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AN EVALUATION OF THE POTENTIAL IMPACTS OF PLANT AND ANIMAL INTRUSION INTO DISPOSED WASTE AT TA-54, MDA G

September 1999

Prepared for

LOS ALAMOS NATIONAL LABORATORY

ROGERS & ASSOCIATES ENGINEERING
A DAMES & MOORE COMPANY

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AN EVALUATION OF THE POTENTIAL IMPACTS OF PLANT AND ANIMAL INTRUSION INTO DISPOSED WASTE AT TA-54, MDA G

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Prepared for

Los Alamos National Laboratory

September 1999

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1. INTRODUCTION

The Los Alamos National Laboratory (LANL) generates radioactive waste as a result of various activities. Operational waste is generated by nuclear weapons research and development, energy production, and medical research, while environmental restoration (ER) and decontamination and decommissioning (D&D) waste is generated as contaminated sites and facilities at LANL undergo cleanup or remediation. Much of this waste is low-level radioactive waste (LLW) and is disposed of at the Technical Area 54 (TA-54) Material Disposal Area G (MDA G) disposal facility.

U.S. Department of Energy (DOE) Order 5820.2A (DOE, 1988) requires that radioactive waste be managed, treated, and disposed of in a manner that protects public health and safety, and the environment. To comply with this order, DOE field sites must prepare and maintain site-specific radiological performance assessments for all active LLW disposal facilities. Each DOE site must also prepare and maintain a composite analysis which accounts for the cumulative impact of active and planned LLW disposal facilities and other sources of radioactive contamination that could interact with these facilities (DOE, 1996).

LANL completed the latest drafts of the MDA G performance assessment and composite analysis in March of 1997 (LANL, 1997). These analyses estimated rates of radionuclide release from the waste disposed of at the facility, simulated the movement of radionuclides through the environment, and projected potential radiation doses to humans for several on- and off-site exposure scenarios. The assessments were based on site and disposal facility data, and assumptions about future rates and methods of waste disposal.

The accuracy of the performance assessment and composite analysis depends upon the validity of the data and assumptions made in conducting the modeling analyses. DOE field sites are required to implement a performance assessment maintenance program. The purpose of this program is to ensure the continued applicability of the analysis through

incremental improvement of the level of understanding of the disposal site and facility. Site personnel are required to conduct field and experimental work to reduce the uncertainty in the data and models used in the assessment. While the maintenance program is designed to specifically address the performance assessment, the same modeling methodology was used to conduct the composite analysis for MDA G. Consequently, insight gained from activities conducted under the LANL performance assessment maintenance program will also generally apply to the composite analysis.

The projected off-site exposures for the performance assessment and composite analysis were greatest for the atmospheric pathway. The peak dose for the composite analysis was projected to occur as a result of plants and animals intruding into the disposed waste and moving contamination to the ground surface. This contamination was suspended and transported to downwind locations with the prevailing winds, exposing hypothetical receptors to contamination at those sites. A peak dose of approximately 6 mrem/yr was estimated for the composite analysis at the receptor location in Canada del Buey. A worst-case estimate of the potential dose, taking into account modeling and data uncertainties, exceeded the 30 and 100 mrem/yr performance objectives for this pathway.

The magnitude of the uncertainty associated with the composite analysis atmospheric pathway doses is unacceptable. Consequently, efforts were initiated under the performance assessment maintenance program to refine the modeling conducted for this pathway, where deemed appropriate, and to reevaluate the sources of uncertainty associated with the pathway. This analysis will address the models and data used to estimate radionuclide releases due to plant and animal intrusion, to estimate resuspension rates for surface contamination, and to simulate atmospheric transport of airborne contaminants from MDA G to human receptor locations in Canada del Buey and White Rock.

This report documents the reevaluation of the biotic intrusion modeling performed to estimate the transport of contaminants to the ground surface. The approach adopted to conduct this analysis is discussed in Section 2. The results of the analysis are presented in

Section 3, and discussed relative to the results presented in the March 1997 draft of the performance assessment and composite analysis (LANL, 1997).

2. METHODOLOGY

The extent to which radionuclides are transported to the surface of MDA G as a result of biotic intrusion is proportional to the degree to which the roots of plants and burrows of animals penetrate into the disposed waste. Estimates of the extent of intrusion following closure of the disposal site were prepared for the MDA G performance assessment and composite analysis. However, assumptions about the long-term maintenance of the disposal site played a critical role in the manner in which this modeling was conducted. The assumption was made that measures would be taken during the 100-year active institutional control period to effectively prevent intrusion into the waste. Furthermore, only moderate disturbance of the site was assumed to occur between the end of active institutional control and the end of the 1,000-year compliance period because of actions taken under the site's long-term maintenance plan.

Consistent with the assumption of long-term preventative maintenance, the plant root distributions used to model contaminant transport to the ground surface were selected to represent species with moderate rooting characteristics (LANL, 1997). A maximum rooting depth of 2 m was used to model disruption of the waste under baseline, or expected, conditions following the end of active institutional control. While the majority of plant root biomass is, in fact, expected to occur within 2 m of the ground surface (Tierney and Foxx, 1987), the use of this maximum rooting depth does not explicitly account for roots that extend to greater depths. The potential for disruption of the waste by burrowing animals was evaluated for the performance assessment and composite analysis using a single representative species patterned after the deer mouse. Although this species is the most common burrowing species observed at MDA G, trapping studies and visual observations have indicated the presence of other species that may pose a greater risk to the integrity of the disposed waste.

All in all, the approach used to estimate the impacts of biotic intrusion for the MDA G performance assessment and composite analysis relied on long-term site maintenance to

confine disturbance of the waste to moderate levels. This approach is valid if it can be assured that the maintenance program will be effective in ridding the site of deep rooting plants and minimizing disturbance of the site by burrowing species. However, if the maintenance program is ineffective or is not implemented properly, the results of the biotic intrusion modeling conducted for the MDA G performance assessment and composite analysis may be invalidated. Recognizing this, a more comprehensive evaluation of the impacts of biotic intrusion was undertaken.

The approach adopted for the updated biotic intrusion modeling is described in three sections. Section 2.1 provides a general overview of the approach, and some of the background information required to understand the models and data used to implement them. The models used to project rates of biotic intrusion into the disposed waste are described in Section 2.2. The data used to implement the models are presented and discussed in Section 2.3.

2.1 GENERAL APPROACH

The updated biotic intrusion modeling analysis is based on the assumption that no steps are taken to prevent the establishment of plant and animal species at the closed disposal facility. Under these conditions, the degree to which the disposed waste is disrupted will be determined by the natural rooting characteristics of the plant species that inhabit the site and the burrowing habits of the resident animals. Plant species inhabiting the site shortly after facility closure will consist of annual and perennial grasses and forbs that are used to seed the site or that invade from surrounding areas. Over time, however, the site will undergo ecological succession, giving rise to dramatically different plant communities. The species and numbers of animals living in the area will shift in response to changes in the plant community.

The effects of ecological succession on the potential for biotic intrusion were taken into account in the updated biotic intrusion modeling. This was done using two separate simulations. The first simulation considered the potential for biotic intrusion shortly after the disposal facility is closed. Closure activities, including the application of final covers, contouring of the site, and the establishment of vegetation, were assumed to leave the site in a disturbed condition. Initially, dominant species of vegetation will include annual and perennial grasses and forbs. Shrubs and half-shrubs will invade the area within a few years time. The second simulation modeled biotic intrusion after the site has undergone ecological succession and has achieved, or is close to, a climax condition. Trees will be the dominant feature of the plant community, although grasses, forbs, and shrubs will continue to play a role in ecosystem functioning.

The use of two separate simulations to estimate the effects ecological succession has upon biotic intrusion potential is artificial insofar as there will be a gradual transition from the site in its early successional stages to the climax condition. Modeling this transition would be difficult at best, and subject to a high degree of uncertainty. Given this, the use of two discrete simulations was considered to be a reasonable approach to understanding the effects that site dynamics may have upon biotic intrusion.

Baseline simulations of biotic intrusion were conducted for early successional stages and climax conditions using best estimates for each of the model input parameters. Sources of uncertainty associated with the models and data used to conduct these simulations, however, will introduce errors into the projected impacts of biotic intrusion. The nature and magnitude of these errors need to be understood in order to conclude, with a reasonable level of confidence, that MDA G is capable of safely isolating the waste from the environment. Towards this end, uncertainty and sensitivity analyses were performed.

While the models used to simulate biotic intrusion at MDA G have uncertainties associated with them, they are expected to provide reasonably conservative estimates of the actual impacts of plants and animals at the site. In contrast, the uncertainty associated with many of the input parameters used to implement the models is great. Errors introduced into the modeling by these uncertainties may also be significant. Given this, the

uncertainty analysis focused on model parameter uncertainty and its impact on the ability of MDA G to satisfy regulatory requirements. The sensitivity analysis was performed to identify the input parameters to which the projected impacts of biotic intrusion were most sensitive. The analysis took into consideration the uncertainties associated with the model parameters, as defined for the uncertainty analysis.

2.2 BIOTIC INTRUSION MODELS

The biotic intrusion models developed for the MDA G performance assessment and composite analysis (LANL, 1997) were also used for this updated modeling. The model for plant intrusion is based on the assumption that roots of plants growing over the site extend into the buried waste. The roots extending into the waste take up radionuclides, which are assimilated by the plants. Litter is formed as plants die and shed their leaves, contamination in the plant material enters the soil as the litter decomposes. Animals constructing burrows for cover or as a means of foraging may also penetrate the disposed waste. Contamination transported to the surface may be mixed with clean soil excavated from the portions of the burrows that lie above the waste, and spread over the ground surface.

The biotic intrusion models project radionuclide concentrations in the surface soils overlying the disposal units, normalized to contaminant concentrations in the disposed waste. The model for plant intrusion is given by:

$$\frac{C_{(p,i,j)}(t)}{C_{w,i}} = f_{p,j} B_{i,j} e^{-\lambda_{p,i,j} t} \qquad (1)$$

where

$C_{(p,i,j)}(t)$	=	soil concentration of radionuclide i due to uptake by plant species j (Ci/m^3)
$C_{w,i}$	=	original waste concentration of radionuclide i (Ci/m^3)
$f_{p,j}$	=	fraction of root volume of plant species j that penetrates into the waste
$B_{i,j}$	=	biota transfer factor for radionuclide i in plant species j
$\lambda_{p,i,j}$	=	buildup constant for radionuclide i in plant species j (yr^{-1})

t = time since facility closure (yr).

The biota transfer factor for plants, $B_{i,j}$, is the radionuclide-specific plant uptake factor. The buildup constant, $\lambda_{p,i,j}$, is given by:

$$\lambda_{p,i,j} = \frac{m_{p,j} f_{p,j} B_{i,j}}{\rho_s (L_{p,j} - L_c)} \tag{2}$$

where

$m_{p,j}$	=	mass transfer rate to the surface soil for plant species j (kg/m^2/yr)
ρ_s	=	bulk density of soil/waste (kg/m^3)
$L_{p,j}$	=	maximum rooting depth of plant species j (m)
L_c	=	cover thickness (m).

The mass transfer rate, $m_{p,j}$, represents the decay of contaminated plant litter and the formation of soil. All litter is assumed to decay at the site of formation within a year of the time it is generated.

Normalized surface concentrations of radionuclides resulting from animal intrusion are given by:

$$\frac{C_{sa,i,j}(t)}{C_{iwo,i}} = f_{a,j} + \left(\frac{L_{a,j} - L_c}{L_{a,j}} - f_{a,j}\right)\left(1 - e^{-\lambda_{a,j} t}\right) \tag{3}$$

where

$C_{sa,i,j}$	=	soil concentration of radionuclide i due to intrusion by animals species j (Ci/m^3)
$f_{a,j}$	=	fraction of burrow volume of animal species j that penetrates into the waste
$L_{a,j}$	=	maximum burrowing depth of animal species j (m)
$\lambda_{a,j}$	=	buildup constant for animal species j (yr^{-1}).

The buildup constant for animal intrusion is given by:

$$\lambda_{a,j} = \frac{m_{a,j} f_{a,j}}{\rho_s (L_{a,j} - L_c)} \tag{4}$$

where

$m_{a,j}$ = mass transfer rate to the surface soil for animal species j (kg/m²/yr).

The mass transfer rate for animals is simply equal to the amount of soil brought to the surface each year.

The total soil concentration of a given radionuclide is determined by summing the contributions from the various species of plants and animals. These contributions are weighted by the mass transfer rates as shown below:

$$\frac{C_{st,i}}{C_{ww,i}} = \frac{\sum_{j=1}^{n} \frac{C_{sp,i,j}(t)}{C_{ww,i}} * m_{p,j} + \sum_{j=1}^{m} \frac{C_{su,i,j}(t)}{C_{ww,i}} * m_{a,j}}{\sum_{j=1}^{n} m_{p,j} + \sum_{j=1}^{m} m_{a,j}} \tag{5}$$

where

$C_{st,i}$ = total surface soil concentration of radionuclide i (Ci/m³)
n = number of plant species growing over the closed disposal units
m = number of animal species burrowing into the closed disposal units.

Surface erosion may, over extended periods of time, significantly reduce the thickness of the cover material over the disposed units, thereby allowing a greater proportion of plant roots and animal burrows access to the waste. The effects of erosion were included in the models presented above in an ad hoc fashion when they were used in the MDA performance assessment and composite analysis. Including the effects of erosion in the modeling had little effect on the projected impacts. In fact, it was found that the projected rate of erosion for the site would need to increase almost two orders of magnitude to significantly affect the projected exposures during the 1,000-year compliance period. Given this, the impacts of surface erosion were not taken into account in this updated modeling.

2.3 BIOTIC INTRUSION MODEL DATA

This update of the biotic intrusion modeling is based on the assumption that no attempt is made to exclude plants and animals from inhabiting the closed disposal site. Under these conditions, the potential impacts of biotic intrusion upon site performance will depend upon the plant and animal communities that naturally inhabit MDA G after the facility is closed. Consequently, characterizing these communities and how they may change over time as the site undergoes ecological succession was an important part of the revised modeling effort. Once these communities were identified, the data required to implement the biotic intrusion models were collected.

Data were collected to support the baseline modeling and to develop parameter distributions for use in the probabilistic uncertainty analysis. The approaches and sources of information used to support the selection of input parameters for the baseline assessment are discussed in Section 2.3.1. Section 2.3.2 discusses the information and rationale that went into developing distributions for the uncertainty analysis.

2.3.1 Baseline Simulation Data

The baseline simulation projects the impacts of plant and animal intrusion on the ability of MDA G to safely isolate the waste from the environment using best estimates of the model parameters discussed in Section 2.2. Given the basic differences in the manner in which plants and animals exert their influence, it is not surprising that the data required to model their respective impacts are also quite different. These data are discussed in the following sections.

Plant Intrusion Data

The input data required to model the potential for plant intrusion at MDA G are the same whether the site is assumed to be in its early successional stages or its climax condition. They include:

- Plant species present at the disposal site.
- Plant root distributions with depth.
- Radionuclide uptake factors for plants.
- Plant litter production rates.
- Thickness of cover over the disposed waste.

The ecological characteristics of the LANL and, more specifically, MDA G must be understood in order to determine the plant species that will be present at the disposal site. A diverse array of plants and animals is found in the Los Alamos region, owing to the large elevational gradient between the Rio Grande River (1500 m above mean sea level) and the Jemez Mountains (7000 m above mean sea level), and the canyon and mesa terrain (DOE, 1979). Six major vegetative community types are found in Los Alamos County including juniper-grassland, pinyon-juniper, ponderosa pine, mixed conifer, spruce-fir, and subalpine grassland. The juniper-grassland, pinyon-juniper, and ponderosa pine communities predominate, with each occupying about one-third of the Laboratory. The juniper-grassland occurs along the Rio Grande and the eastern border of the Pajarito Plateau, extending up to elevations of 1,700 to 1,900 m on the south-facing sides of canyons. The pinyon-juniper community covers large portions of mesa tops at elevations ranging from 1,900 to 2,100 m. Ponderosa pines are found at elevations ranging from 2,100 to 2,300 m in the western portion of the plateau.

Undisturbed areas on Mesita del Buey, the mesa upon which MDA G is located, are dominated by pinyon-juniper woodland. Pinyon pines (*Pinus edulis*) and one-seed juniper (*Juniperus monosperma*) are the dominant tree species, while common shrub species

include big sagebrush (*Artemisia tridentata*), four-wing salt bush (*Atriplex canescens*), currant (*Ribes cereum*), and mountain mahogany (*Cercocarpus montanus*). Blue grama grass (*Bouteloua gracilis*), cryptogamic soil crust, and prickly pear cactus (*Opuntia polyacantha*) are among the most common understory plants on the mesa top. Others include snakeweed (*Gutierrezia sarothrae*), pinque (*Hymenoxys richardsonii*), wild chrysanthemum (*Bahia dissecta*), leafy golden aster (*Chrysopsis filiosa*), purple horned-toothed moss (*Ceratadon purpureus*), several lichen species, three-awn grass (*Aristida* spp.), bottlebrush squirreltail (*Sitanion hystrix*), bluegrass (*Poa* spp.), and false tarragon (*Artemisia dracunculus*). Waste management operations at MDA G have replaced a number of the understory plants native to the area. Recently disturbed areas support plants such as goosefoot (*Chenopodium fremontii*), Russian thistle (*Salsola kali*), cutleaf evening primrose (*Oenothera caespitosa*), common sunflower (*Helianthus anuus*), and other colonizing species. Vegetation introduced as disposal pits are closed consists of native grasses, including blue grama, buffalo grass (*Buchloe dactyloides*), western wheatgrass (*Agropyron smithii*), and dropseed (*Sporobolus* spp.), and forbs such as alfalfa (*Medicago sativa*).

Based on the information provided above, the disposal units at MDA G are expected to undergo ecological succession from their disturbed state shortly after closure to a pinyon-juniper woodland climax, characteristic of the undisturbed portions of Mesita del Buey. Annual and perennial grasses and forbs will predominate when the site is in its early successional stages, becoming established as covers over disposal units are seeded and as grasses and forbs invade from surrounding areas on the mesa. Over time, shrubs and trees will take hold and become established at the site. While some species of grasses and forbs will die out, others will continue to thrive. Given enough time, a condition approximating the climax pinyon-juniper woodland will be attained.

The length of time required for the site to pass from the early stages of succession to a climax condition depends upon conditions at the disposal site, many of which are difficult to predict with certainty. However, information gathered by Arnold et al. (1964) provides some insight into the temporal aspects of succession in these communities. These investigators compared transect measurements from burned areas to measurements taken

in adjacent unburned pinyon-juniper stands at three locations in northern Arizona. Measurements were taken in 1954, approximately 70 to 90 years after the fires had occurred. At a site that had been burned about 80 years prior, tree canopy intercept was about 0.1 percent of that in the adjacent unburned areas; intercepts of mid-grasses and shrub cover were 2.8 and 5.8 times greater in the burned area than in the adjacent mature woodland. Tree cover in 70 and 90 year-old burns were 12 and 15 percent of the cover measured in adjacent unburned communities, grasses and shrubs were still much more abundant in the burned areas.

Based on the data from Arnold et al. (1964), it appears that more than 100 years will be required to establish the mature pinyon-juniper woodland at MDA G. This conclusion is supported by the work of Tress and Klopatek (1987), who estimated the rate of succession in pinyon-juniper woodlands in north-central Arizona based on several post-fire communities. Based on crown cover estimates, it was estimated that slightly more than 200 years would be required for the community to return to an "equivalent" state. Longer periods of time for recovery were indicated by other measures of community structure.

While the general nature of the vegetative cover at MDA G can be predicted for the site in its early successional stages and climax condition, predicting the species-specific composition with any degree of accuracy is difficult. Given this, the plant communities present at the site were identified in terms of the general growth-forms present under each set of conditions, rather than on a species-specific basis. Four growth-forms were identified to implement this approach, including annual and perennial grasses, annual and perennial forbs, shrubs (including subshrubs), and trees. Grasses, forbs, and shrubs were included in the simulation of the site when it is in the early stages of succession. As stated above, grasses and forbs are expected to be early colonizers at the site. While few, if any, shrubs will occur over the recently closed disposal units, they are expected to begin colonization of MDA G within a matter of years. Consequently, the growth-form was included in the simulation of the site. The simulation conducted to address climax conditions at the disposal site included all four vegetation growth-forms.

Plant rooting depth distributions provide the information required for estimating the proportion of the root mass that penetrates the waste and the maximum depth of plant intrusion, the parameters f_{ij} and L_{ij} in Equations (1) and (2). Insofar as the plant communities present at MDA G were defined on the basis of growth-forms, the root distributions used in the modeling had to have a similar basis. These growth-form specific distributions were developed using a combination of LANL-specific data and information taken from the open literature.

Foxx et al. (1984) conducted an extensive review of the rooting characteristics of native and crop plants that occur within the United States. Most of the references included in the review were studies performed in states west of the Mississippi River, a number of species native to the Laboratory were included in the citations. The authors constructed a database of plant rooting depths, and used this information to estimate rooting depth distributions for individual plant species. A distribution was developed for a species if there were at least six data points for the plant. If there were insufficient species-specific data, distributions were developed on the basis of plant genera. All told, Foxx et al. (1984) developed rooting depth distributions for 12 species or genera of grasses, 10 species or genera of forbs, and two species or genera of shrubs.

Least-squares regression analyses of the data developed by Foxx et al. (1984) for grasses and forbs were performed to estimate composite rooting depth distributions for the two growth-forms. The data used to conduct these analyses and the predicted composite distributions for grasses and forbs are shown in Figures 2-1 and 2-2, respectively. The rooting distributions developed by Foxx et al. (1984) for shrubs were considered inadequate for estimating a composite rooting depth distribution for the growth-form. Consequently, these data were supplemented with data for two additional shrub species, four-wing saltbush and rabbitbrush (*Chrysothamnus* spp.), to develop the composite distribution shown in Figure 2-3. These data were taken from Foxx et al. (1984) and Foxx and Tierney. (1987), the latter a report on plant root lengths growing on Laboratory lands. Rooting distributions were not provided by Foxx et al. (1984) for tree species, reflecting a general lack of root distribution data for these plants. In their absence, rooting data for ponderosa pine (*Pinus ponderosa*), pitch pine (*Pinus rigida*), jack pine (*Pinus banksiana*), and juniper

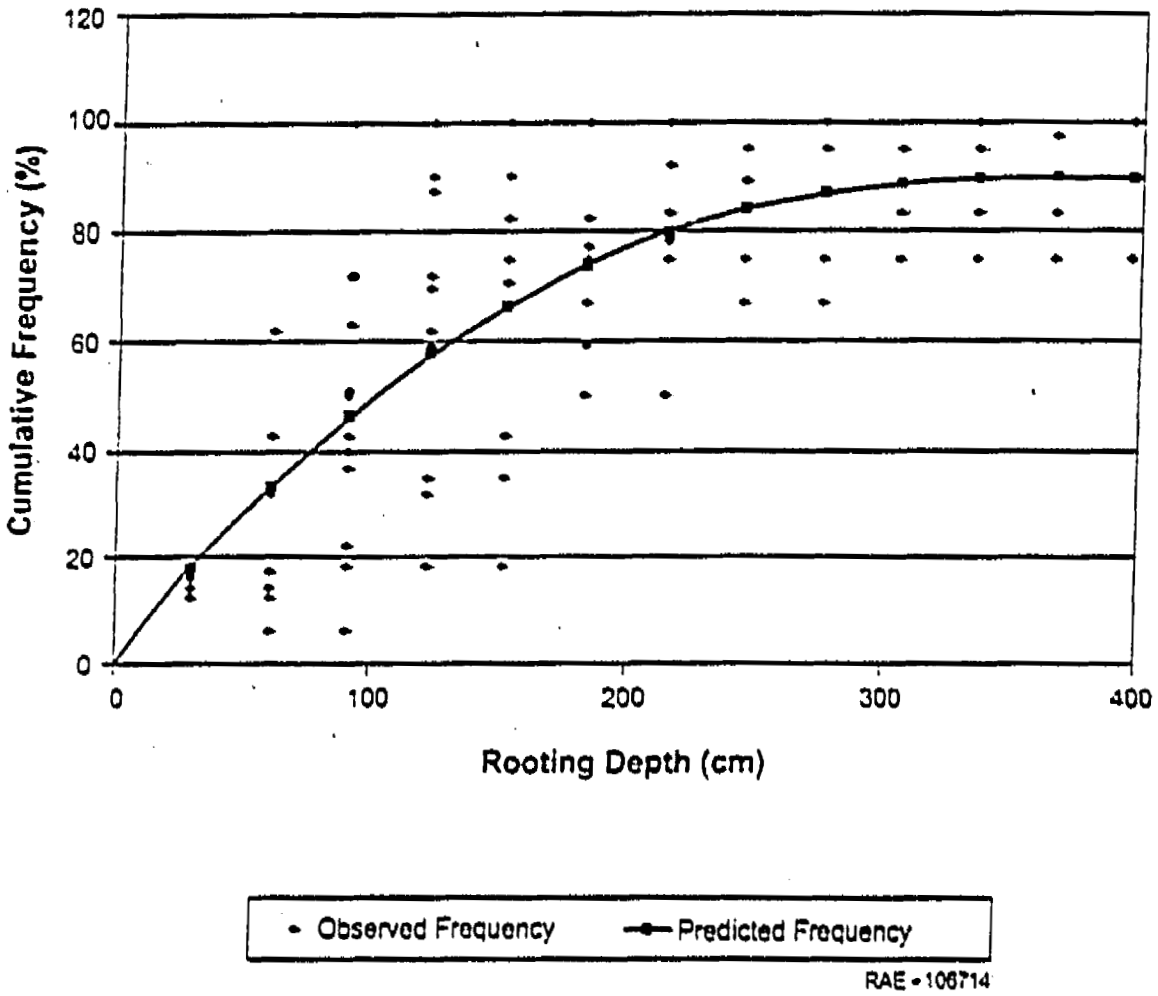

Figure 2-1. Composite rooting depth distribution for grasses.

