After determining exposure points, identify probable exposure routes (i.e., ingestion, inhalation, dermal contact) based on the media contaminated and the anticipated activities at the exposure points. In some instances, an exposure point may exist but an exposure route may not (e.g., a person touches contaminated soil but is wearing gloves). Exhibit 6-7 presents a population/exposure route matrix that can be used in determining potential exposure routes at a site.

6.3.4 INTEGRATE INFORMATION ON SOURCES, RELEASES, FATE AND TRANSPORT, EXPOSURE POINTS, AND EXPOSURE ROUTES INTO EXPOSURE PATHWAYS

Assemble the information developed in the previous three steps and determine the complete exposure pathways that exist for the site. A pathway is complete if there is (1) a source or chemical release from a source, (2) an exposure point where contact can occur, and (3) an exposure route by which contact can occur. Otherwise, the pathway is incomplete, such as the situation where there is a source releasing to air but there are no nearby people. If available from ATSDR, human monitoring data indicating chemical accumulation or chemical-related effects in the site area can be used as evidence to support conclusions about which exposure pathways are complete; however, negative data from such studies should not be used to conclude that a pathway is incomplete.

From all complete exposure pathways at a site, select those pathways that will be evaluated further in the exposure assessment. If exposure to a sensitive subpopulation is possible, select that pathway for quantitative evaluation. All pathways should be selected for further evaluation unless there is sound justification (e.g., based on the results of a screening analysis) to eliminate a pathway from detailed analysis. Such a justification could be based on one of the following:

- the exposure resulting from the pathway is much less than that from another pathway involving the same medium at the same exposure point;
- the potential magnitude of exposure from a pathway is low; or
- the probability of the exposure occurring is very low and the risks associated with the occurrence are not high (if a pathway has catastrophic consequences, it should be selected for evaluation even if its probability of occurrence is very low).

Use professional judgment and experience to make these decisions. Before deciding to exclude a pathway from quantitative analysis, consult with the RPM. If a pathway is excluded from further analysis, clearly document the reasons for the decision in the exposure assessment section of the risk assessment report.

For some complete pathways it may not be possible to quantify exposures in the subsequent steps of the analysis because of a lack of data on which to base estimates of chemical release, environmental concentration, or human intake. Available modeling results should complement and supplement the available monitoring data to minimize such problems. However, uncertainties associated with the modeling results may be too large to justify quantitative exposure assessment in the absence of monitoring data to validate the modeling results. These pathways should nevertheless be carried through the exposure assessment so that risks can be qualitatively evaluated or so that this information can be considered during the uncertainty analysis of the results of the exposure assessment (see Section 6.8) and the risk assessment (see Chapter 8).

6.3.5 SUMMARIZE INFORMATION ON ALL COMPLETE EXPOSURE PATHWAYS

Summarize pertinent information on all complete exposure pathways at the site by identifying potentially exposed populations, exposure media, exposure points, and exposure routes. Also note if the pathway has been selected for quantitative evaluation; summarize the justification if a pathway has been excluded. Summarize pathways for current land use and any alternate future land use separately. This summary information is useful for defining the scope of the next step (quantification of exposure) and also is useful as documentation of the exposure pathway analysis. Exhibit 6-8 provides a sample format for presenting this information.

6.4 STEP 3: QUANTIFICATION OF EXPOSURE: GENERAL CONSIDERATIONS

The next step in the exposure assessment process is to quantify the magnitude, frequency and duration of exposure for the populations and exposure pathways selected for quantitative evaluation. This step is most often conducted in two stages: first, exposure concentrations are estimated, then, pathway-specific intakes are quantified. The specific methodology for calculating exposure concentrations and pathway-specific exposures are presented in Sections 6.5 and 6.6, respectively. This section describes some of the basic concepts behind these processes.

6.4.1 QUANTIFYING THE REASONABLE MAXIMUM EXPOSURE

Exposure is defined as the contact of an organism with a chemical or physical agent. If exposure occurs over time, the total exposure can be divided by a time period of interest to obtain an average exposure rate per unit time. This average exposure rate also can be expressed as a function of body weight. For the purposes of this manual, exposure normalized for time and body weight is termed "intake", and is expressed in units of mg chemical/kg body weight-day.

Exhibit 6-9 presents a generic equation for calculating chemical intakes and defines the intake variables. There are three categories of variables that are used to estimate intake:

(1) chemical-related variable -- exposure concentration;

(2) variables that describe the exposed population -- contact rate, exposure frequency and duration, and body weight; and

(3) assessment-determined variable -- averaging time.

Each intake variable in the equation has a range of values. For Superfund exposure assessments, intake variable values for a given pathway should be selected so that the combination of all intake variables results in an estimate of the reasonable maximum exposure for that pathway. As defined previously, the reasonable maximum exposure (RME) is the maximum exposure that is reasonably expected to occur at a site. Under this approach, some intake variables may not be at their individual maximum values but when in combination with other variables will result in estimates of the RME. Some recommendations for determining the values of the individual intake variables are discussed below. These recommendations are based on EPA's determination of what would result in an estimate of the RME. As discussed previously, a determination of "reasonable" cannot be based solely on quantitative information, but also requires the use of professional judgment. Accordingly, the recommendations below are based on a combination of quantitative information and professional judgment. These are general recommendations, however, and could change based on site-specific information or the particular needs of the risk manager. Consult with the RPM before varying from these recommendations.

Exposure concentration. The concentration term in the intake equation is the arithmetic average of the concentration that is contacted over the exposure period. Although this concentration does not reflect the maximum concentration that could be contacted at any one time, it is regarded as a reasonable estimate of the concentration likely to be contacted over time. This is because in most situations, assuming long-term contact with the maximum concentration is not reasonable. (For exceptions to this generalization, see discussion of hot spots in Section 6.5.3.)

