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MARTIN MARIETTA

OPERATING LIMIT STUDY FOR THE PROPOSED SOLID WASTE LANDFILL AT PADUCAH GASEOUS DIFFUSION PLANT

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MARTIN MARIETTA ENERGY SYSTEMS, INC.
FOR THE UNITED STATES
DEPARTMENT OF ENERGY

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Prepared for

Environmental Management and Enrichment Facilities
Paducah Gaseous Diffusion Plant
Paducah, Kentucky

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ABSTRACT

A proposed solid waste landfill at Paducah Gaseous Diffusion Plant (PGDP) would accept wastes generated during normal operations that are identified as non-radioactive. These wastes may include small amounts of radioactive material from incidental contamination during plant operations. A site-specific analysis of the new solid waste landfill is presented to determine a proposed operating limit that will allow for waste disposal operations to occur such that protection of public health and the environment from the presence of incidentally contaminated waste materials can be assured. Performance objectives for disposal were defined from existing regulatory guidance to establish reasonable dose limits for protection of public health and the environment. Waste concentration limits were determined consistent with these performance objectives for the protection of off-site individuals and inadvertent intruders who might be directly exposed to disposed wastes. Exposures of off-site individuals were estimated using a conservative, site-specific model of the groundwater transport of contamination from the wastes. Direct intrusion was analyzed using an agricultural homesteader scenario. The most limiting concentrations from direct intrusion or groundwater transport were used to establish the concentration limits for radionuclides likely to be present in PGDP wastes. These concentration limits were used to define a proposed operating limit of 30 pCi/g for uranium to be applied to each waste package, based on the U. S. Nuclear Regulatory Commission Branch Technical Position Paper on the management of uranium contaminated wastes. Examination of the application of this operating limit to wastes typical of PGDP suggests that waste disposals would be less than the calculated concentration limits for the solid waste landfill. This site-specific analysis of the long-term performance of the new solid waste landfill at the PGDP provides reasonable assurance that the application of the proposed operating limit will be protective of public health and the environment.

1. INTRODUCTION

1.1 DESCRIPTION OF STUDY

This section provides an overview of the objectives of this study which is to develop proposed operating limits for the proposed solid waste landfill at the Paducah Gaseous Diffusion Plant (PGDP). The radiological performance objectives for landfill disposal that are used in developing the proposed operating limits are presented and the basis for these performance objectives discussed.

PGDP is one of two government-owned, contractor-operated uranium enrichment facilities within the U.S. Department of Energy (DOE) complex. As in any industrial facility, activities at the PGDP generate substantial volumes of waste that are intended for disposal in a solid waste landfill permitted by the Commonwealth of Kentucky in accordance with provisions of Subtitle D of the Resource Conservation and Recovery Act (RCRA).

Nearly all solid waste from any source or facility contains small amounts of radioactive material, due to the presence in most materials of trace quantities of such naturally occurring radionuclides as uranium and thorium. In addition, activities at the PGDP involving the handling and processing of uranium in various isotopic abundances can result in the addition of small amounts of radioactive material to waste that otherwise would be intended for disposal in a solid waste landfill. Such incidentally contaminated waste containing very low levels of radioactive material is quite different, with regard to its potential impacts on public health and the environment, from the kinds of low-level radioactive waste generated directly from handling and processing of uranium and other radioactive materials, which are intended for disposal in dedicated facilities for low-level radioactive waste in accordance with DOE requirements in Order 5820.2A (DOE 1988).

The purpose of this report is to develop proposed operating limits, which are related to protection of public health and the environment, that would allow waste materials to be sent to a new solid waste landfill at the PGDP site even though the materials might contain small amounts of radioactive materials. The operating limits are expressed as limits on concentrations of radionuclides in waste materials that could be sent to the proposed solid waste landfill and are based on a site-specific analysis of the performance of the facility. As indicated above, the radioactive material in the waste could either be naturally occurring or could result from incidental contamination in the course of handling or processing of radioactive materials at the site.

The rationale for allowing waste materials which are slightly contaminated with radioactive material to be sent to a solid waste landfill is twofold. First, in many wastes containing very low levels of contamination, the presence of radionuclides which resulted from activities at the facility would be virtually indistinguishable from background levels that would occur in the absence of any such activities. Second, if the limits on allowable quantities of radionuclides in solid waste are set sufficiently low, the resulting effects on public health and the environment from disposal in a landfill would be negligible, particularly in comparison with the corresponding effects due to the ubiquitous natural background of radiation.

The establishment of operating limits which would permit waste containing small amounts of radioactive material to be sent to a solid waste landfill at the PGDP site would be

advantageous for the site and DOE. The most important advantage is that the cost of on-site waste disposal would be greatly reduced compared with the cost of sending all waste containing or suspected to contain very low levels of radioactivity to a dedicated low-level radioactive waste disposal facility. The levels of radioactivity in waste which would be sent to the PGDP solid waste landfill would be much lower than levels that would be sent to dedicated low-level radioactive waste disposal facilities. If the limits on concentrations of radionuclides in waste that would be acceptable for disposal in a solid waste landfill are sufficiently low, the greatly decreased cost of disposal would not result in any significant increase in health risks to the public or impacts on the environment.

A second advantage is the avoidance of certain liabilities that could result from shipment of solid waste to an off-site commercial facility, which is an option for PGDP waste. For example, a commercial sanitary landfill may have established a policy of not accepting waste to which any radioactive material has been added by activities at the generating site. However, as mentioned previously, for naturally occurring radionuclides including uranium—which is particularly important at the PGDP—and even for certain fission products and transuranic radionuclides which also are ubiquitous in the environment due to fallout from atmospheric nuclear weapons testing, it often is impossible to distinguish unambiguously between small quantities of radioactive material which were added to waste by activities at the generating site and background levels that did not result from such activities. Therefore, off-site shipment of solid waste could leave DOE vulnerable to allegations that radioactive material has been added to the waste when no such additions are allowed.

A third advantage is that disposal operations at the proposed solid waste landfill can be assured to be protective of public health and the environment by having a justified operating limit for accepting waste containing small amounts of radioactive material, instead of relying solely on administrative controls (e.g., accepting waste from some parts of the site but excluding waste from others). Implementation of operating limits for radionuclides in solid waste disposal operations would result in the installation of monitoring equipment with the capability of identifying waste materials inappropriate for disposal in the solid waste landfill and the imposition of administrative controls. Consequently, the possibility of unintended disposals of low-level radioactive wastes would be reduced and the accountability of solid waste disposal operations would be enhanced.

In order to provide a framework for rational and cost-effective waste management which is protective of public health and the environment, it thus appears reasonable to distinguish between waste with quantities of radioactive material sufficiently low that the waste can be managed in all respects as if it were non-radioactive (e.g., in accordance with RCRA requirements) and waste with higher quantities of radioactive material that must be sent to a dedicated facility for low-level radioactive waste in accordance with DOE requirements. However, a national policy for distinguishing between wastes which can be regulated as if they were non-radioactive and wastes which require regulation as radioactive material has not been established. Therefore, in this report, operating limits for making such a distinction are proposed. The operating limits, which again are related to protection of public health and the environment from disposal of radioactive material in the waste, and their implementation in terms of concentration limits of radionuclides that would be acceptable for disposal in a solid waste landfill, are intended to be applied only to the proposed landfill site at the PGDP.

1.2 PERFORMANCE OBJECTIVES FOR LANDFILL DISPOSAL

The term "performance objectives" is used to describe radiological criteria which define acceptable disposals of radioactive materials in waste. Performance objectives for radioactive waste disposal are concerned with protection of public health and the environment. The following discussion presents, and describes in detail, the proposed performance objectives which would be used to define acceptable disposals of waste that is slightly contaminated with radioactive material in the proposed solid waste landfill on the PGDP site.

The acceptability of solid waste landfill disposal for waste materials which may be slightly contaminated with radionuclides is based on the notion that there are levels of radiation exposure so low that they would not be of concern to most individuals. Thus, disposal of slightly contaminated materials in a solid waste landfill would be an acceptable practice if the maximum radiation doses that might be experienced by individuals who reside near or on the disposal site would correspond to risks of adverse health effects which are widely regarded as negligible.

Many exemptions from regulatory requirements for radioactive material, based at least in part on the notion that the resulting doses to individuals or populations should be negligible, have been established by federal regulatory authorities. For example, the NRC's 10 CFR Parts 30 and 40 include many exemptions from licensing requirements for consumer products and other items containing low levels of source material (i.e., uranium or thorium) or by-product material (i.e., fission or activation products). The NRC's 10 CFR Part 20 includes exemptions from licensing requirements for releases of radionuclides into sanitary sewerage systems and disposal of liquid scintillation materials and animal tissues containing low levels of ^{3}H and ^{14}C.

The exemptions in NRC regulations described above have been established only on a case-by-case basis. No general federal regulatory policy as yet exists that is expressed in terms of upper bounds on negligible dose for exposed individuals or populations that could be used to exempt a wide variety of practices and sources, including disposal of any waste, from regulatory requirements for radioactive material. However, as described below, there are regulatory precedents for establishing exempt levels of radionuclides in waste materials for disposal in a solid waste landfill.

Acceptable disposals of radioactive waste in dedicated near-surface facilities normally are determined by specifying performance objectives in the form of limits on radiation dose for potentially exposed individuals, including off-site members of the public and individuals who might inadvertently intrude onto the disposal site, and other requirements for protection of the environment. For example, for disposal of low-level radioactive waste at DOE sites, the performance objectives specified in DOE Order 5820.2A (DOE 1988) include (1) a limit on effective dose equivalent for off-site individuals from all release and exposure pathways of 25 mrem (0.25 mSv) per year, (2) a limit on effective dose equivalent for off-site individuals from atmospheric releases only of 10 mrem (0.10 mSv) per year, (3) limits on effective dose equivalent for inadvertent intruders onto disposal sites after loss of active institutional controls at 100 years after disposal of 100 mrem (1 mSv) per year for continuous exposure or 500 mrem (5 mSv) for a single acute exposure, and (4) protection of groundwater resources consistent with federal, state, and local requirements. The performance objective for protection of groundwater resources usually is interpreted as compliance with standards for radioactivity in public drinking water supplies established by the EPA in 40 CFR Part 141.

The performance objectives described above apply to disposal of radioactive wastes in dedicated facilities. The performance objectives for off-site individuals and inadvertent intruders are intended to correspond to limits on acceptable dose (risk) in each case. However, the performance objectives for off-site individuals and inadvertent intruders are not directly applicable to disposal of slightly contaminated waste in a solid waste landfill, because upper bounds on *negligible* dose that would be appropriate for disposal of waste which is essentially unregulated in regard to its radioactivity content should be substantially less than limits on *acceptable* dose that are appropriate for disposal of waste which is carefully regulated for its radioactivity content (NCRP 1993). On the other hand, as described below, the performance objective for protection of groundwater resources at dedicated disposal facilities for radioactive waste also should be appropriate for disposal of slightly contaminated materials in a solid waste landfill.

In this report, the following performance objectives are proposed for defining acceptable disposals of slightly radioactive materials in a solid waste landfill:

1. For off-site individuals, a limit on effective dose equivalent from direct consumption of contaminated groundwater of 4 mrem (0.04 mSv) per year.

2. For inadvertent intruders onto the disposal site, a limit on effective dose equivalent from all exposure pathways involving direct intrusion into solid waste in the disposal facility of 4 mrem (0.04 mSv) per year.

These radiological performance objectives assume that all other requirements for disposal of waste in a solid waste landfill (e.g., RCRA requirements) will be met. The rationale for the performance objectives is described as follows.

The proposed performance objective for off-site individuals is based, first, on the assumption that releases of radionuclides from the disposal facility to the environment will occur primarily by the groundwater transport pathway and, second, on proposed revisions of EPA standards for radioactivity in public drinking water supplies in 40 CFR Part 141 (EPA 1991a).

The radionuclides which could occur in solid waste at the PGDP are expected to be in particulate form. Therefore, for a disposal facility which is covered with a layer of uncontaminated soil, releases to the atmosphere should not be of concern. Furthermore, for locations near the disposal facility, releases to surface waters that might reasonably be used as a water supply by individuals should not occur. Thus, only releases to groundwater should be of concern for exposures of individuals near the disposal site.