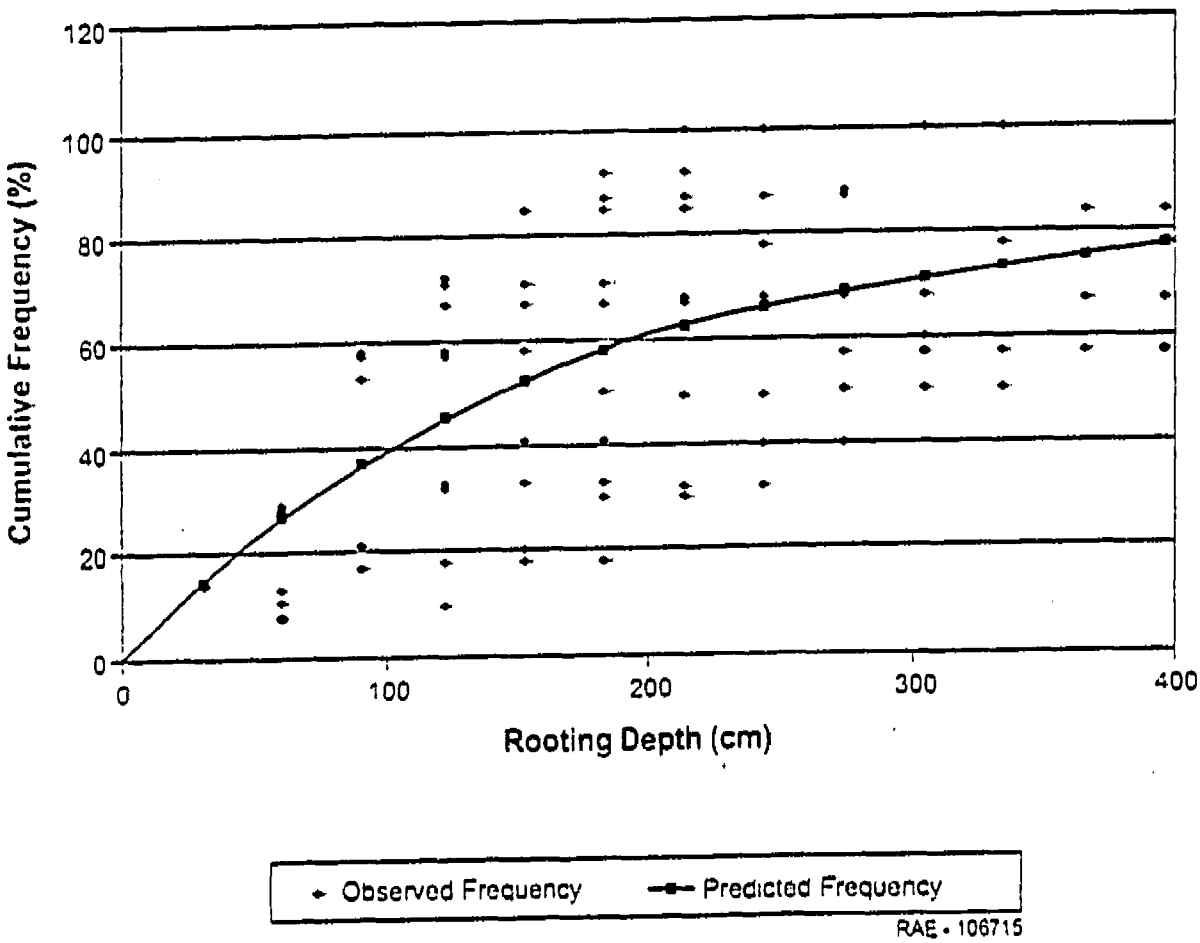

Figure 2-2. Composite rooting depth distribution for forbs.

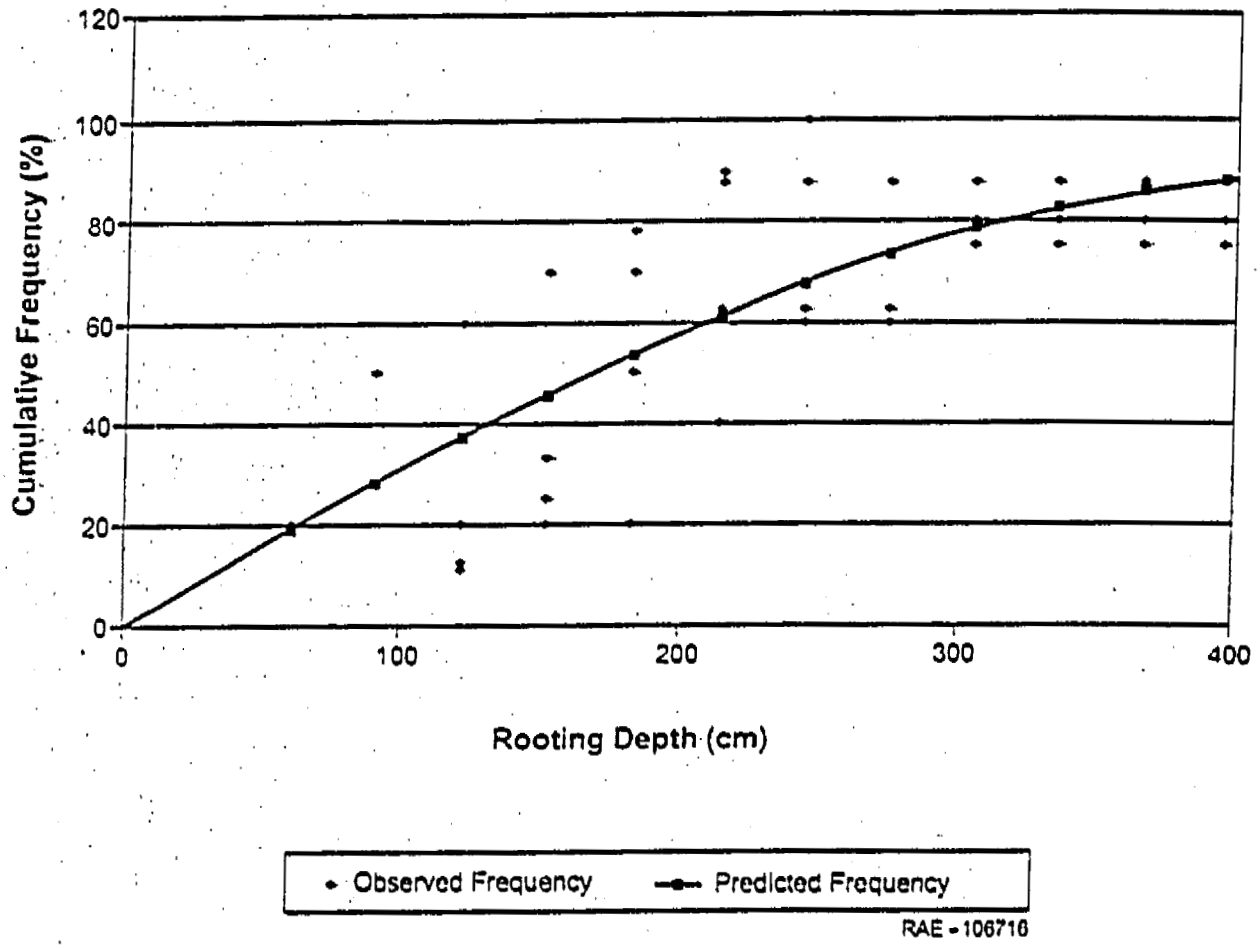

Figure 2-3. Composite rooting depth distribution for shrubs.

(*Juniperus* spp.) were taken from Foxx et al. (1984), Foxx and Tierney (1987), and Foxx (1999), and used to estimate the rooting depth distribution for trees in the pinyon-juniper woodland climax. The data and composite rooting distribution for the trees are shown in Figure 2-4.

The composite rooting distributions shown in Figures 2-1 through 2-4 show the fraction of roots occurring at or above a given depth. The proportion of roots extending into the waste is determined by subtracting the cumulative frequency shown for the cover thickness of interest from 1.0. For instance, the proportion of the roots of grasses penetrating waste disposed beneath a 2-m thick cover is 1.0 minus the cumulative frequency predicted by the regression equation depicted in Figure 2-1. The predicted frequency is 0.77, indicating that 23 percent of the roots of grasses penetrate into waste disposed beneath a 2-m cover. Using this approach, the proportions of roots penetrating into the waste for the four growth-forms and a range of cover depths were calculated. The results of these calculations are shown in Table 2-1.

Maximum rooting depths for the different growth-forms are included in Table 2-1. In general, these depths were based directly on the data used to generate the composite rooting distributions. In the case of the forbs, however, the maximum rooting depth shown excludes the two greatest depths recorded for alfalfa (i.e., 19 and 39 m). These data were not used because the conditions under which the roots were found to penetrate to these depths are not expected to occur at the Laboratory. For example, the roots of the plant extending to a depth of 39 m were found in an underground mine shaft.

Table 2-1. Proportions of plant roots penetrating the disposed waste and maximum rooting depths.

	Cover Thickness (m)			
Plant Growth-Form	1	2	3	Maximum Rooting Depth (m)
Grasses	5.0E-01	2.3E-01	1.2E-01	4.4E+00
Forbs	6.0E-01	3.9E-01	2.9E-01	9.1E+00
Shrubs	6.9E-01	4.2E-01	2.2E-01	7.6E+00
Trees	5.4E-01	3.2E-01	2.8E-01	6.1E+01

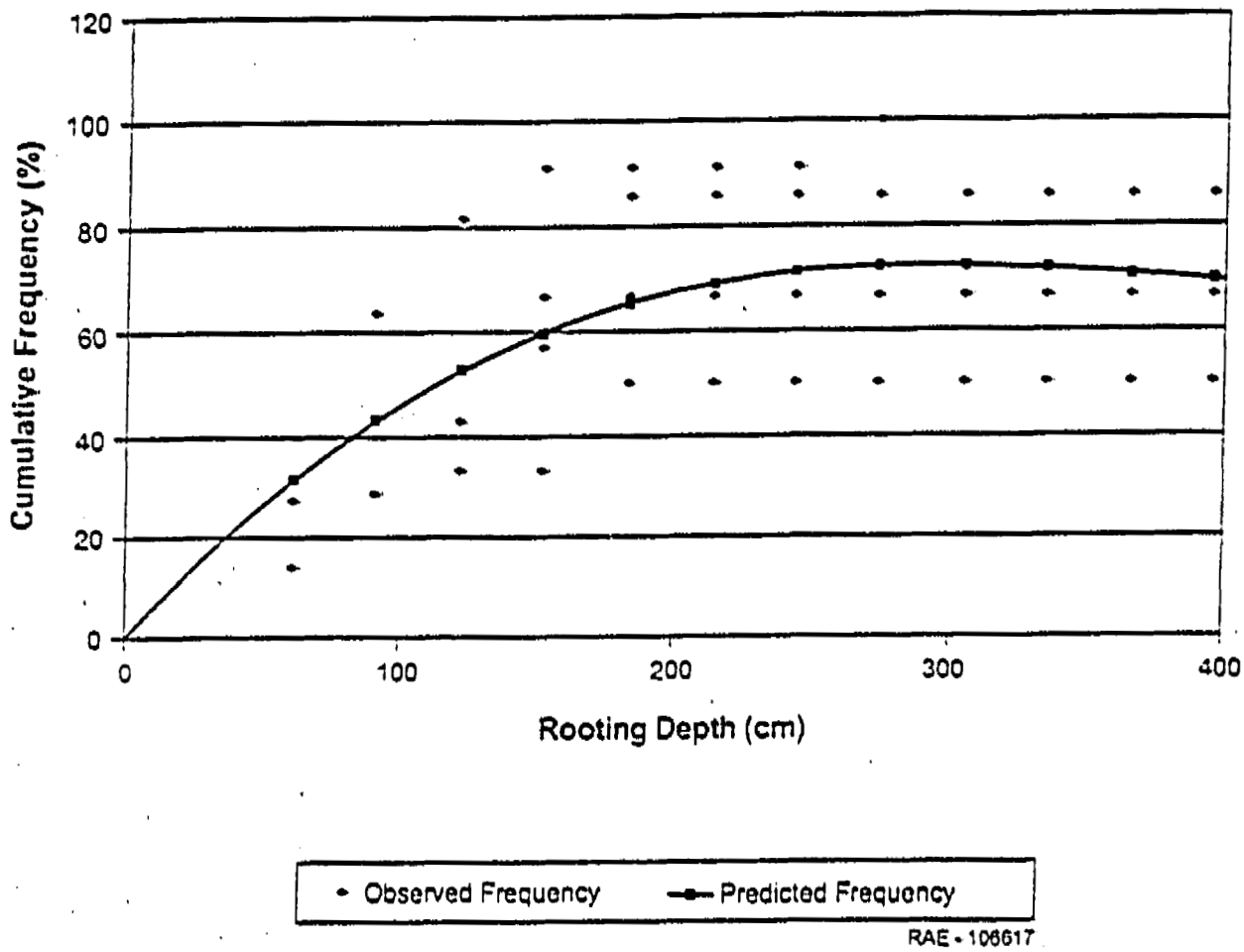

Figure 2-4. Composite rooting depth distribution for trees.

Litter generated by leaf fall and plant death will contribute to soil formation as the organic matter decays. The mass of material added to surface soils annually, m_{ij} in Equation (2), depends upon the above-ground biomass of the plants, the portion of this biomass that dies each year, and the time required for the organic matter to decay. A combination of site-specific data and information from the open literature was used to estimate these parameters and define the rates at which contamination passes from plants to soil for the site when it is in the early stages of succession.

Estimates of above-ground biomass for grasses, forbs, and shrubs were developed based on yield and cover data for each growth-form under conditions expected to resemble MDA G in early succession. When available, yields for the above-ground portions of grasses, forbs, and shrubs were taken directly from the literature. To supplement these data, additional estimates of production were developed by scaling total understory production by the relative cover for each growth-form. This process and the data used in its implementation are discussed in the following paragraphs.

Estimates of understory biomass for the early successional stages at MDA G were based on information from several different sites. In general, these sites consisted of pinyon-juniper woodlands that had been disturbed by fire or physical means, pinyon-juniper grasslands, and areas adjacent to mature woodlands. While MDA G is not expected to look exactly like any one of these sites, the understory production data for these communities are expected to provide reasonable estimates of conditions at the disposal site.

Direct measurements of above-ground production of grasses, forbs, and shrubs were found for a number of sites in the western United States, those used to develop baseline modeling data are summarized in Table 2-2. Clary (1989) evaluated the production of grasses, forbs, and shrubs on areas that had been disturbed by chaining and cabling two to 30 years prior. Average yields across the four sites and the four years in which measurements were conducted were 530 kg/ha for grasses, 33 kg/ha for forbs, and 76 kg/ha for shrubs. Dwyer and Pieper (1967) measured production of grasses and forbs in burned and unburned areas on pinyon-juniper rangeland in south-central New Mexico. The unburned area was free of trees and, as such, was considered to be a reasonable

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Table 2-2. Summary of cover and production studies used to estimate understory biomass for MDA G in its early successional stages.

Community Characteristics	Reference	Cover			Production		
		Grasses	Forbs	Shrubs	Grasses	Forbs	Shrubs
Pinyon-juniper woodland in central Utah	Clary, 1939	—[a]	—[a]	—[a]	Above ground production at four sites ranged from 290 to 770 kg/ha; average production of 570 kg/ha	Above ground production at four sites ranged from 0 to 100 kg/ha; average production of 33 kg/ha	Above ground production at four sites ranged from 7.9 to 160 kg/ha; average production of 76 kg/ha
Pinyon juniper rangeland in south-central New Mexico	Dwyer and Pieper, 1967	Cover at three sites ranged from 13 to 27 percent, average cover of 18 percent	Cover at three sites ranged from 0.17 to 0.87 percent; average cover of 0.45 percent	—[a]	Above-ground production at three sites ranged from 810 to 1000 kg/ha; average production of 890 kg/ha	Above-ground production at three sites ranged from 71 to 240 kg/ha; average production of 140 kg/ha	—[a]
Pinyon juniper rangeland in south-central New Mexico	Pieper, 1968	Cover at three sites ranged from 9.4 to 12 percent; average cover of 11 percent			Above ground production at three sites ranged from 630 to 730 kg/ha; average production of 680 kg/ha		
Grasslands surrounding pinyon-juniper woodlands in Arizona	Arnold et al., 1961	Cover on areas free of trees averaged 3.6 percent	Cover on areas free of trees averaged 0.3 percent	Cover on areas free of trees averaged 3.9 percent	Total herbage production in areas free of trees was approximately 670 kg/ha		
Fallow fields in pinyon-juniper cover type north of LANL	Foxx et al. 1997	Cover estimates for two sites were 3.0 and 10 percent	Cover estimates for two sites were 5.9 and 7.0 percent	Cover estimates for two sites were 0.1 and 3.5 percent	—[a]	—[a]	—[a]
Pinyon juniper woodland disturbed by fire seven years previously	Tress and Klopatek, 1987	Cover on seven-year old burn was 0.31 percent	Cover on seven-year old burn was 1.0 percent	Cover on seven-year old burn was 8.8 percent	—[a]	—[a]	—[a]

a. Cover measurements were not performed.
b. Growth form was not considered in the study.
c. Production measurements were not performed.

approximation of MDA G in its early successional stages. The data for the site, collected over a three-year period, indicated an average herbage production of about 890 kg/ha for grasses and 140 kg/ha for forbs. Additional work conducted by Pieper (1968) in south-central New Mexico examined the effects of grazing on herbage production in pinyon-juniper grasslands. Above-ground production ranged from 630 to 730 kg/ha on three areas protected from grazing. Grasses were the dominant growth-form, accounting for 81 to 98 percent of the vegetation on the basis of cover. Arnold et al. (1964) developed estimates of total herbage yields (i.e., grasses and forbs) for pinyon-juniper woodlands as a function of tree canopy intercept. Yields were about 670 kg/ha of air-dry vegetation in areas that were free of trees.

Several investigators have evaluated plant cover in disturbed pinyon-juniper woodlands and areas adjacent to mature woodlands (Table 2-2). Arnold et al. (1964) examined cover of grasses, forbs, and shrubs in and around pinyon-juniper woodlands in Arizona. The canopy intercept of trees in these woodlands ranged from 0 to 95 percent. In areas free of trees, grass and forb cover were approximately 3.6 and 0.3 percent, respectively. Shrub cover, including half-shrubs, was about 3.9 percent. Tress and Klopatek (1987) examined cover by growth-form in pinyon-juniper woodlands disturbed by fire 7 to 90 years ago, and in a mature woodland that had been free of fire for at least 300 years. Tree cover on the 7- and 35-year old burns was 2 percent or less, these sites were considered suitable for estimating conditions at MDA G in its early successional stages. Grass, forb, and shrub cover on the 7-year-old burn were about 0.40, 1.0, and 8.8 percent, respectively. Cover estimates for the 35-year old burn were 13 percent for grasses, 1.9 percent for forbs, and 22 percent for shrubs.

Foxx et al. (1997) surveyed eight fallow fields of the ponderosa pine and pinyon-juniper cover types to understand plant succession in the greater Los Alamos area. Two of these sites were located on the LANL, while six were located north of the Laboratory. Transects were established at each site to determine species composition, density, and abundance. Cover and frequency were measured for each dominant plant species, and used to develop cover estimates by growth-form (i.e., grasses, forbs, and shrubs).

Two of the fields surveyed by Foxx et al. (1997), Chupaderos and Pumice Mine fields, are found in pinyon-juniper cover types. These fields were removed from active use (i.e., grazing) in 1943, sampling was conducted by Foxx et al. approximately 40 years later in 1982. The total understory cover for Chupaderos Field was 13.5 percent, including 3 percent grass cover, 7 percent coverage by forbs, and 3.5 shrub cover. Small ponderosa pines were scattered throughout the area. The total understory cover for Pumice Mine Field was 16.5 percent, including 10.5 percent grass cover, 5.9 percent forb cover, and 0.1 percent shrub cover. Pinyon pine and juniper occurred at the site, a few ponderosa pine stumps were present as well.

The production and cover data discussed above were used to develop biomass estimates for grasses, forbs, and shrubs for MDA G in the early stages of succession. The understory production data generated by Clary (1989), Dwyer and Pieper (1967), and Pieper (1968) were assumed to apply directly to the disposal site. The production data provided by Pieper were divided between grasses and forbs based on percent cover of each growth-form. Production data for grasses and forbs were estimated for the studies conducted by Foxx et al. (1997), Arnold et al. (1964), and Tress and Klopatek (1987) using the total herbage production of 670 kg/ha developed by Arnold et al. (1964). This total yield was allocated among grasses and forbs based on the percent cover of each growth-form. While scaling in this manner is only approximate, it is expected to provide reasonable estimates of the relative contributions these growth-forms make to the total understory biomass. The yields generated using these approaches are summarized in Table 2-3.

Grass and forb biomass and litter generation rates were assumed to be equivalent to the above-ground yields listed in Table 2-3. The median values of 610 kg/ha for grasses and 47 kg/ha for forbs were adopted for the baseline biotic intrusion modeling. These data are expected to overstate the actual amount of litter that falls to the ground, as a portion of the dead plant material will be left standing for a period of time. For example, Grier et al. (1992) found that approximately 40 percent of the biomass of grasses and other understory vegetation in two pinyon-juniper woodlands consisted of standing dead material.

Table 2-3. Aboveground production of grasses, forbs, and shrubs in early successional stages of pinyon-juniper cover types.

Study	Above-Ground Production (kg/ha)		
	Grasses	Forbs	Shrubs
Clary, 1989			
Church Hills Site – 1982	3.9E+02	9.5E+01	1.8E+02
Church Hills Site – 1985	4.3E+02	1.0E+02	8.5E+01
Clay Springs Site – 1980	2.9E+02	3.7E+01	7.5E+01
Clay Springs – 1985	4.9E+02	2.0E+01	2.0E+01
Eight Mile Site No. 1 – 1980			
Side Slope	7.7E+02	3.4E+00	1.4E+01
Ridge Crest	4.1E+02	1.5E+01	8.6E+01
Eight Mile Site No. 1 – 1982			
Side Slope	5.6E+02	1.1E+00	2.4E+01
Ridge Crest	4.1E+02	3.8E+01	1.1E+02
Eight Mile Site No. 1 – 1985			
Side Slope	7.1E+02	0.0E+00	7.9E+00
Ridge Crest	5.5E+02	1.5E+01	5.8E+01
Eight Mile Site No. 2 – 1980			
Side Slope	6.4E+02	1.7E+01	4.5E+01
Ridge Crest	5.7E+02	1.6E+01	4.2E+01
Eight Mile Site No. 1 – 1981			
Side Slope	6.9E+02	4.7E+01	1.2E+02
Ridge Crest	7.7E+02	1.9E+01	1.2E+02
Eight Mile Site No. 1 – 1985			
Side Slope	7.6E+02	8.6E+01	1.5E+02
Ridge Crest	6.7E+02	2.2E+01	8.1E+01
Dwyer and Pieper, 1967			
Unburned Site – 1964	8.1E+02	7.1E+01	---[a]
Unburned Site – 1965	1.0E+03	2.4E+02	---[a]
Unburned Site – 1966	8.5E+02	1.1E+02	---[a]
Pieper, 1968			
Stony Hills	6.2E+02[b]	1.1E+01[b]	---[a]
Loamy Bottomland	6.2E+02[b]	6.0E+01[b]	---[a]
Loamy Upland	5.9E+02[b]	1.4E+02[b]	---[a]
Arnold et al., 1964	6.4E+02	5.4E+01	---[a]
Foxx et al., 1997			
Chupaderos Field	2.1E+02	4.9E+02	---[a]
Pumice Mine Field	4.5E+02	2.5E+02	---[a]
Tress and Klopatek, 1987			
7-year old burn	1.8E+02	5.2E+02	---[a]
35-year old burn	6.1E+02	8.7E+01	---[a]

a. Indicates no estimate of production was available for growth-form.

b. Production was presented in the report as total herbage (i.e., grass and forbs) production. These yields were divided between grasses and forbs based on percent cover.