Because of the uncertainty associated with any estimate of exposure concentration, the upper confidence limit (i.e., the 95 percent upper confidence limit) on the arithmetic average will be used for this variable. There are standard statistical methods which can be used to calculate the upper confidence limit on the arithmetic mean. Gilbert (1987, particularly sections 11.6 and 13.2) discusses methods that can be applied to data that are distributed normally or log normally. Kriging is another method that potentially can be used (Clark 1979 is one of several reference books on kriging). A statistician should be consulted for more details or for assistance with specific methods.

If there is great variability in measured or modeled concentration values (such as when too few samples are taken or when model inputs are uncertain), the upper confidence limit on the average concentration will be high, and conceivably could be above the maximum detected or modeled value. In these cases, the maximum detected or modeled value should be used to estimate exposure concentrations. This could be regarded by some as too conservative an estimate, but given the uncertainty in the data in these situations, this approach is regarded as reasonable.

For some sites, where a screening level analysis is regarded as sufficient to characterize potential exposures, calculation of the upper confidence limit on the arithmetic average is not required. In these cases, the maximum detected or modeled concentration should be used as the exposure concentration.

Contact rate. Contact rate reflects the amount of contaminated medium contacted per unit time or event. If statistical data are available for a contact rate, use the 95th percentile value for this variable. (In this case and throughout this chapter, the 90th percentile value can be used if the 95th percentile value is not available.) If statistical data are not available, professional judgment should be used to estimate a value which approximates the 95th percentile value. (It is recognized that such estimates will not be precise. They should, however, reflect a reasonable estimate of an upper-bound value.)

Sometimes several separate terms are used to derive an estimate of contact rate. For example, for dermal contact with chemicals in water, contact rate is estimated by combining information on exposed skin surface area, dermal permeability of a chemical, and exposure time. In such instances, the combination of variables used to estimate intake should result in an estimate approximating the 95th percentile value. Professional judgment will be needed to determine the appropriate combinations of variables. (More specific guidance for determining contact rate for various pathways is given in Section 6.6.)

Exposure frequency and duration. Exposure frequency and duration are used to estimate the total time of exposure. These terms are determined on a site-specific basis. If statistical data are available, use the 95th percentile value for exposure time. In the absence of statistical data (which is usually the case), use reasonable conservative estimates of exposure time. National statistics are available on the upper-bound (90th percentile) and average (50th percentile) number of years spent by individuals at one residence (EPA 1989d). Because of the data on which they are based, these values may underestimate the actual time that someone might live in one residence. Nevertheless, the upper-bound value of 30 years can be used for exposure duration when calculating reasonable maximum residential exposures. In some cases, however, lifetime exposure (70 years by convention) may be a more appropriate assumption. Consult with the RPM regarding the appropriate exposure duration for residential exposures. The exposure frequency and duration selected must be appropriate for the contact rate selected. If a long-term average contact rate (e.g., daily fish ingestion rate averaged over a year) is used, then a daily exposure frequency (i.e., 365 days/year) should be assumed.

Body weight. The value for body weight is the average body weight over the exposure period. If exposure occurs only during childhood years, the average child body weight during the exposure period should be used to estimate intake. For some pathways, such as soil ingestion, exposure can occur throughout the lifetime but the majority of exposure occurs during childhood (because of higher contact rates). In these cases, exposures should be calculated separately for age groups with similar contact rate to body weight ratios; the body weight used in the intake calculation for each age group is the average body weight for that age group. Lifetime exposure is then calculated by taking the time-weighted average of exposure estimates over all age groups. For pathways where contact rate to body weight ratios are fairly constant over a lifetime (e.g., drinking water ingestion), a body weight of 70 kg is used.

A constant body weight over the period of exposure is used primarily by convention, but also because body weight is not always independent of the other variables in the exposure equation (most notably, intake). By keeping body weight constant, error from this dependence is minimized. The average body weight is used because, when combined with the other variable values in the intake equation, it is believed to result in the best estimate of the RME. For example, combining a 95th percentile contact rate with a 5th percentile body weight is not considered reasonable because it is unlikely that smallest person would have the highest intake. Alternatively, combining a 95th percentile intake with a 95th percentile body weight is not considered a maximum because a smaller person could have a higher contact rate to body weight ratio.

Averaging time. The averaging time selected depends on the type of toxic effect being assessed. When evaluating exposures to developmental toxicants, intakes are calculated by averaging over the exposure event (e.g., a day or a single exposure incident). For acute toxicants, intakes are calculated by averaging over the shortest exposure period that could produce an effect, usually an exposure event or a day. When evaluating longer-term exposure to noncarcinogenic toxicants, intakes are calculated by averaging intakes over the period of exposure (i.e., subchronic or chronic daily intakes). For carcinogens, intakes are calculated by prorating the total cumulative dose over a lifetime (i.e., chronic daily intakes, also called lifetime average daily intake). This distinction relates to the currently held scientific opinion that the mechanism of action for each category is different (see Chapter 7 for a discussion). The approach for carcinogens is based on the assumption that a high dose received over a short period of time is equivalent to a corresponding low dose spread over a lifetime (EPA 1986b). This approach becomes problematic as the exposures in question become more intense but less frequent, especially when there is evidence that the agent has shown dose-rate related carcinogenic effects. In some cases, therefore, it may be necessary to consult a toxicologist to assess the level of uncertainty associated with the exposure assessment for carcinogens. The discussion of uncertainty should be included in both the exposure assessment and risk characterization chapters of the risk assessment report.