Current EPA standards for radioactivity in drinking water include (1) a concentration limit of 5 pCi/L (0.2 Bq/L) for ^{226}Ra plus ^{228}Ra, (2) a concentration limit of 15 pCi/L (0.6 Bq/L) for gross alpha-particle activity, including ^{226}Ra but excluding radon and uranium, and (3) a limit on dose equivalent to whole body or any organ of 4 mrem (0.04 mSv) per year for all man-made, beta/gamma-emitting radionuclides. The dose limit for beta/gamma-emitting radionuclides normally is converted to radionuclide-specific concentration limits by assuming that an individual consumes 2 liters per day of drinking water. The EPA's proposed revisions of the primary drinking water standards (EPA 1991a) include (1) separate concentration limits of 20 pCi/L (0.7 Bq/L) for ^{226}Ra and ^{228}Ra, (2) a concentration limit of 20 μg/L for uranium, based primarily on prevention of chemical toxicity in the kidney but with additional consideration of the cancer risk from radiation

exposure, (3) a concentration limit of 300 pCi/L (11 Bq/L) for ^{222}Rn, (4) a concentration limit of 15 pCi/L (0.6 Bq/L) for gross alpha-particle activity, excluding ^{226}Ra, uranium, and ^{222}Rn, and (5) a limit on effective dose equivalent of 4 mrem (0.04 mSv) per year for all beta/gamma-emitting radionuclides, excluding ^{228}Ra.

For radionuclides that should be the principal constituents of solid waste at the PGDP, the most important changes in the proposed revisions of the drinking water standards are the inclusion of a concentration limit for uranium and the replacement of a limit on dose equivalent to whole body or any organ for beta/gamma-emitting radionuclides by a limit on effective dose equivalent. For uranium in its natural isotopic abundance, the proposed concentration limit of 20 μg/L corresponds to 14 pCi/L (0.5 Bq/L). This limit is about four times greater than the average concentration of uranium in groundwater in the U.S. (Drury et al. 1981).

The proposed performance objective for protection of off-site individuals from direct consumption of contaminated groundwater [i.e., a limit on effective dose equivalent of 4 mrem (0.04 mSv) per year for the drinking water pathway] is based primarily on the proposed revisions of drinking water standards. However, this performance objective is applied to all radionuclides in the waste, which differs from the approach in drinking water standards of applying the dose limit to beta/gamma-emitting radionuclides only but applying separate concentration limits to alpha-emitting radionuclides. The proposed revisions of the concentration limits for radium and gross alpha-particle activity listed previously are not used in the performance objective for two reasons, first, because these limits include contributions from natural background as well as from waste disposals and, second, because they allow effective dose equivalents that exceed 25 mrem (0.25 mSv) per year. Thus, these limits exceed the performance objective for off-site individuals from all exposure pathways at dedicated low-level radioactive waste disposal facilities. The EPA's proposed concentration limit for uranium corresponds to an effective dose equivalent of about 3 mrem (0.03 mSv) per year for natural uranium, which is reasonably consistent with the performance objective of 4 mrem (0.04 mSv) per year. By applying the proposed performance objective for the drinking water pathway to all radionuclides, the same level of health protection would be achieved for any waste composition.

The basic premise of this report is that disposal of slightly contaminated materials in a solid waste landfill would be acceptable only if the radiation doses to potentially exposed individuals would not exceed levels regarded as negligible. Drinking water standards for radionuclides clearly define an upper bound on negligible risk for the drinking water pathway, because there is no requirement to reduce levels of radioactivity below the standards even if it would be cost-effective to do so. Thus, the proposed performance objective for off-site individuals from the drinking water pathway provides a negligible dose and risk, because it corresponds to, or is more restrictive than, proposed drinking water standards for radionuclides.

For continuous exposure, an effective dose equivalent of 4 mrem (0.04 mSv) per year corresponds to a lifetime risk of fatal cancers of about 10^{-4} (EPA 1991a). This risk level is consistent with the risk goal of 10^{-4} for cleanup of Superfund sites under authority of the Comprehensive Environmental Response, Compensation, and Liability Act (EPA 1990, 1991b). Also, this level is considerably more restrictive than the risk corresponding to proposed standards for cleanup of radioactively contaminated sites under authority of the Atomic Energy Act (NRC 1994; EPA 1994), which specify a limit on effective dose equivalent of 15 mrem (0.15 mSv) per year and are intended to permit unrestricted use of such sites by the public. Thus, there are

several precedents in environmental regulations which indicate that the proposed performance objective for off-site individuals for the drinking water pathway corresponds to a negligible dose and risk and would be appropriate for slightly contaminated materials intended for disposal in a solid waste landfill.

The proposed performance objective for off-site individuals applies to the drinking water pathway only. Off-site individuals also could be exposed by other pathways involving use of contaminated groundwater. These pathways include, for example, consumption of vegetables from a garden which is irrigated with contaminated water and consumption of milk and meat obtained from livestock which drink contaminated water or consume pasture which is irrigated with contaminated water. However, for relatively humid sites, such as the PGDP, and for the radionuclides of concern to waste disposals at the PGDP, including ^{99}Tc as well as any actinide and transuranic radionuclides, previous analyses have shown that the dose from all other exposure pathways involving use of contaminated groundwater would be much less than the dose from direct consumption of drinking water from the same source (ORNL 1994; MMES et al. 1994). That is, at humid sites, doses resulting from bioaccumulation of radionuclides in terrestrial food chains generally are insignificant compared with doses resulting from direct ingestion of radionuclides from a contaminated water source. Thus, the performance objective for the drinking water pathway only would ensure that the dose for off-site individuals from all exposure pathways would not exceed negligible levels.

The proposed performance objective for inadvertent intruders also is based on achieving a negligible dose and risk for such individuals. The performance objective of 4 mrem (0.04 mSv) per year provides a negligible dose and risk for inadvertent intruders because, as discussed previously, it is consistent with current and proposed standards for radioactivity in drinking water and risk goals for cleanup of Superfund sites and it is considerably more restrictive than proposed cleanup standards for radioactively contaminated sites which would permit unrestricted use of such sites by the public.

The performance objective for inadvertent intruders applies only to exposure scenarios involving direct intrusion into solid waste in the disposal facility, but potential exposure pathways involving the use of contaminated groundwater on the disposal site are not included. The exclusion of doses from the groundwater pathway is justified on the following grounds. First, estimates of dose to inadvertent intruders are based on the assumption that no activity is leached from the waste into groundwater. Thus, to the extent that leaching occurs, this approach provides conservative estimates of dose from direct intrusion into solid waste. Second, for most of the radionuclides of concern to waste disposals at the PGDP, the peak concentration in groundwater on the disposal site would occur long after the peak dose from direct intrusion into solid waste. Thus, the maximum doses from the two different pathways would not be additive. Finally, the performance objective for off-site individuals from the groundwater pathway would ensure that doses from direct consumption of groundwater on the disposal site would not greatly exceed 4 mrem (0.04 mSv) per year for any radionuclide. Therefore, the dose to an inadvertent intruder, taking into account releases to groundwater and direct intrusion into solid waste, would not greatly exceed the performance objective even for relatively mobile radionuclides.

Thus, even though the proposed performance objective for inadvertent intruders applies only to direct intrusion into solid waste in the disposal facility, the dose from all exposure pathways would not greatly exceed the performance objective and still would achieve a negligible

level of risk. Furthermore, this approach provides a clear separation between the performance objective for off-site individuals, which is concerned only with releases of radionuclides to the environment, and the performance objective for inadvertent intruders, which would be concerned only with radionuclides which are not released to the environment. Thus, decisions on acceptable disposals of slightly contaminated materials in a solid waste landfill would be based on achieving a reasonable balance between acceptable releases to the environment and acceptable residual levels of radioactivity in the disposal facility.

In summary, the proposed performance objectives for disposal of waste which is slightly contaminated with radionuclides in a solid waste landfill specify a limit on effective dose equivalent of 4 mrem (0.04 mSv) per year for exposures of off-site individuals by the drinking water pathway, and for exposures of inadvertent intruders resulting from direct intrusion into solid waste in the disposal facility. The performance objectives are based on precedents for regulating radioactivity in public drinking water supplies and cleanup of contaminated sites. The use of these regulatory precedents would ensure that risks to off-site individuals and inadvertent intruders from disposal of slightly contaminated materials would not exceed levels regarded as negligible. The use of a single performance objective for off-site individuals and inadvertent intruders which applies to all radionuclides in the waste also ensures that consistent levels of risk would be applied to any individuals and to wastes with any composition of radionuclides.

The proposed performance objectives are applied in the following way. For each radionuclide which is expected to be present in solid waste at the PGDP, a site-specific analysis of the performance of the disposal facility is used to develop an estimate of the maximum concentration in groundwater at off-site locations per unit concentration in disposed waste. This factor then is combined with an estimate of the dose from the drinking water pathway per unit concentration in groundwater and the performance objective for off-site individuals to obtain an estimate of the concentration limit in waste that would be required to ensure compliance with the performance objective. Similarly, an analysis of an exposure scenario for inadvertent intruders is used to develop estimates of the dose to an inadvertent intruder per unit concentration of the radionuclides in the disposal facility, which is then combined with the performance objective for inadvertent intruders to obtain an estimate of the concentration limit in waste that would be required to ensure compliance with the performance objective. For each radionuclide, the concentration limit for disposal in the solid waste landfill is the more restrictive of the two concentration limits obtained in this manner. These limits then provide the basis for the proposed operating limit for the proposed solid waste landfill.

2. DISPOSAL FACILITY DESCRIPTION

2.1 SITE DESCRIPTION

This section describes the location of the proposed solid waste landfill and the topography, geology, hydrogeology, and hydrology of the surrounding area.

2.1.1 Location and Topography

The PGDP resides within a 303-ha (748-ac) fenced area on a 1385-ha (3423-ac) federal reservation located about 14 km (9 miles) west of the city of Paducah, Kentucky, and 5 km (3 miles) south of the Ohio River. The nominal elevation in the main plant area is 116 m (380 ft) above mean sea level (MSL) and 22 m (73 ft) above the average pool level of the Ohio River near PGDP (Fig. 2-1). The proposed landfill is located 3.2 km (2 miles) southwest of the Ohio River near the northern boundary of PGDP. It has an area of 9.19 ha (22.7 ac) and is located 210 m (700 ft) east and 370 m (1200 ft) south of the plant boundary. The ground surface elevation of the "waste limit" area ranges from 110 to 113 m (360 to 370 ft) MSL.

2.1.2 Geology and Hydrogeology

Proceeding from the ground surface downward, the PGDP site is underlain by loess, Pliocene-Pleistocene continental deposits, Tertiary and Cretaceous sediments (Porters Creek clay and McNairy formation), and Mississippian bedrock. The loess deposits range in thickness from 1.5 to 6 m (5 to 25 ft) with an average thickness of 4.6 m (15 ft). The continental deposits subdivide into different hydrogeologic units (Clausen et al. 1992). The upper continental system includes laterally discontinuous but correlatable sands with occasional pebbles, referred to as the shallow sand on Fig. 2-2, which are underlain by a sedimentary unit which varies from all clay to all sand but is predominantly clay, silt, or clayey silt (labeled as upper clay in Fig. 2-2). The upper clay forms a confining layer separating the unconfined groundwater system in the shallow sand from the underlying regional gravel aquifer (RGA). The RGA is in the lower facies of continental deposits and is predominantly gravel with silt and sand. These deposits are sometimes found locally as all sand or as a cobbley gravel. The RGA has an average thickness of 9 m (30 ft) and is the major aquifer in the area. The RGA terminates at the slope formed by the Porters Creek clay beneath the southern part of the PGDP.

Near the PGDP, the continental deposits rest unconformably on the McNairy formation, which has a total thickness ranging from 69 m (225 ft) to 91 m (300 ft). The bottom of the Cretaceous sediments slopes southward on the truncated subcrop of Mississippian bedrock, which dips moderately to the northeast toward the Illinois Basin. Table 2-1 relates the regional stratigraphic column to hydrogeologic units.