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The estimates of shrub production included in Table 2-3 represent several years of accumulated growth, corresponding to the development of the supporting structures of the plants (e.g., woody tissues). Given this, it is unrealistic to take the approach adopted for grasses and forbs and equate litterfall with plant production. Lacking specific data on litter production rates in shrubs found in pinyon-juniper cover types, data from the ecological literature were used to estimate the fraction of the above-ground shrub biomass that may reasonably go into litter production. This fraction was multiplied by the median of the production data provided above to estimate annual litterfall for shrubs.

The proportion of above-ground shrub biomass that goes into annual litter production was estimated on the basis of biomass accumulation ratios. The biomass accumulation ratio is defined as the ratio of dry-weight biomass to annual net primary productivity. Whittaker (1975) provides ranges of the accumulation ratio (for above-ground parts of plants) for different terrestrial communities; normal ranges are from 2 to 10 in the desert, 1 to 3 in grasslands, 3 to 12 in shrublands, 10 to 30 in woodlands, and 20 to 50 in mature forests. If it is assumed that these ratios remain relatively constant after establishment of the plant community, the inverse of these ratios estimates the amount of the above-ground biomass that goes into annual litter production. Using this approach and the data provided by Whittaker for shrublands, it is estimated that 8 to 33 percent of the above-ground biomass of shrubs will enter into annual litter production. A bioaccumulation ratio of 7.5 was adopted for the baseline biotic intrusion modeling, the midpoint of the range provided by Whittaker. This ratio translates to a biomass contribution of 13 percent.

The rate at which litter decays to form soil will depend upon the climatic conditions under which decomposition occurs and the nature of the plant material. In general, rates of decomposition typically increase with increased temperature and moisture. For example, rates of decomposition tend to be much greater in warm, moist tropical forests than in a more temperate, arid environment such as that found at LANL. In terms of plant characteristics, wood is more resistant to decay than leaves, while evergreen leaves generally decompose more slowly than deciduous leaves. Given these relationships, it is reasonable to expect that litter from grasses and forbs will tend to decompose relatively quickly, while litter produced from shrubs and trees will take longer to decay.

Rates of litter decomposition are not widely available in the open literature. Whittaker (1975) discusses litter decay in terms of the decomposition half-life, or the time required for half of the original dry mass of litter to decompose. Half-lives for litter in boreal conifer forests, temperate deciduous forests, and grasslands were estimated at 7.0, 1.0, and 2.8 years, respectively. Millar (1974) cites work by others in which 10 or more years were required in order for *Pinus* needles to reach the humus layer of the soil.

Murphy et al. (1998) examined rates of litter decomposition at five sites along an elevational gradient in northern Arizona. These sites included Great Basin Desert scrub, pinyon-juniper woodland, ponderosa pine forest, and transition zones between these communities. Litter decomposition rates were measured for leaf litter collected from ponderosa pine, pinyon pine, one-seed juniper, and blue grama grass; and for leaves and stems from snakeweed. Based on these measurements, the investigators calculated annual decomposition rate constants for each species, separate constants were estimated for the first and second years of the two-year study.

The litter decomposition constants estimated by Murphy et al. (1998) for the five study sites are summarized in Table 2-4. Decomposition was greatest in the first year, as litter with the most favorable decay characteristics was consumed. Decay of more recalcitrant litter components proceeded at slower rates. If assumed to remain constant over time, the rate constants for the first year of the study infer a litter half-life of 1.7 years for snakeweed, 2 years for blue grama, and 3.3 years for pinyon and juniper trees (averaged over the two species). Similarly, if the rate constants for the second year were assumed to be constant, litter half-lives for the shrub, grass, and tree species are 2.8, 5.4, and 4.7 years, respectively. Even longer half-lives may be anticipated for the more substantial woody litter from mature shrubs and trees.

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Table 2-4. Litter decomposition rate constants.[a]

Plant Species	Decomposition Rate Constant (yr^{-1})	
	First Year	Second Year
Snakeweed	-4.2E-01	-2.5E-01
Blue Grama	-3.4E-01	-1.3E-01
One-Seed Juniper	-2.2E-01	-1.5E-01
Pinyon Pine	-2.0E-01	-1.5E-01
Ponderosa Pine	-2.2E-01	-1.3E-01

a. Source: Murphy et al. (1998).

Biotic intrusion modeling conducted for the MDA G performance assessment and composite analysis assumed that litter decomposed to form soil in the year in which it was produced. Given the decomposition rates listed in Table 2-4, this approach will overestimate the rate at which radionuclides build up in the surface soil. This level of conservatism will be most important for radionuclides that are strongly assimilated by plants and that have relatively short radiological half-lives. The radionuclides that were responsible for the peak composite analysis dose following biotic intrusion, however, have low plant uptake factors and generally have half-lives on the order of hundreds or thousands of years. Under these conditions, the effect of reduced rates of litter decomposition on the projected impacts of biotic intrusion is expected to be small. Consequently, this aspect of the modeling approach was not changed for the updated analysis.

Biomass estimates for grasses, forbs, and shrubs in the mature pinyon-juniper woodland were developed in a manner similar to that described above for the disturbed site. The data used to generate these estimates are summarized in Table 2-5, which includes a description of the sites that were investigated and a summary of the production and cover information that was taken from the reports. Some of these studies provide direct estimates of understory production in the mature woodland. Clary (1971) studied the effects of tree removal on herbage yields in pinyon-juniper woodlands in northern Arizona. Herbage yields of grasses, forbs, and shrubs averaged 71, 77, and 100 kg/ha/yr, respectively, in areas where no trees were removed. In other work, Clary (1989) reported yields ranging from 20 to 140 kg/ha for grasses, 10 to 45 kg/ha for forbs, and 30 to 130 kg/ha for shrubs in mature pinyon-juniper woodlands in central Utah. O'Rourke and Ogden (1969) evaluated

Table 2-5. Summary of cover and production studies used to estimate understory biomass for MDA G in its climax condition.

Community Characteristics	Reference	Cover				Production		
		Grasses	Forbs	Shrubs	Trees	Grasses	Forbs	Shrubs
Pinyon-juniper woodland in northern Arizona	Clary, 1971	---[a]	---[a]	---[a]	Crown cover on study sites ranged from 10 to 45 percent	Above-ground production averaged 71 kg/ha over a four-year period	Above-ground production averaged 77 kg/ha over a four-year period	Above-ground production averaged 100 kg/ha/ over a four-year period
Pinyon-juniper woodland in central Utah	Clary, 1989	---[a]	---[a]	---[a]	---[a]	Above-ground production at two sites ranged from 20 to 140 kg/ha; average production of 70 kg/ha	Above-ground production at two sites ranged from 10 to 45 kg/ha; average production of 26 kg/ha	Above-ground production at two sites ranged from 30 to 130 kg/ha; average production of 72 kg/ha
Pinyon-juniper woodland at four sites in north-central Arizona	O'Rourke and Ogden, 1969	Cover at four sites ranged from 8 to 22 percent; average cover of 16 percent	---[a]	---[a]	Crown cover at four sites ranged from 13 to 44 percent	Above-ground production at four sites ranged from 60 to 230 kg/ha; average production of 130 kg/ha	---[a]	---[a]
Pinyon-juniper woodland communities in northern Arizona	Arnold et al., 1964	---[a]	---[a]	---[a]	---[a]	Developed estimates of total herbage production as a function of tree cover, shown in Figure 2-5		

Table 2-5. Continued.

Community Characteristics	Reference	Cover				Production		
		Grasses	Forbs	Shrubs	Trees	Grasses	Forbs	Shrubs
Pinyon-juniper, woodland in north-central Arizona	Tress and Klopatek, 1987	Cover in mature woodland was 8.4 percent	Cover in mature woodland was 0.4 percent	Cover in mature woodland was 0.4 percent	Crown cover in mature woodland was 35 percent	---[c]	---[c]	---[c]
Pinyon-juniper woodland communities in northern Arizona	Arnold et al., 1964	Cover ranged from 1.4 to 1.7 percent among three sites; average cover of 1.6 percent	Cover ranged from 0.2 to 0.4 percent among three sites; average cover of 0.3 percent	Cover ranged from 1.9 to 2.8 percent among three sites; average cover of 2.2 percent	Crown cover at three sites ranged from 35 to 55 percent	---[c]	---[c]	---[c]
Pinyon-juniper woodland at 10 sites in central Nevada	Everett and Koniak, 1981	Cover at 10 sites ranged from 0 to 4.1 percent; average cover of 1.3 percent	Cover at 10 sites ranged from 0.0 to 7.2 percent; average cover of 1.4 percent	Cover at 10 sites ranged from 0 to 4.3 percent; average cover of 0.7 percent	Crown cover at 10 sites ranged from 26 to 63 percent	---[c]	---[c]	---[c]
Pinyon-juniper woodland in western Colorado	Haufler and Nagy, 1984	Cover in mature woodland was 1.2 percent	Cover in mature woodland was 0.94 percent	Cover in mature woodland was 4.3 percent	Crown cover in mature woodland was 38 percent	---[c]	---[c]	---[c]

a. Cover measurements were not performed.
b. Growth-form was not considered in the study.
c. Production measurements were not performed.

the effects of pinyon-juniper control on herbage production at four woodland sites in north-central Arizona. The two-year study included measurements of grass cover and production in undisturbed woodlands. Cover ranged from 13 to 44 percent, while yields ranged from 53 to 210 kg/ha. As discussed earlier, Arnold et al. (1964) developed estimates of total herbage yields (i.e., grasses and forbs) for pinyon-juniper woodlands as a function of tree canopy intercept. Percent tree cover had a significant effect on above-ground herbage yields, as shown in Figure 2-5. Clary (1971) found a similar relationship between tree cover and understory production in his work in northern Arizona.

Mature pinyon-juniper woodlands have also been characterized in terms of cover by vegetative growth-form. Tress and Klopatek (1987) found total understory coverage of about 9.1 percent in a 300-year old woodland. Arnold et al. (1964) developed cover estimates for grasses, forbs, and shrubs for mature stands of pinyon-juniper with a range of tree canopy intercepts. Work conducted by Padien and Lajtha (1992) suggest that total canopy cover at MDA G may be on the order of 37 to 50 percent. The data from Arnold et al. for canopy intercepts in this range show grass cover ranging from 1.4 to 1.7 percent, forb cover ranging from 0.2 to 0.4 percent, and shrub cover ranging from 1.8 to 2.8 percent. Everett and Koniak (1981) observed an average total plant cover of about 3.5 percent in mature stands. Haufler and Nagy (1984) conducted small mammal studies in mature and disturbed pinyon-juniper woodlands. Cover on the undisturbed woodland consisted of 1.2 percent grasses, 0.94 percent forbs, 4.3 percent shrubs, and 38 percent trees

The data summarized in Table 2-5 and Figure 2-5 were used to generate estimates of above-ground production for grasses, forbs, and shrubs. Production data were adopted directly from the studies conducted by Clary (1971, 1989), and O'Rourke and Ogden (1969). Herbage yields were estimated for the studies conducted by Tress and Klopatek (1987), Arnold et al. (1964), and Everett and Koniak (1981) using the relationship between total herbage production and tree cover developed by Arnold et al. (1964). Total understory yields predicted by Figure 2-5 for the tree coverages indicated in these studies were

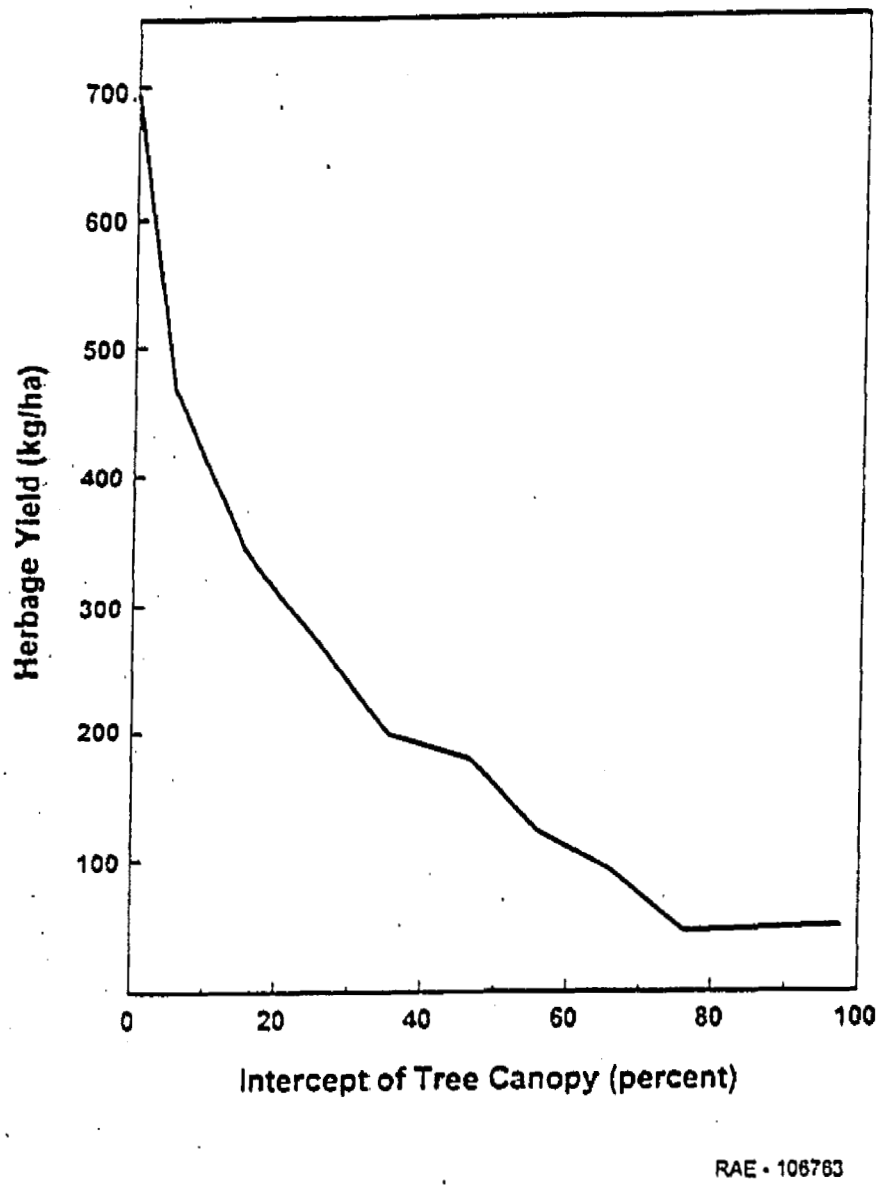

Figure 2-5. Total herbage yield as a function of tree canopy intercept (adapted from Arnold et al., 1964).

allocated among grasses and forbs based on the percent cover of each growth-form. The production estimates generated using these approaches are summarized in Table 2-6.

Grass and forb biomass estimates were set equal to the above-ground yields listed in Table 2-6. The median values of the production data were used for baseline modeling, yielding biomass estimates of 87 and 40 kg/ha for grasses and forbs, respectively. The litter generation rates for these growth-forms were set equal to the biomass estimates. All litter was conservatively assumed to decay in the year in which it was generated.

The litter production rate for shrubs was estimated using the method described earlier. Simply, the annual litter generation rate was assumed to be equal to the product of the inverse of the biomass accumulation ratio and above-ground shrub biomass. The midpoint of the range of accumulation ratios provided by Whittaker (1975) for shrublands and the median shrub production value listed in Table 2-6 were used for the baseline analysis. Shrub litter was assumed to decay in the year in which it was generated.

Estimation of the rate at which litter is generated by trees in the pinyon-juniper woodland presents unique challenges. Tree biomass in the woodlands will increase over very long periods of time as stands become established and grow at MDA G. This complicates the task of establishing representative data for the baseline modeling. While the rate of litter production will be affected by the magnitude of the above-ground biomass, the nature of this relationship is not entirely clear. Rates of litter production will tend to increase in early years in concert with rises in biomass, but may assume a variety of patterns during later stages of growth.

Litter production rates for trees in the mature woodland were estimated using limited biomass and litterfall data for pinyon-juniper woodlands in conjunction with general information about litter production in woodlands and forest communities. The data for the pinyon-juniper woodland were collected by Grier et al. (1992), who determined above-ground biomass and above-ground net productivity for 90- and 350-year old pinyon-juniper woodlands in northern Arizona. Tree biomass was estimated using regression

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Table 2-6. Aboveground production of grasses, forbs, and shrubs in mature pinyon-juniper woodlands.

Study	Production (kg/ha)		
	Grasses	Forbs	Shrubs
Clary, 1971	7.1E+01	7.7E+01	1.0E+02
Clary, 1989			
Church Hills Site - 1980	4.7E+01	4.5E+01	5.7E+01
Church Hills Site - 1982	1.4E+02	1.6E+01	1.3E+02
Church Hills Site - 1985	7.6E+01	3.5E+01	7.2E+01
Clay Springs Site - 1980	2.0E+01	1.0E+01	3.0E+01
O'Rourke and Ogden, 1969			
Boundary Site – 1961	2.3E+02	---[a]	---[a]
Boundary Site – 1962	1.5E+02	---[a]	---[a]
Ryan Site – 1961	8.6E+01	---[a]	---[a]
Ryan Site – 1962	1.9E+02	---[a]	---[a]
Second Site – 1961	9.3E+01	---[a]	---[a]
Second Site – 1962	5.9E+01	---[a]	---[a]
Chevelon Site – 1961	1.3E+02	---[a]	---[a]
Chevelon Site – 1962	7.5E+01	---[a]	---[a]
Tress and Klopatek, 1987	1.8E+02	8.7E+00	---[b]
Arnold et al., 1964	1.3E+02	2.4E+01	---[b]
Everett and Koniak, 1981			
Mt. Wilson Site	9.4E+00	1.0E+02	---[b]
Fredricks Site	0.0E+00	1.9E+02	---[b]
House Canyon Site	3.2E+01	2.4E+02	---[b]
Willow Creek Site	3.0E+01	1.5E+02	---[b]
Camel Springs Site	5.3E+01	4.8E+01	---[b]
Paperback Site	7.6E+01	3.6E+01	---[b]
Monitor Site	6.6E+01	2.4E+01	---[b]
Austin Site	1.3E+02	4.5E+01	---[b]
Ridge Site	1.2E+02	1.6E+01	---[b]
Lowry Springs Site	9.3E+01	6.4E+01	---[b]

a. No production measurements were conducted for the indicated growth-form.
b. Indicates no estimate of production was available for growth-form.

equations developed from destructive analysis of pinyon pine and juniper, taking into account the full diameter range of the trees found at the study sites. Net productivity was estimated as the sum of the annual biomass increment and litterfall. Litterfall was measured using litter traps over a three-year period.

The above-ground biomass and productivity estimates developed for trees by Grier et al. (1992) are summarized in Table 2-7. Total tree biomass of the mature woodland was

2.3 times greater than that estimated for the 90-year old stand. Living branches and bark accounted for approximately 50 percent of the total biomass in both communities. The canopy (i.e., foliage and foliage bearing twigs) was the next greatest contributor to biomass in the young stand, stems and the associated bark were more significant than the canopy in the mature woodland. Above-ground net primary productivity of trees in the young stand was 64 percent of that for the mature stand, litterfall for the young stand was 60 percent of that estimated for the mature woodland.

Table 2-7. Above-ground biomass and net primary productivity estimates for trees in pinyon-juniper woodlands.[a]

	Young Stand			Mature Stand		
Component	Pinyon Pine	Juniper	Total	Pinyon Pine	Juniper	Total
Biomass (Mg/ha)						
Canopy						
Foliage	2.7E+00	1.7E+00	4.4E+00	2.4E+00	3.3E+00	5.7E+00
Foliage-bearing twigs	1.6E+00	1.6E-01	1.7E+00	1.0E+00	4.1E-01	1.4E+00
Living Branches (incl. bark)	6.5E+00	4.2E+00	1.1E+01	1.3E+01	1.5E+01	2.9E+01
Dead Branches	6.9E-01	2.7E-01	9.6E-01	2.2E+00	1.4E+00	3.5E+00
Stem (wood and bark)	3.4E+00	1.5E+00	4.9E+00	7.2E+00	6.5E+00	1.4E+01
Total	1.5E+01	7.9E+00	2.3E+01	2.6E+01	2.7E+01	5.3E+01
Productivity (Mg/ha/yr)						
Biomass Increment						
Living wood	2.0E-01	5.0E-02	2.5E-01	3.2E-01	1.1E-01	4.3E-01
Attached dead material	2.0E-02	Trace	2.0E-02	7.0E-02	1.0E-02	8.0E-02
Bark	4.0E-02	1.0E-02	5.0E-02	6.0E-02	2.0E-02	8.0E-02
Foliage production	7.0E-01	4.5E-01	1.2E+00	9.0E-01	8.2E-01	1.7E+00
Total			1.5E+00			2.3E+00
Litterfall			3.0E-02			5.0E-02

a. Source: Grier et al. (1992).

The biomass estimates developed by Grier et al. (1992) fall at the low end of the range reported for pinyon-juniper woodlands in other regions. Meeuwig (1979), cited in Grier et al., reported above-ground biomass for Great Basin woodlands consisting of singleleaf pinyon (*Pinus monophylla*) and Utah juniper (*Juniperus osteosperma*) ranging from 60 to 121 Mg/ha. The oldest trees in these stands were about 320 and 260 years old, respectively. Grier et al. observed that the productivities they measured are also low compared to most other forests and woodlands in North America. Productivity data for other pinyon-juniper woodlands could not be found to make comparisons between similar ecosystems.

The litter production rates estimated by Grier et al. (1992) are approximately 0.1 percent of the above-ground biomass estimates for the two woodland communities. This relative rate of production cannot be compared to data for other pinyon-juniper woodlands because such data are unavailable. Nevertheless, it appears this production rate is significantly smaller than rates indicated for other forest communities. This conclusion is based on several lines of evidence.

Hinesley et al. (1991) measured litter production in four stands of oak-hickory-pine forests in northern Mississippi. Annual litter production ranged from 3 to 6.5 percent of the tree biomass, and was highest in the stands that were predominantly pine. Whittaker (1975) reported that litter production in a young mixed deciduous forest in the northeastern United States was approximately 3.7 percent of the above-ground biomass. Whittaker also reported a range of biomass accumulation ratios of 10 to 30 for above-ground parts of plants in woodlands. These ratios are largely determined by the presence of trees in the communities. If it is assumed that these ratios remain relatively constant over time, taking the inverse of the ratios yields estimates of the portion of the biomass that goes into litter production each year. Performing this calculation yields litter generation rates that are 3 to 10 percent of the above-ground biomass. Finally, Ricklefs (1979) reported that leaves comprise between 1 and 10 percent of the above-ground biomass of forests, evergreen forests tended to occupy the high end of this range. Grier et al. found that leaves were 11 and 19 percent of the total above-ground biomass in the two pinyon-juniper woodlands they studied. If it is assumed that leaves comprise 30 percent of the forest litter for gymnosperms (Williams and Gray, 1974) and that the lifespan of these leaves is 2 to 3 years, the data from Ricklefs and Grier et al. suggest annual litter production rates that are about 4 to 14 percent of the above-ground biomass.

Based on the preceding discussion, annual litter production in the coniferous forests appears to be on the order of 3 to 10 percent of the above-ground biomass. Lacking the data required to confirm the much lower rates of production observed by Grier et al. (1992) and wishing to provide reasonably conservative estimates of the impacts of plant intrusion upon MDA G, the midpoint of this range was selected for the baseline modeling. This relative rate of production was multiplied by the biomass estimate measured by Grier et al. for the

mature stand on pinyon-juniper to arrive at an above-ground litter production rate of 3.4 Mg/ha/yr.

A comprehensive list of LANL-specific, plant uptake factors does not exist. In their absence, the plant uptake factors developed by Baes et al. (1984) were adopted to model the uptake and assimilation of radionuclides from the disposed waste. That study does not contain data for either H-3 or C-14. It was assumed that H-3 is not extracted from soils by plants because the radionuclide sorbs minimally to the soil particles. The uptake factor used for C-14 was adopted from work performed by Shepherd et al. (1991). Plant uptake factors for a given element were assumed to be the same for grasses, forbs, shrubs, and trees.

The thickness of the cover placed over the disposed waste has varied over the operational lifetime of MDA G. The thickness of the cover placed over the earliest disposal pits is estimated to be about 1 m, while the earliest disposal shafts have approximately 1.4 m of cover. Additional cover material may have been placed over some of these units as waste storage structures were built on the ground surface. Starting in 1995, changes in operational procedures called for the placement of 2 m of cover over all disposal units at the time of closure. Depending upon the nature of the waste disposed in a given unit, this total cover thickness may be increased to as much as 3 m. For modeling purposes, cover depths ranging from 1 to 3 m were evaluated.