6.4.2 TIMING CONSIDERATIONS

At many Superfund sites, long-term exposure to relatively low chemical concentrations (i.e., chronic daily intakes) are of greatest concern. In some situations, however, shorter-term exposures (e.g., subchronic daily intakes) also may be important. When deciding whether to evaluate short-term exposure, the following factors should be considered:

- the toxicological characteristics of the chemicals of potential concern;
- the occurrence of high chemical concentrations or the potential for a large release;
- persistence of the chemical in the environment; and
- the characteristics of the population that influence the duration of exposure.

Toxicity considerations. Some chemicals can produce an effect after a single or very short-term exposure to relatively low concentrations. These chemicals include acute toxicants such as skin irritants and neurological poisons, and developmental toxicants. At sites where these types of chemicals are present, it is important to assess exposure for the shortest time period that could result in an effect. For acute toxicants this is usually a single exposure event or a day, although multiple exposures over several days also could result in an effect. For developmental toxicants, the time period of concern is the exposure event. This is based on the assumption that a single exposure at the critical time in development is sufficient to produce an adverse effect. It should be noted that the critical time referred to can occur in almost any segment of the human population (i.e., fertile men and women, the conceptus, and the child up to the age of sexual maturation [EPA 1989e]).

Concentration considerations. Many chemicals can produce an effect after a single or very short-term exposure, but only if exposure is to a relatively high concentration. Therefore, it is important that the assessor identify possible situations where a short-term exposure to a high concentration could occur. Examples of such a situation include sites where contact with a small, but highly contaminated area is possible (e.g., a source or a hot spot), or sites where there is a potential for a large chemical release (e.g., explosions, ruptured drums, breached lagoon dikes). Exposure should be determined

for the shortest period of time that could produce an effect.

Persistence considerations. Some chemicals may degrade rapidly in the environment. In these cases, exposures should be assessed only for that period of time in which the chemical will be present at the site. Exposure assessments in these situations may need to include evaluations of exposure to the breakdown products, if they are persistent or toxic at the levels predicted to occur at the site.

Population considerations. At some sites, population activities are such that exposure would occur only for a short time period (a few weeks or months), infrequently, or intermittently. Examples of this would be seasonal exposures such as during vacations or other recreational activities. The period of time over which exposures are averaged in these instances depends on the type of toxic effect being assessed (see previous discussion on averaging time, Section 6.4.1).

6.5 QUANTIFICATION OF EXPOSURE: DETERMINATION OF EXPOSURE CONCENTRATIONS

This section describes the basic approaches and methodology for determining exposure concentrations of the chemicals of potential concern in different environmental media using available monitoring data and appropriate models. As discussed in Section 6.4.1, the concentration term in the exposure equation is the average concentration contacted at the exposure point or points over the exposure period. When estimating exposure concentrations, the objective is to provide a conservative estimate of this average concentration (e.g., the 95 percent upper confidence limit on the arithmetic mean chemical concentration).

This section provides an overview of the basic concepts and approaches for estimating exposure concentrations. It identifies what type of information is needed to estimate concentrations, where to find it, and how to interpret and use it. This section is not designed to provide all the information necessary to derive exposure concentrations and, therefore, does not detail the specifics of potentially applicable models nor provide the data necessary to run the models or support concentration estimates. However, sources of such information, including the *Superfund Exposure Assessment Manual* (SEAM; EPA 1988b) are referenced throughout the discussion.

6.5.1 GENERAL CONSIDERATIONS FOR ESTIMATING EXPOSURE CONCENTRATIONS

In general, a great deal of professional judgment is required to estimate exposure concentrations. Exposure concentrations may be estimated by (1) using monitoring data alone, or (2) using a combination of monitoring data and environmental fate and transport models. In most exposure assessments, some combination of monitoring data and environmental modeling will be required to estimate exposure concentrations.

Direct use of monitoring data . Use of monitoring data to estimate exposure concentrations is normally applicable where exposure involves direct contact with the monitored medium (e.g., direct contact with chemicals in soil or sediment), or in cases where monitoring has occurred directly at an exposure point (e.g., a residential drinking water well or public water supply). For these exposure pathways, monitoring data generally provide the best estimate of <u>current</u> exposure concentrations.

As the first step in estimating exposure concentrations, summarize available monitoring data. The manner in which the data are summarized depends upon the site characteristics and the pathways being evaluated. It may be necessary to divide chemical data from a particular medium into subgroups based on the location of sample points and the potential exposure pathways. In other instances, as when the sampling point is an exposure point (e.g., when the sample is from an existing drinking water well) it may not be appropriate to group samples at all, but may be most appropriate to treat the sample data separately when estimating intakes. Still,

in other instances, the assessor may wish to use the maximum concentration from a medium as the exposure concentration for a given pathway as a screening approach to place an upper bound on exposure. In these cases it is important to remember that if a screening level approach suggests a potential health concern, the estimates of exposure should be modified to reflect more probable exposure conditions.

In those instances where it is appropriate to group sampling data from a particular medium, calculate for each exposure medium and each chemical the 95 percent upper confidence limit on the arithmetic average chemical concentration. See Chapter 5 for guidance on how to treat sample concentrations below the quantitation limit.

Modeling approaches . In some instances, it may not be appropriate to use monitoring data alone, and fate and transport models may be required to estimate exposure concentrations. Specific instances where monitoring data alone may not be adequate are as follows.

- Where exposure points are spatially separate from monitoring points. Models may be required when exposure points are remote from sources of contamination if mechanisms for release and transport to exposure points exist (e.g., ground-water transport, air dispersion).

