of concern.
- For portable instrumentation, calibrations should account for the substrate of concern (*i.e.*, concrete, steel) or appropriate correction factors developed for the substrates relative to the actual calibration standard substrate. This is especially important for beta emitters because backscatter is significant and varies with the composition of the substrate. Conversion factors developed during the calibration process should be for the same counting geometry to be used during the actual use of the detector.

For cleanup standards for building surfaces, the contamination level is typically expressed in terms of the particle emission rate per unit time per unit area, normally Bq/m^2 or disintegrations per minute (dpm) per 100 cm^2. In many facilities, surface contamination is assessed by converting the instrument response (in counts per minute) to surface activity using one overall total efficiency. The total efficiency may be considered to represent the product of two factors, the instrument (detector) efficiency, and the source efficiency. Use of the total efficiency is not a problem provided that the calibration source exhibits characteristics similar to the surface contamination (*i.e.*, radiation energy, backscatter effects, source geometry, self-absorption). In practice, this is hardly the case; more likely, instrument efficiencies are determined with a clean, stainless steel source, and then those efficiencies are used to determine the level of contamination on a dust-covered concrete surface. By separating the efficiency into two components, the surveyor has a greater ability to consider the actual characteristics of the surface contamination.

The instrument efficiency is defined as the ratio of the net count rate of the instrument and the surface emission rate of a source for a specified geometry. The surface emission rate is defined as the number of particles of a given type above a given energy emerging from the front face of the source per unit time. The surface emission rate is the 2π particle fluence that embodies both the absorption and scattering processes that effect the radiation emitted from the source. Thus, the instrument efficiency is determined by the ratio of the net count rate and the surface emission rate.

The instrument efficiency is determined during calibration by obtaining a static count with the detector over a calibration source that has a traceable activity or surface emission rate. In many cases, a source emission rate is measured by the manufacturer and certified as NIST traceable. The source activity is then calculated from the surface emission rate based on assumed backscatter and self-absorption properties of the source. The maximum value of instrument efficiency is 1.

The source efficiency is defined as the ratio of the number of particles of a given type emerging from the front face of a source and the number of particles of the same type created or released within the source per unit time. The source efficiency takes into account the increased particle emission due to backscatter effects, as well as the decreased particle emission due to self-absorption losses. For an ideal source (*i.e.*, no backscatter or self-absorption), the value of the source efficiency is 0.5. Many real sources will exhibit values less than 0.5, although values greater than 0.5 are possible, depending on the relative importance of the absorption and backscatter processes.

Source efficiencies may be determined experimentally. Alternatively, ISO-7503-1 (ISO 1988) makes recommendations for default source efficiencies. A source efficiency of 0.5 is recommended for beta emitters with maximum energies above 0.4 MeV. Alpha emitters and beta emitters with maximum beta energies between 0.15 and 0.4 MeV have a recommended source efficiency of 0.25. Source efficiencies for some common surface materials and overlaying material are provided in NUREG-1507 (NRC 1997b).

Instrument efficiency may be affected by detector-related factors such as detector size (probe surface area), window density thickness, geotropism, instrument response time, counting time (in static mode), scan rate (in scan mode), and ambient conditions such as temperature, pressure, and humidity. Instrument efficiency also depends on solid angle effects, which include source-to-detector distance and source geometry.

Source efficiency may be affected by source-related factors such as the type of radiation and its energy, source uniformity, surface roughness and coverings, and surface composition (*e.g.*, wood, metal, concrete).

The calibration of gamma detectors for the measurement of photon radiation fields should also provide reasonable assurance of acceptable accuracy in field measurements. Use of these instruments for demonstration of compliance with cleanup standards is complicated by the fact that most cleanup levels produce exposure rates of at most a few µR/h. Several of the portable survey instruments currently available in the United States for exposure rate measurements of ~1 µR/h (often referred to as micro-R meters) have full scale intensities of ~3 to 5 µR/h on the first range. This is below the ambient background for most low radiation areas and most calibration laboratories. (A typical background dose equivalent rate of 100 mrem/y gives a

background exposure rate of about 10 μR/h.) Even on the second range, the ambient background in the calibration laboratory is normally a significant part of the range and must be taken into consideration during calibration. The instruments commonly are not energy-compensated and are very sensitive to the scattered radiation that may be produced by the walls and floor of the room or additional shielding required to lower the ambient background.

Low intensity sources and large distances between the source and detector can be used for low-level calibrations if the appropriate precautions are taken. Field characterization of low-level sources with traceable transfer standards is difficult because of the poor signal-to-noise ratio in the standard chamber. In order to achieve adequate ionization current, the distance between the standard chamber and the source generally will be as small as possible while still maintaining good geometry (5 to 7 detector diameters). Generally it is not possible to use a standard ionization chamber to characterize the field at the distance necessary to reduce the field to the level required for calibration. A high quality GM detector, calibrated as a transfer standard, may be useful at low levels.

Corrections for scatter can be made using a shadow-shield technique in which a shield of sufficient density and thickness to eliminate virtually all the primary radiation is placed about midway between the source and the detector. The dimensions of the shield should be the minimum required to reduce the primary radiation intensity at the detector location to less than 2% of its unshielded value. The change in reading caused by the shield being removed is attributed to the primary field from the source at the detector position.

In some instruments that produce pulses (GM counters or scintillation counters), the detector can be separated electronically from the readout electronics and the detector output can be simulated with a suitable pulser. Caution must be exercised to ensure that either the high voltage is properly blocked or that the pulser is designed for this application. If this can be accomplished, the instrument can first be calibrated on a higher range that is not affected by the ambient background and in a geometry where scatter is not a problem and, after disconnecting the detector, to provide the pulse-rate from the pulser which will give the same instrument response. The pulse rate can then be related to field strength and reduced to give readings on lower ranges (with the detector disconnected) even below the ambient background. This technique does not take account of any inherent detector background independent of the external background.

Ionization chambers are commonly used to measure radiation fields at very low levels. In order to obtain the sensitivity necessary to measure these radiation levels, the instruments are frequently very large and often pressurized. These instruments have the same calibration problems as the more portable micro-R meters described above. The same precautions (shadow shield) must be taken to separate the response of the instrument to the source and to scattered radiation. Generally, it is not possible to substitute an electronic pulser for the radiation field in these instruments.

For energy-dependent gamma scintillation instruments, such as NaI(Tl) detectors, calibration for the gamma energy spectrum at a specific site may be accomplished by comparing the instrument response to that of a pressurized ionization chamber, or equivalent detector, at different locations on the site. Multiple radionuclides with various photon energies may also be used to calibrate the system for the specific energy of interest.

In the interval between calibrations, the instrument should receive a performance check prior to use. In some cases, a performance check following use may also provide valuable information. This calibration check is merely intended to establish whether or not the instrument is operating within certain specified, rather large, uncertainty limits. The initial performance check should be conducted following the calibration by placing the source in a fixed, reproducible location and recording the instrument reading. The source should be identified along with the instrument, and the same check source should be used in the same fashion to demonstrate the instrument's operability on a daily basis when the instrument is in use. For analog readout (count rate) instruments, a variation of ± 20% is usually considered acceptable. Optionally, instruments that integrate events and display the total on a digital readout typically provide an acceptable average response range of 2 or 3 standard deviations. This is achieved by performing a series of repetitive measurements (10 or more is suggested) of background and check source response and determining the average and standard deviation of those measurements. From a practical standpoint, a maximum deviation of ± 20% is usually adequate when compared with other uncertainties associated with the use of the equipment. The amount of uncertainty allowed in the response checks should be consistent with the level of uncertainty allowed in the final data. Ultimately the decision maker determines what level of uncertainty is acceptable.

Instrument response, including both the background and check source response of the instrument, should be tested and recorded at a frequency that ensures the data collected with the equipment is reliable. For most portable radiation survey equipment, MARSSIM recommends that a response check be performed twice daily when in use—typically prior to beginning the day's measurements and again following the conclusion of measurements on that same day. Additional checks can be performed if warranted by the instrument and the conditions under which it is used. If the instrument response does not fall within the established range, the instrument is removed from use until the reason for the deviation can be resolved and acceptable response again demonstrated. If the instrument fails the post-survey source check, all data collected during that time period with the instrument must be carefully reviewed and possibly adjusted or discarded, depending on the cause of the failure. Ultimately, the frequency of response checks must be balanced with the stability of the equipment being used under field conditions and the quantity of data being collected. For example, if the instrument experiences a sudden failure during the course of the day's work due to physical harm, such as a punctured probe, then the data collected up until that point is probably acceptable even though a post-use performance check cannot be performed. Likewise, if no obvious failure occurred but the instrument failed the post-use response check, then the data collected with that instrument since

the last response check should be viewed with great skepticism and possibly re-collected or randomly checked with a different instrument. Additional corrective action alternatives are presented in Section 9.3. If re-calibration is necessary, acceptable response ranges must be reestablished and documented.

Record requirements vary considerably and depend heavily on the needs of the user. While Federal and State regulatory agencies all specify requirements, the following records should be considered a minimum.

Laboratory Quality Control

- records documenting the traceabililty of radiological standards
- records documenting the traceability of electronic test equipment

Records for Instruments to be Calibrated

- date received in the calibration laboratory
- initial condition of the instrument, including mechanical condition (*e.g.*, loose or broken parts, dents, punctures), electrical condition (*e.g.*, switches, meter movement, batteries), and radiological condition (presence or absence of contamination)
- calibrator's records including training records and signature on calibration records
- calibration data including model and serial number of instrument, date of calibration, recommended recalibration date, identification of source(s) used, "as found" calibration results, and final calibration results—"as returned" for use.

In addition, records of instrument problems, failures, and maintenance can be included and are useful in assessing performance and identifying possible needs for altered calibration frequencies for some instruments. Calibration records should be maintained at the facility where the instruments are used as permanent records, and should be available either as hard copies or in safe computer storage.

6.6 Data Conversion

This section describes methods for converting survey data to appropriate units for comparison to radiological criteria. As stated in Chapter 4, conditions applicable to satisfying decommissioning requirements include determining that any residual contamination will not result in individuals being exposed to unacceptable levels of radiation and/or radioactive materials.

Radiation survey data are usually obtained in units, such as the number of counts per unit time, that have no intrinsic meaning relative to DCGLs. For comparison of survey data to DCGLs, the survey data from field and laboratory measurements should be converted to DCGL units.

6.6.1 Surface Activity

When measuring surface activity, it is important to account for the physical surface area assessed by the detector in order to make probe area corrections and report data in the proper units (*i.e.*, Bq/m^2, dpm/100 cm^2). This is termed the *physical probe area*. A common misuse is to make probe area corrections using the *effective probe area* which accounts for the amount of the physical probe area covered by a protective screen. Figure 6.1 illustrates the difference between the physical probe area and the effective probe area. The physical probe area is used because the reduced detector response due to the screen is accounted for during instrument calibration.