The RGA is recharged by infiltration of precipitation through the shallow sand and upper clay layers and discharges to the Ohio River (Fig. 2-2). Typically, the heaviest rainfall occurs in the spring and the lightest in the fall. From 1951 to 1980, the mean annual precipitation at Paducah, Kentucky, was 116 cm (45.8 in). Davis et al. (1973) estimates that groundwater

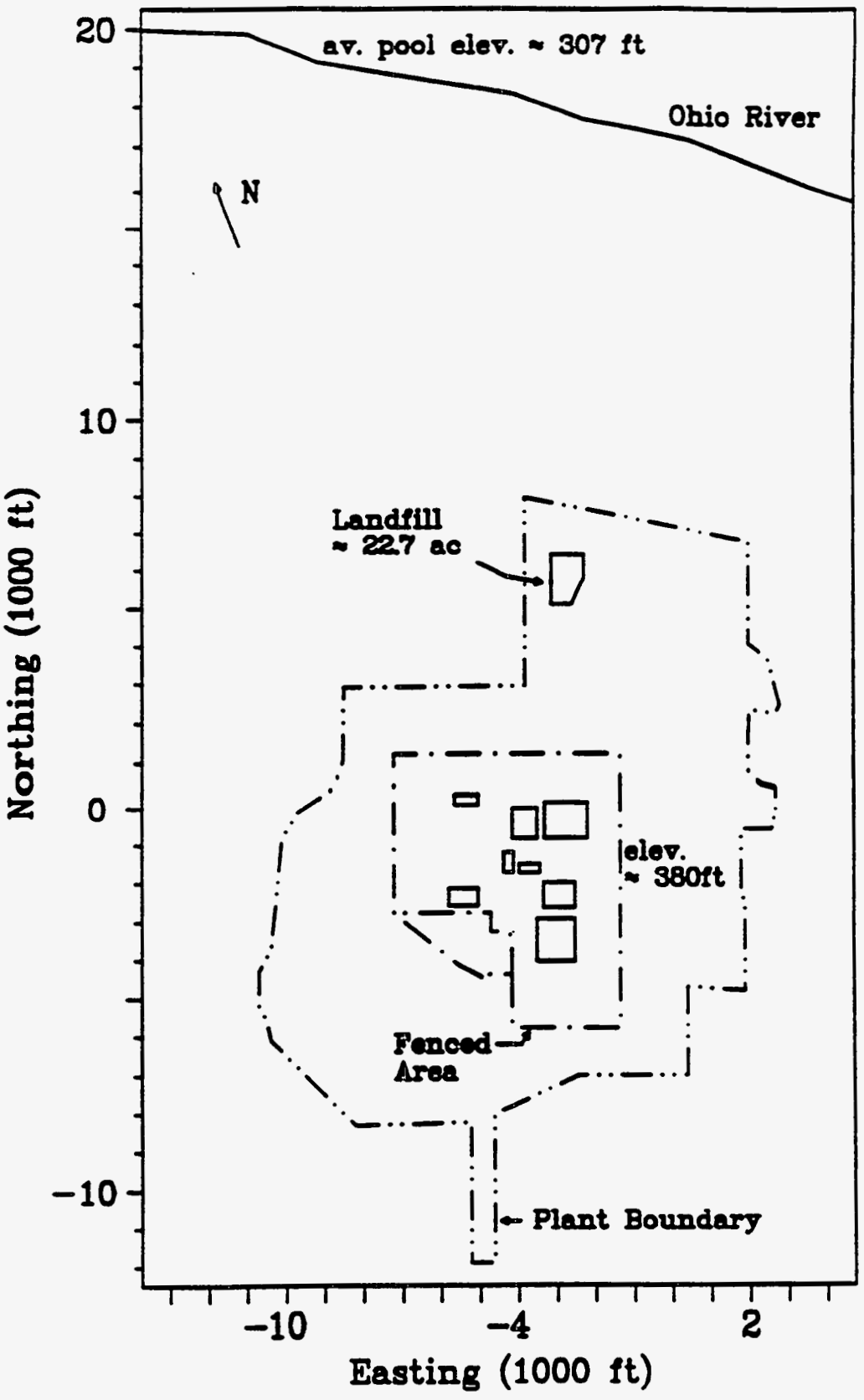


Fig. 2-1. Proposed landfill site at the Paducah Gaseous Diffusion Plant. (1000 ft = 305 m; 1 ac = 0.4 ha)

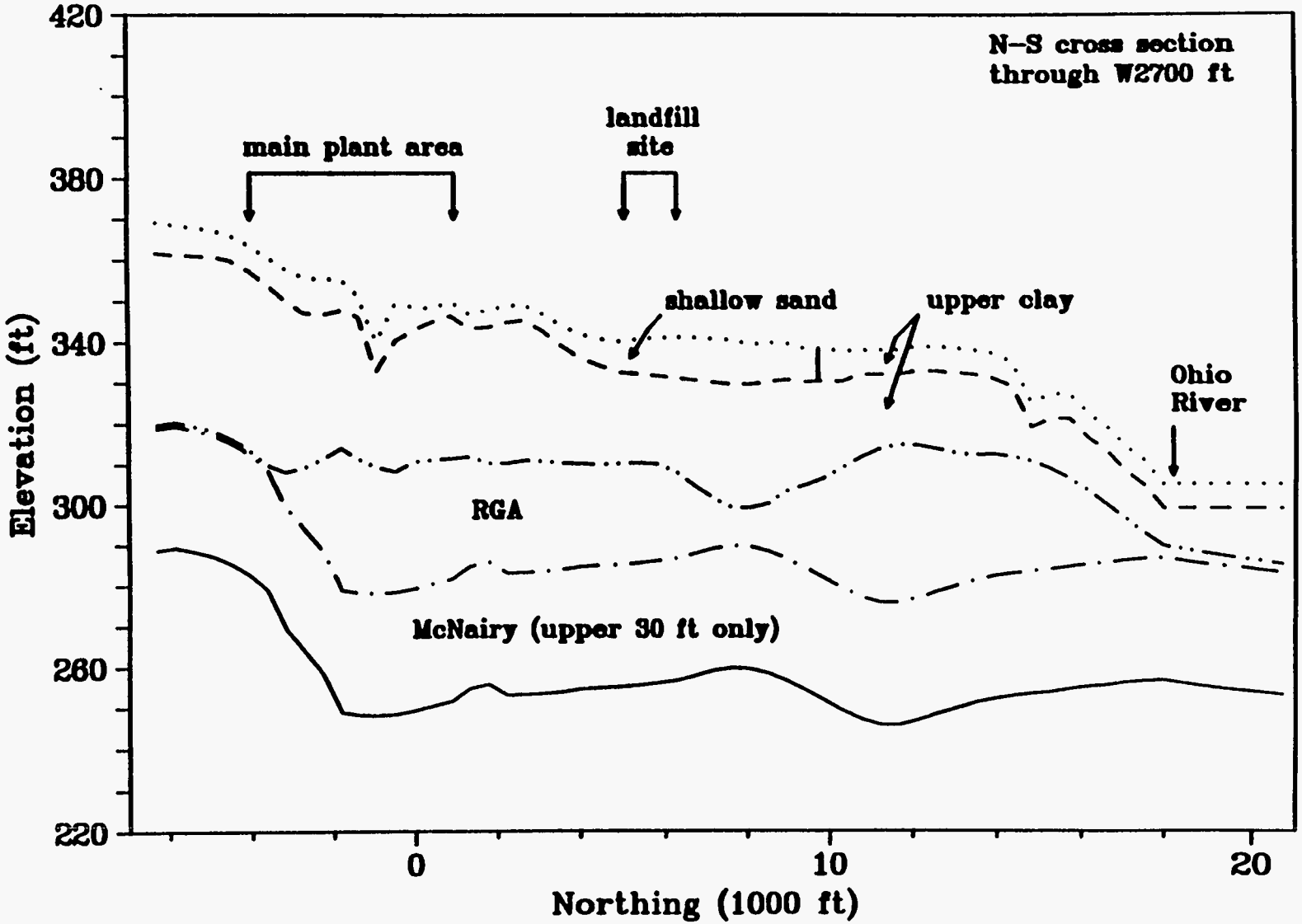
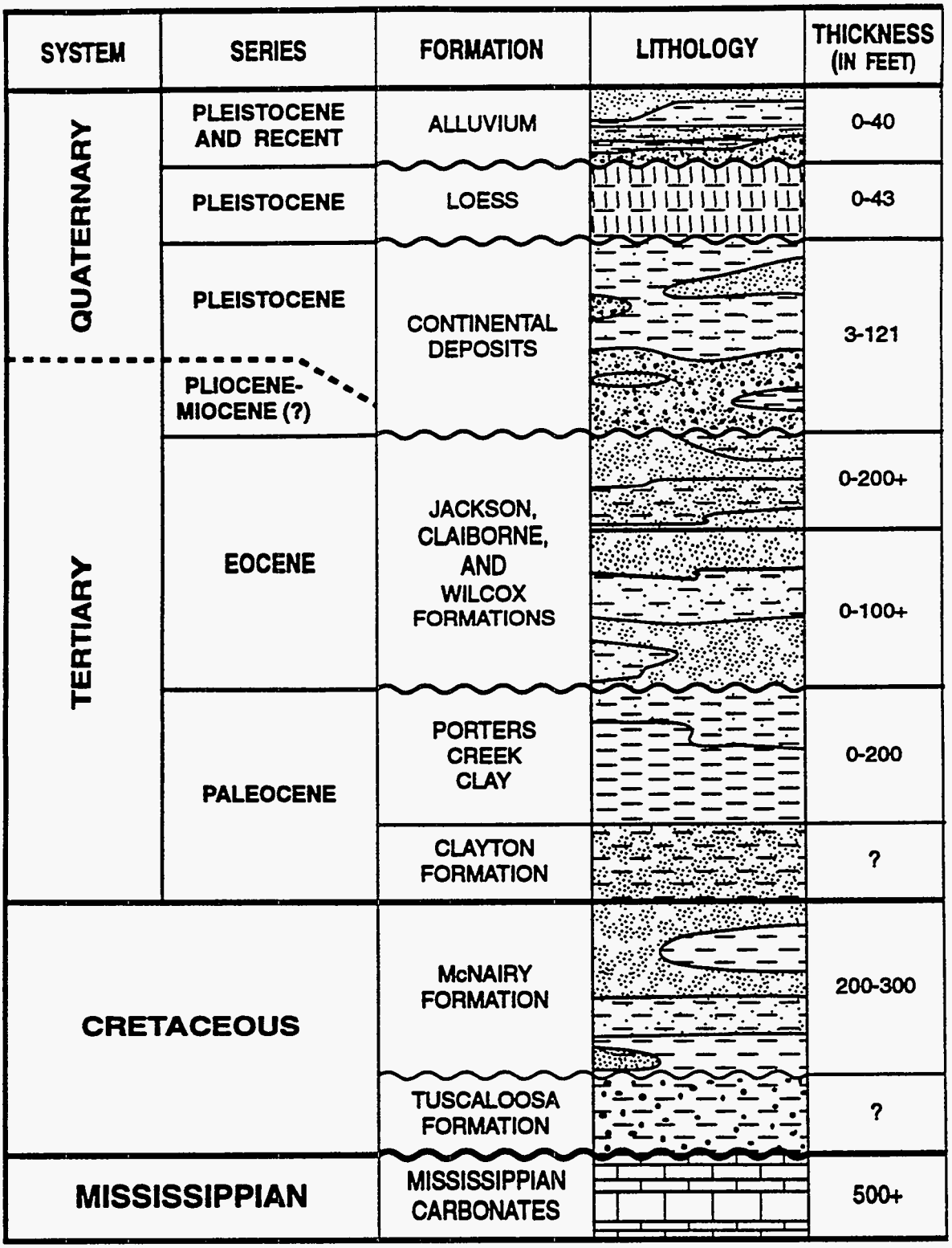


Fig. 2-2. The north-south stratigraphic cross section at West-2700 ft (see Fig. 2-1). The RGA is the main aquifer in the area. The shallow sand layer ends somewhere north of the Paducah Gaseous Diffusion Plant boundary. (1000 ft = 305 m)

Table 2-1. Columnar section of the region around the Paducah Gaseous Diffusion Plant



Source: Clausen et al. 1992.

recharge in the area is approximately 10–15% of precipitation, or about 12 to 30 cm/yr (4.7 to 7.0 in./yr). North of the PGDP, the RGA is in direct contact with the underlying McNairy Formation. The hydraulic communication between the two formations is uncertain, but because the McNairy Formation has a much lower hydraulic conductivity, the amount of contaminant entering it from the RGA is expected to be small. Approximately 120 m (400 ft) of Mississippian carbonate bedrock underlies the McNairy Formation.

Table 2-2 presents the ranges of hydraulic conductivities in the major lithologic units encountered at the PGDP and the surrounding area. These values are based on both field and laboratory measurements. Very low conductivities are observed in the loess, clay facies of the continental deposits, Porters Creek clay, and McNairy Formation. Higher conductivities are observed in the gravel facies that form the RGA.

Table 2-2. Ranges of hydraulic conductivity values for major lithologies near the Paducah Gaseous Diffusion Plant

Major lithographic units	Hydraulic conductivity values (m/d)[a]		
Loess	10^{-3}	to	10^{0}
Upper continental deposits (clay)	10^{-4}	to	10^{1}
Upper continental deposits (sand)	10^{-1}	to	10^{1}
Regional gravel aquifer	10^{0}	to	10^{3}
Porters creek clay	10^{-6}	to	10^{-3}
McNairy formation	10^{-4}	to	10^{1}

Source: Early et al. 1989.
[a]1 m = 3.28 ft

Although the PGDP does not use groundwater for supply purposes, nearby residents within one kilometer (0.6 miles) of the eastern site boundary use wells for their domestic needs (DOE 1989). The rates of groundwater withdrawal from these wells are not available. However, the per capita usage is estimated to range from 19 to 160 L/d (5 to 43 gpd) (DOE 1982). In August 1988, off-site groundwater contamination was discovered north of the PGDP (Rogers and Jett 1989). A community water line was extended to the residents with contaminated wells to provide a long-term water supply. At the present time, the RGA is not utilized north (down gradient) of the plant (Rogers et al. 1988).