- Where temporal distribution of data is lacking. Typically, data from Superfund investigations are collected over a relatively short period of time. This generally will give a clear indication of current site conditions, but both long-term and short-term exposure estimates usually are required in Superfund exposure assessments. Although there may be situations where it is reasonable to assume that concentrations will remain constant over a long period of time, in many cases the time span of the monitoring data is not adequate to predict future exposure concentrations. Environmental models may be required to make these predictions.

- Where monitoring data are restricted by the limit of quantitation. Environmental models may be needed to predict concentrations of contaminants that may be present at concentrations that are below the quantitation limit but that may still cause toxic effects (even at such low concentrations). For example, in the case of a ground-water plume discharging into a river, the dilution afforded by the river may be sufficient to reduce the concentration of the chemical to a level that could not be detected by direct monitoring. However, as discussed in Section 5.3.1, the chemical may be sufficiently toxic or bioaccumulative that it could present a health risk at concentrations below the limit of quantitation. Models may be required to make exposure estimates in these types of situations.

A wide variety of models are available for use in exposure assessments. SEAM (EPA 1988b) and the *Exposure Assessment Methods Handbook* (EPA 1989f) describe some of the models available and provide guidance in selecting appropriate modeling techniques. Also, the Center for Exposure Assessment Modeling (CEAM -- Environmental Research Laboratory (ERL) Athens), the Source Receptor Analysis Branch (Office of Air Quality Planning and Standards, or OAQPS), and modelers in EPA regional offices can provide assistance in selecting appropriate models. Finally, Volume IV of the NTGS (EPA 1989c) provides guidance for air and atmospheric dispersion modeling for Superfund sites. Be sure to discuss the fate and transport models to be used in the exposure assessment with the RPM.

The level of effort to be expended in estimating exposure concentrations will depend on the type and quantity of data available, the level of detail required in the assessment, and the resources available for the assessment. In general, estimating exposure concentrations will involve analysis of site monitoring data and application of simple, screening-level analytical models. The most important factor in determining the level of effort will be the quantity and quality of the available data. In general, larger data sets will support the use of more sophisticated models.

Other considerations . When evaluating chemical contamination at a site, it is important to review the spatial distribution of the data and evaluate it in ways that have the most relevance to the pathway being assessed. In short, consider where the contamination is with respect to known or anticipated population activity patterns. Maps of both concentration distribution and activity patterns will be useful for the exposure assessment. It is the intersection of activity patterns and contamination that defines an exposure area. Data from random sampling or from systematic grid pattern sampling may be more representative of a given exposure pathway than data collected only from hot spots.

Generally, verified GC/MS laboratory data with adequate quality control will be required to support quantitative exposure assessment. Field screening data generally cannot be incorporated when estimating exposure concentrations because they are derived using less sensitive analytical methods and are subject to less stringent quality control.

Other areas to be considered in estimating exposure concentrations are as follows.

- Steady-state vs. non-steady-state conditions. Frequently, it may be necessary to assume steady-state conditions because the information required to estimate non-steady-state conditions (such as source depletion rate) is not readily available. This is likely to overestimate long-term exposure concentrations for certain pathways.

- Number and type of exposure parameters that must be assumed. In developing exposure models, values for site-specific parameters such as hydraulic conductivity, organic carbon content of soil, wind speed and direction, and soil type may be required. These values may be generated as part of the RI. In cases where these values are not available, literature values may be substituted. In the absence of applicable literature values, the assessor must consider if a reliable exposure concentration estimate can be made.

- Number and type of fate processes to be considered. In some cases, exposure modeling may be limited to considerations of mass balance, dilution, dispersion, and equilibrium partitioning. In other cases, models of more complex fate processes, such as chemical reaction, biodegradation, and photolysis may be needed. However, prediction of such fate processes requires significantly larger quantities of model calibration and validation data than required for less complex fate processes. For those sites where these more complex fate processes need to be modeled, be sure to consult with the RPM regarding the added data requirements.

6.5.2 ESTIMATE EXPOSURE CONCENTRATIONS IN GROUND WATER

Exposure concentrations in ground water can be based on monitoring data alone or on a combination of monitoring and modeling. In some cases, the exposure assessor may favor the use of monitoring data over the use of complex models to develop exposure concentrations. It is most appropriate to use ground-water sampling data as estimates of exposure concentrations when the sampling points correspond to exposure points, such as samples taken from a drinking water tap. However, samples taken directly from a domestic well or drinking water tap should be interpreted cautiously. For example, where the water is acidic, inorganic chemicals such as lead or copper may leach from the distribution system. Organic chemicals such as phthalates may migrate into water from plastic piping. Therefore, interpretations of these data should consider the type and operation of the pumping, storage, and distribution system involved.

Most of the time, data from monitoring wells will be used to estimate chemical concentrations at the exposure point. Several issues should be considered when using monitoring well data to estimate these concentrations. First, determine if the aquifer has sufficient production capacity and is of sufficient quality to support drinking water or other uses. If so, it generally should be assumed that water could be drawn from anywhere in the aquifer, regardless of the location of existing wells relative to the

contaminant plume. In a few situations, however, it may not be reasonable to assume that water will be drawn from directly beneath a specific source (e.g., a waste management unit such as a landfill) in the future. In these cases, it should be assumed that water could be drawn from directly adjacent to the source. Selection of the location(s) used to evaluate future ground-water exposures should be made in consultation with the RPM. Second, compare the construction of wells (e.g., drinking water wells) in the area with the construction of the monitoring wells. For example, drinking water wells may draw water from more than one aquifer, whereas individual monitoring wells are usually screened in a specific aquifer. In some cases it may be appropriate to separate data from two aquifers that have very limited hydraulic connection if drinking water wells in the area draw water from only one of them. Consult a hydrogeologist for assistance in the above considerations.