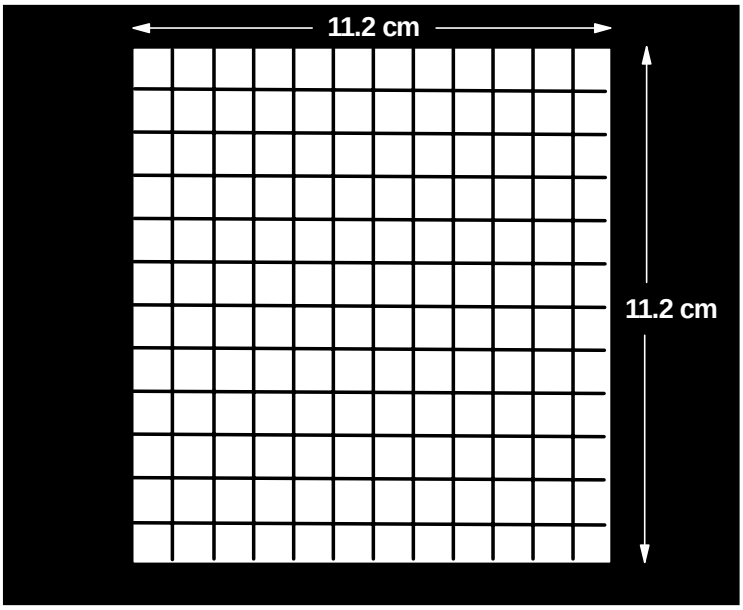

Figure 6.1 The Physical Probe Area of a Detector

The conversion of instrument display in counts to surface activity units is obtained using the following equation.

$$Bq/m^2 = \frac{\frac{C_s}{T_s}}{(\varepsilon_T \times A)} \qquad (6\text{-}1)$$

where

C_s	=	integrated counts recorded by the instrument
T_s	=	time period over which the counts were recorded in seconds
ε_T	=	total efficiency of the instrument in counts per disintegration, effectively the product of the instrument efficiency (ε_i) and the source efficiency (ε_s)
A	=	physical probe area in m^2

To convert instrument counts to conventional surface activity units, Equation 6-1 can be modified as shown in Equation 6-2.

$$\frac{dpm}{100\ cm^2} = \frac{\frac{C_s}{T_s}}{(\varepsilon_T) \times (A/100)} \qquad (6\text{-}2)$$

where T_s is recorded in minutes instead of seconds, and A is recorded in cm^2 instead of m^2.

Some instruments have background counts associated with the operation of the instrument. A correction for instrument background can be included in the data conversion calculation as shown in Equation 6-3. Note that the instrument background is not the same as the measurements in the background reference area used to perform the statistical tests described in Chapter 8.

$$Bq/m^2 = \frac{\frac{C_s}{T_s} - \frac{C_b}{T_b}}{(\varepsilon_T \times A)} \qquad (6\text{-}3)$$

where

C_b	=	background counts recorded by the instrument
T_b	=	time period over which the background counts were recorded in seconds

Equation 6-3 can be modified to provide conventional surface activity units as shown in Equation 6-4.

$$\frac{dpm}{100\ cm^2} = \frac{\frac{C_s}{T_s} - \frac{C_b}{T_b}}{(\varepsilon_T) \times (A/100)} \qquad (6\text{-}4)$$

where T_s and T_b are recorded in minutes instead of seconds and A is recorded in cm^2 instead of m^2.

The presence of multiple radionuclides at a site requires additional considerations for demonstrating compliance with a dose- or risk-based regulation. As demonstrated in Section 4.3.2, a gross activity DCGL should be determined. For example, consider a site contaminated with ^{60}Co and ^{63}Ni, with ^{60}Co representing 60% of the total activity. The relative fractions are 0.6 for ^{60}Co and 0.4 for ^{63}Ni. If the DCGL for ^{60}Co is 8,300 Bq/m^2 (5,000 dpm/100 cm^2) and the DCGL for ^{63}Ni is 12,000 Bq/m^2 (7,200 dpm/100 cm^2), the gross activity DCGL is 9,500 Bq/m^2 (5,700 dpm/100 cm^2) calculated using Equation 4-4.

When using the gross activity DCGL, it is important to use an appropriately weighted total efficiency to convert from instrument counts to surface activity units using Equations 6-1 through 6-4. In this example, the individual efficiencies for ^{60}Co and ^{63}Ni should be independently evaluated. The overall efficiency is then determined by weighting each individual efficiency by the relative fraction of each radionuclide.

6.6.2 Soil Radionuclide Concentration and Exposure Rates

Analytical procedures, such as alpha and gamma spectrometry, are typically used to determine the radionuclide concentration in soil in units of Bq/kg. Net counts are converted to soil DCGL units by dividing by the time, detector or counter efficiency, mass or volume of the sample, and by the fractional recovery or yield of the chemistry procedure (if applicable). Refer to Chapter 7 for examples of analytical procedures.

Instruments, such as a PIC or micro-R meter, used to measure exposure rate typically read directly in mSv/h. A gamma scintillation detector (*e.g.*, NaI(Tl)) provides data in counts per minute and conversion to mSv/h is accomplished by using site-specific calibration factors developed for the specific instrument (Section 6.5.4).

In situ gamma spectrometry data may require special analysis routines before the spectral data can be converted to soil concentration units or exposure rates.

6.7 Detection Sensitivity

The detection sensitivity of a measurement system refers to a radiation level or quantity of radioactive material that can be measured or detected with some known or estimated level of confidence. This quantity is a factor of both the instrumentation and the technique or procedure being used.

The primary parameters that affect the detection capability of a radiation detector are the background count rate, the detection efficiency of the detector and the counting time interval. It is important to use actual background count rate values and detection efficiencies when determining counting and scanning parameters, particularly during final status and verification surveys. When making field measurements, the detection sensitivity will usually be less than what can be achieved in a laboratory due to increased background and, often times, a significantly lower detection efficiency. It is often impossible to guarantee that pure alpha emitters can be detected *in situ* since the weathering of aged surfaces will often completely absorb the alpha emissions. NRC report NUREG-1507 (NRC 1997b) contains data on many of the parameters that affect detection efficiencies *in situ*, such as absorption, surface smoothness, and particulate radiation energy.

6.7.1 Direct Measurement Sensitivity

Prior to performing field measurements, an investigator must evaluate the detection sensitivity of the equipment proposed for use to ensure that levels below the DCGL can be detected (see Section 4.3). After a direct measurement has been made, it is then necessary to determine whether or not the result can be distinguished from the instrument background response of the measurement system. The terms that are used in this manual to define detection sensitivity for fixed point counts and sample analyses are:

Critical level (L_C)
Detection limit (L_D)
Minimum detectable concentration (MDC)

The critical level (L_C) is the level, in counts, at which there is a statistical probability (with a predetermined confidence) of incorrectly identifying a measurement system background value as "greater than background." Any response above this level is considered to be greater than background. The detection limit (L_D) is an *a priori* estimate of the detection capability of a measurement system, and is also reported in units of counts. The minimum detectable concentration (MDC) is the detection limit (counts) multiplied by an appropriate conversion factor to give units consistent with a site guideline, such as Bq/kg.

The following discussion provides an overview of the derivation contained in the well known publication by Currie (Currie 1968) followed by a description of how the resulting formulae should be used. Publications by Currie (Currie 1968, NRC 1984) and Altshuler and Pasternack (Altshuler and Pasternak 1963) provide details of the derivations involved.

The two parameters of interest for a detector system with a background response greater than zero are:

L_C the net response level, in counts, at which the detector output can be considered "above background"

L_D the net response level, in counts, that can be expected to be seen with a detector with a fixed level of certainty

Assuming that a system has a background response and that random uncertainties and systematic uncertainties are accounted for separately, these parameters can be calculated using Poisson statistics. For these calculations, two types of decision errors should be considered. A Type I error (or "false positive") occurs when a detector response is considered to be above background when, in fact, only background radiation is present. A Type II error (or "false negative") occurs when a detector response is considered to be background when in fact radiation is present at levels above background. The probability of a Type I error is referred to as α (alpha) and is associated with L_C; the probability of a Type II error is referred to as ß (beta) and is associated with L_D. Figure 6.2 graphically illustrates the relationship of these terms with respect to each other and to a normal background distribution.

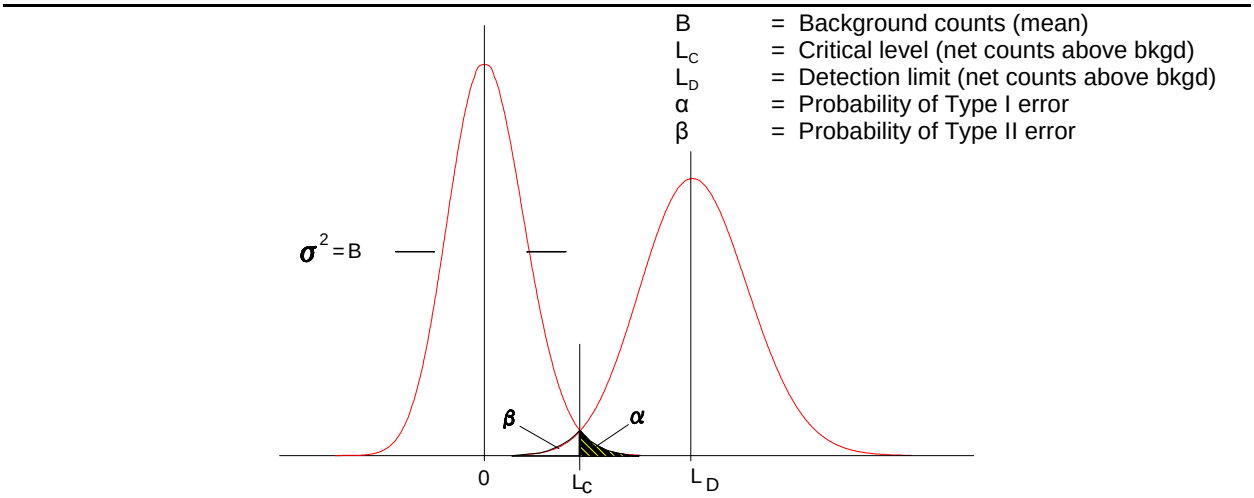

Figure 6.2 Graphically Represented Probabilities for Type I and Type II Errors in Detection Sensitivity for Instrumentation with a Background Response

If α and β are assumed to be equal, the variance (σ^2) of all measurement values is assumed to be equal to the values themselves. If the background of the detection system is not well known, then the critical detection level and the detection limit can be calculated by using the following formulae:

$$L_C = k\sqrt{2B}$$
$$L_D = k^2 + 2k\sqrt{2B} \quad (6\text{-}5)$$

where

L_C	=	critical level (counts)
L_D	=	detection limit (counts)
k	=	Poisson probability sum for α and β (assuming α and β are equal)
B	=	number of background counts that are expected to occur while performing an actual measurement

The curve to the left in the diagram is the background distribution minus the mean of the background distribution. The result is a Poisson distribution with a mean equal to zero and a variance, σ^2, equal to B. Note that the distribution accounts only for the expected statistical variation due to the stochastic nature of radioactive decay. Currie assumed "paired blanks" when deriving the above stated relationships (Currie 1968), which is interpreted to mean that the sample and background count times are the same.