2.1.3 Surface Water

The PGDP is located within the drainage areas of Big and Little Bayou Creeks, which meet about 5.6 km (3.5 miles) north of the site and discharge into the Ohio River near River Mile 948. As can be seen from Fig. 2-3, in the main plant area, surface water drainage is to the east and northeast toward Little Bayou Creek, and to the west and northwest toward Big Bayou Creek. Big Bayou Creek is a perennial stream whose drainage extends from approximately 4 km

(2.5 miles) south of the PGDP to the Ohio River. It flows toward the river along a 14-km (9-mile) course that passes along the western boundary of the plant. Little Bayou Creek originates in the neighboring wildlife management area and flows north toward the Ohio River along a 10-km (6.5-mile) course that includes part of the eastern boundary of the plant.

Big Bayou Creek is a 4820-ha (11,910-ac) watershed, and the watershed for Little Bayou Creek is 2400 ha (6000 ac) (Kornegay et al. 1991). In the absence of discharge from the plant, Big Bayou and Little Bayou Creeks would be essentially dry on numerous days from May–June

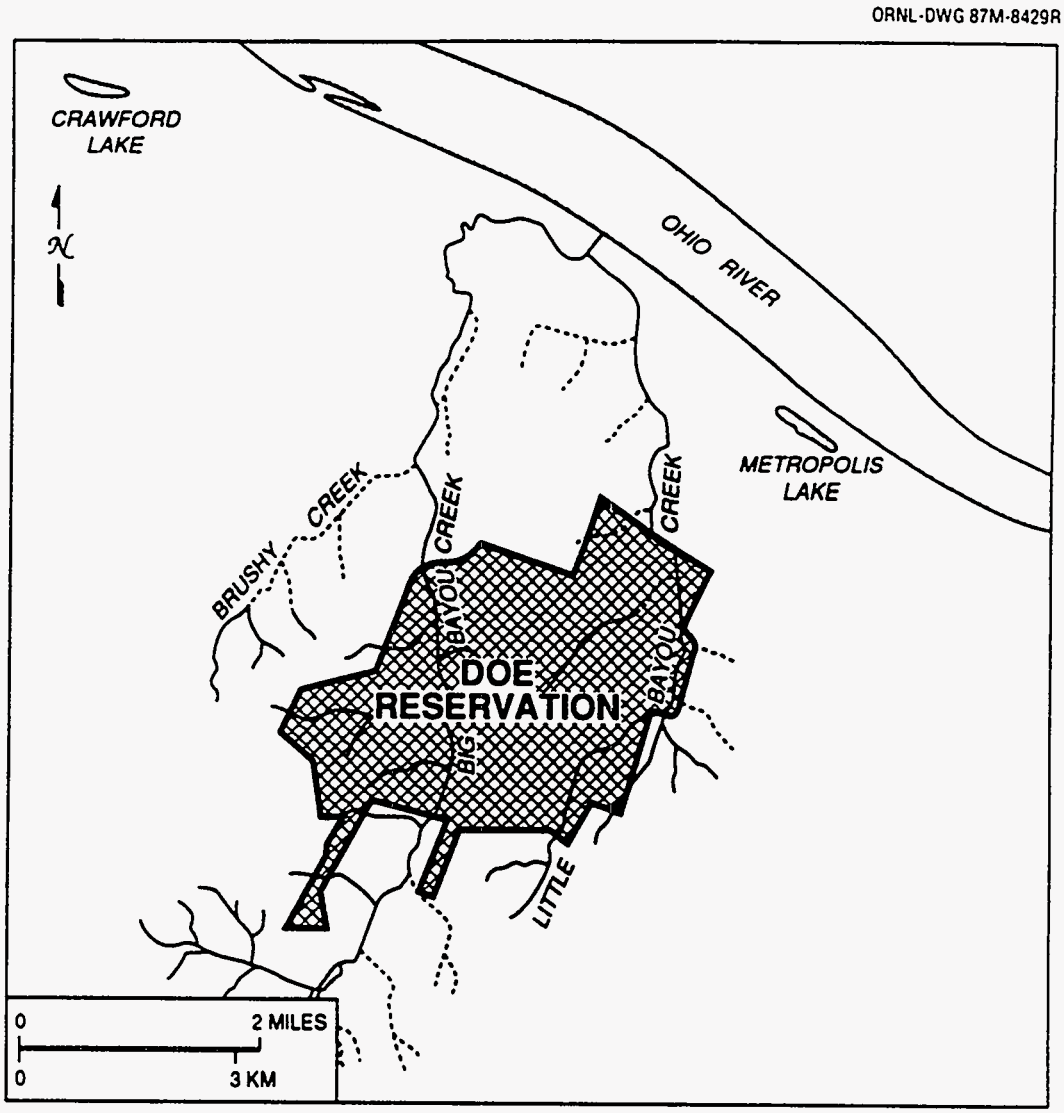


Fig. 2-3. Paducah Gaseous Diffusion Plant site surface hydrology systems.
Source: Kornegay et al. 1991.

through October–November and, on others, would discharge many millions of gallons per day to the Ohio River (DOE 1982). Effluents from PGDP operations constitute about 85% of the normal flow in Big Bayou Creek and 100% of the normal flow in Little Bayou Creek (Kornegay et al. 1991). The average discharge from the PGDP to the Ohio River via the two creeks is 0.18 m^3/s (6.3 ft^3/s), including blowdown from the cooling towers, treated sewage discharge, once-through cooling, and storm water runoff (DOE 1982; Saylor et al. 1990). Little Bayou Creek also receives an average of 0.003 m^3/s (0.1 ft^3/s) as storm water runoff from the Shawnee Power Plant (Saylor et al. 1990).

Both the Big Bayou and Little Bayou watersheds are predominantly rural in nature, with population densities on the order of 40 to 45 persons per km^2 (100 to 110 per $mile^2$) (DOE 1982). No domestic, commercial, or industrial water withdrawals are known from either Big Bayou or Little Bayou Creek (DOE 1982; Saylor et al. 1990). These creeks cannot be relied upon for a non-interruptible source of water supply. The creeks have limited interaction with the groundwater system near and beneath the PGDP. The interaction depends locally on the surface water levels and on the widths, slopes, and hydraulic properties of the creek bottoms.

2.2 WASTE CHARACTERISTICS

The wastes to be disposed of at the proposed solid waste landfill are generated during normal operations and are identified as non-radioactive wastes. Typical wastes to be disposed of include soils, wood, concrete, roofing and construction debris, and sanitary wastes generated in offices or similar areas. These waste materials generated within the PGDP may have radioactive material contents that include the naturally occurring radionuclides uranium and thorium, and may become slightly contaminated with uranium, thorium, and other radionuclides from unintended contact with radioactive material during plant operations. Wastes with the potential for being contaminated from plant operations are regarded as environmentally sensitive wastes and are managed separately. These wastes are transported to a trash sorting and monitoring facility for screening. Wastes registering above a prescribed operating limit that is above background are considered radioactive contaminated and unacceptable for disposal in the solid waste landfill. These wastes are returned to the generator for proper disposal as radioactive wastes (PGDP 1992). Wastes less than the prescribed operating limit are considered non-radioactive and disposed of in the existing solid waste landfill.

Operating limits at the proposed solid waste landfill are to be applied in the management of environmentally sensitive wastes to ensure that wastes disposed of at the landfill do not constitute an undue risk to public health and the environment, and can be ensured to be non-radioactive, consistent with the regulations of the Commonwealth of Kentucky. As a uranium enrichment facility, PGDP wastes have the potential to be contaminated primarily with uranium. Smaller amounts of ^{99}Tc, ^{230}Th, ^{232}Th, ^{237}Np, ^{238}Pu, ^{239}Pu, and ^{241}Am may also be present in PGDP wastes. The transuranic nuclides are derived from historical chemical operations and storage activities at the PGDP. Technetium at PGDP originated as part of the breeder reactor program (Feldman et al. 1993). Table 2-3 lists the half-lives and specific activities of radionuclides considered in this study. Other radionuclides of potential concern to protection of public health or the environment are not known or suspected to be in wastes generated at the PGDP that could be considered for disposal in the solid waste landfill.

Table 2-3. Half-lives and specific activities of the radionuclides considered in this analysis

Nuclide	Half-life (year)	Specific activity (Ci/g)
^{99}Tc	2.13×10^{5}	1.70×10^{-2}
^{230}Th	7.54×10^{4}	2.06×10^{-2}
^{232}Th	1.40×10^{10}	1.10×10^{-7}
^{234}U	2.45×10^{5}	6.24×10^{-3}
^{235}U	7.04×10^{8}	2.16×10^{-6}
^{236}U	2.34×10^{7}	6.49×10^{-5}
^{238}U	4.47×10^{9}	3.35×10^{-7}
^{237}Np	2.14×10^{6}	7.05×10^{-4}
^{238}Pu	8.77×10^{1}	1.71×10^{1}
^{239}Pu	2.41×10^{4}	6.21×10^{-2}
^{241}Am	4.33×10^{2}	3.43

2.3 WASTE CERTIFICATION AND DISPOSAL

As discussed in Sect. 2.2, wastes to be disposed of in the proposed solid waste landfill consist primarily of soils and construction and demolition debris. These waste materials have the potential to be contaminated with radionuclides present in plant operations. Wastes would be certified by waste operations staff to ensure that any radionuclides present are in quantities less than prescribed operating limits.

The location of the proposed solid waste disposal facility is discussed in Sect. 2.1.1. The plan view of the disposal facility is shown in Fig. 2-4. An east-west elevation of the landfill is shown in Fig. 2-5, which shows the final configuration of the landfill after filling with waste and emplacement of the cap. Leachate collected above the clay liner by the leachate collection system would be discharged to sediment ponds located within the permitted boundary after monitoring. The water table elevation beneath the facility fluctuates seasonally. The high water table elevation is estimated conservatively to be 105 m (345 ft) MSL, or about 3 m (10 ft) below the bottom of the clay liner. The approximate total waste volume of the facility is 1,190,000 m^3 (1,560,000 yd^3).

The proposed solid waste landfill would be filled in several phases from the southern to the northern end. Historically, landfilling of wastes at the PGDP has occurred at the rate of about 4000 m^3 (5000 yd^3) per year. Existing materials identified for disposal are estimated to amount to 19,000 m^3 (25,000 yd^3). This large volume of waste materials to be managed is expected to increase the disposal rate to 6100 m^3 (8000 yd^3) per year. At an average disposal rate of 5500 m^3

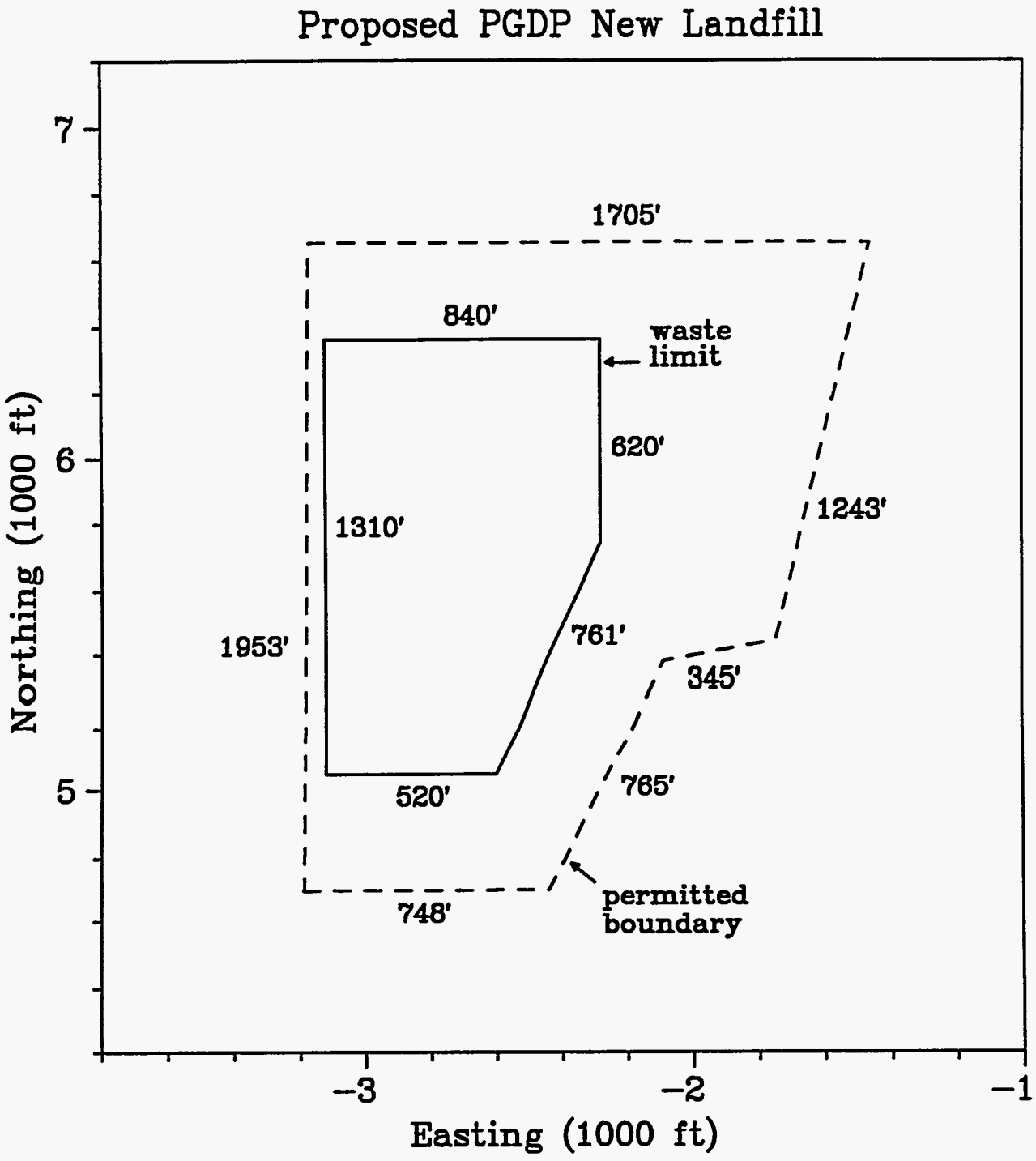


Fig. 2-4. Dimensions of the proposed landfill. (1000 ft = 305 m)