Another issue to consider is filtration of water samples. While filtration of ground-water samples provides useful information for understanding chemical transport within an aquifer (see Section 4.5.3 for more details), the use of filtered samples for estimating exposure is very controversial because these data may underestimate chemical concentrations in water from an unfiltered tap. Therefore, data from unfiltered samples should be used to estimate exposure concentrations. Consult with the RPM before using data from filtered samples.

Ground-water monitoring data are often of limited use for evaluating long-term exposure concentrations because they are generally representative of current site conditions and not long-term trends. Therefore, ground-water models may be needed to estimate exposure concentrations. Monitoring data should be used when possible to calibrate the models.

Estimating exposure concentrations in ground water using models can be a complex task because of the many physical and chemical processes that may affect transport and transformation in ground water. Among the important mechanisms that should be considered when estimating exposure concentrations in ground water are leaching from the surface, advection (including infiltration, flow through the unsaturated zone, and flow with ground water), dispersion, sorption (including adsorption, desorption, and ion exchange), and transformation (including biological degradation, hydrolysis, oxidation, reduction, complexation, dissolution, and precipitation). Another consideration is that not all chemicals may be dissolved in water, but may be present instead in nonaqueous phases that float on top of ground water or sink to the bottom of the aquifer.

The proper selection and application of soil and ground-water models requires a thorough understanding of the physical, chemical, and hydrogeologic characteristics of the site. SEAM (EPA 1988b) provides a discussion of the factors controlling soil and ground-water contaminant migration as well as descriptions of various soil and ground-water models. For more in-depth guidance on the selection and application of appropriate ground-water models, consult *Selection Criteria for Mathematical Models Used in Exposure Assessments: Ground-water Models* (EPA 1988c). As with all modeling, the assessor should carefully evaluate the applicability of the model to the site being evaluated, and should consult with a hydrogeologist as necessary.

If ground-water modeling is not used, current concentrations can be used to represent future concentrations in ground water assuming steady-state conditions. This assumption should be noted in the exposure assessment chapter and in the uncertainties and conclusions of the risk assessment.

6.5.3 ESTIMATE EXPOSURE CONCENTRATIONS IN SOIL

Estimates of current exposure concentrations in soil can be based directly on summarized monitoring data if it is assumed that concentrations remain constant over time. Such an assumption may not be appropriate for some chemicals and some sites where leaching, volatilization, photolysis, biodegradation, wind erosion, and surface runoff will reduce chemical concentrations over time. Soil monitoring data and site conditions should be carefully screened to identify situations where source depletion is likely to be important. SEAM (EPA 1988b) gives steady-state equations for estimating many of these processes. However, incorporating these processes into the calculation of exposure concentrations for soil involves considerable effort. If a modeling approach is not adopted in these situations, assume a constant concentration over time and base exposure concentrations on monitoring data. This assumption should be clearly documented.

In evaluating monitoring data for the assessment of soil contact exposures, the spatial distribution of the data is a critical factor. The spatial distribution of soil contamination can be used as a basis for estimating the average concentrations contacted over time if it is assumed that contact with soil is spatially random (i.e., if

contact with soil in all areas of the site is equally probable). Data from random sampling programs or samples from evenly spaced grid networks generally can be considered as representative of concentrations across the site. At many sites however, sampling programs are designed to characterize only obviously contaminated soils or hot spot areas. Care must be taken in evaluating such data sets for estimating exposure concentrations. Samples from areas where direct contact is not realistic (such as where a steep slope or thick vegetation prevents current access) should not be considered when estimating current exposure concentrations for direct contact pathways. Similarly, the depth of the sample should be considered; surface soil samples should be evaluated separately from subsurface samples if direct contact with surface soil or inhalation of wind blown dust are potential exposure pathways at the site.

In some cases, contamination may be unevenly distributed across a site, resulting in hot spots (areas of high contamination relative to other areas of the site). If a hot spot is located near an area which, because of site or population characteristics, is visited or used more frequently, exposure to the hot spot should be assessed separately. The area over which the activity is expected to occur should be considered when averaging the monitoring data for a hot spot. For example, averaging soil data over an area the size of a residential backyard (e.g., an eighth of an acre) may be most appropriate for evaluating residential soil pathways.

6.5.4 ESTIMATE EXPOSURE CONCENTRATIONS IN AIR

There are three general approaches to estimating exposure concentrations in air: (1) ambient air monitoring, (2) emission measurements coupled with dispersion modeling, and (3) emission modeling coupled with dispersion modeling. Whichever approach is used, the resulting exposure concentrations should be as representative as possible of the specific exposure pathways being evaluated. If long-term exposures are being evaluated, the exposure concentrations should be representative of long-term averages. If short-term exposures are of interest, measured or modeled peak concentrations may be most representative.

If monitoring data have been collected at a site, their adequacy for use in a risk assessment should be evaluated by considering how appropriate they are for the exposures being addressed. Volume II of the NTGS (EPA 1989b) provides guidance for measuring emissions and should be consulted when evaluating the appropriateness of emission data. See Chapter 4 (Section 4.5.5) for factors to consider when evaluating the appropriateness of ambient air monitoring data. As long as there are no significant analytical problems affecting air sampling data, background levels are not significantly higher than potential site-related levels, and site-related levels are not below the instrument detection limit, air monitoring data can be used to derive exposure concentrations. There still will be uncertainties inherent in using these data because they usually are not representative of actual long-term average air concentrations. This may be because there were only a few sample collection periods, samples were collected during only one type of meteorological or climatic condition, or because the source of the chemicals will change over time. These uncertainties should be mentioned in the risk assessment.