Animal Intrusion Data

The types of information required to evaluate the impacts of animal intrusion at MDA G are also the same for the early stages of succession and the climax condition. They include:

- Animal species present at the site.
- Animal burrow distributions with depth.
- Soil mass transfer rates for animals.

- Thickness of cover over disposed waste.

The wide range of plant communities at the Laboratory provides a diversity of habitats for wildlife species. The species of burrowing animals included in the updated biotic intrusion modeling were selected to address changes at the site as it underwent ecological succession from a disturbed state at the time of closure to its climax condition. A summary of the information upon which species selection was based is provided in the following paragraphs.

Harvester ants (*Pogonomyrmex* spp.) are found in a variety of grassland habitats throughout the western United States. The insects are a common inhabitant at MDA G, and have been observed in the disturbed portions of the site as well as in the mature pinyon-juniper woodland that surrounds the disposal area. Harvester ants have also been observed in pinyon-juniper and ponderosa pine communities near the LANL (Carlson and Whitford, 1991). Zarn (1977) noted that ants (Formicidae) were predominant among the invertebrates identified in pinyon-juniper woodlands in Arizona.

Small mammals have been trapped at two waste burial locations at MDA G and at control or background sites from 1994 through 1997 in order to evaluate radionuclide concentrations in tissues of the organisms (Biggs et al, 1995, 1997; Bennett et al., 1997, 1998). One of the burial locations (Site 5) was recently disturbed, with poorly established vegetation consisting of species typical of disturbed sites; the other burial location (Site 7) has been partially disturbed and has a mix of native plant species and species associated with disturbed areas. The background or control sites sampled throughout the period consist of undisturbed pinyon-juniper woodland.

The majority of the animals trapped at the two burial sites over the years have been deer mice. From 1994 through 1996, the deer mouse (*Peromyscus maniculatus*) was the only species caught at Site 5, two western harvest mice (*Reithrodontomys megalotis*) and a pocket gopher (*Thomomys* spp.) were caught there in 1997. In addition to deer mice, a single western harvest mouse was trapped at Site 7 in 1995, 1996, and 1997; four harvest mice and one brush mouse (*Peromyscus boylii*) were caught in 1997. Deer and pinyon mice

(*Peromyscus truei*) were the predominant species collected at the pinyon-juniper woodland sites. Other species collected include western harvest mice, brush mice, and silky pocket mice (*Perognathus flavus*). While the trapping efforts of Biggs et al. and Bennett et al. have indicated relatively little in the way of pocket gopher activity, the presence of these animals at MDA G has also been verified through visual observations. Gopher mounds have been found near several disposal shafts, and have been intercepted in the course of conducting vegetation transects at the site.

Additional information about the species of small mammals that are likely to inhabit pinyon-juniper woodlands can be found in the form of distribution maps developed for Los Alamos County. These maps show common inhabitants of the six major plant communities found at the Laboratory (DOE, 1979). Species shown for the pinyon-juniper woodlands include the deer mouse, pinyon mouse, Colorado chipmunk (*Eutamias quadrivittatus*), woodrat (*Neotoma* spp.), and mountain cottontail (*Sylvilagus nuttalli*).

Zarn (1977) summarized the work of several investigators who characterized the species of small mammals found in pinyon-juniper woodlands in Utah, western Colorado, and Arizona. Species found in those woodlands include deer mice, western harvest mice, brush mice, Great Basin pocket mice (*Perognathus parvus*), pinyon mice, northern grasshopper mice (*Onychomys leucogaster*), longtail voles (*Microtus longicaudus*), Mexican voles (*Microtus mexicanus*), sagebrush voles (*Lagurus curtatus*), Great Basin kangaroo rats (*Dipodomys microps*), least chipmunks (*Eutamias minimus*), rock squirrels (*Citellus variegatus*), and several species of woodrats and lagomorphs (*Lepus* spp., *Sylvilagus* spp.). In a food habits study conducted in western Colorado, Haufler and Nagy (1984) found deer mice, plains pocket mice (*Perognathus flavescens*), golden-mantled ground squirrels (*Citellus lateralis*), bushytail woodrats (*Neotoma cinerea*), least chipmunks, and mountain cottontails associated with disturbed and undisturbed pinyon-juniper woodlands.

Based on the information summarized above, four species of animals were identified for inclusion in the updated biotic intrusion modeling. These include the harvester ant, deer mouse, pocket gopher, and least or Colorado chipmunk. Other species observed in the mature pinyon-juniper woodland are expected to pose little or no threat in terms of

biointrusion. Voles tend to restrict burrowing activities to depths less than 1 m; woodrats generally live among rocks, cliffs and vegetation, and do not establish burrows for cover or foraging. While cottontail rabbits may establish burrows for cover, these excavations are not expected to extend to great depths.

The animal species included in the biotic intrusion modeling were evaluated in terms of their likelihood of occurrence at MDA G in its early successional stages and once it has reached a climax condition. The harvester ant was assumed to be present at MDA G during both periods. This is consistent with observations of the species in both habitats at the LANL and the occurrence of the insects in the pinyon-juniper woodland studied by Carlson and Whitford (1991).

The deer mouse was selected to represent the various species of mice (e.g., *Peromyscus* spp., *Perognathus* spp.) that may inhabit MDA G over time. The deer mouse is currently the most common mouse species at MDA G, and is expected to be a common inhabitant during early successional stages at the site. As suggested by the trapping data of Biggs et al. (1995, 1997) and Bennett et al. (1997, 1998), other species of mice (e.g., the pinyon mouse) may become more common as the site progresses towards pinyon-juniper woodland. However, rather than try to model the temporal dynamics of several species of mice, the deer mouse was used to represent the general mouse population. This approach is valid as long as the modeled burrowing characteristics of the deer mouse (i.e., burrow distribution with depth, burrow volume, and burrow density) provide reasonable estimates of the collective impacts of the mice species that are actually present at any given time.

Pocket gophers are expected to reside at MDA G as long as the site is in a relatively disturbed condition. Visual observations at the site indicate the species prefers disturbed areas, the animals are generally less common in the pinyon-juniper woodland. Where pocket gophers do occur in the woodland habitat, they tend to occur either in very limited areas or in disturbed areas such as those that occur along the sides of roads. Based on these observations and the faunal studies described above, the pocket gopher was not assumed to be present at MDA G when the site was in its climax condition. Chipmunks were assumed to be present only in the mature woodland. These animals have not been

observed or trapped near the disposal units at MDA G, but are common inhabitants in mature pinyon-juniper woodlands (see discussion above).

Animal burrow distributions with depth provide the basis for determining the proportion of burrows that penetrate the waste and the maximum depth of intrusion. However, relatively little information of this type exists. McKenzie et al. (1982) conducted a literature review of existing information for species of burrowing animals that may occur at low-level waste disposal sites. Some of that information addresses species included in the updated biotic intrusion modeling, and is summarized in Table 2-8.

Table 2-8. Burrow depth distributions for MDA G animal species.[a]

	Percent of Burrow Systems Within Indicated Depth Interval				
Animal Species	0 to 0.5 m	0.5 to 1.0 m	1.0 to 1.5 m	1.5 to 2.0 m	> 2.0 m
Harvester Ants	7.0E+01	1.0E+01	1.0E+01	5.0E+00	5.0E+00
Pocket Mice and Kangaroo Rats	5.0E+01	4.0E+01	5.0E+00	5.0E+00	0.0E+00
Pocket Gophers	8.5E+01	1.5E+01	0.0E+00	0.0E+00	0.0E+00
Ground Squirrels	5.0E+01	3.0E+01	1.5E+01	5.0E+00	0.0E+00

a. Source: McKenzie et al. (1982).

More recent evaluations of animal burrows add to the data assembled by McKenzie et al. (1982). In their work in southeastern Idaho, Reynolds and Laundre (1988) characterized the distribution of burrows for four species of rodents, including three relevant to the present study. Of the 43 deer mouse burrows examined, none extended deeper than 50 cm. Burrows of Ord's kangaroo rat (*Dipodomys ordi*) and Townsend's ground squirrels (*Spermophilus townsendii*) extended to depths of 90 and 140 cm, respectively. Williams and Cameron (1990) measured the depths and diameters of 399 burrow systems of Attwater's pocket gopher (*Geomys attwateri*) near Corpus Christi, Texas. Depths to the tops of feeding tunnels ranged from 18 to 21 cm. Kawamichi (1989) examined the burrow structure of Siberian chipmunks (*Eutamias sibiricus*) as a function of season. The deepest burrows were excavated when the animals were preparing to hibernate. These burrows averaged 79 to 83 cm in depth, and extended as deep as 107 cm. Sources cited in Gano and States (1982) indicate that the burrows of the eastern chipmunk (*Tamias*

striatus) and western species of chipmunks (*Eutamias* spp.) generally extend to depths of 30 to 40 cm.

Some observations exist on the depth of burrowing in pocket gophers that are specific to the Laboratory. Tierney and Foxx (1987) observed gopher burrows at depths of 95 to 110 cm during the excavation of plant roots. Gonzales et al. (1995) evaluated the impacts of pocket gopher burrowing on erosion using simulated waste covers. Pocket gopher activity extended to a depth of 1.5 m in that study. On the other hand, results from a study conducted by Hakonson et al. (1981) suggest that gophers had not penetrated into waste below a 1.25-m cover over a four-year period.

The burrow distribution data presented by McKenzie et al. (1982), shown in Table 2-8, were relied upon heavily to model the impacts of animal intrusion at MDA G. The burrow distribution shown for harvester ants was applied directly to the MDA G modeling. The burrowing data for pocket mice and kangaroo rats were assumed to represent the burrow distributions of the deer mouse and all other mice species present at MDA G. As indicated by the work of Reynolds and Laundre (1988), this distribution is expected to overestimate the depths to which deer mice, and possibly other species of mice, burrow. However, given the lack of site-specific burrowing data and the desire to represent several different species of mice with a single burrow distribution, these data were adopted for use in the biotic intrusion modeling.

The burrow distribution presented by McKenzie et al. (1982) for pocket gophers indicates no penetration to depths greater than 1 m. However, as discussed above, burrows have been observed at depths in excess of 1 m under natural and experimental conditions. Based on these observations, the burrow distribution presented by McKenzie et al. (1982) was modified for the MDA G biotic intrusion modeling. Specifically, the maximum depth of penetration was increased to 1.5 m and the fractions of the burrow systems falling within the 0 to 0.5 m, 0.5 to 1 m, and 1 to 1.5 m intervals were set equal to 80, 15, and 5 percent, respectively.

Comprehensive data sufficient to describe the burrowing distribution of chipmunks were not available in the open literature. Lacking this information, the burrow distribution data presented by McKenzie et al. (1982) for ground squirrels were used to represent the impacts of these animals. These data are expected to overestimate the depths to which these animals burrow, as suggested by the data collected by Kawamichi (1989) and the sources cited by Gano and States (1982).

Using the burrow data discussed above, least squares regression analyses were conducted to generate continuous burrow distributions as a function of depth. These functions were developed in order to evaluate the impacts of biotic intrusion for cover depths that were not even multiples of 0.5-m intervals. The burrow distribution curves generated by these analyses are shown in Figures 2-6 through 2-9.

The burrow depth distributions shown in Figures 2-6 through 2-9 show the fraction of burrows occurring within a given interval. The proportion of burrows that extends into the waste is calculated by subtracting the cumulative frequency shown for the cover thickness of interest from 1.0. For instance, the proportion of the burrows of harvester ant burrows that penetrate a 1-m cover is 1.0 minus the cumulative frequency predicted by the regression equation depicted in Figure 2-6. The predicted frequency is 0.83, indicating that 17 percent of the burrows of this species will penetrate into waste disposed beneath a 1-m cover. This approach was used to estimate the proportions of burrows penetrating into the waste for the four animal species over a range of cover depths. The results of these calculations are shown in Table 2-9. Maximum burrowing depths for the different species are also included in the table, and were taken from the burrow data discussed above.

The soil mass transfer rate for a species of burrowing animal, m_{aj} in Equation (4), is a function of the burrow density, the mass of material removed from each burrow, and the proportion of new burrows excavated each year. If it is assumed that each animal (or colony in the case of harvester ants) constructs a single burrow, the density of burrows at MDA G is equivalent to the animal (or colony) density. The product of the burrow density and the mass of the excavated soil yields the soil mass transfer rate for the year in which the burrow systems are established. Transfer rates for succeeding years may differ

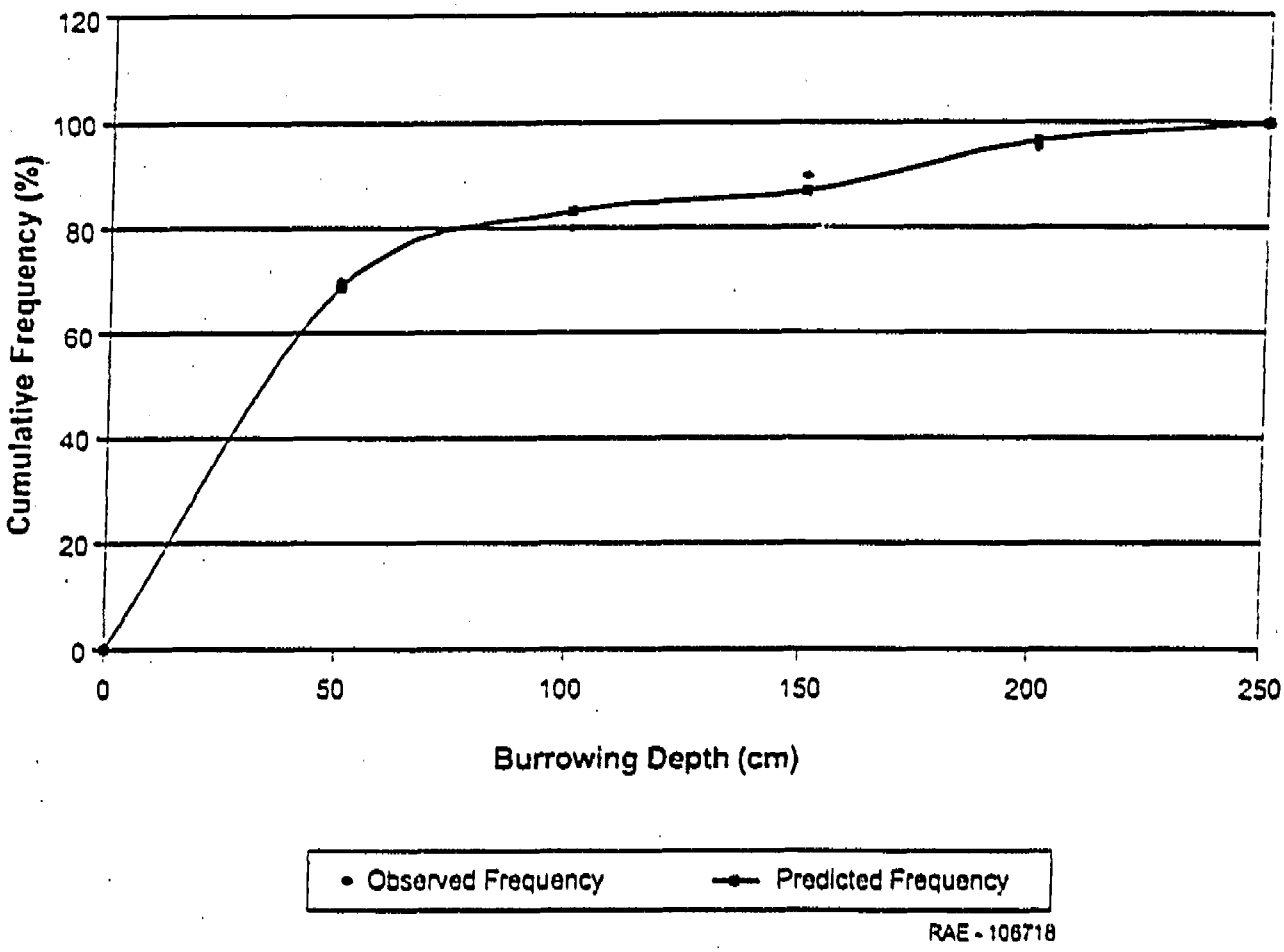

Figure 2-6. Burrowing depth distribution for harvester ants.

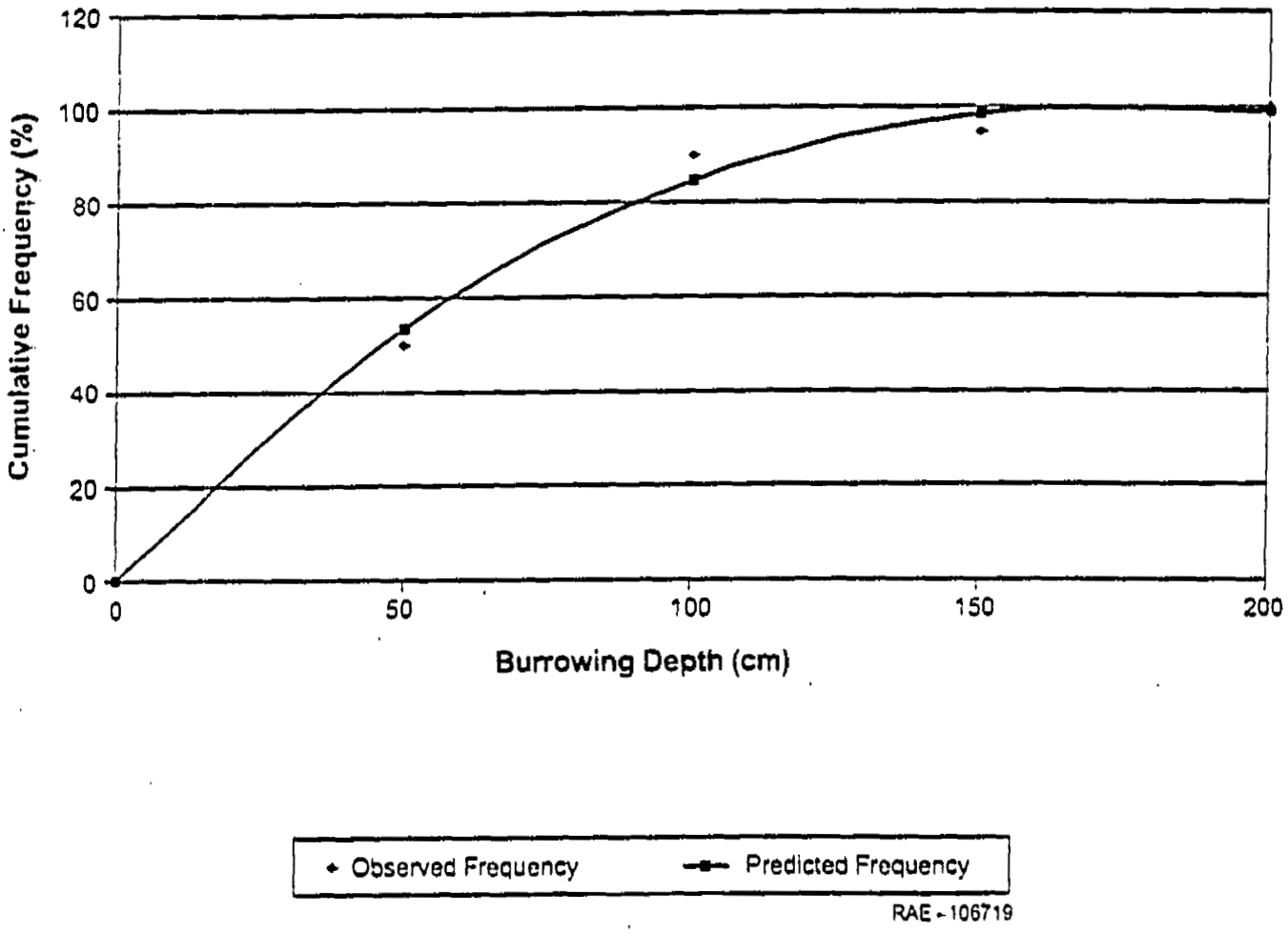

Figure 2-7. Burrowing depth distribution for mice.

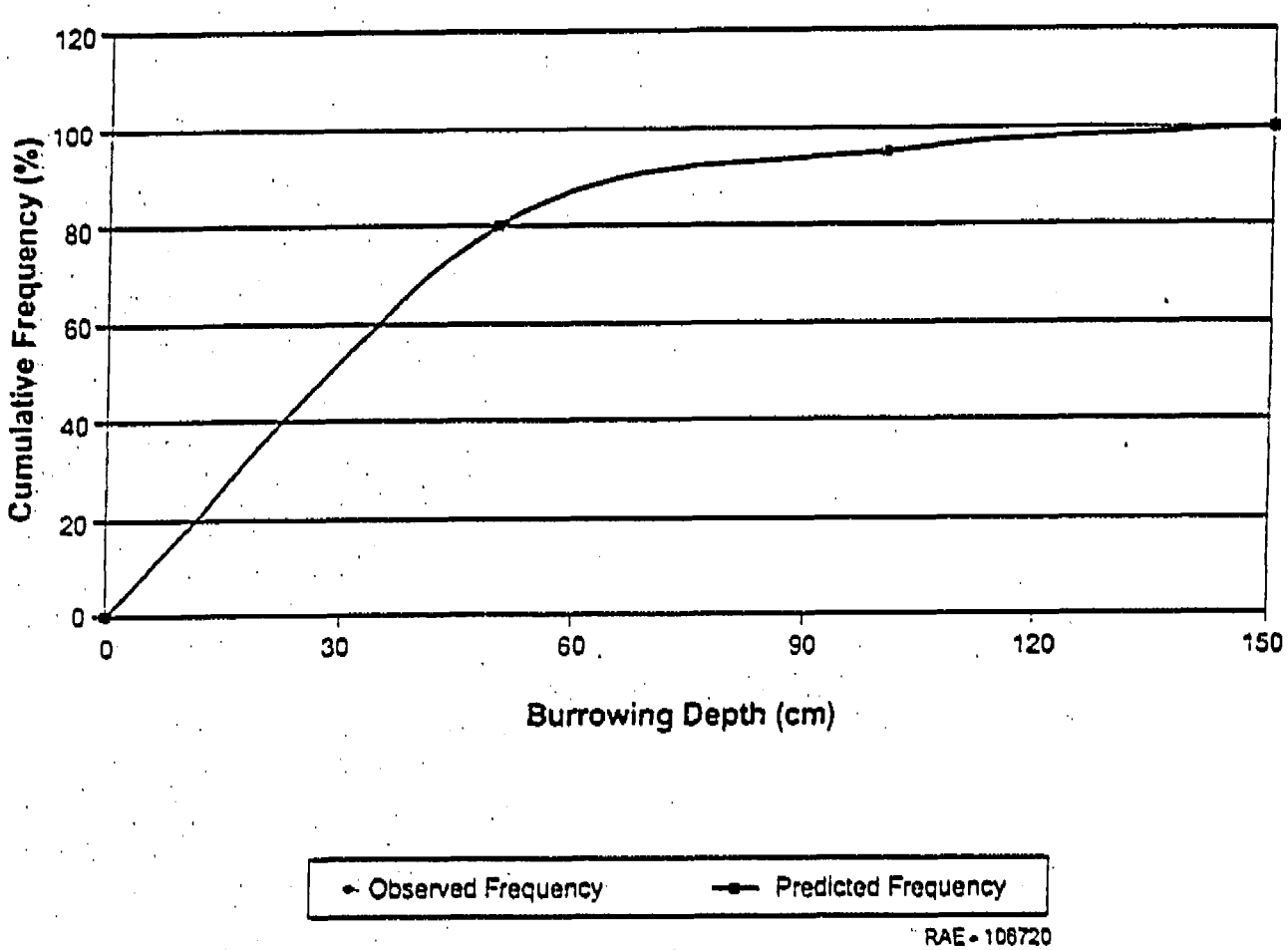

Figure 2-8. Burrowing depth distribution for pocket gophers.