If values of 0.05 for both α and β are selected as acceptable, then k = 1.645 (from Appendix I, Table I.1) and Equation 6-5 can be written as:

$$L_C = 2.33\sqrt{B}$$
$$L_D = 3 + 4.65\sqrt{B} \quad (6\text{-}6)$$

Note: In Currie's derivation, the constant factor of 3 in the L_D formula was stated as being 2.71, but since that time it has been shown (Brodsky 1992) and generally accepted that a constant factor of 3 is more appropriate. If the sample count times and background count times are different, a slightly different formulation is used.

For an integrated measurement over a preset time, the MDC can be obtained from Equation 6-6 by multiplying by the factor, C. This factor is used to convert from counts to concentration as shown in Equation 6-7:

$$MDC = C \times (3 + 4.65\sqrt{B}) \quad (6\text{-}7)$$

The total detection efficiency and other constants or factors represented by the variable C are usually not truly constants as shown in Equation 6-7. It is likely that at least one of these factors

will have a certain amount of variability associated with it which may or may not be significant. These varying factors are gathered together into the single constant, C, by which the net count result will be multiplied when converting the final data. If C varies significantly between measurements, then it might be best to select a value, C′, from the observed distribution of C values that represents a conservative estimate. For example, a value of C might be selected to ensure that at least 95% of the possible values of C are less than the chosen value, C′. The MDC calculated in this way helps assure that the survey results will meet the Data Quality Objectives. This approach for including uncertainties into the MDC calculation is recommended in both NUREG/CR-4007 (NRC 1984) and Appendix A to ANSI N13.30 (ANSI 1996a). Underestimating an MDC can have adverse consequences, especially if activity is later detected at a level above the stated MDC.

Summary of Direct Measurement Sensitivity Terms

- The MDC is the *a priori* net activity level above the critical level that an instrument can be expected to detect 95% of the time. This value should be used when stating the detection capability of an instrument. The MDC is the detection limit, $L_{D,}$ multiplied by an appropriate conversion factor to give units of activity. Again, this value is used before any measurements are made and is used to estimate the level of activity that can be detected using a given protocol.

- The critical level, L_C, is the lower bound on the 95% detection interval defined for L_D and is the level at which there is a 5% chance of calling a background value "greater than background." This value should be used when actually counting samples or making direct radiation measurements. Any response above this level should be considered as above background (*i.e.*, a net positive result). This will ensure 95% detection capability for L_D.

- From a conservative point of view, it is better to overestimate the MDC for a measurement method. Therefore, when calculating MDC and L_C values, a measurement system background value should be selected that represents the high end of what is expected for a particular measurement method. For direct measurements, probes will be moved from point to point and, as a result, it is expected that the background will most likely vary significantly due to variations in background, source materials, and changes in geometry and shielding. Ideally, the MDC values should be calculated for each type of area, but it may be more economical to simply select a background value from the highest distribution expected and use this for all calculations. For the same reasons, realistic values of detection efficiencies and other process parameters should be used when possible and should be reflective of the actual conditions. To a great degree, the selection of these parameters will be based on judgment and will require evaluation of site-specific conditions.

MDC values for other counting conditions may be derived from Equation 6-7 depending on the detector and contaminants of concern. For example, it may be required to determine what level of contamination, distributed over 100 cm^2, can be detected with a 500 cm^2 probe or what contamination level can be detected with any probe when the contamination area is smaller than the probe active area. Table 6.4 lists several common field survey detectors with estimates of MDC values for ^{238}U on a smooth, flat plane. As such, these represent minimum MDC values and may not be applicable at all sites. Appropriate site-specific MDC values should be determined using the DQO Process.

Table 6.4 Examples of Estimated Detection Sensitivities for Alpha and Beta Survey Instrumentation

(Static one minute counts for ^{238}U calculated using Equations 6-6 and 6-7)

Detector	Probe area (cm^2)	Background (cpm)	Efficiency (cpm/dpm)	Approximate Sensitivity		
				L_C (counts)	L_D (counts)	MDC (Bq/m^2) [a]
Alpha proportional	50	1	0.15	2	7	150
Alpha proportional	100	1	0.15	2	7	83
Alpha proportional	600	5	0.15	5	13	25
Alpha scintillation	50	1	0.15	2	7	150
Beta proportional	100	300	0.20	40	83	700
Beta proportional	600	1500	0.20	90	183	250
Beta GM pancake	15	40	0.20	15	32	1800

[a] Assumes that the size of the contamination area is at least as large as the probe area.

Sample Calculation 1:

The following example illustrates the calculation of an MDC in Bq/m^2 for an instrument with a 15 cm^2 probe area when the measurement and background counting times are each one minute:

B = 40 counts
C = (5 dpm/count)(Bq/60 dpm)(1/15 cm^2 probe area)(10,000 cm^2/m^2)
= 55.6 Bq/m^2-counts

The MDC is calculated using Equation 6-7:

$$MDC = 55.6 \times (3 + 4.65\sqrt{40}) = 1{,}800\ Bq/m^2\ (1{,}100\ dpm/100\ cm^2)$$

The critical level, L_c, for this example is calculated from Equation 6-6:

$$L_C = 2.33\sqrt{B} = 15 \text{ counts}$$

Given the above scenario, if a person asked what level of contamination could be detected 95% of the time using this method, the answer would be 1,800 Bq/m^2 (1,100 dpm/100 cm^2). When actually performing measurements using this method, any count yielding greater than 55 total counts, or greater than 15 net counts (55-40=15) during a period of one minute, would be regarded as greater than background.

6.7.2 Scanning Sensitivity

The ability to identify a small area of elevated radioactivity during surface scanning is dependent upon the surveyor's skill in recognizing an increase in the audible or display output of an instrument. For notation purposes, the term "scanning sensitivity" is used throughout this section to describe the ability of a surveyor to detect a pre-determined level of contamination with a detector. The greater the sensitivity, the lower the level of contamination that can be detected.

Many of the radiological instruments and monitoring techniques typically used for occupational health physics activities may not provide the detection sensitivities necessary to demonstrate compliance with the DCGLs. The detection sensitivity for a given application can be improved (*i.e.*, lower the MDC) by: 1) selecting an instrument with a higher detection efficiency or a lower background, 2) decreasing the scanning speed, or 3) increasing the size of the effective probe area without significantly increasing the background response.

Scanning is usually performed during radiological surveys in support of decommissioning to identify the presence of any areas of elevated activity. The probability of detecting residual contamination in the field depends not only on the sensitivity of the survey instrumentation when used in the scanning mode of operation, but is also affected by the surveyor's ability—*i.e.*, human factors. The surveyor must make a decision whether the signals represent only the

background activity, or residual contamination in excess of background. The greater the sensitivity, the lower the level of contamination that may be detected by scanning. Accounting for these human factors represents a significant change from the traditionally accepted methods of estimating scanning sensitivities.

An empirical method for evaluating the detection sensitivity for contamination surveys is by actual experimentation or, since it is certainly feasible, by simulating an experimental setup using computer software. The following steps provide a simple example of how one can perform this empirical evaluation:

1) A desired nuclide contamination level is selected.
2) The response of the detector to be used is determined for the selected nuclide contamination level.
3) A test source is constructed which will give a detector count rate equivalent to what was determined in step 2. The count rate is equivalent to what would be expected from the detector when placed on an actual contamination area equal in value to that selected in step 1.
4) The detector of choice is then moved over the source at different scan rates until an acceptable speed is determined.

The most useful aspect of this approach is that the source can then be used to show surveyors what level of contamination is expected to be targeted with the scan. They, in turn, can gain experience with what the expected response of the detector will be and how fast they can survey and still feel comfortable about detecting the target contamination level. The person responsible for the survey can then use this information when developing a fixed point measurement and sampling plan.

The remainder of this section is dedicated to providing the reader with information pertaining to the underlying processes involved when performing scanning surveys for alpha, beta, and gamma emitting radionuclides. The purpose is to provide relevant information that can be used for estimating realistic scanning sensitivities for survey activities.

6.7.2.1 Scanning for Beta and Gamma Emitters

The minimum detectable concentration of a scan survey (scan MDC) depends on the intrinsic characteristics of the detector (efficiency, physical probe area, *etc.*), the nature (type and energy of emissions) and relative distribution of the potential contamination (point versus distributed source and depth of contamination), scan rate, and other characteristics of the surveyor. Some factors that may affect the surveyor's performance include the costs associated with various outcomes—*e.g.*, fatigue, noise, level of training, experience—and the survey's *a priori* expectation of the likelihood of contamination present. For example, if the surveyor believes that

the potential for contamination is very low, as in a Class 3 area, a relatively large signal may be required for the surveyor to conclude that contamination is present. NRC draft report NUREG/CR-6364 (NRC 1997d) provides a complete discussion of the human factors as they relate to the performance of scan surveys.

Signal Detection Theory. Personnel conducting radiological surveys for residual contamination at decommissioning sites must interpret the audible output of a portable survey instrument to determine when the signal ("clicks") exceeds the background level by a margin sufficient to conclude that contamination is present. It is difficult to detect low levels of contamination because both the signal and the background vary widely. Signal detection theory provides a framework for the task of deciding whether the audible output of the survey meter during scanning is due to background or signal plus background levels. An index of sensitivity (d') that represents the distance between the means of the background and background plus signal (refer to Figure 6.2 for determining L_D), in units of their common standard deviation, can be calculated for various decision errors (correct detection and false positive rate). As an example, for a correct detection rate of 95% (complement of a false negative rate of 5%) and a false positive rate of 5%, d' is 3.29 (similar to the static MDC for the same decision error rates). The index of sensitivity is independent of human factors, and therefore, the ability of an ideal observer (theoretical construct), may be used to determine the minimum d' that can be achieved for particular decision errors. The ideal observer makes optimal use of the available information to maximize the percent correct responses, providing an effective upper bound against which to compare actual surveyors. Table 6.5 lists selected values of d'.

Two Stages of Scanning. The framework for determining the scan MDC is based on the premise that there are two stages of scanning. That is, surveyors do not make decisions on the basis of a single indication, rather, upon noting an increased number of counts, they pause briefly and then decide whether to move on or take further measurements. Thus, scanning consists of two components: continuous monitoring and stationary sampling. In the first component, characterized by continuous movement of the probe, the surveyor has only a brief "look" at potential sources, determined by the scan speed. The surveyor's willingness to decide that a signal is present at this stage is likely to be liberal, in that the surveyor should respond positively on scant evidence, since the only "cost" of a false positive is a little time. The second component occurs only after a positive response was made at the first stage. This response is marked by the surveyor interrupting his scanning and holding the probe stationary for a period of time, while comparing the instrument output signal during that time to the background counting rate. Owing to the longer observation interval, sensitivity is relatively high. For this decision, the criterion should be more strict, since the cost of a "yes" decision is to spend considerably more time taking a static measurement or a sample.