In the absence of monitoring data, exposure concentrations often can be estimated using models. Two kinds of models are used to estimate air concentrations: emission models that predict the rate at which chemicals may be released into the air from a source, and dispersion models that predict associated concentrations in air at potential receptor points.

Outdoor air modeling. Emissions may occur as a result of the volatilization of chemicals from contaminated media or as a result of the suspension of onsite soils. Models that predict emission rates for volatile chemicals or dust require numerous input parameters, many of which are site-specific. For volatile chemicals, emission models for surface water and soil are available in SEAM (EPA 1988b). Volume IV of the NTGS (EPA 1989c) also provides guidance for evaluating volatile emissions at Superfund sites. Emissions due to suspension of soils may result from wind erosion of exposed soil particles and from vehicular disturbances of the soil. To predict soil or dust emissions, EPA's fugitive dust models provided in AP42 (EPA 1985b) or models described in SEAM (1988b) may be used. Volume IV of the NTGS (EPA 1989c) also will be useful in evaluating fugitive dust emissions at Superfund sites. Be sure to critically review all models before use to determine their applicability to the situation and site being evaluated. If necessary, consult with air modelers in EPA regional offices, the Exposure

Assessment Group in EPA headquarters or the Source Receptor Analysis Branch in OAQPS.

After emissions have been estimated or measured, air dispersion models can be applied to estimate air concentrations at receptor points. In choosing a dispersion model, factors that must be considered include the type of source and the location of the receptor relative to the source. For area or point sources, EPA's Industrial Source Complex model (EPA 1987a) or the simple Gaussian dispersion models discussed in SEAM (EPA 1988b) can provide air concentrations around the source. Other models can be found in Volume IV of the NTGS (EPA 1989c). The Source Receptor Analysis Branch of OAQPS also can be contacted for assistance. Again, critically review all models for their applicability.

Indoor air modeling. Indoor emissions may occur as a result of transport of outdoor-generated dust or vapors indoors, or as a result of volatilization of chemicals indoors during use of contaminated water (e.g., during showering, cooking, washing). Few models are available for estimating indoor air concentrations from outside sources. For dust transport indoors, it can generally be assumed that indoor concentrations are less than those outdoors. For vapor transport indoors, concentrations indoors and outdoors can be assumed to be equivalent in most cases. However, at sites where subsurface soil gas or ground-water seepage are entering indoors, vapor concentrations inside could exceed those outdoors. Vapor concentrations resulting from indoor use of water may be greater than those outdoors, depending on the emission source characteristics, dispersion indoors, and indoor-outdoor air exchange rates. Use models discussed in the *Exposure Assessment Methods Handbook* (EPA 1989f) to evaluate volatilization of chemicals from indoor use of water.

6.5.5 ESTIMATE EXPOSURE CONCENTRATIONS IN SURFACE WATER

Data from surface water sampling and analysis may be used alone or in conjunction with fate and transport models to estimate exposure concentrations. Where the sampling points correspond to exposure points, such as at locations where fishing or recreational activities take place, or at the intake to a drinking water supply, the monitoring data can be used alone to estimate exposure concentrations. However, the data must be carefully screened. The complexity of surface water processes may lead to certain limitations in monitoring data. Among these are the following.

- **Temporal representativeness.** Surface water bodies are subject to seasonal changes in flow, temperature, and depth that may significantly affect the fate and transport of contaminants. Releases to surface water bodies often depend on storm conditions to produce surface runoff and soil erosion. Lakes are subject to seasonal stratification and changes in biological activity. Unless the surface water monitoring program has been designed to account for these phenomena, the data may not represent long-term average concentrations or short-term concentrations that may occur after storm events.

- **Spatial representativeness.** Considerable variation in concentration can occur with respect to depth and lateral location in surface water bodies. Sample locations should be examined relative to surface water mixing zones. Concentrations within the mixing zone may be significantly higher than at downstream points where complete mixing has taken place.

- **Quantitation limit limitations.** Where large surface water bodies are involved, contaminants that enter as a result of ground-water discharge or runoff from relatively small areas may be significantly diluted. Although standard analytical methods may not be able to detect chemicals at these levels, the toxic effects of the chemicals and/or their potential to bioaccumulate may nevertheless require that such concentrations be assessed.

- **Contributions from other sources.** Surface water bodies are normally subject to contamination from many sources (e.g., pesticide runoff, stormwater, wastewater discharges, acid mine drainage). Many of the chemicals associated with these sources may be difficult to distinguish from site-related chemicals. In many cases background samples will be useful in assessing site-related contaminants from other contaminants (see Section 4.4). However, there may be other cases where a release and transport model may be required to make the distinction.

Many analytical and numerical models are available to estimate the release of contaminants to surface water

and to predict the fate of contaminants once released. The models range from simple mass balance relationships to numerical codes that contain terms for chemical and biological reactions and interactions with sediments. In general, the level of information collected during the RI will tend to limit the use of the more complex models.

There are several documents that can be consulted when selecting models to estimate surface water exposure concentrations, including SEAM (EPA 1988b), the *Exposure Assessment Methods Handbook* (EPA 1989f), and *Selection Criteria for Mathematical Models Used in Exposure Assessments: Surface Water Models* (EPA 1987b). SEAM lists equations for surface water runoff and soil erosion and presents the basic mass balance relationships for estimating the effects of dilution. A list of available numerical codes for more complex modeling also is provided. The selection criteria document (EPA 1987b) provides a more in-depth discussion of numerical codes and other models. In addition, it provides guidelines and procedures for evaluating the appropriate level of complexity required for various applications. The document lists criteria to consider when selecting a surface water model, including: (1) type of water body, (2) presence of steady-state or transient conditions, (3) point versus non-point sources of contamination, (4) whether 1, 2, or 3 spatial dimensions should be considered, (5) the degree of mixing, (6) sediment interactions, and (7) chemical processes. Each of the referenced documents should be consulted prior to any surface water modeling.