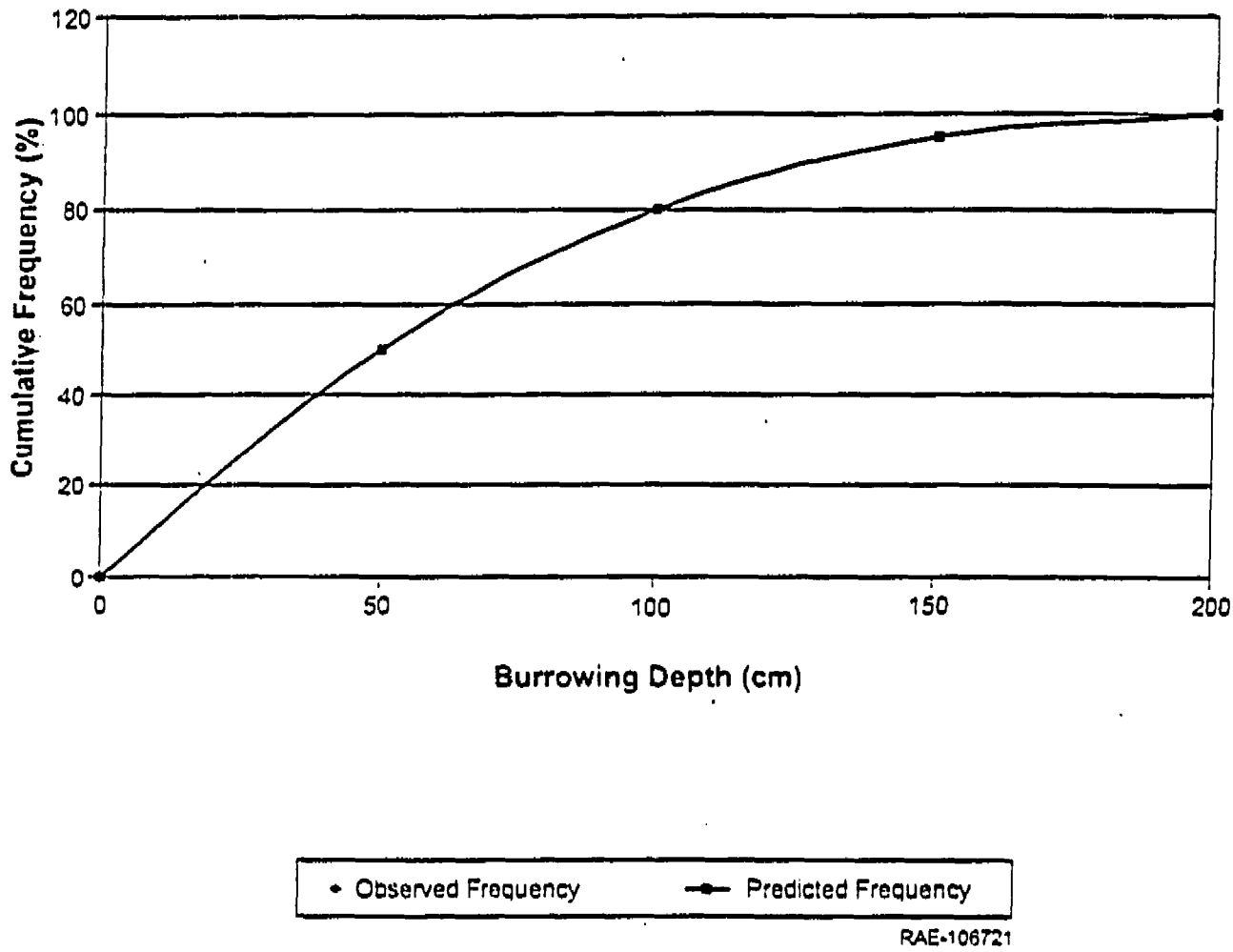

Figure 2-9. Burrowing depth distribution for chipmunks.

Table 2-9. **Proportions of animal burrows that penetrate into the disposed waste and maximum burrowing depths.**

Animal Species	Cover Thickness (m)			Maximum Burrowing Depth (m)
	1	2	3	
Harvester Ants	1.7E-01	3.5E-02	0.0E+00	2.5E+00
Deer Mice	1.5E-01	0.0E+00	0.0E+00	2.0E+00
Pocket Gophers	5.0E-02	0.0E+00	0.0E+00	1.5E+00
Chipmunks	2.0E-01	0.0E+00	0.0E+00	2.0E+00

depending upon whether animals use burrows for more than one year and whether animal (or colony) densities are changing at the site.

The baseline biotic intrusion modeling did not attempt to simulate animal population dynamics at MDA G as the site passed through the early stages of succession and into the climax condition. Rather, two simulations were used to represent what may be best described as steady-state representations of the site as it exists as grass- and shrub-land, and mature woodland. Consistent with this approach, no distinction was made between soil mass transfer rates for the first and subsequent years of the site simulations. Instead, the long-term burrow renewal fraction for each species was assumed to apply for the first and all succeeding years of the simulations.

The soil mass transfer rate calculated for harvester ants is 3.9×10^{-3} kg/m^2/yr. The burrow density was set equal to 31 burrows/ha, based on the median value of ant colony densities reported in the open literature (Blom et al., 1991; Keeler, 1993; Porter and Jorgensen, 1988; Soule and Knapp, 1996). Work conducted by Carlson and Whitford (1991) indicated that each colony of harvester ants inhabiting pinyon-juniper woodlands near Los Alamos brought an average of 38 kg of soil to the surface. This value was used as the basis for the baseline modeling. The proportion of burrows established each year will depend upon the life-span of the individual colonies and whether the number of colonies at MDA G is increasing, decreasing, or remaining constant over time. Data from the open literature indicate life-spans for harvester ant colonies ranging from 5 to 58 years under a variety of

conditions (Keeler, 1993; Porter and Jorgensen, 1988). Colonies evaluated for a number of studies survived for about 30 years, this life-span was adopted for the baseline modeling. If it is assumed that the colony population is constant, a life-span of 30 years translates to an average rate of colony replacement of 3.3 percent of the burrows per year.

The soil mass transfer rates for the species of small mammals included in the biotic intrusion modeling were developed using the information shown in Table 2-10. Soil mass transfer rates for mice were calculated using animal densities estimated by Biggs et al. (1995, 1997) and Bennett et al. (1997, 1998) and the average burrow volume calculated by McKenzie et al. (1982) for pocket mice and kangaroo rats. As discussed earlier, Biggs et al. and Bennett et al. trapped small mammals over closed MDA G disposal units and in undisturbed pinyon-juniper woodlands from 1994 through 1997. Estimates of total animal densities at these sites are summarized in Table 2-11. For the disposal site in the early stages of succession, the total density of mice was estimated as the average of the density estimates for the two waste burial sites. This yielded a density of 19.2 animals/ha. The density of mice in the mature pinyon-juniper woodland was set equal to the average density at the control and background sampling locations, or 5.2 animals/ha. The volume of soil excavated per burrow was defined using the average volume specified by McKenzie et al. for pocket mice and kangaroo rats, this volume was converted to a mass using an average soil density of 1,300 kg/m^3. The proportion of new burrow systems per year was set equal to 0.90, which is the approximate midpoint of the range estimated by McKenzie et al. (1982) for pocket mice and kangaroo rats.

The soil mass transfer rate for pocket gophers was based directly on the data provided by McKenzie et al. (1982), shown in Table 2-10. It is equal to the product of the average soil removal rate (8.3 m^3/ha) and the proportion of burrows replaced each year. A burrow renewal rate of 0.9 was used, the approximate midpoint of the range given in McKenzie et al. The amount of soil excavated per burrow by chipmunks was assumed to be 0.5 m^3/ha, the average value listed by McKenzie et al. (1982) for ground squirrels. The proportion of new burrows excavated each year was set equal to 0.75, the midpoint of the range provided by McKenzie et al. for ground squirrels. Transfer rates for pocket gophers and chipmunks were converted to a mass basis using a soil bulk density of 1,300 kg/m^3.

Table 2-10. Animal density and burrow data used to estimate soil mass transfer rates.

Animal Species	Animal Density (individuals/ha)		Burrow Volume (m^3)		Soil Mass Transfer Rate for First Year (m^3/ha)	Proportion of New Burrow Systems per Year
	Range	Average	Range	Average		
Deer Mice at Disturbed Site	1.0E+01 to 3.2E+01	1.9E+01	3.0E-03 to 1.0E-01	1.4E-02	2.7E-01	7.5E-01 to 1.0E+00
Mice in Mature Woodland	1.5E+00 to 8.7E+00	5.2E+00	3.0E-03 to 1.0E-01	1.4E-02	7.3E-02	7.5E-01 to 1.0E+00
Pocket Gophers	2.0E+00 to 1.2E+02	---	5.1E-01 to 8.2E+01[a]	8.3E+00[a]	8.3E+00	7.5E-01 to 1.0E+00
Chipmunks	5.7E+00 to 7.4E+01	2.5E+01	8.0E-03 to 7.7E-02	2.0E-02	5.0E-01	5.0E-01 to 1.0E+00

a. Estimate of volume of soil excavated in m^3/ha.

Table 2-11. Small mammal densities at MDA G and background/control areas.[a]

Site[b]	Animal Density (individuals/ha)				Mean Density (individuals/ha)
	1994	1995	1996	1997	
Waste Burial Site (5)	2.4E+01	3.2E+01	2.3E+01	1.0E+01	2.2E+01
Waste Burial Site (7)	2.1E+01	1.4E+01	1.1E+01	1.9E+01[c]	1.6E+01
Control Site (8)	3.3E+00	---	1.5E+00	7.7E+00	4.2E+00
Background Site (9)	---	8.7E+00	---	4.9E+00	6.8E+00

a. Source: Biggs et al. (1995, 1997) and Bennett et al. (1997, 1998)
b. Values in parentheses indicate the site number, as assigned for the 1997 trapping. Characteristics of these sites are provided in the text.
c. Number of animals does not include one pocket gopher trapped at the site.

As discussed earlier, the thickness of the cover placed over the disposed waste at MDA G has varied over the lifetime of the facility. The amount of cover that has been placed over disposal units in the past has ranged from 1 to 2 m. Current operational plans call for all units to be covered with a minimum of 2 m of clean material. However, this total cover thickness may be increased to as much as 3 m if the need arises. Given this, the baseline modeling evaluated the impacts of animal intrusion for cover depths ranging from 1 to 3 m.

2.3.2 Uncertainty Analysis Data Selection

The uncertainty analysis conducted in support of the biotic intrusion modeling for the MDA G performance assessment and composite analysis examined the impacts of a greater proportion of plant roots penetrating the disposed waste and intrusion into the waste by pocket gophers. As discussed earlier, these uncertainties and those associated with other aspects of the atmospheric pathway led to the conclusion that the performance objectives for the composite analysis may be exceeded under worst-case conditions. Given the unacceptable nature of this conclusion, a more extensive analysis of the uncertainties associated with the biotic intrusion source release model was undertaken as part of the updated modeling. The results of this analysis, in conjunction with updated analyses on other aspects of the atmospheric pathway, will permit a more rigorous evaluation of the disposal facility's ability to satisfy the performance objective for this exposure scenario.

A probabilistic approach was taken to evaluate the impacts of parameter uncertainties on the projected rates of radionuclide release following biotic intrusion. Adoption of this approach required the estimation of parameter distributions for the variables that enter into the plant and animal intrusion models shown in Section 2.2. These include the fractions of plant roots and animal burrows that penetrate into the disposed waste, plant radionuclide uptake factors, litter production rates, and mass transfer coefficients for burrowing animals. The waste cover thickness and soil bulk density are also input parameters for the biotic intrusion models, but were assumed to remain constant for the probabilistic uncertainty analysis.

The proportion of plant roots that penetrate the waste, f_{nj} in Equations (1) and (2), is a function of the rooting characteristics of the vegetation and the thickness of the cover. As discussed earlier, regression analyses were conducted to estimate composite rooting depth distributions for grasses, forbs, shrubs, and trees. The variability in the genera and species-specific data used to generate these distributions was used to estimate distributions of the proportion of roots penetrating the waste for each vegetation type.

The process used to estimate distributions for f_{pj} may be illustrated with the data used to develop the composite rooting distribution for grasses. These data, and the regression curve developed to represent them, are shown in Figure 2-1. The distribution for f_{pj} for a given cover thickness was estimated by the distribution of genera or species-specific root frequencies at the two depths bracketing that cover thickness. For example, the distribution of rooting frequencies at 91 and 122 cm were used to estimate the variability in f_{pj} for a cover thickness of 1 m. The minimum and maximum cumulative rooting frequencies at these depths were identified and used to estimate the minimum and maximum proportions of roots that penetrate into the waste. The parameter f_{pj} was assumed to have a triangular distribution with endpoints defined by these minimum and maximum proportions. The fraction of roots projected to penetrate the waste by the regression equation was assumed to be the most likely value. This process was repeated to estimate distributions of f_{pj} for each plant growth-form for cover depths of 1, 2, and 3 m.

Variability in litter production and decay rates will lead to uncertainties in projections of the rate at which contamination assimilated by plants enters into the surface soil. The baseline modeling assumed that the amount of grass and forb litter decaying to form soil each year was equivalent to the measured or estimated yields for these growth-forms. Consistent with this approach, distributions for litter production rates at the recently closed and climax sites were defined using the production data presented in Tables 2-3 and 2-6. For the uncertainty analysis, distributions of these production data were developed and used to represent distributions for litter production. Yields of grasses and forbs were assumed to have triangular distributions with most likely values set equal to the median yields used in the baseline modeling. The minimum and maximum values were defined based on the range of the data for each growth-form. All litter was assumed to decompose to form soil in the year in which it was generated.

As discussed earlier, shrub litter production rates for the early successional and climax stages of MDA G were estimated as the product of the inverse of the biomass accumulation ratio for shrublands and the production values for the two communities. This same approach was used for the uncertainty analysis. The biomass accumulation ratios were assumed to have a triangular distribution. The endpoints of the distribution were

defined using the range of biomass accumulation ratios provided by Whittaker (1975) for shrublands, while the most likely value was set equal to the mid-point of this range. Shrub biomass was also assumed to have a triangular distribution. The minimum, most likely, and maximum values were defined using the minimum, median, and maximum production values listed in Tables 2-3 and 2-6.

Few data exist to characterize the quantity of litter generated annually by trees in mature pinyon-juniper woodlands, making the development of a distribution for this parameter particularly difficult. As discussed earlier, annual litter production rates for coniferous trees appear to be on the order of a few percent of the above-ground biomass. For the uncertainty analysis, the relative rate of production was assumed to have a triangular distribution with minimum, most likely, and maximum values of 3, 6.5, and 10 percent. These proportions were multiplied by the above-ground biomass of the woodland to determine annual litter production rates. The distribution for above-ground biomass was estimated using the data of Grier et al. (1992) and Meeuwig (1979), discussed in Section 2.3.1. Biomass was also assumed to have a triangular distribution with a most likely value of 53 Mg/ha, the value used as the basis of the baseline analysis. The minimum and maximum values of the distribution were 22 and 121 Mg/ha.

The radionuclide plant uptake factors characterize the extent to which contaminants in the waste are taken up by the roots and assimilated by the vegetation. Baes et al. (1984) conducted an extensive literature review of terrestrial transport and agricultural parameters to estimate the uptake factors used in the March 1997 performance assessment and composite analysis. In general, plant uptake factors were found by these investigators to be lognormally distributed. For those elements for which sufficient data existed, geometric means and standard deviations of the plant uptake factors were provided. In most cases, the geometric means of the distributions were adopted as the best estimate of plant uptake factors for leafy vegetation.

The data developed by Baes et al. (1984) were used as the basis for defining distributions for root uptake factors for the biotic intrusion modeling. While geometric means and standard deviations could be adopted directly from the report for many

elements, this information did not exist for others. Included among the elements for which this information was absent were those that made significant contributions to the atmospheric pathway exposures projected for human receptors following biotic intrusion. Lacking the desired data, the root uptake factors for all elements were assumed to be lognormally distributed with a geometric mean equal to the factors used in the baseline modeling and a geometric standard deviation of 3. The standard deviation of 3 is the approximate average of the standard deviations reported by Baes et al. for the elements for which distributions could be estimated.

As discussed in Section 2.2.1, there is little in the way of detailed data describing the distribution of burrows with depth for burrowing species. Consequently, it was not possible to develop distributions for the proportions of animal burrows that penetrate into the disposed waste, parameter f_{s_j} in Equations (3) and (4). However, the impacts of uncertainties associated with the degree of intrusion into the disposed waste were evaluated in terms of the soil mass transfer rate.

The soil mass transfer rate for a given species of animal is a function of the burrow density, the mass of soil excavated per burrow, and the proportion of new burrows excavated each year. Distributions of these parameters were estimated for the harvester ant, and used to develop a distribution for the soil transfer rate. A distribution of harvester ant burrow, or colony, densities was developed using measured colony densities from the open literature (Blom et al., 1991; Keeler, 1993; Porter and Jorgensen, 1988; Soule and Knapp, 1996). A triangular distribution was used to represent these data, setting the most likely value to the density used in the baseline modeling and the minimum and maximum to the lowest and highest densities reported by these studies.

Limited information exists on the mass of soil excavated by each ant colony. Rogers and Lavigne (1974) estimated that an average of 2.8 kg of soil was excavated by harvester ant colonies in northeastern Colorado. Based on this soil volume, the maximum soil removal rate observed in four colonies was 87 kg/ha. These values are low relative to data collected by other investigators. Carlson and Whitford (1991) estimated that the average mass of ant mounds in pinyon-juniper woodlands near Los Alamos was 38 kg, while the

mass of mounds in a ponderosa pine community averaged 48 kg. The total soil removal rates in both communities were about 650 kg/ha. While per burrow mass data were not provided, Briese (1982; cited in Carlson and Whitford, 1991) reported that ants in a semiarid shrub steppe in Australia turned over 350 to 420 kg of soil annually. Whitford et al. (1986) found ants in eastern New Mexico moved an average of 840 kg/ha to the surface on a seasonal basis. While neither average mound masses nor colony densities were provided in the studies by Briese and Whitford et al., the total soil removal rates suggest that the average mass of soil excavated by each colony will be greater than that estimated by Rogers and Lavigne (1974). Based on this rationale, the mound mass per harvester ant colony was assumed to have a triangular distribution with minimum, most likely, and maximum values of 2.8, 38, and 48 kg.

The proportion of new burrows constructed by harvester ants each year was estimated based on the assumption that the density of colonies at MDA G is constant over time. Under these conditions, the proportion of new burrows per year is the inverse of the colony life-span. Life-spans were assigned a triangular distribution using data from the open literature (Keeler, 1993; Porter and Jorgensen, 1988). The 30-year life-span used in the baseline was the most likely value, while the minimum and maximum life-spans were 5 and 58 years.

The variability in soil mass transfer rates for species of mice was represented using animal densities estimated by Biggs et al. (1995, 1997) and Bennett et al. (1997, 1998), and burrow information from McKenzie et al. (1982). Mouse densities at MDA G when the site is in its early successional stages were assumed to have a triangular distribution with minimum and maximum values of 10 and 32 animals/ha. These endpoints represent the range in densities found at the two waste burial sites over the 1994 to 1997 trapping period (Table 2-11). The upper end of this range is in general agreement with information provided in Burt and Grossenheider (1980), which indicates that densities of 25 to 37 animals/ha in the summer months is high. The most likely animal density was assumed to be 19 individuals/ha, the average value used in the baseline modeling. A similar approach was used to generate a distribution of mouse densities for the mature pinyon-juniper woodland. Animal density was assumed to have a triangular distribution with endpoints

equal to 1.5 and 8.7 animals/ha, the range of densities estimated by Biggs et al. and Bennett et al. The most likely value was set equal to 5.2 animals/ha, the average density used in the baseline modeling. The volumes of the burrows constructed by all species of mice were assumed to have a triangular distribution, with endpoints defined by the range of burrow volumes reported by McKenzie et al. for pocket mice and kangaroo rats (Table 2-10). The average burrow volume for these species was used as the most likely value. Burrow volumes were converted to a mass basis using a soil bulk density of 1,300 kg/m^3. Finally, the proportion of new burrows constructed annually was assumed to range from 0.75 to 1.0, with a most likely value of 0.9.

The uncertainty associated with the soil mass transfer rate for pocket gophers was represented using the data presented in McKenzie et al. (1982), shown in Table 2-10. The amount of soil brought to the surface was described using a triangular distribution with minimum, most likely, and maximum values of 0.51, 8.3, and 82 m^3/ha, respectively. The proportion of new burrow systems constructed each year was also assumed to have a triangular distribution, with minimum, most likely, and maximum values of 0.75, 0.9, and 1.0. The data reported by McKenzie et al. for ground squirrels were used to develop distributions of animal densities and burrow volumes for chipmunks. Triangular distributions were assumed for both parameters, the endpoints of which were defined by the ranges provided in Table 2-10. The most likely values were set equal to the average values reported by McKenzie et al. (1982). The proportion of new burrow systems constructed each year was assumed to have a triangular distribution, with a minimum value of 0.5, a most likely value of 0.75, and a maximum value of 1.0. Mass transfer rates for the pocket gopher and chipmunk were converted to a mass basis using a soil bulk density of 1,300 kg/m^3.

3. BIOTIC INTRUSION MODELING RESULTS

The results of the updated biotic intrusion modeling for MDA G are presented below. Section 3.1 presents the baseline modeling results, and compares them to modeling results obtained using the data developed for the March 1997 draft of the MDA G performance assessment and composite analysis (LANL 1997). Results from the probabilistic uncertainty analysis and sensitivity analysis are discussed in Section 3.2. Some general conclusions about the impacts of biotic intrusion on the ability of MDA G to safely contain the waste are provided in Section 3.3.

3.1 BASELINE MODELING RESULTS

The baseline modeling considered the impacts of biotic intrusion upon MDA G under two sets of conditions. These conditions address the disposal units during early succession and following the establishment of pinyon-juniper woodland across Mesita del Buey. Projected impacts were evaluated for radionuclides with plant uptake factors ranging from 10^{-5} to 100, for a period of 10,000 years.

The normalized soil radionuclide concentrations projected for MDA G during early succession are shown in Figure 3-1 for disposal units with 1 m of cover. Depletion of soil contaminant concentrations due to radioactive decay is not accounted for in these results. Plant intrusion is the dominant mode of radionuclide transport for isotopes that have high plant uptake factors, these factors are denoted by the variable B_i in the legend of the figure. Soil concentrations projected at early times in the simulation decline about an order of magnitude as the plant uptake factor decreases from 100 to 10. Rates of decline slow with further decreases in plant uptake factor, as contamination excavated by burrowing animals plays a greater role in determining surface soil concentrations. Animal burrowing is the

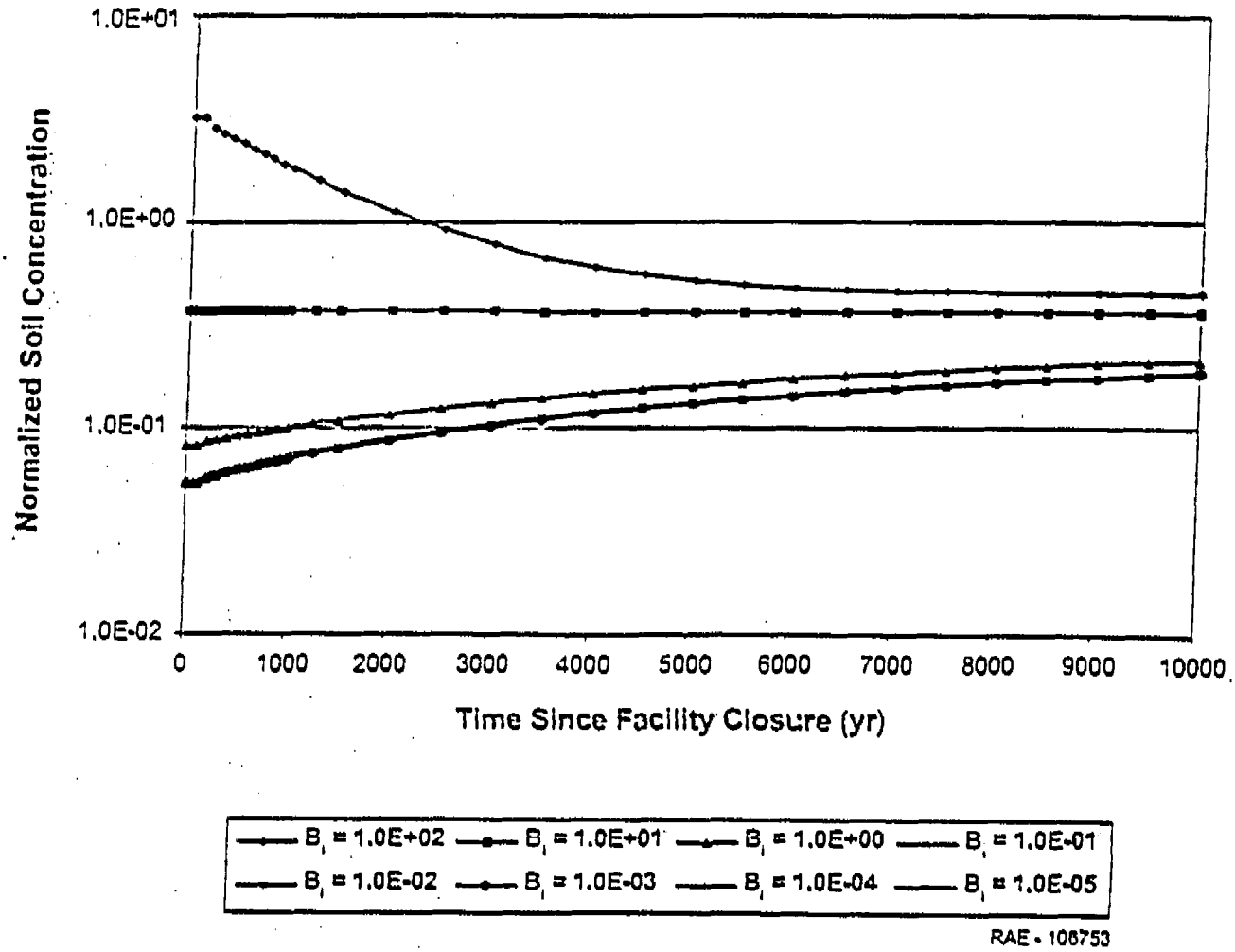

Figure 3-1. Projected soil concentrations for disposal units with 1 m of cover - early succession.

dominant mode of radionuclide transport for isotopes with plant uptake factors of 0.1 and less. The concentration curves for these isotopes overlap with one another because the rates of soil excavation are independent of the properties of the radionuclides.