Table 6.5 Values of *d′* for Selected True Positive and False Positive Proportions

False Positive Proportion	True Positive Proportion							
	0.60	**0.65**	**0.70**	**0.75**	**0.80**	**0.85**	**0.90**	**0.95**
0.05	1.90	2.02	2.16	2.32	2.48	2.68	2.92	3.28
0.10	1.54	1.66	1.80	1.96	2.12	2.32	2.56	2.92
0.15	1.30	1.42	1.56	1.72	1.88	2.08	2.32	2.68
0.20	1.10	1.22	1.36	1.52	1.68	1.88	2.12	2.48
0.25	0.93	1.06	1.20	1.35	1.52	1.72	1.96	2.32
0.30	0.78	0.91	1.05	1.20	1.36	1.56	1.80	2.16
0.35	0.64	0.77	0.91	1.06	1.22	1.42	1.66	2.02
0.40	0.51	0.64	0.78	0.93	1.10	1.30	1.54	1.90
0.45	0.38	0.52	0.66	0.80	0.97	1.17	1.41	1.77
0.50	0.26	0.38	0.52	0.68	0.84	1.04	1.28	1.64
0.55	0.12	0.26	0.40	0.54	0.71	0.91	1.15	1.51
0.60	0.00	0.13	0.27	0.42	0.58	0.82	1.02	1.38

Since scanning can be divided into two stages, it is necessary to consider the survey's scan sensitivity for each of the stages. Typically, the minimum detectable count rate (MDCR) associated with the first scanning stage will be greater due to the brief observation intervals of continuous monitoring—provided that the length of the pause during the second stage is significantly longer. Typically, observation intervals during the first stage are on the order of 1 or 2 seconds, while the second stage pause may be several seconds long. The greater value of MDCR from each of the scan stages is used to determine the scan sensitivity for the surveyor.

Determination of MDCR and Use of Surveyor Efficiency. The minimum detectable number of net source counts in the interval is given by s_i. Therefore, for an ideal observer, the number of source counts required for a specified level of performance can be arrived at by multiplying the square root of the number of background counts by the detectability value associated with the desired performance (as reflected in *d′*) as shown in Equation 6-8:

$$s_i = d' \sqrt{b_i} \qquad (6\text{-}8)$$

where the value of *d′* is selected from Table 6.5 based on the required true positive and false positive rates and b_i is the number of background counts in the interval.

For example, suppose that one wished to estimate the minimum count rate that is detectable by scanning in an area with a background of 1,500 cpm. Note that the minimum detectable count rate must be considered for both scan stages—and the more conservative value is selected as the minimum count rate that is detectable. It will be assumed that a typical source remains under the probe for 1 second during the first stage, therefore, the average number of background counts in the observation interval is 25 ($b_i = 1500 \times (1/60)$). Furthermore, as explained earlier, it can be assumed that at the first scanning stage a high rate (*e.g.*, 95%) of correct detections is required, and that a correspondingly high rate of false positives (*e.g.*, 60%) will be tolerated. From Table 6.5, the value of *d′*, representing this performance goal, is 1.38. The net source counts needed to support the specified level of performance (assuming an ideal observer) will be estimated by multiplying 5 (the square root of 25) by 1.38. Thus, the net source counts per interval, s_i, needed to yield better than 95% detections with about 60% false positives is 6.9. The minimum detectable source count rate, in cpm, may be calculated by:

$$MDCR = s_i \times (60/i) \tag{6-9}$$

For this example, MDCR is equivalent to 414 cpm (1,914 cpm gross). Table 6.6 provides the scan sensitivity for the ideal observer (MDCR) at the first scanning stage for various background levels, based on an index of sensitivity (*d′*) of 1.38 and a 2-second observation interval.

Table 6.6 Scanning Sensitivity (MDCR) of the Ideal Observer for Various Background Levels[a]

Background (cpm)	MDCR (net cpm)	Scan Sensitivity (gross cpm)
45	50	95
60	60	120
260	120	380
300	130	430
350	140	490
400	150	550
1,000	240	1,240
3,000	410	3,410
4,000	480	4,480

*The sensitivity of the ideal observer during the first scanning stage is based on an index of sensitivity (d′) of 1.38 and a 2-second observation interval.

The minimum number of source counts required to support a given level of performance for the final detection decision (second scan stage) can be estimated using the same method. As explained earlier, the performance goal at this stage will be more demanding. The required rate of true positives remains high (*e.g.*, 95%), but fewer false positives (*e.g.*, 20%) can be tolerated, such that d' (from Table 6.5) is now 2.48. One will assume that the surveyor typically stops the probe over a suspect location for about 4 seconds before making a decision, so that the average number of background counts in an observation interval is 100 ($b_i = 1{,}500 \times (4/60)$). Therefore, the minimum detectable number of net source counts, s_i, needed will be estimated by multiplying 10 (the square root of 100) by 2.48 (the d' value); so s_i equals 24.8. The MDCR is calculated by $2.48 \times (60/4)$ and equals 372 cpm. The value associated with the first scanning stage (this example, 414 cpm) will typically be greater, owing to the relatively brief intervals assumed.

Laboratory studies using simulated sources and backgrounds were performed to assess the abilities of surveyors under controlled conditions. The methodology and analysis of results for these studies are described in draft NUREG/CR-6364 (NRC 1997d) and NUREG-1507 (NRC 1997b). The surveyor's actual performance as compared with that which is ideally possible (using the ideal observer construct) provided an indication of the efficiency of the surveyors. Based on the results of the confidence rating experiment, this surveyor efficiency (p) was estimated to be between 0.5 and 0.75.

MARSSIM recommends assuming an efficiency value at the lower end of the observed range (*i.e.*, 0.5) when making MDC estimates. Thus, the required number of net source counts for the surveyor, $MDCR_{surveyor}$, is determined by dividing the MDCR by the square root of p. Continuing with this example, the surveyor MDCR is calculated by 414 cpm/0.707, or 585 cpm (2,085 cpm gross).

Scan MDCs for Structure Surfaces and Land Areas. The survey design for determining the number of data points for areas of elevated activity (see Section 5.5.2.4) depends on the scan MDC for the selected instrumentation. In general, alpha or beta scans are performed on structure surfaces to satisfy the elevated activity measurements survey design, while gamma scans are performed for land areas. Because of low background levels for alpha emitters, the approach described here is not generally applied to determining scan MDCs for alpha contaminants—rather, the reader is referred to Section 6.7.2.2 for an appropriate method for determining alpha scan MDCs for building surfaces. In any case, the data requirements for assessing potential elevated areas of direct radiation depend on the scan MDC of the survey instrument (*e.g.*, floor monitor, GM detector, NaI scintillation detector).

Scan MDCs for Building/Structure Surfaces. The scan MDC is determined from the minimum detectable count rate (MDCR) by applying conversion factors that account for detector and surface characteristics and surveyor efficiency. As discussed above, the MDCR accounts for the background level, performance criteria (d'), and observation interval. The observation interval

during scanning is the actual time that the detector can respond to the contamination source—this interval depends on the scan speed, detector size in the direction of the scan, and area of elevated activity. Because the actual dimensions of potential areas of elevated activity in the field cannot be known *a priori*, MARSSIM recommends postulating a certain area (*e.g.*, perhaps 50 to 200 cm^2), and then selecting a scan rate that provides a reasonable observation interval.

Finally, the scan MDC for structure surfaces may be calculated:

$$Scan\ MDC = \frac{MDCR}{\sqrt{p}\ \varepsilon_i\ \varepsilon_s\ \frac{probe\ area}{100\ cm^2}} \qquad (6\text{-}10)$$

where

MDCR	=	minimum detectable count rate
ε_i	=	instrument efficiency
ε_s	=	surface efficiency
p	=	surveyor efficiency

As an example, the scan MDC (in dpm/100 cm^2) for ^{99}Tc on a concrete surface may be determined for a background level of 300 cpm and a 2-second observation interval using a hand-held gas proportional detector (126 cm^2 probe area). For a specified level of performance at the first scanning stage of 95% true positive rate and 60% false positive rate (and assuming the second stage pause is sufficiently long to ensure that the first stage is more limiting), d' equals 1.38 (Table 6.5) and the MDCR is 130 cpm (Table 6.6). Using a surveyor efficiency of 0.5, and assuming instrument and surface efficiencies of 0.36 and 0.54, respectively, the scan MDC is calculated using Equation 6-10:

$$Scan\ MDC = \frac{130}{\sqrt{0.5}\ (0.36)\ (0.54)\ (1.26)} = 750\ dpm/100\ cm^2$$

Additional examples for calculating the scan MDC may be found in NUREG-1507 (NRC 1997b).

Scan MDCs for Land Areas. In addition to the MDCR and detector characteristics, the scan MDC (in pCi/g) for land areas is based on the area of elevated activity, depth of contamination, and the radionuclide (*i.e.*, energy and yield of gamma emissions). If one assumes constant parameters for each of the above variables, with the exception of the specific radionuclide in question, the scan MDC may be reduced to a function of the radionuclide alone. NaI scintillation detectors are generally used for scanning land areas.

An overview of the approach used to determine scan MDCs for land areas follows. The NaI(Tl) scintillation detector background level and scan rate (observation interval) are postulated, and the MDCR for the ideal observer, for a given level of performance, is obtained. After a surveyor efficiency is selected, the relationship between the surveyor MDCR ($MDCR_{surveyor}$) and the radionuclide concentration in soil (in Bq/kg or pCi/g)is determined. This correlation requires two steps—first, the relationship between the detector's net count rate to net exposure rate (cpm per µR/h) is established, and second, the relationship between the radionuclide contamination and exposure rate is determined.

For a particular gamma energy, the relationship of NaI(Tl) scintillation detector count rate and exposure rate may be determined analytically (in cpm per µR/h). The approach used to determine the gamma fluence rate necessary to yield a fixed exposure rate (1 µR/h)—as a function of gamma energy—is provided in NUREG-1507 (NRC 1997b). The NaI(Tl) scintillation detector response (cpm) is related to the fluence rate at specific energies, considering the detector's efficiency (probability of interaction) at each energy. From this, the NaI(Tl) scintillation detector versus exposure rates for varying gamma energies are determined. Once the relationship between the NaI(Tl) scintillation detector response (cpm) and the exposure rate is established, the $MDCR_{surveyor}$ (in cpm) of the NaI(Tl) scintillation detector can be related to the minimum detectable net exposure rate. The minimum detectable exposure rate is used to determine the minimum detectable radionuclide concentration (*i.e.*, the scan MDC) by modeling a specified small area of elevated activity.

Modeling (using Microshield™) of the small area of elevated activity (soil concentration) is used to determine the net exposure rate produced by a radionuclide concentration at a distance 10 cm above the source. This position is selected because it relates to the average height of the NaI(Tl) scintillation detector above the ground during scanning.

The factors considered in the modeling include:

- radionuclide of interest (considering all gamma emitters for decay chains)
- expected concentration of the radionuclide of interest
- areal dimensions of the area of elevated activity
- depth of the area of elevated activity
- location of dose point (NaI(Tl) scintillation detector height above the surface)
- density of soil

Modeling analyses are conducted by selecting a radionuclide (or radioactive material decay series) and then varying the concentration of the contamination. The other factors are held constant—the areal dimension of a cylindrical area of elevated activity is 0.25 m^2 (radius of 28 cm), the depth of the area of elevated activity is 15 cm, the dose point is 10 cm above the surface, and the density of soil is 1.6 g/cm^3. The objective is to determine the radionuclide concentration that is correlated to the minimum detectable net exposure rate.