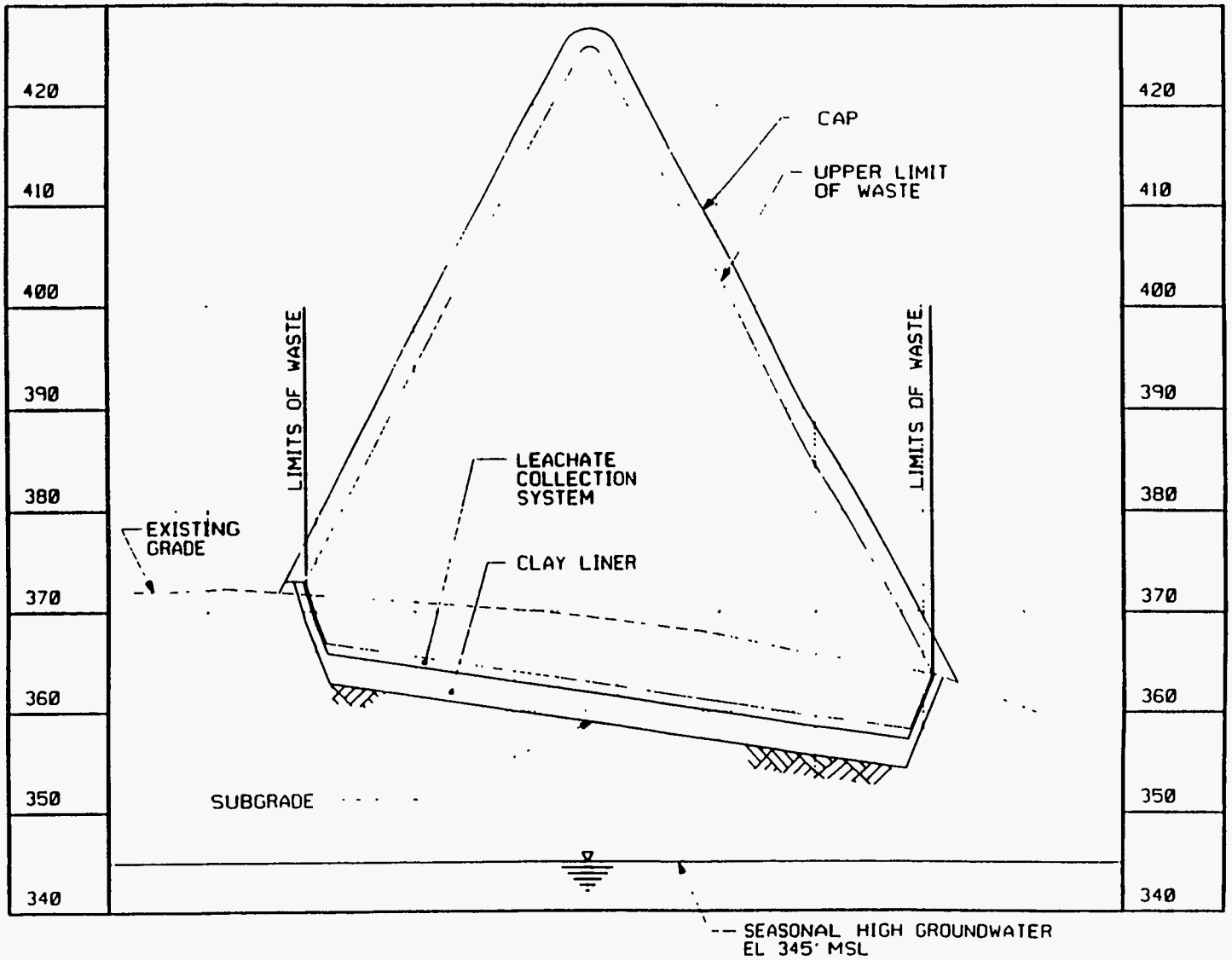


Fig. 2-5. The east-west cross section at N=5300 ft (see Fig. 2-4) showing the landfill configuration when filled and capped. Each horizontal and vertical scale is 100 and 10 ft, respectively. (1000 ft = 305 m)

(7200 yd^3) per year, the waste disposal area has an approximate lifetime of 200 years. For the purposes of this analysis, the southern half of the proposed site with an area of 4.4 ha (11 ac) is treated as a disposal site for the next 100 years (Fig. 2-6). The site is subdivided into five parts being filled sequentially from the southern end. Each part is assumed to have a 20-year capacity. Each part is assumed to be closed after 20 years and have an institutional control period of 30 years. The generation and release of leachate to groundwater from the successive parts of the landfill are superimposed in the analysis. Further discussion of the modeling of the disposal facility is presented in Sections 3.1 and 3.3.1.

Wastes disposed of in the solid waste landfill would be compacted and covered with excavated soils, consistent with the permit application submitted to the Kentucky Department of Solid Waste. The overall mixing of wastes and cover material results in a 1:1 ratio of waste to soil within the landfill. The details of the final cover for the landfill are identified in the permit application. For the purposes of this analysis, the operating period for each part of the landfill is assumed to be 20 years. The final cover is assumed to be installed over each part at the end of the operating period. The final cover is assumed to operate, as designed, for 30 years after emplacement and then to degrade. While this assumed mode of operations is not entirely consistent with the permit application for the solid waste landfill, the actual operation of the landfill over the next 100 to 200 years is uncertain. The assumed operation of the landfill in this analysis was selected to provide a reasonably conservative representation of landfill performance for developing operating limits.

Inadvertent intrusion onto the disposal site would not be a credible occurrence as long as disposal operations are taking place at the site. In this analysis, exposures of inadvertent intruders are assumed to occur immediately after closure of the entire facility, that is, after all disposals have been completed and the 30-year institutional control period of the last part of the landfill has ended. No credit is taken for the possibility that extended active or passive institutional controls may be maintained over the disposal site to preclude inadvertent intrusion. Additionally, the actual time when the entire disposal facility will be closed is approximate and depends on future waste generation rates. However, since the radionuclides of concern to waste disposals at the landfill have relatively long half-lives, and the models for estimating dose to inadvertent intruders assume that radioactivity is not removed from the disposal facility by leaching, the results of a dose analysis for inadvertent intrusion would not depend significantly on the time after disposal when the exposures are assumed to occur.

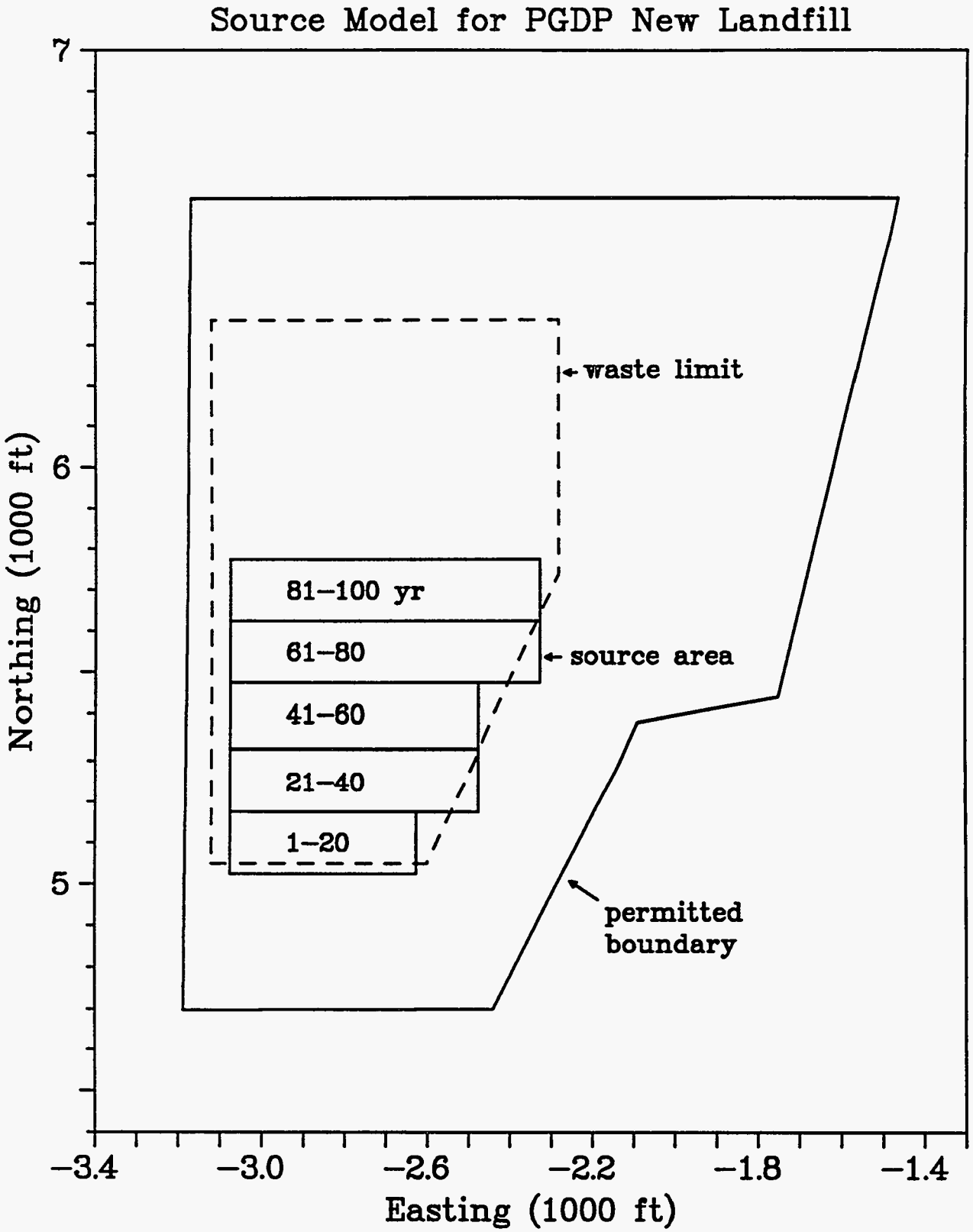


Fig. 2-6. Schematic diagram showing, for simulation purposes, the division of the southern half of the landfill site (source area) into five parts. (1000 ft = 305 m)

3. ANALYSIS OF PERFORMANCE

This section describes the methods used to analyze the performance of the proposed solid waste landfill and provides an overview of the data used in the analysis. The release of radionuclides from the proposed landfill is treated as the source term for the analysis of off-site transport. A qualitative discussion of the source term is given in Sect. 3.1. The quantitative details are presented in Sect. 3.3, where the models used, assumptions made, and the analysis methodologies for groundwater flow and mass transport are discussed in detail. Pathways for the radionuclides to travel from the landfill to off-site individuals and scenarios for exposure of inadvertent intruders are presented in Sect. 3.2.

3.1 SOURCE TERM

The release of radionuclides with time from the proposed solid waste landfill was calculated to provide a source term for the contaminant transport modeling. The modeling was based on the finite difference code FTWORK developed by GeoTrans (1990a). For modeling purposes, the proposed landfill site was subdivided into parts (roughly representing different phases of the landfill) which were filled in succession over time. The source term model for each part assumed that waste was placed uniformly over each part at a constant rate during its period of operation of 20 years and then was covered at closure to reduce infiltration. The cover and liner of each part were assumed to remain intact for a period of 30 years after covering, and the hydraulic conductivities were assumed to deteriorate exponentially afterwards to become those of local soils (see Sect. 3.3.1).