Projected soil concentrations for radionuclides with very high plant uptake factors decrease over time as the waste inventory is depleted and as relatively clean soil brought to the surface by animals dilutes the contamination deposited by plants. In contrast, soil concentrations of radionuclides that are primarily transported to the ground surface by burrowing animals increase over the 10,000-year period, due to the slow rate of waste excavation. Concentrations of these radionuclides are diluted somewhat by relatively uncontaminated plant litter that falls and decays to form soil.

Radionuclide concentrations in surface soils decrease as the cover thickness over the disposed waste is increased. This is shown in Figures 3-2 and 3-3, which present normalized concentrations over disposal units with 2 and 3 m of cover, respectively. For the disposal units with 2 m of cover, soil concentrations of radionuclides with plant uptake factors greater than 10^{-3} are dominated by plant processes. Animal intrusion has an increasing role for radionuclides with plant uptake factors of 10^{-3} and less. Projected soil concentrations for disposal units with 3 m of cover are directly proportional to the plant uptake factors over the entire range of factors evaluated. This is due to the fact that animals are not projected to penetrate into the waste placed in these disposal units. For both sets of disposal units, concentrations of radionuclides with plant uptake factors of 100 decline slowly due to inventory depletion and dilution of contaminated litter with soil excavated by animals. Soil concentrations of radionuclides with plant uptake factors of 10 or less remain essentially constant over the simulation period.

The results shown in Figures 3-1 through 3-3 will tend to overestimate the actual impacts of biotic intrusion at the site for the first few years of the simulations. This is because the modeling assumes that the plant and animal communities are present at the

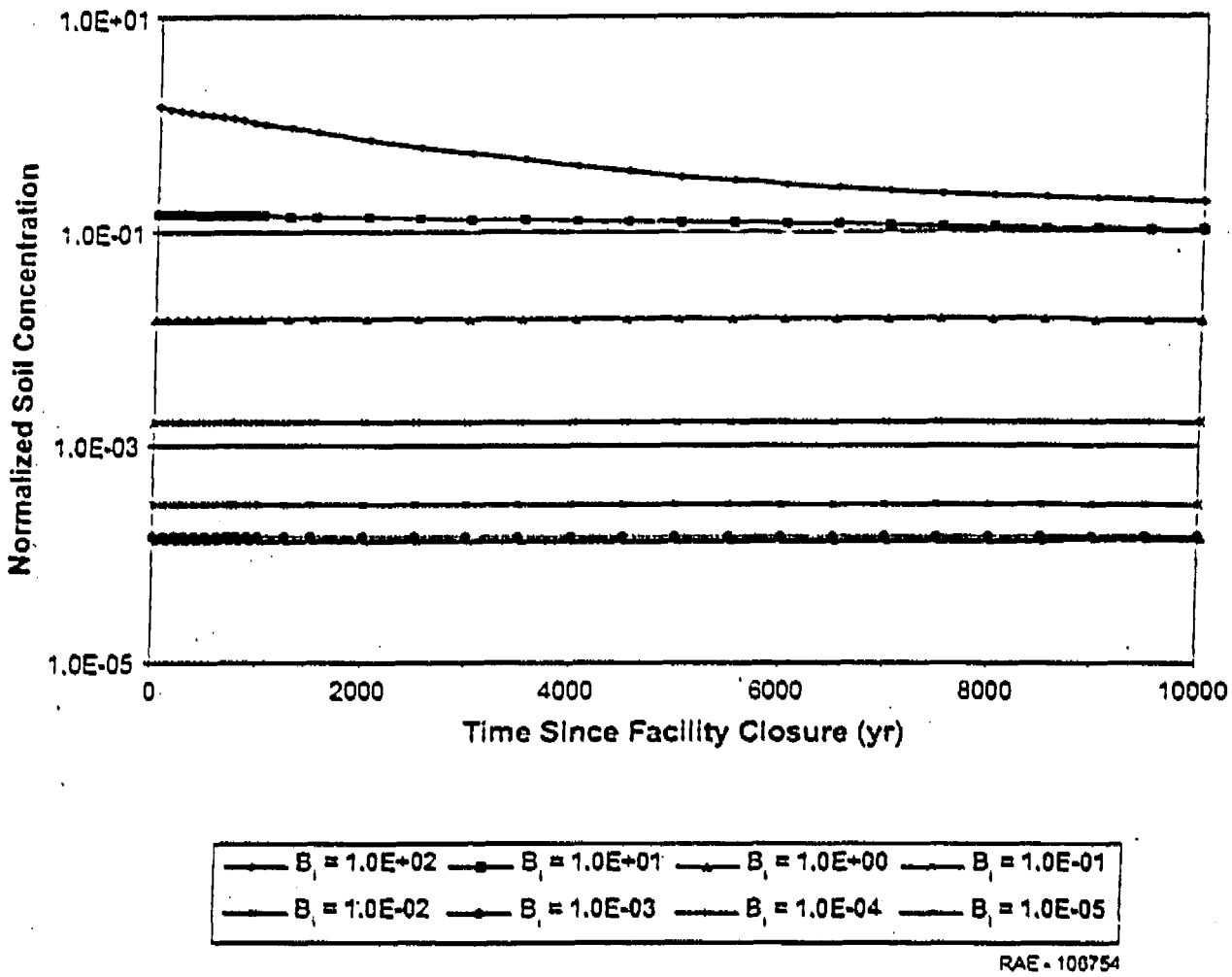

Figure 3-2. Projected soil concentrations for disposal units with 2 m of cover - early succession

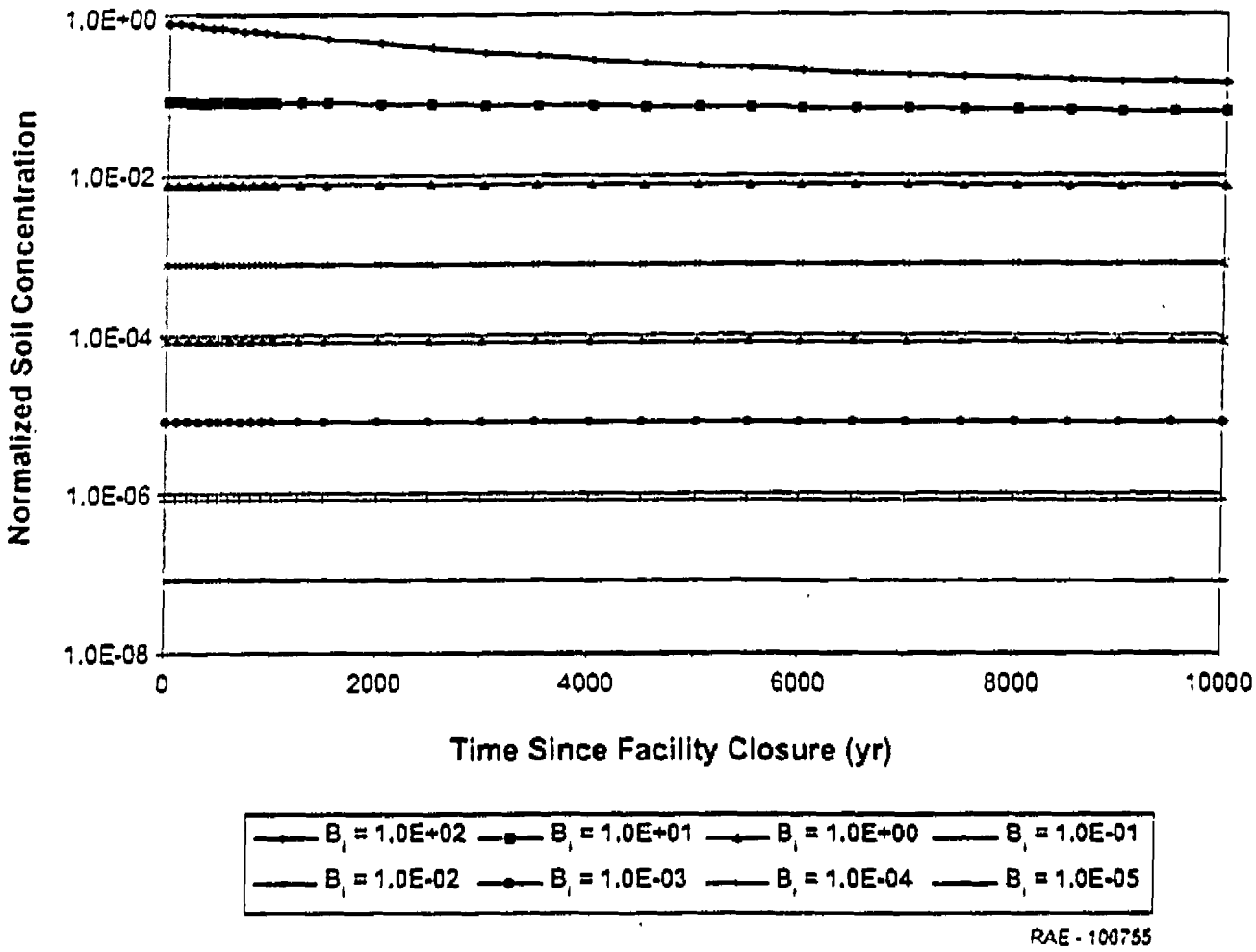

Figure 3-3. Projected soil concentrations for disposal units with 3 m of cover - early succession.

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indicated levels as soon as the facility closes. In fact, while annuals will invade the site in a rapid fashion, perennial grasses and forbs will become established over a period of a few years; shrubs will require even longer periods of time to become established at the levels estimated for the biotic intrusion modeling. Similarly, burrowing animals will gradually colonize the site and establish burrow systems over a period of time.

The projected soil concentrations for MDA G in its climax condition are shown in Figures 3-4 through 3-6. These concentrations do not account for the depletion of radionuclides by radioactive decay. The times shown in the figures refer to the amount of time since the start of the climax condition. As discussed earlier, the amount of time required for the site to approach or attain a climax condition is expected to be at least 100 years, and more likely on the order of 200 years or more. Consequently, the times shown in Figures 3-4 through 3-6 occur well after the end of facility operations.

For the disposal units with 1 m of cover, concentrations of radionuclides with high plant uptake factors are 12 to 14 times greater at the climax site than they are during early succession (Figures 3-1 and 3-4). This difference is the result of the presence of trees in the plant community, and the absence of pocket gophers in the woodland. While the degree to which the roots of trees penetrate the waste is comparable to that of the other growth-forms, the amount of litter generated by the trees is projected to be much larger than the amount produced by grasses, forbs, and shrubs combined. As a result, larger amounts of contamination will be incorporated into the soil as plant material decomposes. The effect of the pocket gopher is related to the degree to which contamination deposited on the ground in the form of litter is diluted in relatively uncontaminated soil. When present, the gopher excavates large quantities of soil that has radionuclide concentrations that are much lower than those found in the litter. Consequently, the soil effectively dilutes the contamination deposited as litter. When the species is absent, as it is assumed to be in the mature woodland, less soil is brought to the surface resulting in less dilution and higher soil concentrations.

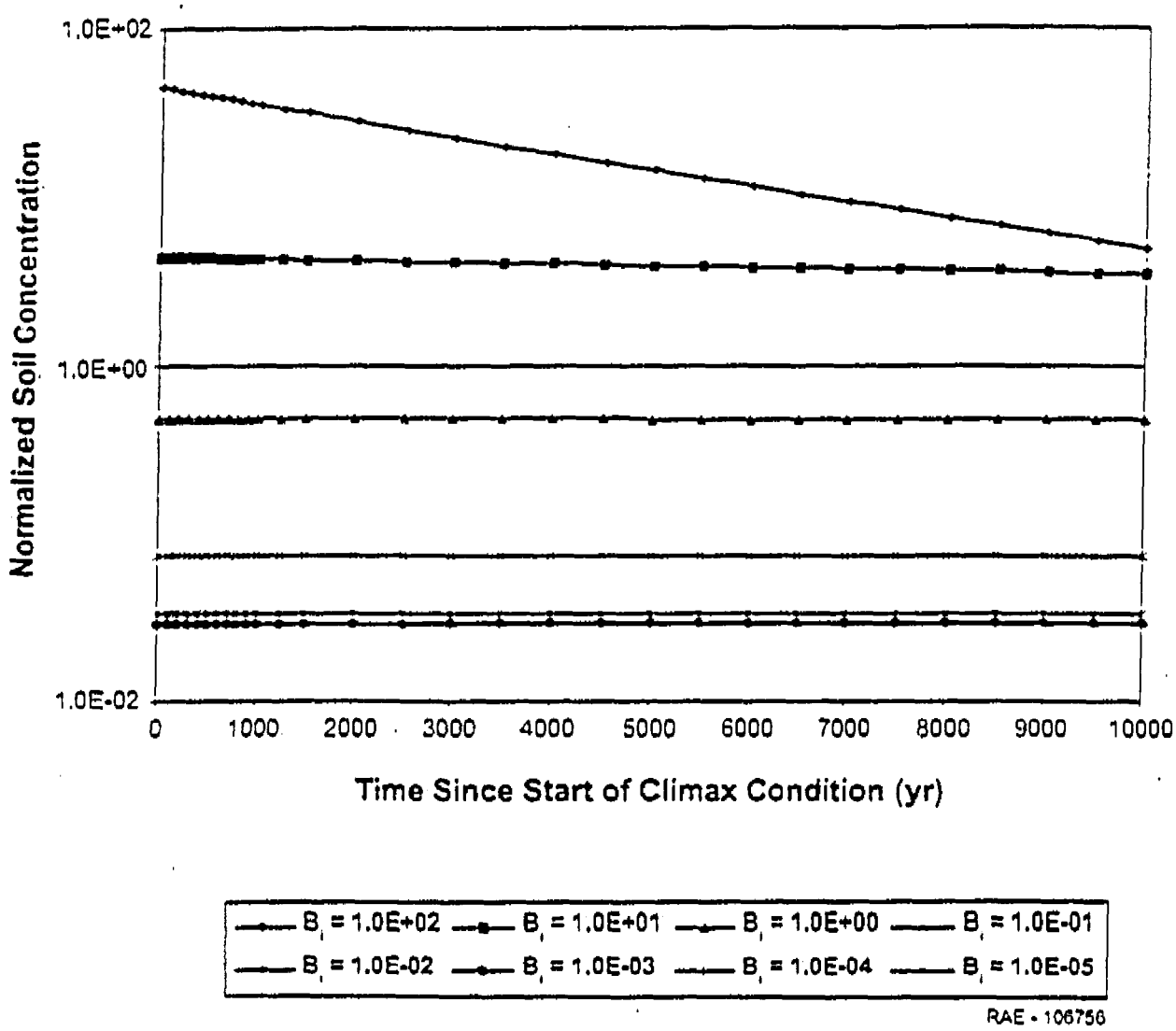

Figure 3-4. Projected soil concentrations for disposal units with 1 m of cover - climax condition.

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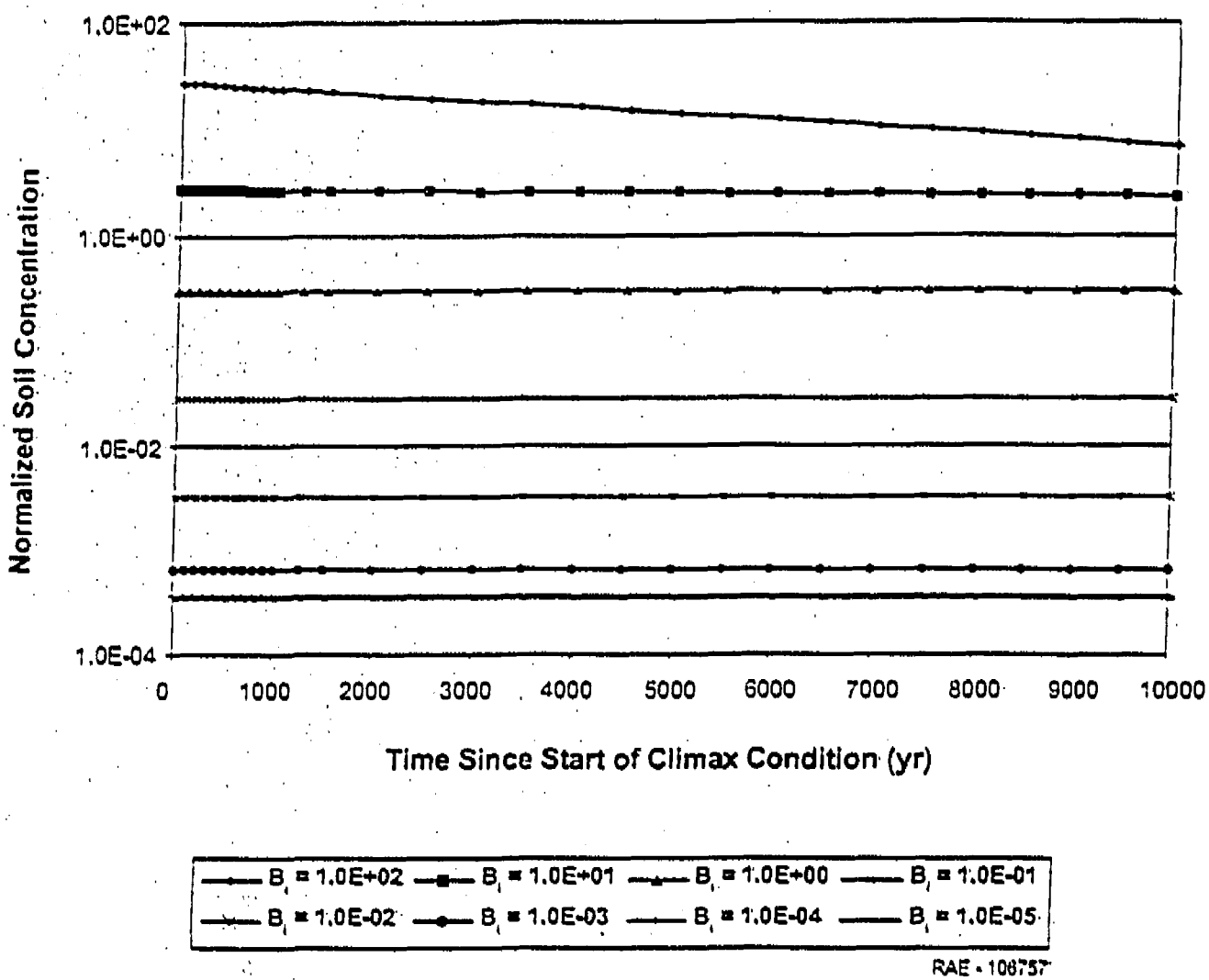

Figure 3-5. Projected soil concentrations for disposal units with 2 m of cover - climax condition.

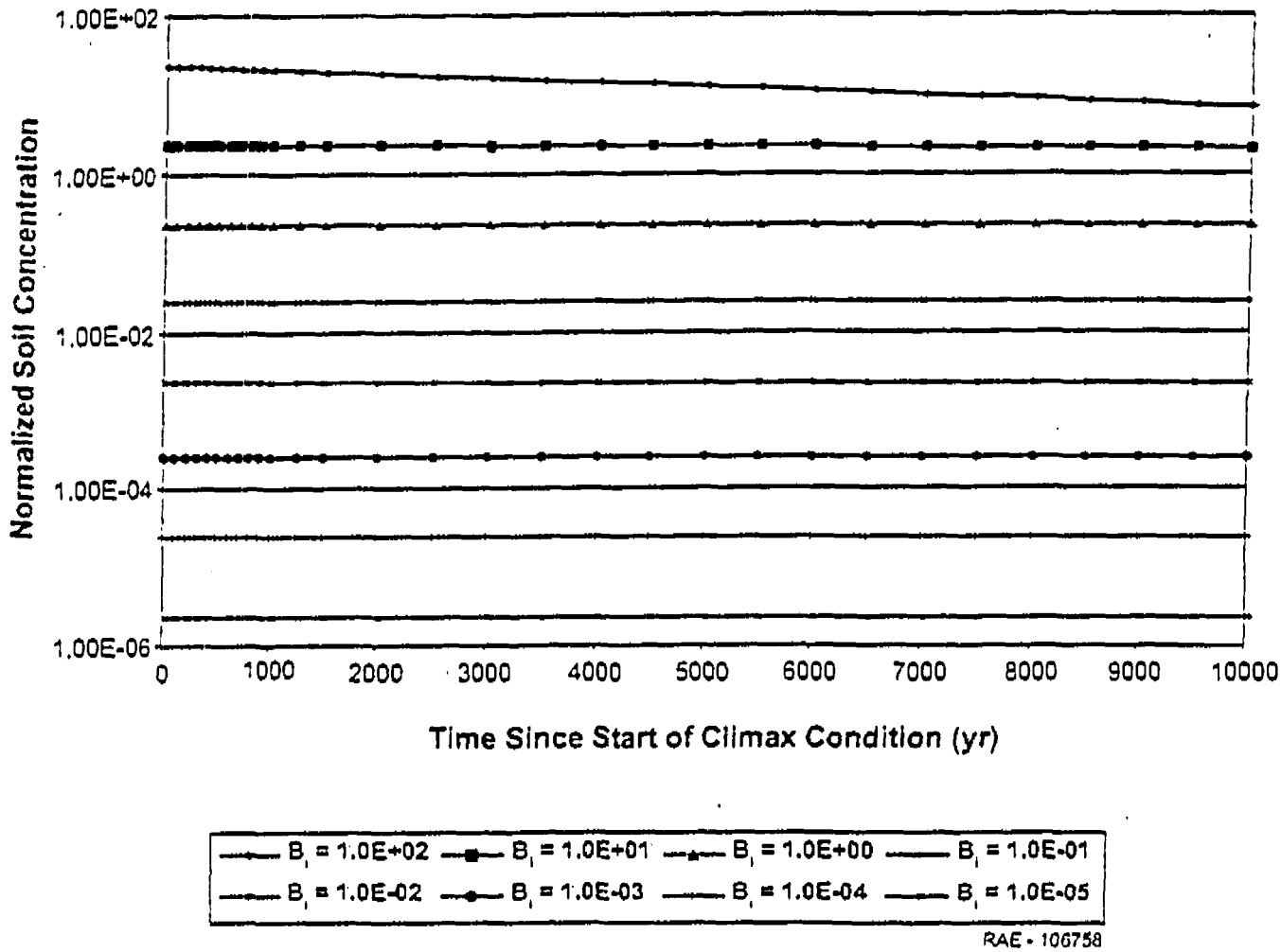

Figure 3-6. Projected soil concentrations for disposal units with 3 m of cover - climax condition.

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Projected soil concentrations of radionuclides with low plant uptake factors are 10 to 44 percent less in the climax condition than for disposal units with 1 m of cover during early succession. Less total contamination is brought to the surface by the animals in the woodland community. Furthermore, the large quantities of litter generated by trees in the woodland have low concentrations of these radionuclides. As such, the soil formed by the decomposition of litter dilutes the contamination brought to the surface by animals.

Large differences also exist between the soil concentrations projected for climax conditions and early succession for the disposal units with 2 and 3 m of cover. For the units with 2 m of cover, concentrations of radionuclides with high plant uptake factors under climax conditions are almost 20 times greater than the levels estimated for the early stages of succession (Figures 3-2 and 3-5). Unlike the situation noted above for disposal units with 1 m of cover, concentrations of radionuclides with low plant uptake factors are also greater than the concentrations projected for the early successional stages. In this case, the contamination that is deposited as tree litter plays an important role in the overall impacts of biotic intrusion even when the plants have only a limited ability to assimilate the radionuclides. Climax concentrations of all radionuclides over the disposal units with 3 m of cover are almost 30 times greater than the concentrations projected for the site during early succession (Figures 3-3 and 3-6). Plant uptake is the only way in which contamination may be brought to the surface over these units. The contamination deposited as litter undergoes less dilution in the climax woodland due largely to the absence of the pocket gopher.