As an example, the scan MDC for ^{137}Cs using a 1.5 in. by 1.25 in. NaI(Tl) scintillation detector is considered in detail. Assume that the background level is 4,000 cpm and that the desired level of performance, 95% correct detections and 60% false positive rate, results in a d' of 1.38. The scan rate of 0.5m/s provides an observation interval of 1-second (based on a diameter of about 56 cm for the area of elevated activity). The $MDCR_{surveyor}$ may be calculated assuming a surveyor efficiency (p) of 0.5 as follows:

1) $b_i = (4{,}000 \text{ cpm}) \times (1 \text{ sec}) \times (1 \text{ min}/60 \text{ sec}) = 66.7 \text{ counts}$

2) $MDCR = (1.38) \times (\sqrt{66.7}) \times (60 \text{ sec}/1 \text{ min}) = 680 \text{ cpm}$

3) $MDCR_{surveyor} = 680/\sqrt{0.5} = 960 \text{ cpm}$

The corresponding minimum detectable exposure rate is determined for this detector and radionuclide. The manufacturer of this particular 1.5 in. by 1.25 in. NaI(Tl) scintillation detector quotes a count rate to exposure rate ratio for ^{137}Cs of 350 cpm per µR/h. The minimum detectable exposure rate is calculated by dividing the count rate (960 cpm) by the count rate to exposure rate ratio for the radionuclide of interest (350 cpm per µR/h). The minimum detectable exposure rate for this example is 2.73 µR/h.

Both ^{137}Cs and its short-lived progeny, ^{137m}Ba, were chosen from the Microshield™ library. The source activity and other modeling parameters were entered into the modeling code. The source activity was selected based on an arbitrary concentration of 5 pCi/g. The modeling code performed the appropriate calculations and determined an exposure rate of 1.307 µR/h (which accounts for buildup). Finally, the radionuclide concentrations of ^{137}Cs and ^{137m}Ba (scan MDC) necessary to yield the minimum detectable exposure rate (2.73 μR/h) may be calculated using the following formula.

$$scan\ MDC = \frac{(5\ pCi/g)(2.73\ \mu R/h)}{1.307\ \mu R/h} = 10.4\ pCi/g \qquad (6\text{-}11)$$

It must be emphasized that while a single scan MDC value can be calculated for a given radionuclide—other scan MDC values may be equally justifiable depending on the values chosen for the various factors, including the MDCR (background level, acceptable performance criteria, observation interval), surveyor efficiency, detector parameters and the modeling conditions of the contamination. It should also be noted that determination of the scan MDC for radioactive materials—like uranium and thorium—must consider the gamma radiation emitted from the entire decay series. NUREG-1507 (NRC 1997b) provides a detailed example of how the scan MDC can be determined for enriched uranium.

Table 6.7 provides scan MDCs for common radionuclides and radioactive materials in soil. It is important to note that the variables used in the above examples to determine the scan MDCs for the 1.25 in. by 1.5 in. NaI(Tl) scintillation detector—*i.e.*, the $MDCR_{surveyor}$ detector parameters (*e.g.*, cpm per µR/h), and the characteristics of the area of elevated activity—have all been held constant to facilitate the calculation of scan MDCs provided in Table 6.7. The benefit of this approach is that generally applicable scan MDCs are provided for different radioactive contaminants. Additionally, the relative detectability of different contaminants is evident because the only variable in Table 6.7 is the nature of the contaminant.

As noted above, the scan MDCs calculated using the approach in this section are dependent on several factors. One way to validate the appropriateness of the scan MDC is by tracking the residual radioactivity (both surface activity and soil concentrations) levels identified during investigations performed as a result of scanning surveys. The measurements performed during these investigations may provide an *a posteriori* estimate of the scan MDC that can be used to validate the *a priori* scan MDC used to design the survey.

6.7.2.2 Scanning for Alpha Emitters

Scanning for alpha emitters differs significantly from scanning for beta and gamma emitters in that the expected background response of most alpha detectors is very close to zero. The following discussion covers scanning for alpha emitters and assumes that the surface being surveyed is similar in nature to the material on which the detector was calibrated. In this respect, the approach is purely theoretical. Surveying surfaces that are dirty, non-planar, or weathered can significantly affect the detection efficiency and therefore bias the expected MDC for the scan. The use of reasonable detection efficiency values instead of optimistic values is highly recommended. Appendix J contains a complete derivation of the alpha scanning equations used in this section.

Since the time a contaminated area is under the probe varies and the background count rate of some alpha instruments is less than 1 cpm, it is not practical to determine a fixed MDC for scanning. Instead, it is more useful to determine the probability of detecting an area of contamination at a predetermined DCGL for given scan rates.

For alpha survey instrumentation with backgrounds ranging from <1 to 3 cpm, a single count provides a surveyor sufficient cause to stop and investigate further. Assuming this to be true, the probability of detecting given levels of alpha surface contamination can be calculated by use of Poisson summation statistics.

Table 6.7 NaI(Tl) Scintillation Detector Scan MDCs for Common Radiological Contaminants[a]

Radionuclide/Radioactive Material	1.25 in. by 1.5 in. NaI Detector		2 in. by 2 in. NaI Detector	
	Scan MDC (Bq/kg)	Weighted cpm/μR/h	Scan MDC (Bq/kg)	Weighted cpm/μR/h
Am-241	1,650	5,830	1,170	13,000
Co-60	215	160	126	430
Cs-137	385	350	237	900
Th-230	111,000	4,300	78,400	9,580
Ra-226 (in equilibrium with progeny)	167	300	104	760
Th-232 decay series (Sum of all radionuclides in he thorium decay series)	1,050	340	677	830
Th-232 (In equilibrium with progeny in decay series)	104	340	66.6	830
Depleted Uranium[b] (0.34% U-235)	2,980	1,680	2,070	3,790
Natural Uranium[b]	4,260	1,770	2,960	3,990
3% Enriched Uranium[b]	5,070	2,010	3,540	4,520
20% Enriched Uranium[b]	5,620	2,210	3,960	4,940
50% Enriched Uranium[b]	6,220	2,240	4,370	5,010
75% Enriched Uranium[b]	6,960	2,250	4,880	5,030

[a] Refer to text for complete explanation of factors used to calculate scan MDCs. For example, the background level for the 1.25 in. by 1.5 in. NaI detector was assumed to be 4,000 cpm, and 10,000 cpm for the 2 in. by 2 in. NaI detector. The observation interval was 1-sec and the level of performance was selected to yield d' of 1.38.

[b] Scan MDC for uranium includes sum of ^{238}U, ^{235}U, and ^{234}U.

Given a known scan rate and a surface contamination DCGL, the probability of detecting a single count while passing over the contaminated area is

$$P(n \geq 1) = 1 - e^{\frac{-GEd}{60v}} \quad (6\text{-}12)$$

where

$P(n \geq 1)$	=	probability of observing a single count
G	=	contamination activity (dpm)
E	=	detector efficiency (4π)
d	=	width of detector in direction of scan (cm)
v	=	scan speed (cm/s)

Note: Refer to Appendix J for a complete derivation of these formulas.

Once a count is recorded and the guideline level of contamination is present the surveyor should stop and wait until the probability of getting another count is at least 90%. This time interval can be calculated by

$$t = \frac{13{,}800}{CAE} \quad (6\text{-}13)$$

where

t	=	time period for static count (s)
C	=	contamination guideline (dpm/100 cm^2)
A	=	physical probe area (cm^2)
E	=	detector efficiency (4π)

Many portable proportional counters have background count rates on the order of 5 to 10 cpm, and a single count should not cause a surveyor to investigate further. A counting period long enough to establish that a single count indicates an elevated contamination level would be prohibitively inefficient. For these types of instruments, the surveyor usually will need to get at least 2 counts while passing over the source area before stopping for further investigation.

Assuming this to be a valid assumption, the probability of getting two or more counts can be calculated by:

$$P(n \geq 2) = 1 - P(n=0) - P(n=1)$$

$$= 1 - \left(1 + \frac{(GE + B)t}{60}\right)\left(e^{-\frac{(GE + B)t}{60}}\right) \quad (6\text{-}14)$$

where

$P(n \geq 2)$	=	probability of getting 2 or more counts during the time interval t
$P(n=0)$	=	probability of not getting any counts during the time interval t
$P(n=1)$	=	probability of getting 1 count during the time interval t
B	=	background count rate (cpm)

All other variables are the same as for Equation 6-12.

Appendix J provides a complete derivation of Equations 6-12 through 6-14 and a detailed discussion of the probability of detecting alpha surface contamination for several different variables. Several probability charts are included at the end of Appendix J for common detector sizes. Table 6.8 provides estimates of the probability of detecting 300 dpm/100 cm^2 for some commonly used alpha detectors.

Table 6.8 Probability of Detecting 300 dpm/100 cm^2 of Alpha Activity While Scanning with Alpha Detectors Using an Audible Output (calculated using Equation 6-12)

Detector Type	Detection Efficiency cpm/dpm	Probe Dimension in Direction of Scan (cm)	Scan Rate (cm/s)	Probability of detecting 300 dpm/100 cm^2
Proportional	0.20	5	3	80%
Proportional	0.15	15	5	90%
Scintillation	0.15	5	3	70%
Scintillation	0.15	10	3	90%

6.8 Measurement Uncertainty (Error)

The quality of measurement data will be directly impacted by the magnitude of the measurement uncertainty associated with it. Some uncertainties, such as statistical counting uncertainties, can be easily calculated from the count results using mathematical procedures. Evaluation of other

sources of uncertainty require more effort and in some cases is not possible. For example, if an alpha measurement is made on a porous concrete surface, the observed instrument response when converted to units of activity will probably not exactly equal the true activity under the probe. Variations in the absorption properties of the surface for particulate radiation will vary from point to point and therefore will create some level of variation in the expected detection efficiency. This variability in the expected detector efficiency results in uncertainty in the final reported result. In addition, QC measurement results provide an estimate of random and systematic uncertainties associated with the measurement process.

The measurement uncertainty for every analytical result or series of results, such as for a measurement system, should be reported. This uncertainty, while not directly used for demonstrating compliance with the release criterion, is used for survey planning and data assessment throughout the Radiation Survey and Site Investigation (RSSI) process. In addition, the uncertainty is used for evaluating the performance of measurement systems using QC measurement results. Uncertainty can also be used for comparing individual measurements to the DCGL. This is especially important in the early stages of decommissioning (*i.e.*, scoping, characterization, remedial action support) when decisions are made based on a limited number of measurements.