Leachate generation and transport by water infiltrating through the waste was modeled using a simplified mathematical representation discussed in detail in Sect. 3.3.1. The assumed flux of infiltrating water into the wastes was based on the average groundwater recharge of the area of 12 cm/yr. The concentration of radionuclides in the leachate was calculated using an assumed distribution coefficient (K_d) of each radionuclide in the waste. The leachate flux was partitioned between a portion collected by the leachate collection system and a small portion leaking directly into the soils beneath the disposal facility. Leachate collected by the collection system was removed from the model during the operations and institutional control periods. At the end of the 30-year institutional control period, all of the leachate was assumed to infiltrate into the soils beneath the disposal unit, because the leachate collection system was assumed conservatively to fail. The time-dependent contaminant releases of leachate to the groundwater from all parts were superimposed in the transport calculations.

3.2 PATHWAYS AND SCENARIOS

In this section, the time periods of concern are first discussed. The dominant transport pathways for radionuclides in the landfill to reach the public and possible exposure scenarios are then presented. The latter are divided into two general categories: exposures of off-site members of the public and exposure scenarios for inadvertent intruders.

3.2.1 Time Periods of Concern

Three time periods are considered: the operational period, the institutional control period, and the post-institutional control period.

The operational period is the period during which waste disposal operations occur and waste is actively emplaced within the solid waste landfill. This period could extend for as long as 200 years, as discussed in Sect. 2.3. For the purposes of this analysis, a 100-year operational period has been assumed, which is divided into five 20-year parts corresponding to the five parts of the southern portion of the solid waste landfill. The actual division of the landfill into the phases of operations and rates of site utilization may differ from this representation depending on the generation rates of waste. However, an assumption of independent operations of each part of the landfill and the independent closure and degradation of each part of the landfill leads to quicker and larger releases of leachate than are anticipated from the actual landfill.

The institutional control period for each part of the landfill analyzed is the period after the cover is emplaced during which periodic maintenance and monitoring activities are conducted such that the cover and liner remain intact. During this period, leachate is still being collected. The length of this period is assumed to be 30 years. The operational period and institutional control period for the various parts of the landfill overlap. From the beginning of the first part to the end of the institutional control period of the last part, the entire landfill is considered to be controlled by the PGDP such that inadvertent intrusion into the waste by a member of the public is not considered reasonable (see Sect. 3.2.4 for discussion of inadvertent intrusion).

The post-institutional control period begins when the landfill is no longer maintained by the PGDP and could be accessed by the public. It starts at the end of the institutional control period of the last part of the landfill. Because of the potential presence of long-lived radionuclides in the waste, inadvertent intrusion into radioactive materials may occur during this period. During the post-institutional control period for each part, the leachate collection system is assumed to fail. All leachate generation during this period is allowed to infiltrate directly to the underlying aquifers. Also, the cover and liner are assumed to decay with a decay constant of 0.1 y^{-1} (see Sect. 3.3.1). The hydraulic conductivities of each part of the landfill are assumed to return to those of the native soils when the cover and liner are fully degraded. The post-institutional control period for the early part of the landfill overlaps with the operational and institutional control periods of the later parts of the landfill. As a result, the model admits fully degraded conditions in the first parts of the landfill while active operations are being conducted in the last parts of the landfill. This has been incorporated into the analysis to provide a conservative but reasonable representation of the overall performance of the solid waste landfill.

3.2.2 Transport Pathways

Potential transport of radionuclides from the landfill to off-site members of the public may occur by air and water pathways. Off-site transport via the atmospheric pathway is believed to be relatively unimportant because there are no volatile radionuclide species present; leaching tends to transport the radionuclides to the saturated zone, and nonvolatile radionuclides are not readily suspended into the air at a humid site. In addition, annual effective dose equivalents from inhalation are included in the exposure scenarios for inadvertent intruders, and the results for

these scenarios involving exposure on top of the proposed landfill should be more restrictive than those for the air pathway at off-site locations.

As can be seen from Fig. 2-3, the surface waters that might be affected by the proposed landfill operations are Little Bayou Creek and the Ohio River. In the absence of discharge from the PGDP, Little Bayou Creek can be expected to be essentially dry on numerous days from May–June through October–November. Effluents from the PGDP operations typically constitute 100% of the normal flow in Little Bayou Creek (Kornegay et al. 1991). Little Bayou Creek cannot be considered a reliable drinking water source or a suitable location for swimming. Although the Ohio River is considered a source for drinking water and is used for swimming, any releases from Little Bayou Creek to the Ohio River would result in very low doses to individuals because of the dilution provided by the Ohio River, which has a high average flow rate of 7500 m^3/s (265,000 ft^3/s) (DOE 1982).

Thus, this assessment of the proposed solid waste landfill assumes that transport of radionuclides in groundwater is the principal mechanism for removal of radionuclides from the landfill, and doses to off-site individuals are assumed to result mainly from drinking contaminated groundwater. Off-site doses resulting from possible contamination of surface water and from releases to the atmosphere are assumed to be less important.

3.2.3 Exposures of Off-Site Members of the Public and Protection of Groundwater

As discussed in Sect. 3.2.2, radionuclides placed in the proposed landfill are assumed to be released and transported to off-site locations by the groundwater pathway, and other potential pathways, such as releases to air or surface waters, are assumed to be relatively unimportant. In principle, several exposure pathways for off-site individuals involving the use of contaminated groundwater could occur. These include direct ingestion of contaminated waste from the source (i.e., the drinking water pathway), ingestion of vegetables obtained from a garden which is irrigated with contaminated water, and ingestion of milk and meat obtained from livestock which drink contaminated water or consume pasture which is irrigated with contaminated water. However, as discussed in Sect. 1, previous analyses of exposure pathways for off-site individuals (ORNL 1994; MMES et al. 1994), which are directly applicable to the PGDP site, have indicated that the dose from the drinking water pathway is by far the most important for the radionuclides of concern to waste disposals at the site, and that the dose from all other exposure pathways involving the use of contaminated groundwater would be insignificant.

Therefore, in this analysis, estimates of dose for off-site individuals from releases of radionuclides into groundwater consider only the drinking water pathway. As discussed in Sect. 1, the analysis assumes that the effective dose equivalent for off-site individuals from the drinking water pathway should be limited in accordance with the performance objective of 4 mrem (0.04 mSv) per year. This performance objective is used in conjunction with estimates of dose per unit concentration of radionuclides from the drinking water pathway to determine limits on concentrations of radionuclides in groundwater at the assumed off-site receptor location. These limits then are used in conjunction with the models for release and transport of radionuclides to estimate the limits on concentrations of radionuclides in solid waste in the disposal facility which would correspond to the performance objective for the drinking water pathway for off-site individuals.

3.2.4 Exposure Scenarios for Inadvertent Intruders

In performance assessments for low-level radioactive waste disposal facilities, several different scenarios for inadvertent intrusion are typically considered. These scenarios include the agriculture, resident, discovery, and post-drilling scenarios (ORNL 1994; MMES et al. 1994). However, many of these scenarios are appropriate only for facilities which are constructed using engineered barriers which would be effective in deterring inadvertent intrusion into the waste by normal excavation activities that might occur at the site. Since the proposed solid waste landfill will be constructed without engineered barriers, the most appropriate scenario for inadvertent intrusion into solid waste at the site, and the scenario which generally gives the highest estimates of dose to inadvertent intruders, is the agriculture, or homesteader, scenario.

The agriculture-homesteader scenario assumes that an inadvertent intruder arrives at the site immediately after the end of the active institutional control period. The intruder is assumed to construct a house directly on top of the landfill such that the foundation extends into the wastes. Waste materials are assumed to be indistinguishable from native soil. Wastes are assumed to be exhumed in the construction of the house and mixed with native soils in a vegetable garden. The following pathways are assumed to occur:

1. ingestion of vegetables grown in the contaminated garden soil,
2. direct ingestion of contaminated soil from the garden in conjunction with vegetable ingestion,
3. external exposure to contaminated soil while working in the garden or residing in the house, and
4. inhalation of radionuclides suspended into the air from contaminated soil while working in the garden or residing in the house.

As discussed in Sect. 1, the analysis assumes that the effective dose equivalent for any inadvertent intruder should be limited in accordance with the performance objective of 4 mrem (0.04 mSv) per year. The performance objective is used in conjunction with estimates of dose per unit concentration of radionuclides for the agriculture scenario to determine limits on concentrations of radionuclides in solid waste in the disposal facility. These limits then are compared with the corresponding limits obtained from the analysis for the groundwater transport pathway described in Sect. 3.2.3. For any radionuclide, the more restrictive of the concentration limits for the groundwater pathway and exposures of inadvertent intruders then provides the basis for the operating limit for the solid waste landfill.

3.3 MODELS, ASSUMPTIONS, AND ANALYSIS METHODOLOGY

This section describes the models adopted, the assumptions made, and the computational methods used to estimate doses and calculate operating limits. The assumptions are used to implement the conceptual models and to obtain quantitative results.

3.3.1 Source Model

The basic assumptions used in the modeling of the source terms are as follows:

1. the complex physical forms of wastes can be approximated as simple forms (e.g., uniform slabs);
2. the average concentration of each radionuclide in the waste and in the water infiltrating through the waste is related by a distribution coefficient K_d;
3. radioactive decay is accounted for at the end of the transport calculation using the total travel time for the radionuclide; and
4. possible solubility limits of radionuclides are not considered.

As a result of the last assumption, the calculated concentration limits would be conservative for radionuclides whose release is controlled by solubility limits.

For simulation purposes, the southern half of the proposed landfill site was subdivided into five parts which were filled in succession over a period of 100 years (Fig. 2-6). The source model for each part assumed that waste was placed uniformly, without the addition of clean soil for a daily cover, over the whole part at a constant rate during its period of operation, t_o = 20 years, and a final cover was emplaced at closure to reduce infiltration. The cover and liner of each part were assumed to remain functional for 30 years after the closure (from t_o to t_1, with t_1 being 50 years). The hydraulic conductivities of the cover and liner were assumed to degrade afterwards to become those of native soils with an assumed decay constant, a. The value of a was assumed to be 0.1 y^{-1}. The infiltration of water through the waste was considered in three phases. In the first phase, disposal operations, the infiltration is assumed to be limited by the natural infiltration of the soils, because the waste is merely covered by the daily soil cover over the waste. In the second phase of institutional control, the infiltration is substantially reduced as a result of the cover installed over the waste during closure. In the third phase of post-institutional control, the cover degrades and infiltration reverts to that of the native soils.

The volume of the waste for each part of the landfill was assumed to be V(t), with $dV/dt = b = 5500$ m^3/yr (7200 $yard^3/yr$) for $t < t_o$ and $dV/dt = 0$ afterwards. The flux of water from precipitation passing through the waste, F(t), was assumed to have the following mathematical form:

$$F(t) = f_1 \quad , \qquad for \quad t \le t_o$$

$$= f_2 \quad , \qquad for \quad t_o < t \le t_1$$

$$= \frac{f_2 f_3}{f_2 + (f_3 - f_2)\, e^{-a(t-t_1)}} \quad , \qquad for \quad t > t_1 \qquad (1)$$

where f_1, f_2, and f_3 are constants. This assumed form is intended to represent the effects of the liner and cover and their degradation after closure. The value of f_1 was taken to be the average groundwater recharge of the area, 12 cm/yr (4.7 in./yr), developed as part of the calibration of the

groundwater-flow model. This groundwater recharge value is at the lower end of the range reported by Davis et al. (1973) and can be justified by the compaction of soils during facility construction and operation. The value of f_2 was assumed to be $f_1/10$. The reduction in infiltration is an estimate that takes into consideration the intended performance of the cover system designed for the solid waste landfill. Actual reductions in infiltration from cover installation could reasonably exceed the value selected for this analysis. As time increases, $t > t_1$, F(t) increases from f_2 to represent the degradation of the cover and the liner during the post-institutional control period. At large t, Eq. (1) reduces to $F(t) = f_3$, which was also assumed to be 12 cm/yr (4.7 in./yr). This groundwater recharge value is consistent with the lower end of the range of infiltration values for the site (Davis et al. 1973), and is presumed to be a reasonable estimate given the presence of the liner and cover system for the facility. The functional behavior of F(t) is illustrated in Fig. 3-1. For $t < t_1$, a fraction of the leachate, $f_w F(t)$, was assumed to infiltrate to the water table, and the rest of the leachate was assumed to be collected by the leachate collection system and removed from the landfill. The magnitude of f_w was assumed to be 0.1 for $t < t_1$ and 1 for $t > t_1$. The increase of f_w for $t > t_1$ is the result of the assumed failure of the leachate collection system.

The rate of contaminant emplacement in the landfill can be written as

$$I(t) = C_o \frac{dV}{dt} \quad , \tag{2}$$

where C_o is the initial contaminant concentration in waste prior to disposal. The rate of contaminant removal from the waste, S(t), was assumed to be proportional to the average concentration C(t) and the water flux F(t),

$$S(t) = \frac{1}{DK_d} C(t) F(t) \quad , \tag{3}$$

where D is the density of waste and K_d is the distribution coefficient. Note that C(t)/D has dimensions of contaminant per unit mass.