As discussed earlier, it was assumed in the March 1997 draft of the MDA G performance assessment and composite analysis that maintenance of the closed disposal site prevented the establishment of deep-rooting plants and extensive burrowing by animals. The effect this assumption has upon the projected impacts of biotic intrusion may be determined by comparing the results from the updated biotic intrusion modeling with corresponding results from the earlier modeling. These comparisons are shown for disposal units with 1 m of cover in Figures 3-7 and 3-8. Figure 3-7 compares the results of the updated modeling for the site in its early successional stages with the results from the

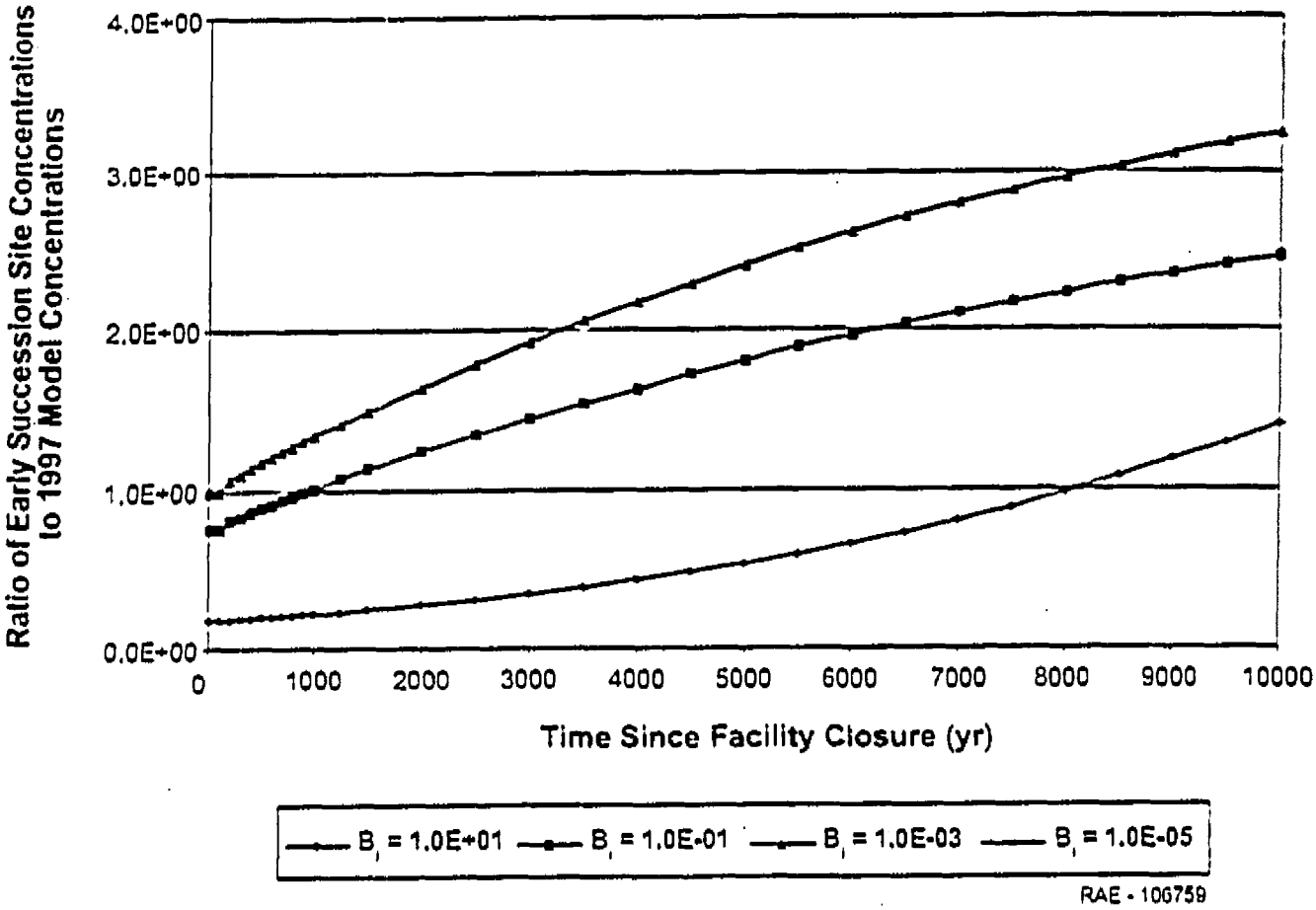

Figure 3-7. Comparison of early succession site concentrations to 1997 model concentrations.

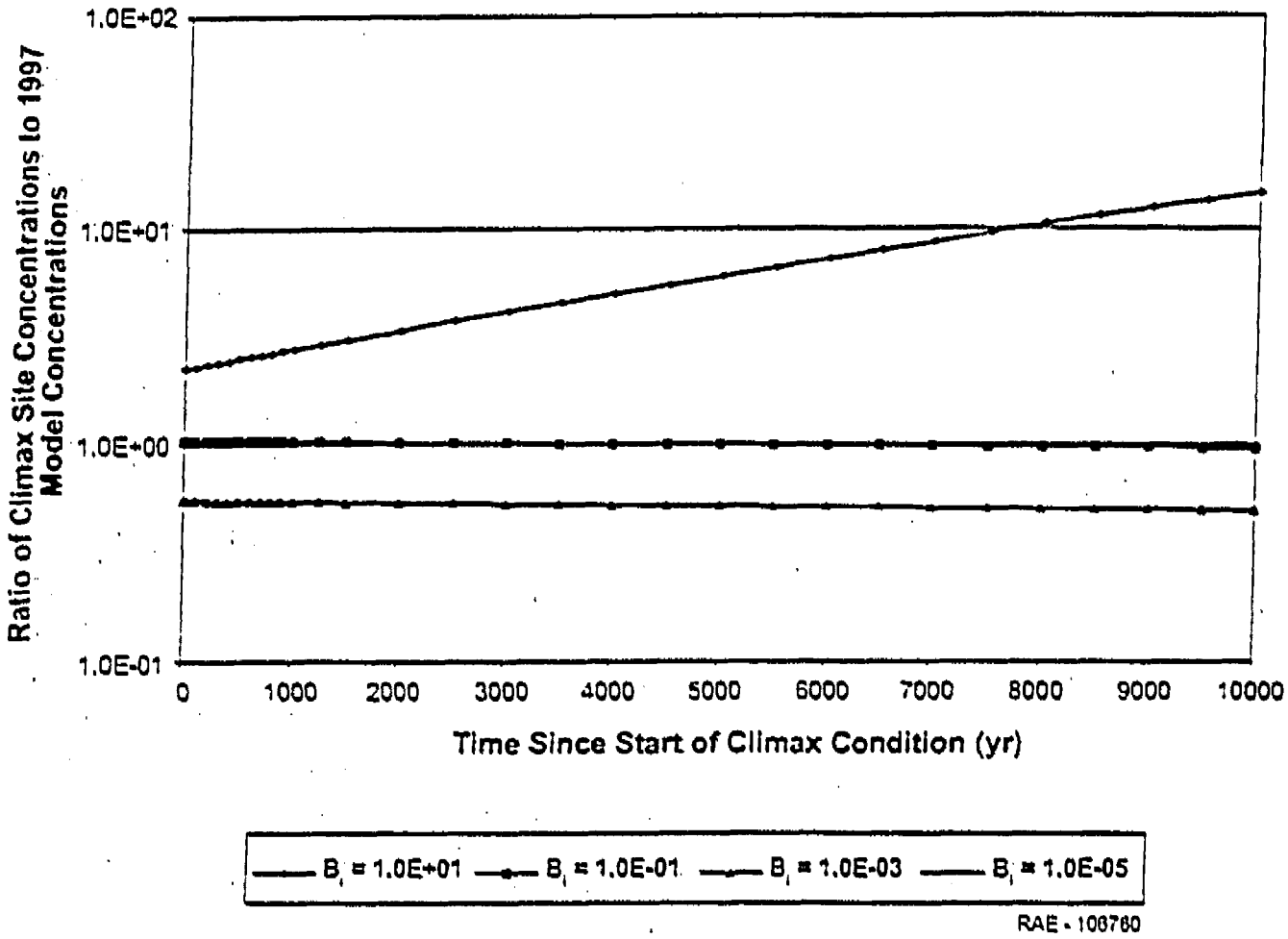

Figure 3-8. Comparison of climax site concentrations to 1997 model concentrations.

modeling used for the MDA G performance assessment and composite analysis. The results for the climax condition of the site are compared to the prior modeling results in Figure 3-8. All comparisons are shown as the ratio of the updated soil projections (i.e., for the site during early succession and in its climax condition) to the soil concentrations projected using the 1997 baseline model. These ratios are shown as a function of time and for a range of plant uptake factors.

Soil concentrations projected for radionuclides with high plant uptake factors during the early stages of succession are less than the concentrations projected using the MDA G performance assessment model for early simulation times (Figure 3-7). While the extent of plant intrusion is greater in the updated modeling, so too is the degree to which animals burrow into the site. The greater amounts of soil brought to the surface by animals in the updated modeling dilutes the additional contamination taken up by plants, yielding soil concentrations lower than those estimated for the performance assessment and composite analysis. Differences in litter production rates and plant rooting depth distributions cause the concentrations projected using the 1997 model to decrease more rapidly than seen for the updated modeling. As a result, the concentrations projected for the site during early succession eventually exceed those estimated in the 1997 modeling.

Concentrations of radionuclides with low plant uptake factors are similar between the site in early succession and the site as it was modeled in 1997 for early simulation times. As time passes, however, surface concentrations projected by the updated modeling mount because more extensive penetration of the waste by burrowing animals occurs. While the updated concentrations are 34 percent greater than those projected by the 1997 models at year 1,000, they are about 3.2 times higher at the end of 10,000 years.

Ratios of site soil concentrations under climax conditions and those projected for the MDA G performance assessment and composite analysis increase over time for radionuclides with very high plant uptake factors (Figure 3-8). Greater rates of litter production in the updated modeling and lower rates of dilution of surface contamination by soil brought to the surface by animals are primarily responsible for this pattern. Climax community concentrations of radionuclides with small plant uptake factors are projected to

be about 45 percent less than the levels estimated using the 1997 model. This is due to less intrusion into the waste by animals and greater dilution of surface contamination by relatively uncontaminated plant litter.

The comparison of projected soil concentrations shown in Figure 3-8 assumes that the start of the climax community simulation coincides with the beginning of the 1997 model output. As discussed earlier, this is not expected to be the case as the site is expected to require a few hundred years to reach or near climax. However, similar relationships are observed even if this temporal shift is taken into account.

As discussed earlier, surface erosion at the levels projected for the MDA G performance assessment and composite analysis has a negligible effect on the projected impacts of biotic intrusion, and was not included in the updated modeling. If an assumption of no erosion is applied to the 1997 modeling, no transport of contamination to the surface of MDA G is projected to occur for disposal units covered with 2 m or more of clean soil. Consequently, modeling results from the updated and 1997 approaches were not compared for cover depths of 2 and 3 m.

As discussed earlier, the impacts of biotic intrusion projected for MDA G in the 1997 draft of the composite analysis were greatest for the atmospheric exposure scenario. In this scenario, contamination deposited on the surface by plants and animals was transported to off-site receptor locations with the prevailing winds. Exposures were received through a variety of exposure pathways, including the inhalation of airborne particulates, ingestion of contaminated soil, ingestion of food crops grown in contaminated soils and contaminated by atmospheric deposition, and direct radiation from contaminated soils and airborne radionuclides.

Ultimately, the impacts of biotic intrusion on the ability of MDA G to safely contain the disposed waste will be judged in terms of human exposures. Consequently, the atmospheric scenario used to project doses for the MDA G composite analysis was implemented using the results from the updated biotic intrusion modeling. This modeling used the existing composite analysis inventory, which includes all radioactive waste

disposed of, and projected to require disposal, at MDA G. The updated exposure modeling considered only the receptor location in Canada del Buey, the point of maximum exposure for the scenario. Depletion of the radionuclide inventories due to radioactive decay was taken into account in these simulations.

The atmospheric scenario dose projections for MDA G in its early successional stages are compared to the 1997 composite analysis projections in Figure 3-9. Doses for the updated modeling are shown starting at the end of facility operations, while the prior modeling results begin 100 years after facility closure. This is consistent with the assumptions upon which the two analyses were based. The updated biotic intrusion modeling assumed that no actions were taken to limit biotic intrusion after site closure, while the composite analysis assumed that intrusion into the waste by plants and animals was prevented during the 100-year active institutional control period.

The peak dose projected using the updated biotic intrusion modeling is 8.4 mrem/yr and occurs at the end of facility closure. This dose is 44 percent greater than the peak dose projected for the composite analysis at the end of active institutional control. The inhalation of airborne contamination is responsible for more than 99 percent of the peak dose projected using the updated biotic intrusion modeling. Isotopes of plutonium and americium are the dominant radionuclides, accounting for 98 percent of the peak dose. The waste disposed of in pits prior to 1971 is the main contributor to the projected doses, accounting for 87 percent of the peak exposure. Pathway, radionuclide, and inventory contributions similar to these were also observed for the 1997 composite analysis exposure projections.

As discussed earlier, modeling of MDA G in its early successional stages assumes that the plant and animal communities are instantly established at the site when the facility closes. In reality, these communities will become established gradually over a period of a few years. Consequently, the doses shown in Figure 3-9 for the early years of the succession are expected to overestimate actual exposures. While further analysis is required to accurately determine how much the exposures are overestimated, the disparity

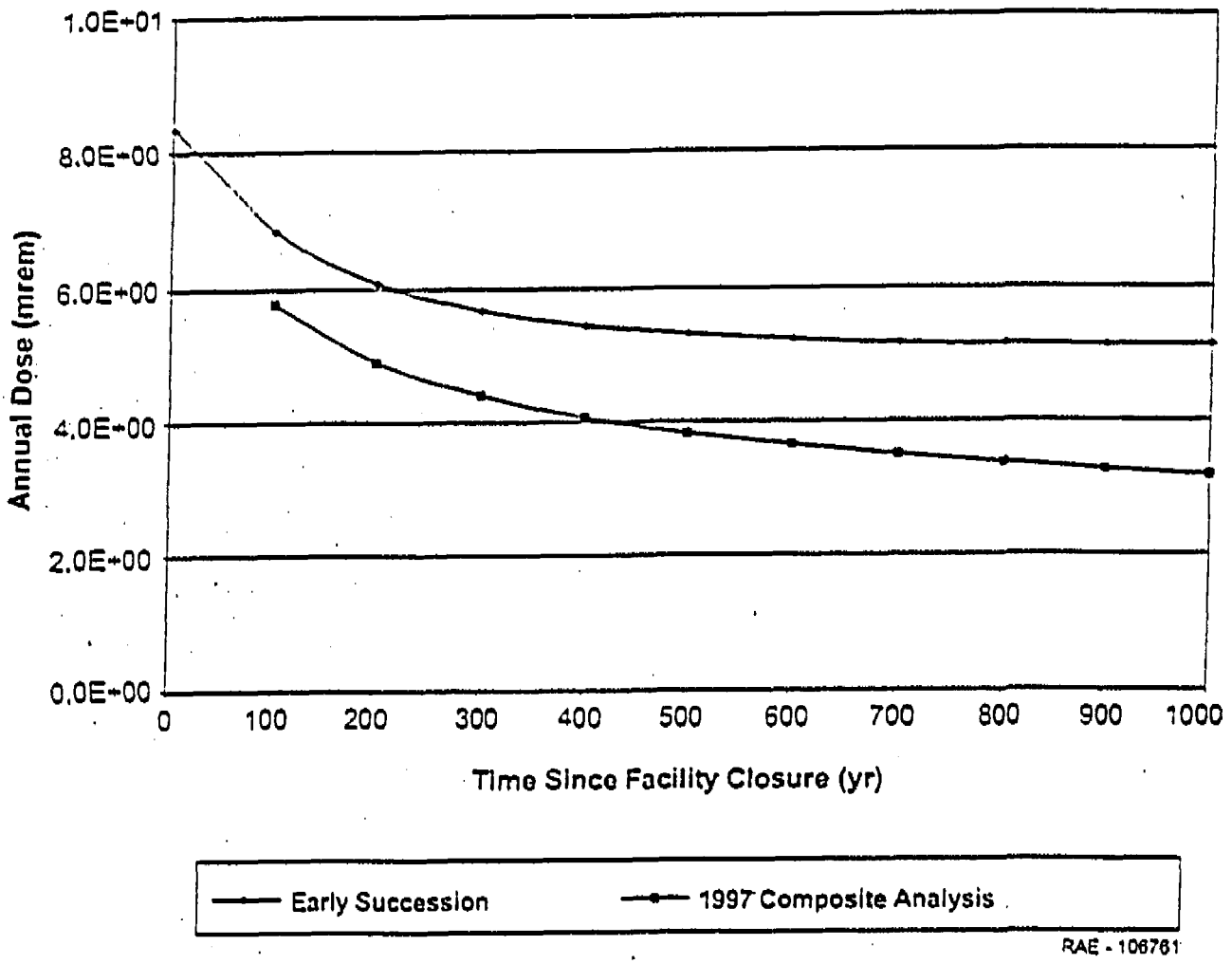

Figure 3-9. Comparison of projected atmospheric scenario doses for the early successional community and the 1997 composite analysis.

is expected to be less than 1.5 mrem/yr, or the difference between the doses projected for year 1 and 100 of the simulation.

The projected atmospheric scenario doses for MDA G in its climax condition are compared to the 1997 composite analysis doses in Figure 3-10. Information from Arnold et al. (1964) and Tress and Klopatek (1987) suggest that the time required for MDA G to achieve climax conditions similar to those modeled will be at least 100 years, and probably much longer. The modeling results for the climax community are shown beginning at 100 years post-closure to cover the range of these estimates.

The peak annual exposure projected for the atmospheric scenario under climax conditions is 3.5 mrem if the site is assumed to reach a climax condition by 100 years post-closure. It declines to 2.7 mrem/yr if it is assumed that climax conditions are not achieved until 300 years after facility closure. These doses are 46 to 60 percent of the peak dose projected for the 1997 composite analysis 100 years after facility closure. The inhalation of airborne contamination accounts for almost all of the doses projected for the site in its climax condition. Isotopes of plutonium and americium are responsible for more than 97 percent of the projected peak dose; waste disposed of between 1957 and 1970 accounts for 87 percent or more of the total exposure. Plutonium and americium in the pre-1971 waste made similar contributions to the doses projected for the 1997 draft of the MDA G composite analysis.

3.2 UNCERTAINTY ANALYSIS RESULTS

Sources of uncertainty associated with the updated biotic intrusion modeling may introduce errors into the soil concentrations projected for MDA G in its early successional stages and its climax condition. A probabilistic uncertainty analysis was conducted to better understand the impact of uncertainties associated with the model input parameters. The results of this analysis provide insight into the potential variability associated with the baseline estimates of surface soil concentrations presented in Section 3.1. A sensitivity

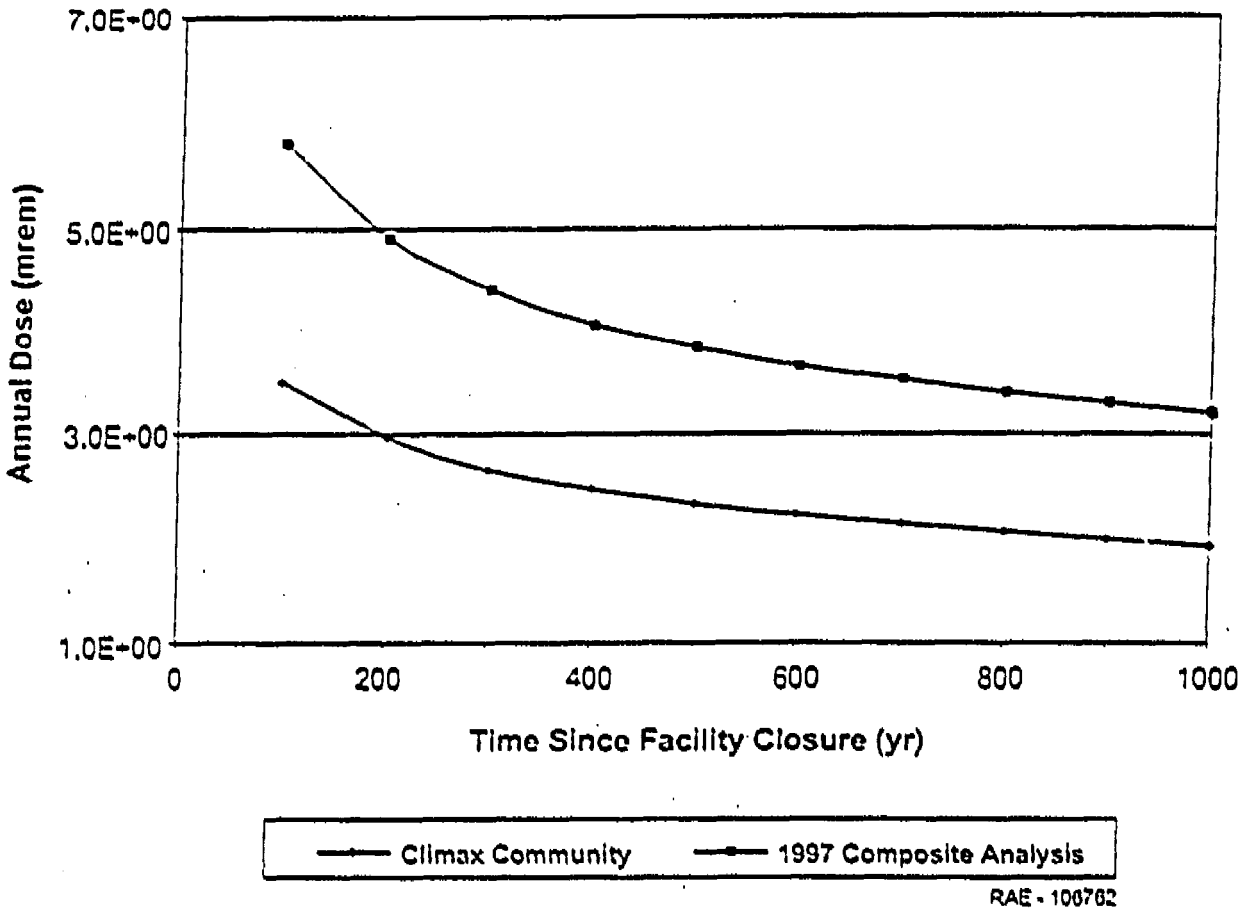

Figure 3-10. Projected atmospheric scenario doses for the climax community and the 1997 composite analysis.

analysis was conducted to help identify the parameters that play major roles in the projected impacts of biotic intrusion.

A 1,000-year compliance period was used to evaluate the ability of MDA G to satisfy the performance objectives for the performance assessment and composite analysis. While the discussion of the baseline modeling in Section 3.1 considers the potential impacts of biotic intrusion over a 10,000-year period, the uncertainty analysis was conducted on the basis of this compliance period. The analysis uses the maximum mesa-top soil concentrations projected during this period as a measure of disposal facility performance, and addresses the impacts of parameter variability on these concentrations for early succession and climax conditions. The analysis considers the range of cover depths and plant uptake factors included in the baseline modeling.

The results of the uncertainty analysis for the disposal site during the early stages of succession are summarized in Table 3-1. This table presents the summary statistics for the maximum projected soil concentrations within 300 years of facility closure, as the site is expected to reach, or near, a climax condition by the end of this period. Concentrations are provided for cover depths ranging from 1 to 3 m and plant uptake factors ranging from 10^{-5} to 100. For each combination of cover depth and uptake factor, the minimum, median, 90th percentile, 95th percentile, and maximum normalized concentrations projected by the model are given. The percentile values indicate the percentage of the generated soil concentrations that are less than or equal to the associated value. For example, the 95th percentile concentration for radionuclides disposed under 1 m of cover and a plant uptake factor of 10 indicates that 95 percent of the calculated normalized concentrations will be less than or equal to 3.9. Conversely, 5 percent of the projected concentrations will be greater than 3.9.

Table 3-1. Summary statistics of the normalized soil concentrations projected by the uncertainty analysis for MDA G in its early successional stages.