For most sites, evaluations of uncertainty associated with field measurements is important only for data being used as part of the final status survey documentation. The final status survey data, which is used to document the final radiological status of a site, should state the uncertainties associated with the measurements. Conversely, detailing the uncertainties associated with measurements made during scoping or characterization surveys may or may not be of value depending on what the data will be used for—*i.e.* the data quality objectives (DQOs). From a practical standpoint, if the observed data are obviously greater than the DCGL and will be eventually cleaned up, then the uncertainty may be relatively unimportant. Conversely, data collected during early phases of a site investigation that may eventually be used to show that the area is below the DCGL—and therefore does not require any clean-up action—will need the same uncertainty evaluation as the final status survey data. In summary, the level of effort needs to match the intended use of the data.

6.8.1 Systematic and Random Uncertainties

Measurement uncertainties are often broken into two sub-classes of uncertainty termed systematic (*e.g.*, methodical) uncertainty and random (*e.g.*, stochastic) uncertainty. Systematic uncertainties derive from a lack of knowledge about the true distribution of values associated with a numerical parameter and result in data that is consistently higher (or lower) than the true value. An example of a systematic uncertainty would be the use of a fixed counting efficiency value even though it is known that the efficiency varies from measurement to measurement but without knowledge of the frequency. If the fixed counting efficiency value is higher than the true

but unknown efficiency—as would be the case for an unrealistically optimistic value—then every measurement result calculated using that efficiency would be biased low. Random uncertainties refer to fluctuations associated with a known distribution of values. An example of a random uncertainty would be a well documented chemical separation efficiency that is known to fluctuate with a regular pattern about a mean. A constant recovery value is used during calculations, but the true value is known to fluctuate from sample to sample with a fixed and known degree of variation.

To minimize the need for estimating potential sources of uncertainty, the sources of uncertainty themselves should be reduced to a minimal level by using practices such as:

- The detector used should minimize the potential uncertainty. For example, when making field surface activity measurements for ^{238}U on concrete, a beta detector such as a thin-window Geiger-Mueller "pancake" may provide better quality data than an alpha detector depending on the circumstances. Less random uncertainty would be expected between measurements with a beta detector such as a pancake since beta emissions from the uranium will be affected much less by thin absorbent layers than will the alpha emissions.

- Calibration factors should accurately reflect the efficiency of a detector being used on the surface material being measured for the contaminant radionuclide or mixture of radionuclides (see Section 6.5.4). For most field measurements, variations in the counting efficiency on different types of materials will introduce the largest amount of uncertainty in the final result.

- Uncertainties should be reduced or eliminated by use of standardized measurement protocols (*e.g.*, SOPs) when possible. Special effort should be made to reduce or eliminate systematic uncertainties, or uncertainties that are the same for every measurement simply due to an error in the process. If the systematic uncertainties are reduced to a negligible level, then the random uncertainties, or those uncertainties that occur on a somewhat statistical basis, can be dealt with more easily.

- Instrument operators should be trained and experienced with the instruments used to perform the measurements.

- QA/QC should be conducted as described in Chapter 9.

Uncertainties that cannot be eliminated need to be evaluated such that the effect can be understood and properly propagated into the final data and uncertainty estimates. As previously stated, non-statistical uncertainties should be minimized as much as possible through the use of good work practices.

Overall random uncertainty can be evaluated using the methods described in the following sections. Section 6.8.2 describes a method for calculating random counting uncertainty. Section 6.8.3 discusses how to combine this counting uncertainty with other uncertainties from the measurement process using uncertainty propagation.

Systematic uncertainty is derived from calibration errors, incorrect yields and efficiencies, non-representative survey designs, and "blunders." It is difficult—and sometimes impossible—to evaluate the systematic uncertainty for a measurement process, but bounds should always be estimated and made small compared to the random uncertainty, if possible. If no other information on systematic uncertainty is available, Currie (NRC 1984) recommends using 16% as an estimate for systematic uncertainties (1% for blanks, 5% for baseline, and 10% for calibration factors).

6.8.2 Statistical Counting Uncertainty

When performing an analysis with a radiation detector, the result will have an uncertainty associated with it due to the statistical nature of radioactive decay. To calculate the total uncertainty associated with the counting process, both the background measurement uncertainty and the sample measurement uncertainty must be accounted for. The standard deviation of the net count rate, or the statistical counting uncertainty, can be calculated by

$$\sigma_n = \sqrt{\frac{C_{s+b}}{T_{s+b}^2} + \frac{C_b}{T_b^2}} \qquad (6\text{-}15)$$

where

σ_n = standard deviation of the net count rate result
C_{s+b} = number of gross counts (sample)
T_{s+b} = gross count time
C_b = number of background counts
T_b = background count time

6.8.3 Uncertainty Propagation

Most measurement data will be converted to different units or otherwise included in a calculation to determine a final result. The standard deviation associated with the final result, or the total uncertainty, can then be calculated. Assuming that the individual uncertainties are relatively small, symmetric about zero, and independent of one another, then the total uncertainty for the final calculated result can be determined by solving the following partial differential equation:

$$\sigma_u = \sqrt{\left(\frac{\partial u}{\partial x}\right)^2 \sigma_x^2 + \left(\frac{\partial u}{\partial y}\right)^2 \sigma_y^2 + \left(\frac{\partial u}{\partial z}\right)^2 \sigma_z^2 + \ldots} \qquad (6\text{-}16)$$

where

u	=	function, or formula, that defines the calculation of a final result as a function of the collected data. All variables in this equation, *i.e.*, x, y, z..., are assumed to have a measurement uncertainty associated with them and do not include numerical constants
σ_u	=	standard deviation, or uncertainty, associated with the final result
σ_x, σ_y,...	=	standard deviation, or uncertainty, associated with the parameters x, y, z, ...

Equation 6-16, generally known as the error propagation formula, can be solved to determine the standard deviation of a final result from calculations involving measurement data and their associated uncertainties. The solutions for common calculations along with their uncertainty propagation formulas are included below.

Data Calculation	Uncertainty Propagation
u = x + y , or u= x - y :	$\sigma_u = \sqrt{\sigma_x^2 + \sigma_y^2}$
u = x ÷ y , or u = x × y :	$\sigma_u = u\sqrt{\left(\frac{\sigma_x}{x}\right)^2 + \left(\frac{\sigma_y}{y}\right)^2}$
u = c × x, where c is a positive constant:	$\sigma_u = c\sigma_x$
u = x ÷ c, where c is a positive constant:	$\sigma_u = \frac{\sigma_x}{c}$

Note: In the above examples, x and y are measurement values with associated standard deviations, or uncertainties, equal to σ_x and σ_y respectively. The symbol "c" is used to represent a numerical constant which has no associated uncertainty. The symbol σ_u is used to denote the standard deviation, or uncertainty, of the final calculated value u.

6.8.4 Reporting Confidence Intervals

Throughout Section 6.8, the term "measurement uncertainty" is used interchangeably with the term "standard deviation." In this respect, the uncertainty is qualified as numerically identical to

the standard deviation associated with a normally distributed range of values. When reporting a confidence interval for a value, one provides the range of values that represent a pre-determined level of confidence (*i.e.*, 95%). To make this calculation, the final standard deviation, or total uncertainty σ_u as shown in Equation 6-16, is multiplied by a constant factor k representing the area under a normal curve as a function of the standard deviation. The values of k representing various intervals about a mean of normal distributions as a function of the standard deviation is given in Table 6.9. The following example illustrates the use of this factor in context with the propagation and reporting of uncertainty values.

Table 6.9 Areas Under Various Intervals About the Mean of a Normal Distribution

Interval ($\bar{\mu} \pm k\sigma$)	Area
$\bar{\mu} \pm 0.674\sigma$	0.500
$\bar{\mu} \pm 1.00\sigma$	0.683
$\bar{\mu} \pm 1.65\sigma$	0.900
$\bar{\mu} \pm 1.96\sigma$	0.950
$\bar{\mu} \pm 2.00\sigma$	0.954
$\bar{\mu} \pm 2.58\sigma$	0.990
$\bar{\mu} \pm 3.00\sigma$	0.997

Example:

Uncertainty Propagation and Confidence Interval: A measurement process with a zero background yields a count result of 28 ± 5 counts in 5 minutes, where the ± 5 counts represents one standard deviation about a mean value of 28 counts. The detection efficiency is 0.1 counts per disintegration ± 0.01 counts per disintegration, again representing one standard deviation about the mean.

Calculate the activity of the sample, in dpm, total measurement uncertainty, and the 95% confidence interval for the result.

1) The total number of disintegrations is:

$$\frac{28\ counts}{0.1\,c/d} = 280$$

2) Using the equation for error propagation for division, total uncertainty is:

$$280\sqrt{\left(\frac{5}{28}\right)^2 + \left(\frac{0.01}{0.1}\right)^2} = 57 \text{ disintegrations}$$

3) The activity will then be 280 ÷ 5 minutes = 56 dpm and the total uncertainty will be 57 ÷ 5 minutes = 11 dpm. (Since the count time is considered to have trivial variance, this is assumed to be a constant.)

Referring to Table 6.9, a k value of ±1.96 represents a confidence interval equal to 95% about the mean of a normal distribution. Therefore, the 95% confidence interval would be 1.96 × 11 dpm = 22 dpm. The final result would be 56 ± 22 dpm.

6.9 Radon Measurements

There are three radon isotopes in nature: ^{222}Rn (radon) in the ^{238}U decay chain, ^{220}Rn (thoron) in the ^{232}Th chain, and ^{219}Rn (actinon) in the ^{235}U chain. ^{219}Rn is the least abundant of these three isotopes, and because of its short half-life of 4 seconds it has the least probability of emanating into the atmosphere before decaying. ^{220}Rn with a 55 second half-life is somewhat more mobile. ^{222}Rn with a 3.8 d half-life is capable of migrating through several decimeters of soil or building material and reaching the atmosphere. Therefore, in most situations, ^{222}Rn should be the predominant airborne radon isotope.

Many techniques have been developed over the years for measuring radon (Jenkins 1986) and radon progeny in air. In addition, considerable attention is given by EPA to measurement of radon and radon progeny in homes (EPA 1992d). Radon and radon progeny emit alpha and beta particles and gamma rays. Therefore, numerous techniques can and have been developed for measuring these radionuclides based on detecting alpha particles, beta particles, or gamma rays, independently or in some combination. It is even difficult to categorize the various techniques that are presently in use. This section contains an overview of information dealing with the measurement of radon and radon progeny. The information is focused on the measurement of ^{222}Rn, however the information may be adapted for the measurement of ^{219}Rn and ^{220}Rn.

Radon concentrations within a fixed structure can vary significantly from one section of the building to another and can fluctuate over time. If a home has a basement, for instance, it is usually expected that a higher radon concentration will be found there. Likewise, a relatively small increase in the relative pressure between the soil and the inside of a structure can cause a significant increase in the radon emanation rate from the soil into the structure. Many factors play a role in these variations, but from a practical standpoint it is only necessary to recognize

that fluctuations are expected and that they should be accounted for. Long term measurement periods are required to determine a true mean concentration inside a structure and to account for the fluctuations.