The contaminant removal rate, S(t), and emplacement rate, I(t), and the average concentration in waste, C(t), are related by

$$C(t) = \frac{\int [I(t) - S(t)]\, dt}{V(t)} \quad . \tag{4}$$

By eliminating C(t) from Eqs. (3) and (4), a differential equation for S(t) is obtained,

$$DK_d \frac{d}{dt} \frac{V(t)S(t)}{F(t)} + S(T) = I(t) \quad . \tag{5}$$

This equation was solved for each of the three time intervals separated by t_o and t_1, and for each of the five parts of the modeled area. The various rates were calculated and used in the transport calculations. The release rate for one part of the modeled area for uranium

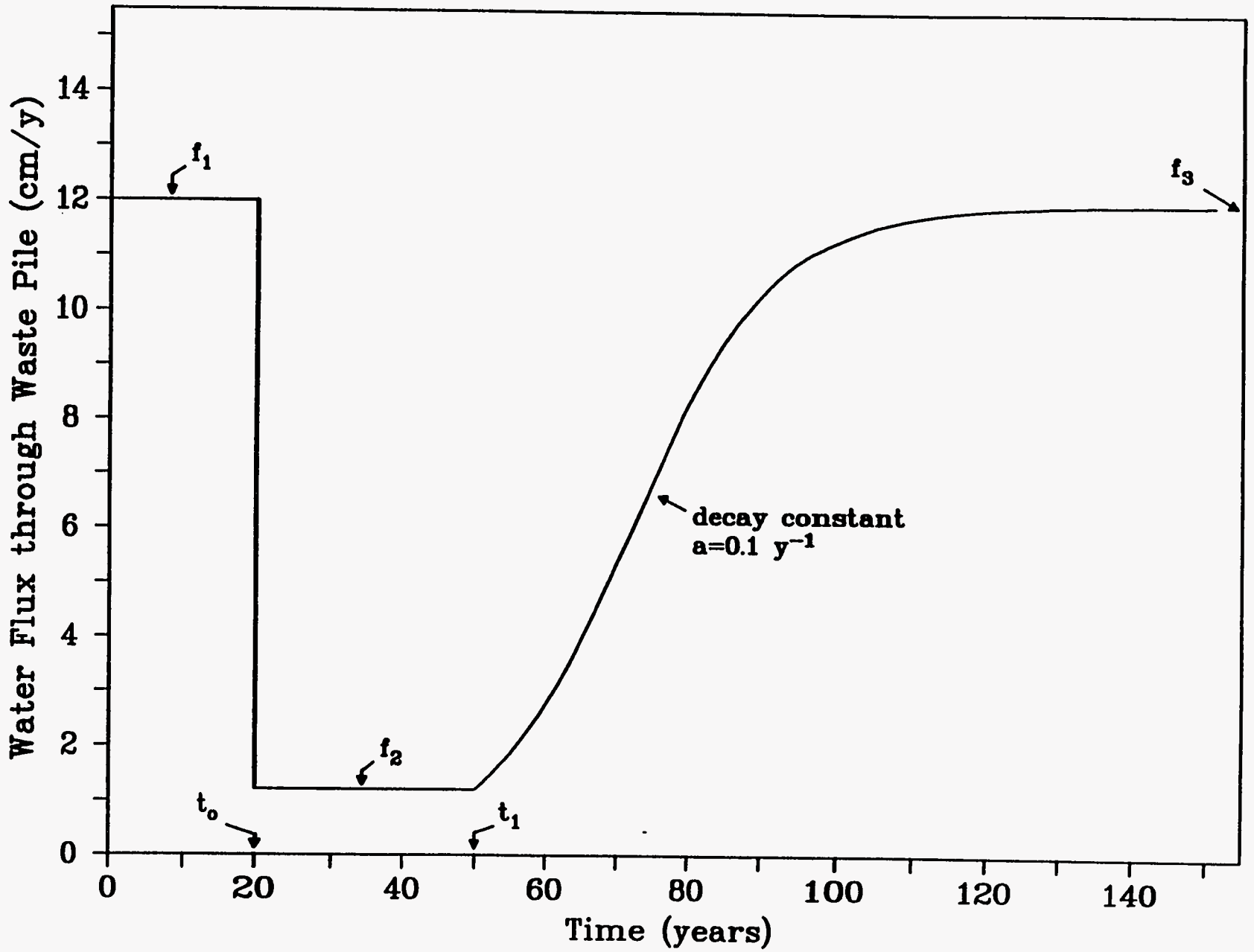


Fig. 3-1. Assumed water flux through the waste pile as a function of time.

(see Fig. 2-6) is shown, as an example, in Fig. 3-2. The distribution coefficient for uranium in waste, K_d=27 mL/g, was estimated from experimental results provided by the PGDP (Story 1993). (K_d values for other isotopes are discussed in Sect. 3.3.2.) This value of K_d is responsible for the slow concentration change shown in the figure. The time dependence of the contaminant release rate to groundwater, $S(t)f_w$, during the first 100 years reflects the assumed effects of the liner and cover.

3.3.2 Groundwater Flow and Mass Transport

The finite difference computer code FTWORK (GeoTrans 1990a) was used for modeling groundwater flow and contaminant transport for this analysis. The size of the rectangular modeled area, as shown in Fig. 3-3, is about 29 km^2 (18 $miles^2$). Also shown in the figure are some of the boundary conditions used in the groundwater flow model. The model uses the Ohio River as the northern boundary and is composed of four layers: the shallow sand, the upper clay, the RGA, and the McNairy Formation. Only the upper 10 m (30 ft) of the McNairy Formation immediately beneath the RGA is included in the model as the fourth layer (Fig. 2-2).

The three-dimensional groundwater flow model grid consists of 41 columns (south-north) and 61 rows (west-east) with a uniform block size of 137 m by 137 m (450 ft by 450 ft) (Fig. 3-3). The model was calibrated by comparing calculated heads with average observed heads (monthly measured data from July 1991 to June 1992) at 45 wells in the RGA and 30 wells in the shallow sand and upper clay layers. Figures 3-4 and 3-5 compare the observed and calculated head contours in the RGA. The root-mean-square difference for the calibrated model is 1.3 m (4.4 ft). Table 3-1 summarizes the hydraulic properties obtained from the model calibration. These values are comparable to the values obtained by GeoTrans (1990b, 1992) in two groundwater flow models for the same general area shown in Fig. 3-3. The recharge for the calibrated model is 12 cm/yr (4.7 in./yr), which is about 10% of the 117-cm (46-in) average annual precipitation.

Table 3-1. Hydraulic conductivities used in the calibrated groundwater flow model

	Conductivity (ft/d)[a]	
Layer	Horizontal	Vertical
Shallow sand (layer 1)	3.0	3.0
Upper clay (layer 2)	1.0	0.00092
RGA (layer 3)	560	560
McNairy (layer 4)	0.01	0.01

[a]1 ft/d = 3.53×10^{-4} cm/s

The 137 m by 137 m (450 ft by 450 ft) block size for the groundwater flow model was chosen so that the total number of blocks in each layer, 41 by 61, was computationally manageable and the model grid covered the area of interest (Fig. 3-3). For contaminant transport

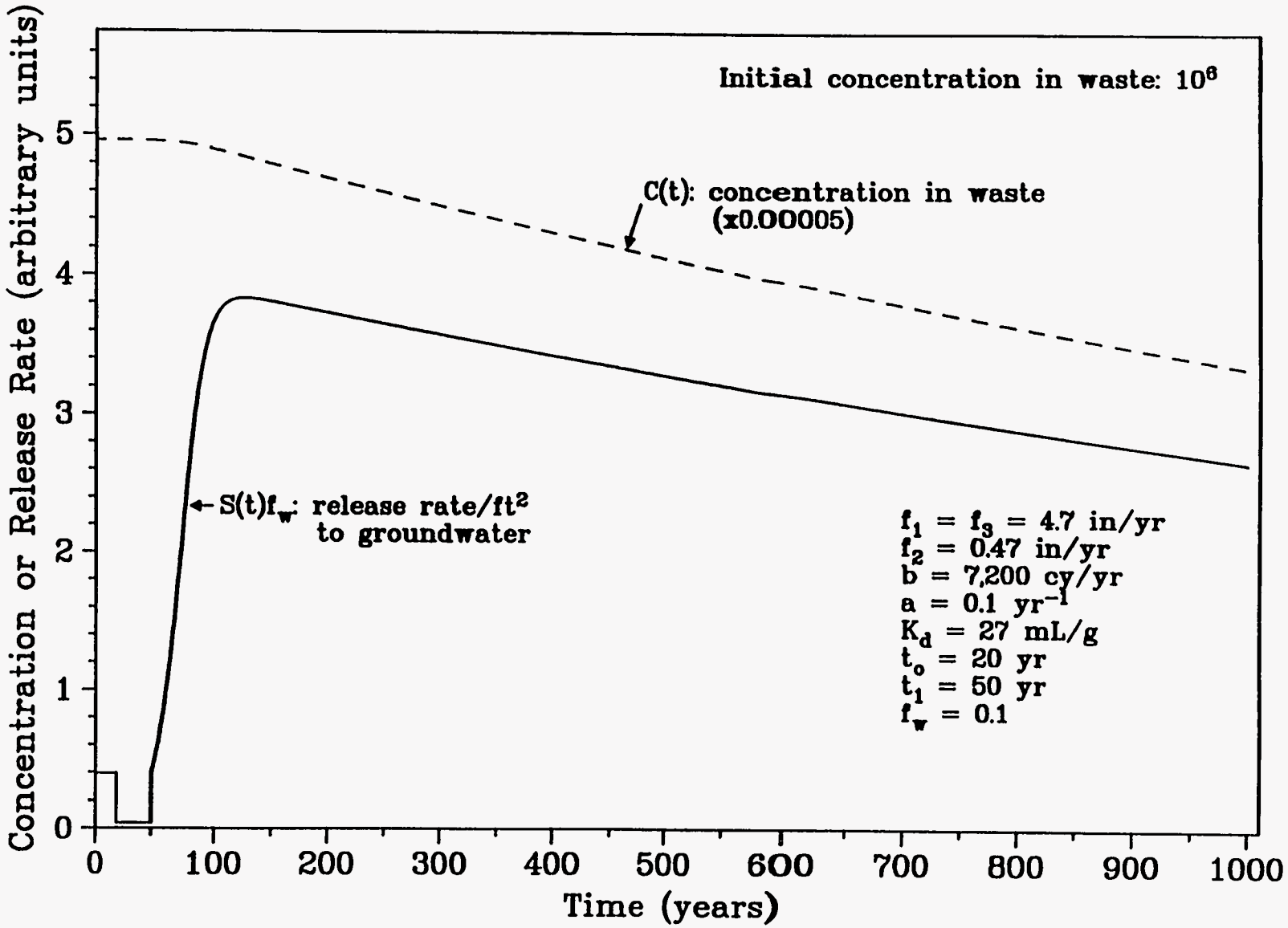


Fig. 3-2. Time-dependent uranium concentration in the waste pile and release rate to the groundwater for Part 1 of the source area. (1 in. = 2.54 cm; 1 cy = 0.76m^3)