6.5.6 ESTIMATE EXPOSURE CONCENTRATIONS IN SEDIMENTS

In general, use sediment monitoring data to estimate exposure concentrations. Sediment monitoring data can be expected to provide better temporal representativeness than surface water concentrations. This will especially be true in the case of contaminants such as PCBs, PAHs, and some inorganic chemicals, which are likely to remain bound to the sediments. When using monitoring data to represent exposure concentrations for direct contact exposures, data from surficial, near-shore sediments should be used.

If modeling is needed to estimate sediment exposure concentrations, consult SEAM (EPA 1988b). SEAM treats surface water and sediment together for the purpose of listing available models for the release and transport of contaminants. Models for soil erosion releases are equally applicable for estimating exposure concentrations for surface water and sediment. Many of the numerical models listed in SEAM and the surface water selection criteria document (EPA 1987b) contain sections devoted to sediment fate and transport.

6.5.7 ESTIMATE CHEMICAL CONCENTRATIONS IN FOOD

Fish and shellfish. Chemical concentrations in fish and shellfish may be measured or estimated. Site-specific measured values are preferable to estimated values, but before using such values, evaluate the sampling plan to determine if it was adequate to characterize the population and species of concern (see Section 4.5.6 for some sampling considerations). Also examine analytical procedures to determine if the quantitation limits were low enough to detect the lowest concentration potentially harmful to humans. Inadequate sampling or high levels of quantitation may lead to erroneous conclusions.

In the absence of adequate tissue measurements, first consider whether the chemical bioconcentrates (i.e., is taken up from water) or bioaccumulates (i.e., is taken up from food, sediment, and water). For example, low molecular weight volatile organic chemicals do not bioaccumulate in aquatic organisms to a great extent. Other chemicals accumulate in some species but not in others. For example, PAHs tend to accumulate in mollusk species but not in fish, which rapidly metabolize the chemicals. For those chemicals that bioconcentrate in aquatic species of concern, use the organism/water partition coefficient (i.e., bioconcentration factor, or BCF) approach to estimate steady-state concentrations. BCFs that estimate concentrations in edible tissue (muscle) are generally more appropriate for assessing human exposures from fish or shellfish ingestion than those that estimate concentrations in the whole body, although this is not true for all aquatic species or applicable to all human populations consuming fish or shellfish. When data from multiple experiments are available, select the BCF from a test that used a species most similar to the species of concern at the site, and multiply the BCF directly by the dissolved chemical concentration in water to obtain estimates of tissue concentrations. Be aware that the study from which the BCF is obtained should reflect a steady state or equilibrium condition, generally achieved over long-term exposures (although some chemicals may reach steady state rapidly in certain species). For some chemicals, BCFs may overestimate tissue levels in fish that may be exposed only for a short period of time.

When no BCF is available, estimate the BCF with a regression equation based on octanol/water partition coefficients (K_{ow}). Several equations are available in the literature. Those developed for chemicals with structural similarities to the chemical of concern should be used in preference to general equations because of better statistical correlations.

The regression equation approach to estimating BCFs can overestimate or underestimate concentrations in fish tissue depending upon the chemical of concern and the studies used to develop the regression equations. For example, high molecular weight PAHs (such as benz(a)pyrene) with high K_{ow} values lead to the prediction of high fish tissue residues. However, PAHs are rapidly metabolized in the liver, and do not appear to accumulate significantly in fish. Regression equations using K_{ow} cannot take into account such pharmacokinetics, and thus may overestimate bioconcentration. On the other hand, studies used to develop regression equations which were not representative of steady-state conditions will tend to underestimate BCFs.

Typical methods for estimating fish tissue concentrations are based on dissolved chemical concentrations in water. While chemicals present in sediment and biota may also bioaccumulate in fish, there are only limited data available to estimate contributions to fish from these sources. However, chemicals that readily adsorb to sediments, such as PCBs, can be present in surface water at concentrations below detection limits and still significantly bioaccumulate. Some models are available to assess the contribution of chemical concentrations in sediment to chemical concentrations in aquatic biota. CEAM (ERL Athens) may be of assistance in choosing and applying an appropriate model.

Plants. Site-related chemicals may be present in plants as a result of direct deposition onto plant surfaces, uptake from the soil, and uptake from the air. When possible, samples of plants or plant products should be used to estimate exposure concentrations. In the absence of monitoring data, several modeling approaches are available for estimating exposure concentrations in plants. Use of these models, however, can introduce substantial uncertainty into an exposure assessment.

If deposition onto plants is the source of the chemical, air deposition modeling can be used in conjunction with plant interception fractions to estimate uptake. The plant interception fraction can be estimated by methods published in the literature or can be developed for a specific crop by considering crop yield and the area of the plant available for deposition.

If soil contamination is the source of the chemical, calculate the concentration in plants by multiplying soil to plant partition coefficients by soil concentrations. Use the open literature or computerized data bases to obtain these coefficients from field, microcosm, or laboratory experiments that are applicable to the type of vegetation or crop of concern (see EPA 1985c sludge documents for some). In the absence of more specific information, use general BCFs published in the literature that are not crop-specific (see Baes *et al*. 1984 for some). When using these parameters, it is important to consider that many site-specific factors affect the extent of uptake. These factors include pH, the amount of organic material present in soil, and the presence of other chemicals.