Two analytical end points are of interest when performing radon measurements. The first and most commonly used is radon concentration, which is stated in terms of activity per unit volume (Bq/m^3 or pCi/L). Although this terminology is consistent with most federal guidance values, it only infers the potential dose equivalent associated with radon. The second analytical end point is the radon progeny working level. Radon progeny usually attach very quickly to charged aerosols in the air following creation. The fraction that remains unattached is usually quite small (*i.e.*, 5-10%). Since most aerosol particles carry an electrical charge and are relatively massive ($\geq$ 0.1 µm), they are capable of attaching to the surfaces of the lung. Essentially all dose or risk from radon is associated with alpha decays from radon progeny attached to tissues of the respiratory system. If an investigator is interested in accurately determining the potential dose or risk associated with radon in the air of a room, the radon progeny concentration must be known.

Radon progeny concentrations are usually reported in units of working levels (WL), where one working level is equal to the potential alpha energy associated with the radon progeny in secular equilibrium with 100 pCi/L of radon. One working level is equivalent to 1.28×10^5 MeV/L of potential alpha energy. Given a known breathing rate and lung attachment probability, the expected mean lung dose from exposure to a known working level of radon progeny can be calculated.

Radon progeny are not usually found in secular equilibrium with radon indoors due to plating out of the charged aerosols onto walls, furniture, *etc.* The ratio of ^{222}Rn progeny activity to ^{222}Rn activity usually ranges from 0.2 to as high as 0.8 indoors (NCRP 1988). If only the ^{222}Rn concentration is measured and it is not practical to measure the progeny concentrations, then general practice is to assume a progeny to ^{222}Rn equilibrium ratio of 0.5 for indoor areas. This allows one to estimate the expected dose or risk associated with a given radon concentration.

In general, the following generic guidelines should be followed when performing radon measurements during site investigations:

- The radon measurement method used should be well understood and documented.
- Long term measurements are used to determine the true mean radon concentration.
- The impact of variable environmental conditions (*e.g.*, humidity, temperature, dust loading, and atmospheric pressure) on the measurement process should be accounted for when necessary. Consideration should be given to effects on both the air collection process and the counting system.

- The background response of the detection system should be accounted for.

- If the quantity of interest is the working level, then the radon progeny concentrations should be evaluated. If this is not practical, then the progeny activities can be estimated by assuming they are 50% of the measured radon activity (NCRP 1988).

For a general overview, a list of common radiation detectors with their usual applications during radon surveys is provided in Table 6.10. Descriptions and costs for specific equipment used for the measurement of radon are contained in Appendix H.

Table 6.10 Radiation Detectors with Applications to Radon Surveys

System	Description	Application	Remarks
Large area activated charcoal collector	A canister containing activated charcoal is twisted into the surface and left for 24 hours.	Short term radon flux measurements	The LLD is 0.007 Bq $m^{-2}s^{-1}$ (0.2 pCi $m^{-2}s^{-1}$).
Continuous radon monitor	Air pump and scintillation cell or ionization chamber.	Track the real time concentration of radon	Takes 1 to 4 hours for system to equilibrate before starting. The LLD is 0.004-0.04 Bq/L (0.1-1.0 pCi/L).
Activated charcoal adsorption	Activated charcoal is opened to the ambient air, then gamma counted on a gamma scintillator or in a liquid scintillation counter.	Measure radon concentration in indoor air	Detector is deployed for 2 to 7 days. The LLD is 0.007-0.04 Bq/L (0.2 to 1.0 pCi/L).
Electret ion chamber	This is a charged plastic vessel that can be opened for air to pass through.	Measure short-term or long-term radon concentration in indoor air	Must correct reading for gamma background concentration. Electret is sensitive to extremes of temperature and humidity. LLD is 0.007-0.02 Bq/L (0.2-0.5 pCi/L).
Alpha track detection	A small piece of special plastic or film inside a small container. Damage tracks from alpha particles are chemically etched and tracks counted.	Measure indoor or outdoor radon concentration in air	LLD is 0.04 Bq $L^{-1}d^{-1}$ (1 pCi $L^{-1}d^{-1}$).

The following provides a general overview of radon sampling and measurement concepts. The intent of this section is to provide an overview of common methods and terminology.

6.9.1 Direct Radon Measurements

Direct radon measurements are performed by gathering radon into a chamber and measuring the ionizations produced. A variety of methods have been developed, each making use of the same fundamental mechanics but employing different measurement processes. The first step is to get the radon into a chamber without collecting any radon progeny from the ambient air. A filter is normally used to capture charged aerosols while allowing the radon gas to pass through. Most passive monitors rely on diffusion of the ambient radon in the air into the chamber to establish an equilibrium between the concentrations of radon in the air and in the chamber. Active monitors use some type of air pump system for the air exchange method.

Once inside the chamber, the radon decays by alpha emission to form ^{218}Po which usually takes on a positive charge within thousandths of a second following formation. Some monitor types collect these ionic molecules and subsequently measure the alpha particles emitted by the radon progeny. Other monitor types, such as the electret ion chamber, measure the ionization produced by the decay of radon in the air within the chamber by directly collecting the ions produced inside the chamber. Simple systems measure the cumulative radon during the exposure period based on the total alpha decays that occur. More complicated systems actually measure the individual pulse height distributions of the alpha and/or beta radiation emissions and derive the radon plus progeny isotopic concentration in the air volume.

Care must be taken to accurately calibrate a system and to understand the effects of humidity, temperature, dust loading, and atmospheric pressure on the system. These conditions create a small adverse effect on some systems and a large influence on others.

6.9.1.1 Integrating Methods for Radon Measurement

With integrating methods, measurements are made over a period of days, weeks, or months and the device is subsequently read by an appropriate device for the detector media used. The most common detectors used are activated charcoal adsorbers, electret ion chamber (EIC), and alpha track plastics. Short term fluctuations are averaged out, thus making the measurement representative of average concentration. Results in the form of an average value provide no way to determine the fluctuations of the radon concentration over the measurement interval. Successive short term measurements can be used in place of single long term measurements to gain better insight into the time dependence of the radon concentration.

6.9.1.2 Continuous Methods for Radon Measurement

Devices that measure direct radon concentrations over successive time increments are generally called continuous radon monitors. These systems are more complex than integrating devices in that they measure the radon concentration and log the results to a data recording device on a real

time basis. Continuous radon measurement devices normally allow the noble gas radon to pass through a filter into a detection chamber where the radon decays and the radon and/or the resulting progeny are measured. The most common detectors used for real time measurements are ion chambers, solid state surface barrier detectors, and ZnS(Ag) scintillation detectors.

Continuous methods offer the advantage of providing successive, short-term results over long periods of time. This allows the investigator not only to determine the average radon concentration, but also to analyze the fluctuations in the values over time. More complicated systems are available that measure the relative humidity and temperature at the measurement location and log the values along with the radon concentrations to the data logging device. This allows the investigator to make adjustments, if necessary, to the resulting data prior to reporting the results.

6.9.2 Radon Progeny Measurements

Radon progeny measurements are performed by collecting charged aerosols onto filter paper and subsequently counting the filter for attached progeny. Some systems pump air through a filter and then automatically count the filter for alpha and/or beta emissions. An equivalent but more labor intensive method is to collect a sample using an air sampling pump and then count the filter in stand alone alpha and/or beta counting systems. The measurement system may make use of any number of different techniques ranging from full alpha and beta spectrometric analysis of the filters to simply counting the filter for total alpha and or beta emissions.

When performing total (gross) counting analyses, the assumption is usually made that the only radioisotopes in the air are due to ^{222}Rn and its progeny. This uncertainty, which is usually very small, can be essentially eliminated when performing manual sampling and analysis by performing a follow up measurement of the filter after the radon progeny have decayed to a negligible level. This value can then be used as a background value for the air. Of course, such a simple approach is only applicable when ^{222}Rn is the isotope of concern. For ^{219}Rn or ^{220}Rn, other methods would have to be used.

Time is a significant element in radon progeny measurements. Given any initial equilibrium condition for the progeny isotopes, an investigator must be able to correlate the sampling and measurement technique back to the true concentration values. When collecting radon progeny, the buildup of total activity on the filter increases asymptotically until the activity on the filter becomes constant. At this point, the decay rate of the progeny atoms on the filter is equal to the collection rate of progeny atoms. This is an important parameter to consider when designing a radon sampling procedure.

Note that the number of charged aerosol particles in the air can affect the results for radon progeny measurements. If the number of particles is few, as is possible when humidity is low and a room is very clean, then most of the progeny will not be attached and can plate out on room

surfaces prior to reaching the sample filter. This is not a problem if the same conditions always exist in the room, however the calculated dose would underestimate the dose that would be received in a higher humidity or dust concentration state with the same radon progeny concentration.

6.9.3 Radon Flux Measurements

Sometimes it is desirable to characterize the source of radon in terms of the rate at which radon is emanating from a surface—that is, soil, uranium mill tailings, or concrete. One method used for measuring radon flux is briefly described here.

The measurement of radon flux can be achieved by adsorption onto charcoal using a variety of methods such as a charcoal canister or a large area collector (*e.g.*, 25 cm PVC end cap). The collector is deployed by firmly twisting the end cap into the surface of the material to be measured. After 24 hours of exposure, the activated charcoal is removed and transferred to plastic containers. The amount of radon adsorbed on the activated charcoal is determined by gamma spectroscopy. Since the area of the surface is well defined and the deployment period is known, the radon flux (in units of Bq/m^2-s or pCi/m^2-s) can be calculated.

This method is reliable for measuring radon flux in normal environmental situations. However, care should be taken if an extremely large source of radon is measured with this method. The collection time should be chosen carefully to avoid saturating the canister with radon. If saturation is approached, the charcoal loses its ability to absorb radon and the collection rate decreases. Even transporting and handling of a canister that is saturated with radon can be a problem due to the dose rate from the gamma rays being emitted. One would rarely encounter a source of radon that is so large that this would become a problem; however, it should be recognized as a potential problem. Charcoal can also become saturated with water, which will affect the absorption of radon. This can occur in areas with high humidity.

An alternative method for making passive radon flux measurements has been developed recently using electret ionization chambers (EICs). EIC technology has been widely used for indoor radon measurements. The passive EIC procedure is similar to the procedures used with large area activated charcoal canisters. In order to provide the data for the background corrections, an additional passive monitor is located side by side on a radon impermeable membrane. These data are used to calculate the net radon flux. The Florida State Bureau of Radiation Protection has compared the results from measurements of several phosphogypsum flux beds using the charcoal canisters and EICs and has shown that the two methods give comparable results. The passive method seems to have overcome some of the limitations encountered in the use of charcoal. The measurement periods can be extended from hours to several days in order to obtain a better average, if needed. EIC flux measurements are not affected by environmental conditions such as temperature, humidity, and air flow. The measured sensitivities are

comparable to the charcoal method but, unlike charcoal, EICs do not become saturated by humidity. Intermediate readings can be made if needed.. In view of the low cost of the EIC reading/analyzing equipment, the cost per measurement can be as much as 50% lower than the charcoal method with additional savings in time.