Cover Thickness/Plant Uptake Factor	Normalized Soil Concentrations					
	Baseline	Minimum	Median	90th Percentile	95th Percentile	Maximum
1 m Cover Thickness						
Plant uptake factor = 10^{2}	3.2E+00	9.5E-02	1.3E+00	7.1E+00	1.1E+01	1.1E+02
Plant uptake factor = 10^{1}	3.7E-01	7.1E-02	2.2E-01	8.9E-01	1.3E+00	1.1E+01
Plant uptake factor = 10^{0}	8.8E-02	6.0E-02	9.5E-02	1.5E-01	2.0E-01	1.5E+00
Plant uptake factor = 10^{-1}	5.9E-02	5.8E-02	8.0E-02	9.8E-02	1.0E-01	1.7E-01
Plant uptake factor = 10^{-2}	5.6E-02	5.6E-02	7.7E-02	9.5E-02	1.0E-01	1.2E-01
Plant uptake factor = 10^{-3}	5.6E-02	5.6E-02	7.7E-02	9.5E-02	1.0E-01	1.2E-01
Plant uptake factor = 10^{-4}	5.6E-02	5.6E-02	7.7E-02	9.5E-02	1.0E-01	1.2E-01
Plant uptake factor = 10^{-5}	5.6E-02	5.6E-02	7.7E-02	9.5E-02	1.0E-01	1.2E-01
2 m Cover Thickness						
Plant uptake factor = 10^{2}	1.5E+00	3.0E-03	7.5E-01	4.1E+00	6.4E+00	8.5E+01
Plant uptake factor = 10^{1}	1.5E-01	1.5E-03	8.4E-02	4.7E-01	7.3E-01	5.6E+00
Plant uptake factor = 10^{0}	1.5E-02	4.5E-04	1.0E-02	4.9E-02	7.8E-02	8.5E-01
Plant uptake factor = 10^{-1}	1.6E-03	7.5E-05	2.1E-03	7.7E-03	1.1E-02	7.1E-02
Plant uptake factor = 10^{-2}	1.8E-04	5.4E-05	1.1E-03	3.5E-03	4.8E-03	1.6E-02
Plant uptake factor = 10^{-3}	1.4E-04	4.7E-05	9.9E-04	3.1E-03	4.3E-03	1.6E-02
Plant uptake factor = 10^{-4}	1.3E-04	2.2E-05	9.7E-04	3.1E-03	4.3E-03	1.6E-02
Plant uptake factor = 10^{-5}	1.3E-04	2.2E-05	9.7E-04	3.1E-03	4.3E-03	1.6E-02
3 m Cover Thickness						
Plant uptake factor = 10^{2}	8.1E-01	2.1E-03	4.9E-01	2.7E+00	4.2E+00	3.6E+01
Plant uptake factor = 10^{1}	8.1E-02	7.5E-04	5.5E-02	3.1E-01	5.0E-01	3.9E+00
Plant uptake factor = 10^{0}	8.1E-03	7.8E-05	5.6E-03	3.1E-02	5.0E-02	5.9E-01
Plant uptake factor = 10^{-1}	8.1E-04	8.0E-06	5.8E-04	3.3E-03	5.3E-03	5.0E-02
Plant uptake factor = 10^{-2}	8.1E-05	7.2E-07	5.4E-05	3.2E-04	5.2E-04	4.2E-03
Plant uptake factor = 10^{-3}	8.1E-06	7.2E-08	5.3E-06	3.1E-05	5.0E-05	1.2E-03
Plant uptake factor = 10^{-4}	8.1E-07	3.8E-09	5.5E-07	3.0E-06	4.6E-06	5.6E-05
Plant uptake factor = 10^{-5}	8.1E-08	1.8E-10	5.6E-08	3.2E-07	5.1E-07	8.1E-06

The maximum normalized soil concentrations projected by the baseline modeling for the 300-year period immediately following facility closure are included in Table 3-1 to permit their comparison to the projected distributions. Baseline concentrations of radionuclides with plant uptake factors equal to 1.0 or greater generally lie between the median and 90th percentile values estimated in the uncertainty analysis for disposal units with 1 or 2 m of cover. The baseline concentrations of radionuclides with low uptake factors tend to lie between the minimum and median values for these units. For disposal units with 3 m of cover, the baseline concentrations of all radionuclides fall between the median and 90th percentile concentrations.

The sensitivity of the projected soil concentrations to the model input parameters was evaluated on the basis of Spearman rank correlation coefficients. The results of this analysis for MDA G in its early successional stages are summarized in Table 3-2, which shows the model parameters for which the absolute values of the correlation coefficients were 0.1 or greater. While many of the correlation coefficients that had absolute values smaller than 0.1 were statistically significant, this cut-off was used to focus attention on those parameters that had greatest effects on the projected outcomes.

Patterns of parameter sensitivity differ significantly with the thickness of the cover placed over the waste and the radionuclide under consideration. The projected soil concentrations for radionuclides with mean plant uptake factors of 1.0 or above are most sensitive to plant uptake factors, the soil removal rate of pocket gophers, and litter production rates for grasses and forbs. The correlation between soil concentrations and gopher soil removal rates is negative, indicating that this parameter is important because of its affect on the dilution of contaminants entering the soil through litter decay. As the projected rate of soil removal increases, the contaminants deposited on the ground surface by plants are diluted to a greater extent. The correlation coefficients between the soil concentrations, and the plant uptake factors and litter production rates are positive, indicating that these quantities move in proportion to one another.

The projected soil concentrations for radionuclides over disposal units with 1 to 2 m of cover are increasingly sensitive to animal burrowing parameters as the mean plant

Table 3-2. Sensitivity analysis results for MDA G in its early successional stages.

Radionuclide/ Plant Uptake Factor	Cover Thickness: 1 m Cover Thickness		2 m Cover Thickness		3 m Cover Thickness	
	Input Parameter	Correlation Coefficient	Input Parameter	Correlation Coefficient	Input Parameter	Correlation Coefficient
Isotope 1, $B_v = 10^{2}$	Plant uptake factor	8.1E-01	Plant uptake factor	8.1E-01	Plant uptake factor	8.1E-01
	Soil removal rate of pocket gophers	-5.0E-01	Soil removal rate of pocket gophers	-4.9E-01	Soil removal rate of pocket gophers	-4.9E-01
	Grass litter production rate	1.8E-01	Grass litter production rate	1.7E-01	Forb litter production rate	1.9E-01
	Forb litter production rate	1.3E-01	Forb litter production rate	1.6E-01	Grass litter production rate	1.6E-01
Isotope 2, $B_v = 10^{1}$	Plant uptake factor	8.3E-01	Plant uptake factor	8.1E-01	Plant uptake factor	8.2E-01
	Soil removal rate of pocket gophers	-4.5E-01	Soil removal rate of pocket gophers	-5.0E-01	Soil removal rate of pocket gophers	-4.9E-01
	Grass litter production rate	1.7E-01	Grass litter production rate	1.5E-01	Forb litter production rate	1.5E-01
			Forb litter production rate	1.3E-01	Grass litter production rate	1.4E-01
Isotope 3, $B_v = 10^{0}$	Plant uptake factor	7.8E-01	Plant uptake factor	7.7E-01	Plant uptake factor	8.1E-01
	Grass litter production rate	1.3E-01	Soil removal rate of pocket gophers	-5.3E-01	Soil removal rate of pocket gophers	-4.7E-01
			Grass litter production rate	1.4E-01	Forb litter production rate	1.6E-01
			Forb litter production rate	1.3E-01	Grass litter production rate	1.4E-01
Isotope 4, $B_v = 10^{-1}$	Soil removal rate of pocket gophers	6.7E-01	Soil removal rate of pocket gophers	-7.1E-01	Plant uptake factor	8.2E-01
	Plant uptake factor	2.5E-01	Plant uptake factor	5.4E-01	Soil removal rate of pocket gophers	-1.8E-01
	Harvester ant colony density	1.5E-01	Harvester ant colony density	2.3E-01	Forb litter production rate	1.7E-01
	Deer mouse burrow volume	1.5E-01	Harvester ant burrow mass	1.7E-01	Grass litter production rate	1.6E-01
	Harvester ant burrow mass	1.3E-01	Grass litter production rate	1.2E-01		
Isotope 5, $B_v = 10^{-2}$	Soil removal rate of pocket gophers	8.1E-01	Soil removal rate of pocket gophers	-7.5E-01	Plant uptake factor	8.2E-01

Table 3-2. Continued.

Radionuclide/ Plant Uptake Factor	Cover Thickness: 1 m Cover Thickness		2 m Cover Thickness		3 m Cover Thickness	
	Input Parameter	Correlation Coefficient	Input Parameter	Correlation Coefficient	Input Parameter	Correlation Coefficient
	Deer mouse burrow volume	1.8E-01	Harvester ant colony density	4.5E-01	Soil removal rate of pocket gophers	-4.9E-01
	Harvester ant colony density	1.8E-01	Harvester ant burrow mass	3.5E-01	Forb litter production rate	1.7E-01
	Harvester ant burrow mass	1.6E-01	Plant uptake factor	1.6E-01	Grass litter production rate	1.4E-01
Isotope 6, $B_v = 10^{3}$	Soil removal rate of pocket gophers	8.2E-01	Soil removal rate of pocket gophers	-7.2E-01	Plant uptake factor	8.2E-01
	Deer mouse burrow volume	1.8E-01	Harvester ant colony density	4.9E-01	Soil removal rate of pocket gophers	-4.8E-01
	Harvester ant colony density	1.8E-01	Harvester ant burrow mass	3.9E-01	Forb litter production rate	1.6E-01
	Harvester ant burrow mass	1.6E-01			Grass litter production rate	1.3E-01
Isotope 7, $B_v = 10^{4}$	Soil removal rate of pocket gophers	8.2E-01	Soil removal rate of pocket gophers	-7.1E-01	Plant uptake factor	8.1E-01
	Deer mouse burrow volume	1.8E-01	Harvester ant colony density	5.0E-01	Soil removal rate of pocket gophers	-4.9E-01
	Harvester ant colony density	1.8E-01	Harvester ant burrow mass	3.9E-01	Grass litter production rate	1.5E-01
	Harvester ant burrow mass	1.6E-01			Forb litter production rate	1.5E-01
Isotope 8, $B_v = 10^{5}$	Soil removal rate of pocket gophers	8.2E-01	Soil removal rate of pocket gophers	-7.2E-01	Plant uptake factor	8.1E-01
	Deer mouse burrow volume	1.8E-01	Harvester ant colony density	5.0E-01	Soil removal rate of pocket gophers	-4.7E-01
	Harvester ant colony density	1.8E-01	Harvester ant burrow mass	4.0E-01	Forb litter production rate	1.6E-01
	Harvester ant burrow mass	1.6E-01			Grass litter production rate	1.3E-01

uptake factor decreases from 0.1 to 10^{-6}. Concentrations are positively correlated with pocket gopher soil removal rates, harvester ant colony density and burrow mass, and deer mouse burrow volume for units with 1 m of cover. While harvester ant colony density and burrow mass are positively correlated with concentrations over units with 2 m of cover, the soil removal rate of pocket gophers is negatively correlated because these animals do not penetrate into the waste. For the disposal units with 3 m of cover, projected soil concentrations are sensitive to plant uptake factors and litter production rates, pocket gopher activity has the effect of diluting the contamination deposited on the ground by litter decay.

Table 3-3 presents the summary statistics for the uncertainty analysis as applied to MDA G in its climax condition. The baseline data and the uncertainty analysis statistics are based on the maximum soil concentrations projected to occur within the 1,000-year compliance period. The baseline concentrations of radionuclides with plant uptake factors of 1.0 or more fall between the median and 90th percentile values calculated for the disposal units with 1 or 2 m of cover. Projected baseline concentrations of radionuclides with lower plant uptake factors tend to be less than the median values for these units. The baseline concentrations of all contaminants disposed in units with 3 m of cover are essentially equal to the median values estimated by the uncertainty analysis.

The results of the sensitivity analysis for the climax condition of MDA G are presented in Table 3-4. Projected concentrations of radionuclides with mean plant uptake factors greater than 0.1 are positively and strongly correlated with the plant uptake factor, tree biomass, and tree litter production rate for disposal units with 1 to 2 m of cover. Tree biomass and litter production rate are important because they play a role in determining how much contaminated litter is produced annually. Parameters specifying the amount of animal intrusion become more important for disposal units with 1 to 2 m of cover as the mean plant uptake factor decreases. For units with 1 m of cover, the densities of harvester ants and chipmunks and the masses and volumes of these species' burrows are positively correlated to the projected soil concentrations. Chipmunk activity is negatively correlated with soil concentrations over units with 2 m of cover because they do not penetrate into the

Table 3-3. Summary statistics of the normalized soil concentrations projected by the uncertainty analysis for MDA G in its climax condition.

Cover Thickness/Plant Uptake Factor	Normalized Soil Concentrations					
	Baseline	Minimum	Median	90th Percentile	95th Percentile	Maximum
1 m Cover Thickness						
Plant uptake factor = 10^{2}	4.5E+01	9.2E-01	3.4E+01	1.3E+02	1.9E+02	1.4E+03
Plant uptake factor = 10^{1}	4.6E+00	1.6E-01	3.6E+00	1.4E+01	2.1E+01	1.1E+02
Plant uptake factor = 10^{0}	4.9E-01	5.3E-02	4.4E-01	1.6E+00	2.4E+00	2.4E+01
Plant uptake factor = 10^{-1}	7.4E-02	2.5E-02	1.2E-01	2.2E-01	3.1E-01	2.3E+00
Plant uptake factor = 10^{-2}	3.2E-02	1.7E-02	8.0E-02	1.1E-01	1.2E-01	2.6E-01
Plant uptake factor = 10^{-3}	2.8E-02	1.4E-02	7.4E-02	1.1E-01	1.1E-01	1.5E-01
Plant uptake factor = 10^{-4}	2.8E-02	1.4E-02	7.4E-02	1.1E-01	1.2E-01	1.5E-01
Plant uptake factor = 10^{-5}	2.8E-02	1.4E-02	7.4E-02	1.1E-01	1.2E-01	1.5E-01
2 m Cover Thickness						
Plant uptake factor = 10^{2}	2.7E+01	5.4E-01	2.3E+01	8.7E+01	1.3E+02	8.0E+02
Plant uptake factor = 10^{1}	2.8E+00	5.9E-02	2.4E+00	1.0E+01	1.4E+01	9.1E+01
Plant uptake factor = 10^{0}	2.8E-01	1.2E-02	2.5E-01	1.0E+00	1.5E+00	1.6E+01
Plant uptake factor = 10^{-1}	2.8E-02	2.2E-03	2.9E-02	1.1E-01	1.7E-01	1.5E+00
Plant uptake factor = 10^{-2}	3.1E-03	6.0E-04	8.5E-03	1.6E-02	2.0E-02	1.1E-01
Plant uptake factor = 10^{-3}	6.0E-04	3.8E-04	5.5E-03	1.0E-02	1.2E-02	2.4E-02
Plant uptake factor = 10^{-4}	3.6E-04	2.6E-04	5.1E-03	9.8E-03	1.1E-02	1.9E-02
Plant uptake factor = 10^{-5}	3.3E-04	2.4E-04	5.1E-03	9.8E-03	1.1E-02	1.9E-02
3 m Cover Thickness						
Plant uptake factor = 10^{2}	2.3E+01	5.7E-01	2.2E+01	8.0E+01	1.2E+02	1.4E+03
Plant uptake factor = 10^{1}	2.3E+00	3.8E-02	2.3E+00	9.3E+00	1.4E+01	6.3E+01
Plant uptake factor = 10^{0}	2.3E-01	5.0E-03	2.3E-01	1.0E+00	1.5E+00	1.2E+01
Plant uptake factor = 10^{-1}	2.3E-02	6.1E-04	2.2E-02	9.8E-02	1.5E-01	1.3E+00
Plant uptake factor = 10^{-2}	2.3E-03	5.9E-05	2.3E-03	9.7E-03	1.5E-02	1.1E-01
Plant uptake factor = 10^{-3}	2.3E-04	3.2E-06	2.3E-04	1.0E-03	1.5E-03	1.8E-02
Plant uptake factor = 10^{-4}	2.3E-05	4.9E-07	2.3E-05	9.7E-05	1.4E-04	8.0E-04
Plant uptake factor = 10^{-5}	2.3E-06	3.0E-08	2.4E-06	9.7E-06	1.5E-05	1.6E-04

Table 3-4. Sensitivity analysis results for MDA G in its climax condition.

Radionuclide/ Plant Uptake Factor	Cover Thickness: 1 m Cover Thickness		2 m Cover Thickness		3 m Cover Thickness	
	Input Parameter	Correlation Coefficient	Input Parameter	Correlation Coefficient	Input Parameter	Correlation Coefficient
Isotope 1, $B_v = 10^{2}$	Plant uptake factor	9.7E-01	Plant uptake factor	9.7E-01	Plant uptake factor	9.7E-01
	Tree biomass	1.1E-01	Tree biomass	1.1E-01	Tree biomass	1.1E-01
Isotope 2, $B_v = 10^{1}$	Plant uptake factor	9.8E-01	Plant uptake factor	9.7E-01	Plant uptake factor	9.7E-01
	Tree biomass	1.1E-01	Tree biomass	1.1E-01	Tree biomass	1.2E-01
			Tree litter production rate	1.0E-01	Tree litter production rate	1.0E-01
Isotope 3, $B_v = 10^{0}$	Plant uptake factor	9.9E-01	Plant uptake factor	9.8E-01	Plant uptake factor	9.7E-01
Isotope 4, $B_v = 10^{-1}$	Plant uptake factor	8.8E-01	Plant uptake factor	9.7E-01	Plant uptake factor	9.7E-01
	Chipmunk burrow volume	5.0E-01	Chipmunk burrow volume	-1.2E-01	Tree biomass	1.1E-01
	Tree biomass	-1.9E-01			Chipmunk burrow volume	-1.0E-01
	Chipmunk density	1.8E-01				
	Tree litter production rate	-1.6E-01				
Isotope 5, $B_v = 10^{-2}$	Tree biomass	-5.0E-01	Plant uptake factor	6.9E-01	Plant uptake factor	9.7E-01
	Chipmunk burrow volume	4.1E-01	Harvester ant colony density	3.9E-01	Tree biomass	1.0E-01
	Chipmunk density	4.0E-01				

Table 3-4. Continued.

Radionuclide/ Plant Uptake Factor	Cover Thickness: 1 m Cover Thickness		2 m Cover Thickness		3 m Cover Thickness	
	Input Parameter	Correlation Coefficient	Input Parameter	Correlation Coefficient	Input Parameter	Correlation Coefficient
	Tree litter production rate	-3.6E-01	Harvester ant burrow mass	3.1E-01		
	Harvester ant colony density	2.6E-01	Chipmunk burrow volume	-1.8E-01		
	Plant uptake factor	2.4E-01	Chipmunk density	-1.5E-01		
	Harvester ant burrow mass	1.8E-01	Tree biomass	-1.4E-01		
			Tree litter production rate	-1.3E-01		
Isotope 6, $B_v = 10^{3}$	Tree biomass	-5.3E-01	Harvester ant colony density	6.6E-01	Plant uptake factor	9.7E-01
	Chipmunk burrow volume	4.4E-01	Harvester ant burrow mass	5.0E-01	Chipmunk density	-1.1E-01
	Chipmunk density	4.2E-01	Tree biomass	-2.9E-01		
	Tree litter production rate	-3.8E-01	Tree litter production rate	-2.0E-01		
	Harvester ant colony density	2.8E-01	Chipmunk density	-1.6E-01		
	Harvester ant burrow mass	2.0E-01	Plant uptake factor	1.6E-01		
			Chipmunk burrow volume	-1.5E-01		
Isotope 7, $B_v = 10^{4}$	Tree biomass	-5.3E-01	Harvester ant colony density	6.7E-01	Plant uptake factor	9.7E-01
	Chipmunk burrow volume	4.4E-01	Harvester ant burrow mass	5.2E-01	Tree biomass	1.3E-01
	Chipmunk density	4.1E-01	Tree biomass	-3.0E-01	Tree litter production rate	1.0E-01

Table 3-4. Continued.

Radionuclide/ Plant Uptake Factor	Cover Thickness: 1 m Cover Thickness		2 m Cover Thickness		3 m Cover Thickness	
	Input Parameter	Correlation Coefficient	Input Parameter	Correlation Coefficient	Input Parameter	Correlation Coefficient
	Tree litter production rate	-3.8E-01	Tree litter production rate	-2.2E-01		
	Harvester ant colony density	2.8E-01	Chipmunk density	-1.5E-01		
	Harvester ant burrow mass	2.0E-01	Chipmunk burrow volume	-1.4E-01		
Isotope 8, $B_p = 10^5$	Tree biomass	-5.3E-01	Harvester ant colony density	6.7E-01	Plant uptake factor	9.7E-01
	Chipmunk burrow volume	4.4E-01	Harvester ant burrow mass	5.2E-01		
	Chipmunk density	4.1E-01	Tree biomass	-3.0E-01		
	Tree litter production rate	-3.8E-01	Tree litter production rate	-2.2E-01		
	Harvester ant colony density	2.8E-01	Chipmunk density	-1.5E-01		
	Harvester ant burrow mass	2.0E-01	Chipmunk burrow volume	-1.4E-01		

waste. Tree biomass and litter production rates are negatively correlated with soil concentrations of radionuclides with small plant uptake factors for units with 1 or 2 m of cover. This is because the relatively uncontaminated litter dilutes the contamination brought to the surface by chipmunks and/or ants. Soil concentrations over the disposal units with 3 m of cover are most sensitive to plant uptake factor, tree biomass, and tree litter production rate.

3.3 DISCUSSION

The primary reason for updating the MDA G biotic intrusion modeling was to evaluate the potential impacts of plants and animals on the ability of the facility to safely contain the waste if there was no long-term maintenance of the site. The results of the analysis indicate that the projected impacts under these conditions are similar to those projected for the 1997 MDA G performance assessment and composite analysis. This is illustrated by the composite analysis doses projected for the atmospheric exposure scenario using the updated biotic intrusion modeling. Assuming the site reaches or nears its climax condition within 300 years of facility closure, these doses are expected to be less than 1.4 times those projected for the 1997 analysis. Doses projected for the site after it has reached the pinyon-juniper climax are actually lower than those projected for the earlier analysis.

The projected impact of parameter uncertainties on the atmospheric exposure scenario doses also appears to be similar between the updated results and the 1997 composite analysis. The uncertainty analysis performed for the 1997 analysis estimated that the projected peak dose of 5.8 mrem/yr could, under worst-case conditions, increase to as much as 17 mrem/yr based *only* on the uncertainties associated with the extent of biotic intrusion into the site. As discussed earlier, plutonium and americium were the main contributors to the doses projected using the updated biotic intrusion modeling. These elements have mean plant uptake factors of 4.5×10^{-4} and 5.5×10^{-3}, respectively. The majority of the plutonium and americium-contaminated waste was disposed in pits with as little as 1 m of cover. Under these conditions, and given that atmospheric scenario doses

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are directly proportional to the normalized soil concentrations, the uncertainty analysis results in Table 3-1 indicate that the maximum doses will be about 2.1 times greater than the exposures shown in Figure 3-9. The product of this factor and the peak dose projected for the site during early succession, 8.4, mrem/yr, is 18 mrem/yr, practically identical to the worst-case dose of 17 mrem/yr estimated for the MDA G composite analysis.

The comparison provided above is based only on the projected levels of biotic intrusion into the disposed waste. As discussed in the Introduction, the activities planned to refine the models and data used for projecting doses for the atmospheric exposure scenario extend beyond the biotic intrusion modeling. They include the examination of the models used to estimate resuspension rates for surface contamination and to simulate atmospheric transport from MDA G to off-site receptor locations. While differences between the 1997 and updated biotic intrusion modeling do not appear to be great, the results from these other analyses will be needed in order to understand the overall impacts of biotic intrusion on MDA G. Once these impacts are understood, the need for corrective actions to lower potentially unacceptable exposures may be addressed.

The results of the updated biotic intrusion modeling and the probabilistic uncertainty analysis provide additional insight into how plant and animal intrusion into waste disposed at MDA G may impact the site. The conclusions drawn from these analyses, however, are specific to the data used to represent the various model input parameters. While information specific to MDA G or the Laboratory was used to develop the baseline values and parameter distributions when it was available, non-site-specific data were used to estimate the values for several model parameters. In some cases (e.g., tree biomass and litter generations rates in shrubs and trees), the data available in these alternate sources were extremely limited. Where possible, then, Laboratory-specific data generated in the future should be used to evaluate the parameter values and distributions used in this analysis, updating them as appropriate.

While all improvements in the quality of any of the data used to model biotic intrusion at MDA G are welcome, particular attention should be paid to parameters to which the projected doses are most sensitive. On the basis of the updated doses presented

in Section 3.1, parameters related to the level of animal intrusion into the site are expected to be most significant. Plutonium and americium placed in pits with as little as 1 m of cover contribute the majority of the projected peak dose for the site during early succession and after it has reached a climax condition. The mean plant uptake factors for these elements are 4.5×10^{-4} and 5.5×10^{-3}, respectively. Under these conditions, soil concentrations projected for the site during early succession are most sensitive to the soil removal rate of pocket gophers, deer mouse and harvester ant activity are also important (Table 3-2). Soil concentrations in the climax woodland are also sensitive to animal activity, although plant parameters play an important role in that community as well (Table 3-4).

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