When literature values are not available, consider equations published in the literature for estimating uptake into the whole plant, into the root, and translocation from the root into above ground parts (see Calamari *et al*. 1987). Such methods require physical/chemical parameters such as K_{ow} or molecular weight and were developed using a limited data base. Scientific judgment must always be applied in the development and application of any partition coefficient, and caution must be applied in using these values in risk assessment.

Terrestrial animals. Use tissue monitoring data when available and appropriate for estimating human exposure to chemicals in the terrestrial food chain. In the absence of tissue monitoring data, use transfer coefficients together with the total chemical mass ingested by an animal per day to estimate contaminant concentrations in meat, eggs, or milk. Data to support modeling of uptake by terrestrial animals generally are not available for birds, but are available for some mammalian species. Terrestrial mammals such as cattle are simultaneously exposed to chemicals from several sources such as water, soil, corn silage, pasture grass, and hay. Cattle ingest varying amounts of these sources per day, each of which will contain a different contaminant concentration. Because all sources can be important with regard to total body burden, an approach based upon the daily mass of chemical ingested per day is recommended because it can be applied to input from many sources.

Obtain transfer coefficients from the literature (see Ng *et al*. 1977, 1979, 1982; Baes *et al*. 1984 for some), or calculate them directly from feeding studies (see Jensen *et al*. 1981; Jensen and Hummel 1982; Fries *et al*. 1973; Van Bruwaene *et al*. 1984). In the absence of this information, use regression equations in the literature for the estimation of transfer coefficients (see Travis and Arms 1988). It is important to be aware that regression equations that use feeding study results from short-term exposures may underestimate meat or milk concentrations. In addition, regression equations which rely on K_{ow} values may overestimate exposures for chemicals such as benz(a)pyrene that are rapidly metabolized. Information on the amount of feed, soil and water ingested by dairy and beef cows is available in the literature and should be combined with chemical concentrations in these media to estimate a daily dose to the animal.

6.5.8 SUMMARIZE EXPOSURE CONCENTRATIONS FOR EACH PATHWAY

Summarize the exposure concentrations derived for each pathway. Exhibit 6-10 presents a sample format.

6.6 QUANTIFICATION OF EXPOSURE: ESTIMATION OF CHEMICAL INTAKE

This section describes the methodology for calculating chemical-specific intakes for the populations and exposure pathways selected for quantitative evaluation. The general equation for estimating intake was shown in Exhibit 6-9. Remember that the intakes calculated in this step are expressed as the amount of chemical at the exchange boundary (e.g., skin, lungs, gut) and available for absorption. Intake, therefore, is not equivalent to absorbed dose, which is the amount of a chemical absorbed into the blood stream.

The sections that follow give standard equations for estimating human intakes for all possible exposure routes at a site. Values for equation variables are presented for use in evaluating residential exposures. Considerations for deriving pathway-specific variable values for populations other than residential (i.e., commercial/industrial or recreational) also are given. In general, both upper-bound (e.g., 95th percentile or maximum values) and average (mean or median) values are presented. These values can be used to calculate the RME or to evaluate uncertainty. A general discussion of which variable values should be used to calculate the RME was provided in Section 6.4.1; more specific guidance follows. A discussion of the uncertainty analysis is presented in Section 6.8.

The information presented below is organized by exposure medium and exposure route.

6.6.1 CALCULATE GROUND-WATER AND SURFACE WATER INTAKES

Individuals may be exposed to chemicals of potential concern in ground water and surface water by the following routes:

(1) ingestion of ground water or surface water used as drinking water;

(2) incidental ingestion of surface water while swimming; and

(3) dermal contact with ground water or surface water.

Inhalation exposures to chemicals that have volatilized from surface or ground water are covered in Section 6.6.3.

Intake from drinking water. Calculate residential intakes from ingestion of ground water or surface water used as drinking water, using the equation and variable values presented in Exhibit 6-11. As discussed in section 6.5.3, chemical concentration in water (CW) should be based on data from unfiltered samples. Develop pathway-specific variable values as necessary. Ingestion rates (IR) could be lower for residents who spend a portion of their day outside the home (e.g., at work). Also, exposure frequency (EF) may vary with land use. Recreational users and workers generally would be exposed less frequently than residents.

Intake from ingestion of surface water while swimming. Calculate intakes from incidental ingestion of surface water while swimming. Use the equation and variable values presented in Exhibit 6-12. Chemical concentration in water (CW) should represent unfiltered concentrations. Incidental ingestion rates (IR) while swimming have not been found in the available literature. SEAM (EPA 1988b) recommends using an incidental ingestion rate of 50 ml/hour of swimming. Exposure duration (ED) will generally be less for recreational users of a surface water compared to residents living near the surface water. Workers are not expected to be exposed via this pathway.

Intake from dermal contact. Calculate intakes from dermal contact with water while swimming, wading, etc., or during household use (e.g., bathing).

Use the equation and variable values presented in Exhibit 6-13. <u>In this case, the calculated exposure is actually the absorbed dose, not the amount of chemical that comes in contact with the skin (i.e., intake)</u>. This is because permeability constants (PC) reflect the movement of the chemical across the skin to the stratum corneum and into the bloodstream. Be sure to record this information in the summary of exposure assessment results so that the calculated intake is compared to an appropriate toxicity reference value in the risk characterization chapter. Note that PC are based on an equilibrium partitioning and likely result in an over-estimation of absorbed dose over short exposure periods (e.g., < 1 hr). The open literature should be consulted for chemical-specific PC values. The values in SEAM (EPA 1988b) are currently being reviewed and should not be used at this time. If chemical-specific PC values are not available, the permeability of water can be used to derive a default value. (See Blank *et al.* [1984] for some values [e.g., 8.4×10^{-4} cm/hr].) Note that this approach may underestimate dermal permeability for some organic chemicals.