6.10 Special Equipment

Various specialized systems have been developed which can be used during the performance of radiation surveys and site investigations. These range from specially designed quick radiation scanning systems to commercial global positioning systems (GPSs). The equipment may be designed to detect radiation directly, detect and locate materials associated with the contamination (*e.g.*, metal containers), or locate the position where a particular measurement is performed (*e.g.*, GPS). Because these specialized systems are continuously being modified and developed for site-specific applications, it is not possible to provide detailed descriptions of every system. The following sections provide examples of specialized equipment that have been applied to radiation surveys and site investigations.

6.10.1 Positioning Systems

As stated in Section 4.8.5, documenting the location of measurements is important for demonstrating the reproducibility of the results. There are a variety of positioning systems available that provide a range of accuracy and precision that can be evaluated during survey planning to determine their applicability to a particular site. These positioning systems can be used to establish a reproducible reference coordinate system or to locate individual measurements using an established reference coordinate system (*e.g.*, longitude and latitude).

6.10.1.1 Differential Global Positioning Systems

A variety of practical and versatile GPSs based on radio signals tracked from satellite beacons are available (e.g., Trimble™, Novatel™, Garmin™). These systems are generally used to aid in recording and retrieving location data with precision on the order of tens of meters. With a stationary base station and a separate moving locator, the system is deployed in the "differential global positioning system" (DGPS) mode. DGPSs can record and retrieve location data with a precision in the centimeter range.

DGPS can be used to provide position information on surface features in areas being surveyed, linking the survey results to previously published maps and aerial photographs. In addition, survey results may be positioned using the DGPS readings to accurately and precisely locate the results as well as the results of any subsequent analyses to these same maps or photographs. A process called waypointing uses the DGPS to locate specific points and allows the user to find

predetermined locations and set up gridded locations for measurements based on location data that are tied into local or state coordinate systems.

Limitations on the use of DGPS are related to the number of satellite beacons available to the system. When three or fewer satellites are available the accuracy and precision of the location data will be reduced. There are short periods of time (usually less than one hour even on the worst days) when a limited number of satellites are overhead in the continental United States. Satellites may also be blocked by excess tree cover or tall buildings. Distance between the moving locator and the stationary base station may be several kilometers or may be limited to line-of-sight. This limitation can be mitigated through the strategic use of repeater stations to re-transmit the signal between the moving locator and the base station.

6.10.1.2 Local Microwave and Sonar Positioning Systems

Local microwave or sonar beacons and receivers may provide useful location data in small areas and tree-covered locales. One example of a sonar-based system is the ultrasonic ranging and data system (USRADS). With a number of fixed beacons in place, a roving unit can be oriented and provide location data with similar accuracy and precision as the DGPS. If the beacons are located at known points, the resulting positions can be determined using simple calculations based on the known reference locations of the beacons.

The logistics of deploying the necessary number of beacons properly and the short range of the signals are the major limitations of the system. In addition, multipathing of signals within wooded areas can cause jumps in the positioning data.

6.10.2 Mobile Systems with Integrated Positioning Systems

In recent years, the advent of new technologies has introduced mobile sensor systems for acquiring data that include fully-integrated positioning systems. Portable and vehicle-based versions of these systems record survey data while moving over surfaces to be surveyed and simultaneously recording the location data from either a roving DGPS receiver or local microwave/sonar receiver. All measurement data are automatically stored and processed with the measurement location for later posting (see Section 8.2.2.2 for a discussion of posting plots) or for mapping the results. These systems are designed with a variety of detectors for different applications. For example, alpha or beta detectors have been mounted on a robot a fixed distance over a smooth surface. The robot moves at a predetermined speed over the surface to provide scanning results, and also records individual direct measurements at predetermined intervals. This type of system not only provides the necessary measurement data, but also reduces the uncertainty associated with human factors. Other systems are equipped with several types of radiation detectors, magnetometers, electromagnetic sensors, or various combinations of multiple sensors. The limitations of each system should be evaluated on a site-specific basis to determine

if the positioning system, the detector, the transport system, or some combination based on site-specific characteristics will represent the limits of the system.

6.10.3 Radar, Magnetometer, and Electromagnetic Sensors

The number of sensors and sensor systems applicable to the detection and location of buried waste have increased in use and reliability in recent years. These systems are typically applicable to scoping and characterization surveys where the identification of subsurface contamination is a primary concern. However, the results of these surveys may be used during final status survey planning to demonstrate that subsurface contamination is not a concern for a particular site or survey unit. Some of the major technologies are briefly described in the following sections.

6.10.3.1 Ground Penetrating Radar

For most sites, ground penetrating radar (GPR) is the only instrument capable of collecting images of buried objects *in situ*, as compared to magnetometers (Section 6.10.3.2) and electromagnetic sensors (Section 6.10.3.3) which detect the strength of signals as measured at the ground surface. Additionally, GPR is unique in its ability to detect both metallic and non-metallic (*e.g.*, plastic, glass) containers.

Subsurface radar detection systems have been the focus of study for locating and identifying buried or submerged objects that otherwise could not be detected. There are two major categories of radar signals: 1) time domain, and 2) frequency domain. Time-domain radar uses short impulses of radar-frequency energy directed into the ground being investigated. Reflections of this energy, based on changes in dielectric properties, are then received by the radar. Frequency-domain radar, on the other hand, uses a continuous transmission where the frequency of the transmission can be varied either stepwise or continuously. The changes in the frequency characteristics due to effects from the ground are recorded. Signal processing, in both cases, converts this signal to represent the location of radar reflectors against the travel time of the return signal. Greater travel time corresponds to a greater distance beneath the surface. Table 6.11 lists the typical penetration depth for various geologic materials (fresh water is included as a baseline for comparison).

Examples of existing GPR technologies currently being applied to subsurface investigations include:

- narrow-band radar
- ultra-wideband radar
- synthetic aperture radar
- frequency modulated continuous radar
- polarized radar waves

Table 6.11 Typical Radar Penetration Depths for Various Geologic Materials

Material	Penetration Depth m (ft)
Fresh Water	100 (330)
Sand (desert)	5 (16)
Sandy Soil	3 (10)
Loam Soil	3 (10)
Clay Soil	2 (6)
Salt Flats (dry)	1 (3)
Coal	20 (66)
Rocks	20 (66)
Walls	0.3 (1)

The major limitation to GPR is the difficulty in interpreting the data, which is often provided in the form of hazy, "waterfall-patterned" data images requiring an experienced professional to interpret. Also, GPR can vary depending on the soil type as shown in Table 6.10. Highly conductive clay soils often absorb a large amount of the radar energy, and may even reflect the energy. GPR can be deployed using ground-based or airborne systems.

6.10.3.2 Magnetometers

Although contaminated soil and most radioactive waste possess no ferromagnetic properties, the containers commonly used to hold radioactive waste (*e.g.*, 55-gallon drums) are made from steel. These containers possess significant magnetic susceptibility making the containers detectable using magnetometry.

Magnetometers sense the pervasive magnetic field of the Earth. This field, when encountering an object with magnetic susceptibility, induces a secondary magnetic field in that object. This secondary field creates an increase or decrease in Earth's ambient magnetic field. Magnetometers measure these changes in the expected strength of the ambient magnetic field. Some magnetometers, called "vector magnetometers," can sense the direction as well as the magnitude of these changes. However, for subsurface investigations only the magnitude of the changes are used.

The ambient magnetic field on Earth averages 55,000 gamma in strength. The variations caused by the secondary magnetic fields typically range from 10 to 1,000 gamma, and average around 100 gamma. Most magnetometers currently in use have a sensitivity in the 0.1 to 0.01 gamma range and are capable of detecting these secondary fields.

An alternate magnetometer survey can be performed using two magnetometers in a gradiometric configuration. This means that the first magnetometer is placed at the ground surface, while the second is mounted approximately 0.5 meters above the first. Data is recorded from both sensors and compared. When the readings from both detectors are nearly the same, it implies that there is no significant disturbance in the Earth's ambient magnetic field or that such disturbances are broad and far away from the gradiometer. When a secondary magnetic field is induced in an object, it affects one sensor more strongly than the other, producing a difference in the readings from the two magnetometers. This approach is similar to the use of a guard detector in anti-coincidence mode in a low-background gas-flow proportional counter in a laboratory (see Appendix H for a description of gas-flow proportional counters). The gradiometric configuration filters out the Earth's ambient magnetic field, large scale variations, and objects located far from the sensor to measure the effects of nearby objects, all without additional data processing.

Fifty-five gallon drums buried 5 to 7 meters below the surface may be detectable using a magnetometer. At many sites, multiple drums have been buried in trenches or pits and detection is straightforward. A single operator carrying a magnetometer with the necessary electronics in a backpack can cover large areas in a relatively small amount of time.

The limitations on the system are related to the size of the objects and their depth below the surface. Objects that are too small or buried too deep will not provide a secondary magnetic field that can be detected at the ground surface.

6.10.3.3 Electromagnetic Sensors

Electromagnetic sensors emit an electromagnetic wave, in either a pulsed or continuous wave mode, and then receive the result of that transmission. The result of the transmission is two signals; quadrature and in-phase. As the wave passes through some material other than air, it is slowed down by a resistive medium or sped up by a conductor through dielectric effects. This produces the quadrature signal. If the electromagnetic wave encounters a highly conductive object it induces a magnetic field in the object. This induced electromagnetic field returns to the sensor as a reflection of the original electromagnetic wave and forms the in-phase signal.

The in-phase signal is indicative of the presence, size, and conductivity of nearby objects (*e.g.*, 55-gallon drums), while the quadrature signal is a measure of the dielectric properties of the nearby objects such as soil. This means that electromagnetic sensors can detect all metallic objects (including steel, brass, and aluminum), such as the metal in waste containers, and also sample the soil for changes in properties, such as those caused by leaks of contaminants.

Depths of interest are largely determined by the spacing between the coil used to transmit the primary electromagnetic wave, and the receiver used to receive that transmission. The rule of thumb is that the depth of interest is on the order of the distance between the transmitter and the receiver. A system designed with the transmitter and receiver placed tens of meters apart can detect signals from tens of meters below the surface. A system with the transmitter and receiver collocated can only detect signals from depths on the order of the size of the coil, which is typically about one meter. The limitations of electromagnetic sensors include a lack of clearly defined signals, and decreasing resolution of the signal as the distance below the surface increases.

6.10.4 Aerial Radiological Surveys

Low–altitude aerial radiological surveys are designed to encompass large areas and may be useful in:

- providing data to assist in the identification of radioactive contaminants and their corresponding concentrations and spatial distributions
- characterizing the nature, extent, and impact of contamination

The measurement sensitivity and data processing procedures provide total area coverage and a detailed definition of the extent of gamma-producing isotopes for a specific area. The gamma radiation spectral data are processed to provide a qualitative and quantitative analysis of the radionuclides in the survey area. Helicopter flights establish a grid pattern (*e.g.*, east–west) of parallel lines approximately 61 m (200 ft) above the ground surface.

The survey consists of airborne measurements of natural and man–made gamma radiation from the terrain surface. These measurements allow for the determination of terrestrial spatial distribution of isotopic concentrations and equivalent gamma exposure rates (*e.g.*, ^{60}Co, ^{234m}Pa, and ^{137}Cs). The results are reported as isopleths for the isotopes and are usually superimposed on scale maps of the area.