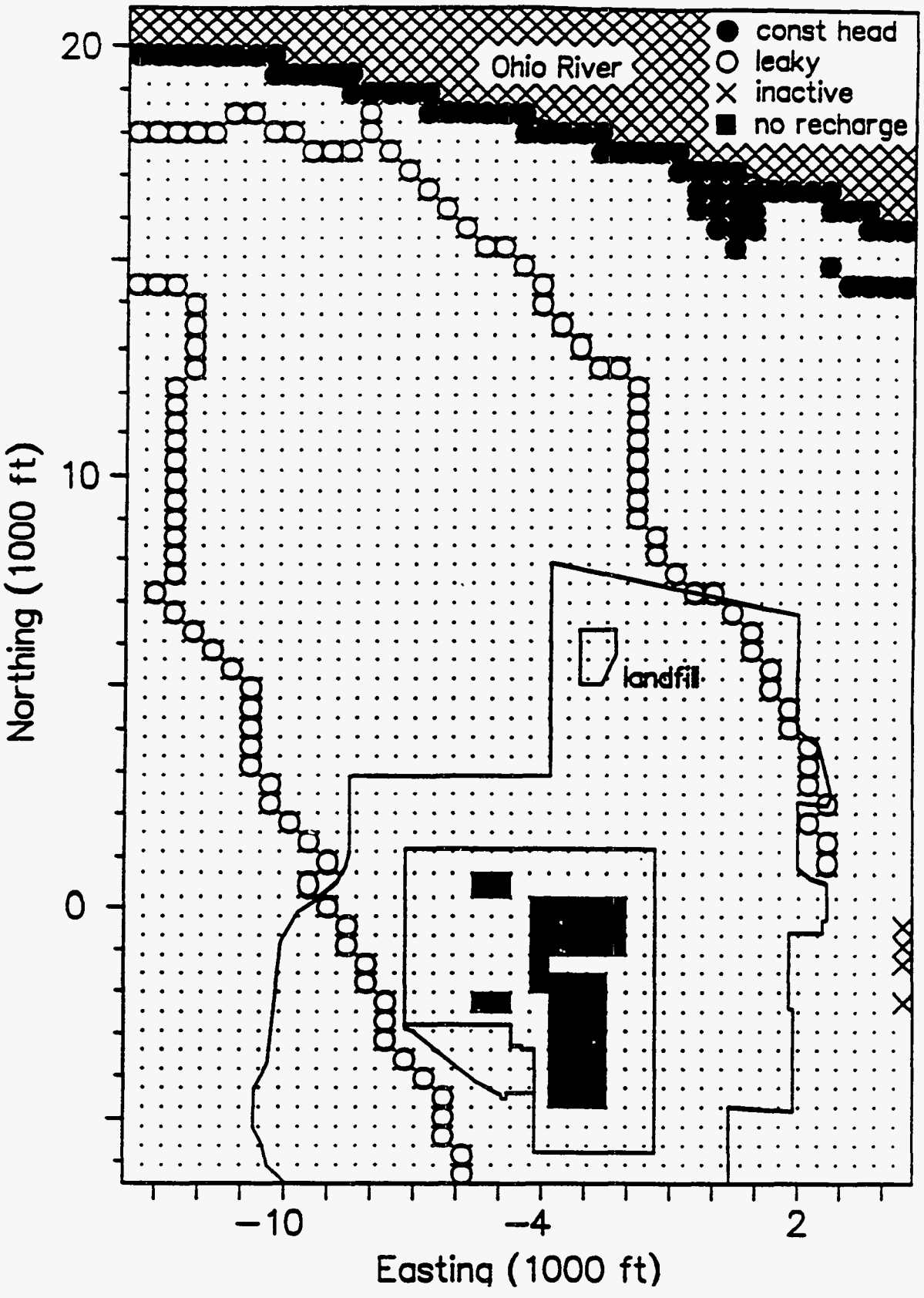


Fig. 3-3. Area included in the groundwater flow model. Boundary conditions include constant head, leaky, inactive, and no recharge grid blocks. (1000 ft = 305 m)

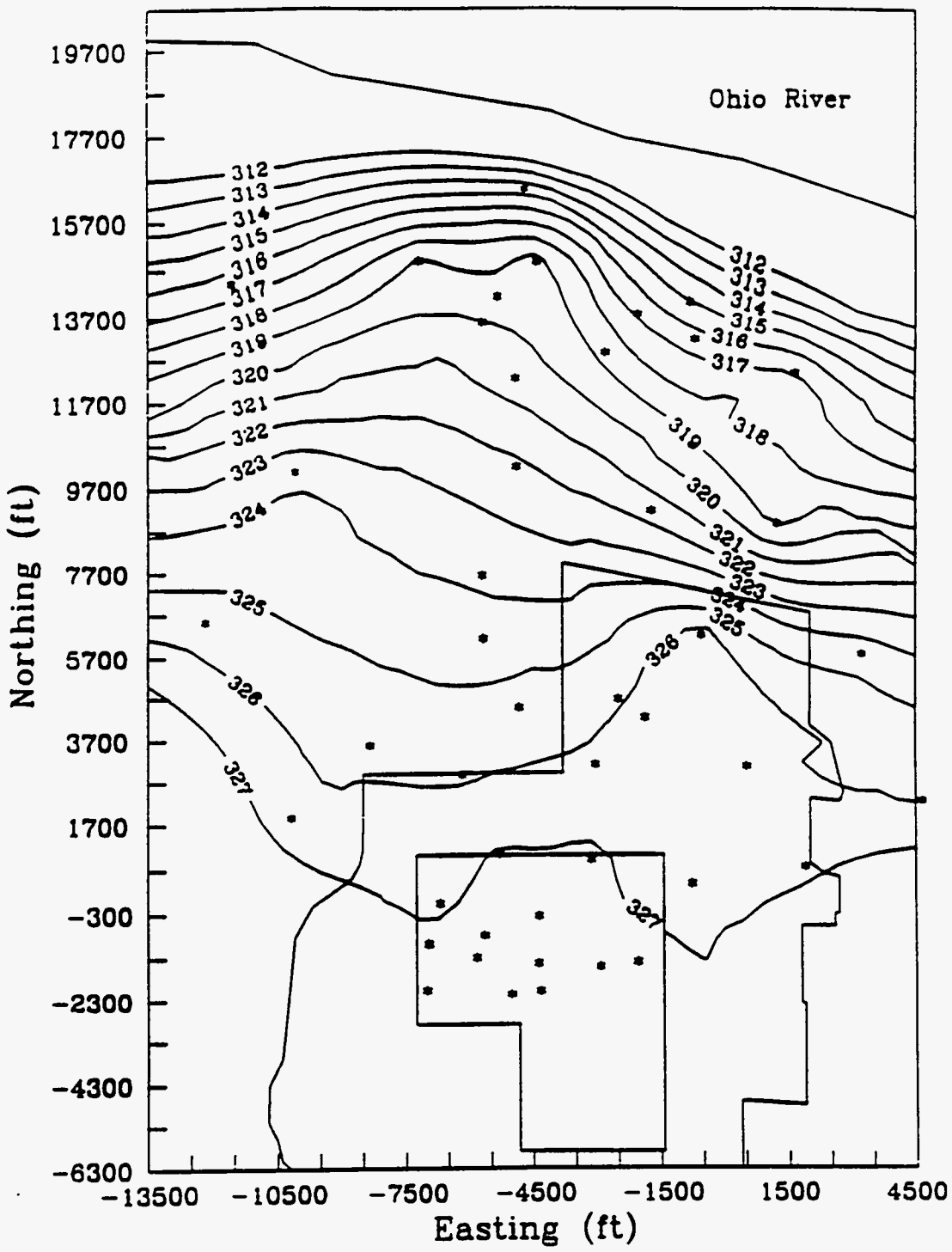


Fig. 3-4. Average head distribution in the RGA constructed from monthly data from July 1991 to June 1992. Contours are in feet and asterisks indicate locations of monitoring wells. (1000 ft = 305 m)

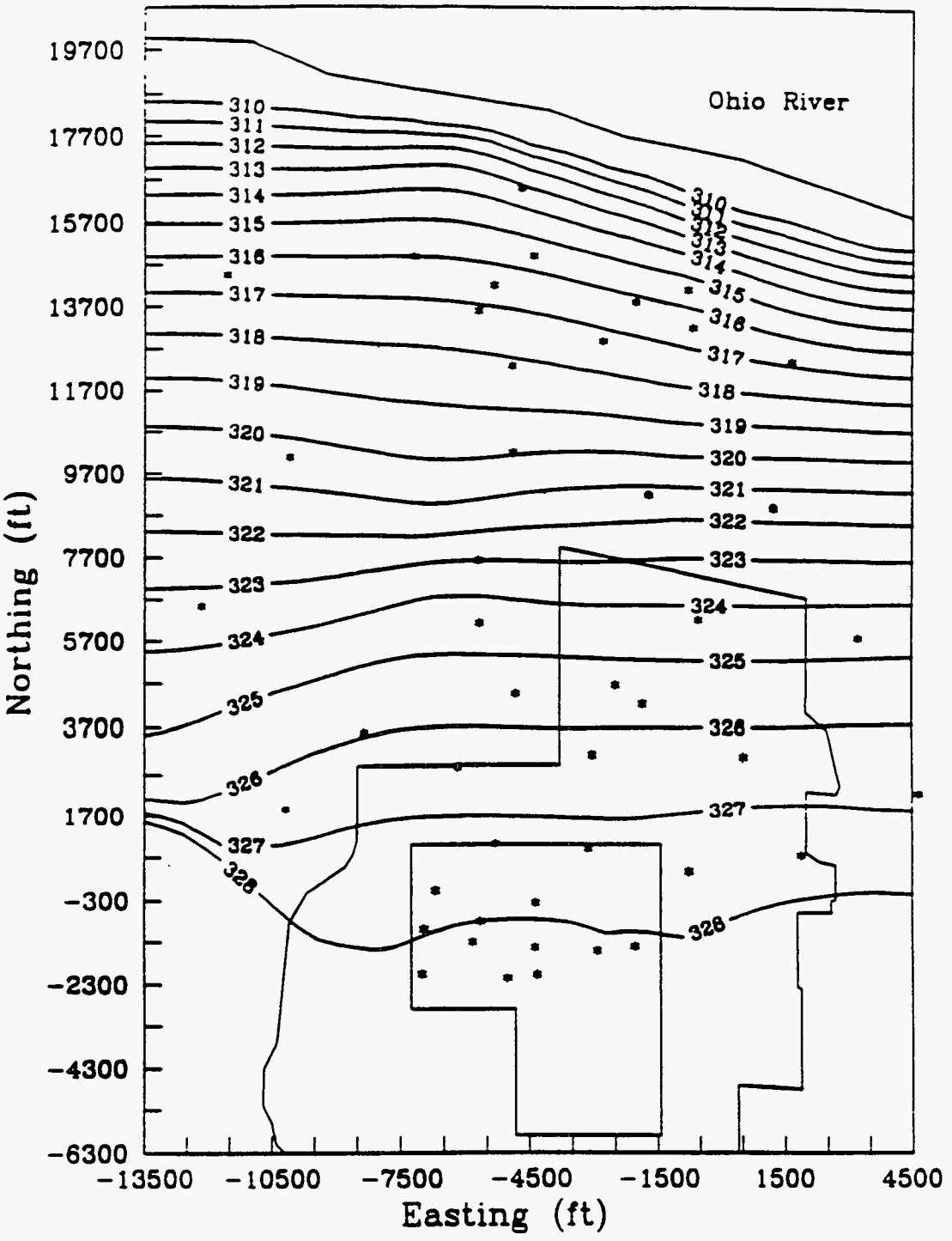


Fig. 3-5. Calculated potentiometric surface in the RGA. Contours are in feet and asterisks indicate locations of monitoring wells. (1000 ft = 305 m)

calculations, this block size can become too large either because of the small source area involved or because of numerical instability. In this analysis, a smaller area of interest was identified (Fig. 3-6) and a smaller block size of 46 m by 46 m (150 ft by 150 ft) was used for the transport calculations. Boundary conditions such as constant head and inactive and leaky blocks (Fig. 3-3) were modified to make them consistent with the smaller model grid. The blocks on the edge of the smaller model grid were treated as constant head blocks with head values specified by the original calibrated model. The transport model was recalibrated before contaminant transport calculations were made, because small changes in calculated heads were introduced by reducing the grid size.

Sufficient data for specifying the dispersivity and porosity for the four soil layers are not available. For the present calculation, the effective porosity for all layers was assumed to be 30%, which is a reasonable value for coarse sand, gravel, sand and gravel mixes, or glacial till (Bouwer 1978; Freeze and Cherry 1979). The dispersivity in the RGA was taken to be 21 m (70 ft) in the longitudinal direction and 2.1 m (7 ft) in the transverse direction. These values are qualitatively in agreement with an observed plume at the PGDP (Clausen et al. 1992), and comparable to those reported by Pinder (1973), and yield a groundwater transport time of about 30 years from the plant to the Ohio River. In other layers, the dispersivity value was taken to be 0 to prevent the calculated plume from artificially spreading into neighboring layers. To further ensure this, a 3.0 cm (0.1 ft) layer of the upper clay and a 3.0 cm layer of the McNairy Formation next to the RGA layer were treated as two additional layers in the contaminant transport calculations, making the 4-layer model a 6-layer model in the calculations.

Radionuclides in leachate and groundwater can be sorbed by waste and soils retarding their transport. This process can be quantified with the distribution or sorption coefficient, K_d. The values of K_d in soils at the PGDP for the relevant radionuclides were estimated by the PGDP Environmental Restoration Program and are listed in Table 3-2. Also listed in the table are corresponding values of the retardation factor, R, used in the contaminant transport calculations. The density of the soils was assumed to be 1.8 g/cm^3.

3.3.3 Models for Dose Estimation

As described in Sect. 1, estimates of dose to off-site individuals from direct consumption of contaminated groundwater and estimates of dose to inadvertent intruders from direct intrusion into solid waste in the landfill provide the basis for the estimates of concentration limits for disposal of radionuclides in the proposed solid waste landfill. This section briefly describes the models used to estimate dose to off-site individuals from the drinking water pathway and dose to inadvertent intruders from direct intrusion into solid waste.

The results of the dose analysis for off-site individuals from the drinking water pathway are given in the form of annual effective dose equivalents per unit concentration of radionuclides in groundwater. These pathway dose conversion factors are estimated as the product of the annual consumption of drinking water from the contaminated source, which is assumed to be 730 L/yr (2 L/d), and the ingestion dose conversion factor for the radionuclide of concern, which is the effective dose equivalent per unit activity ingested. The pathway dose conversion factors for the drinking water pathway for the radionuclides of concern to waste disposals at the PGDP, as well as the source of these data, are presented in Table 4-3 of Sect. 4.3.

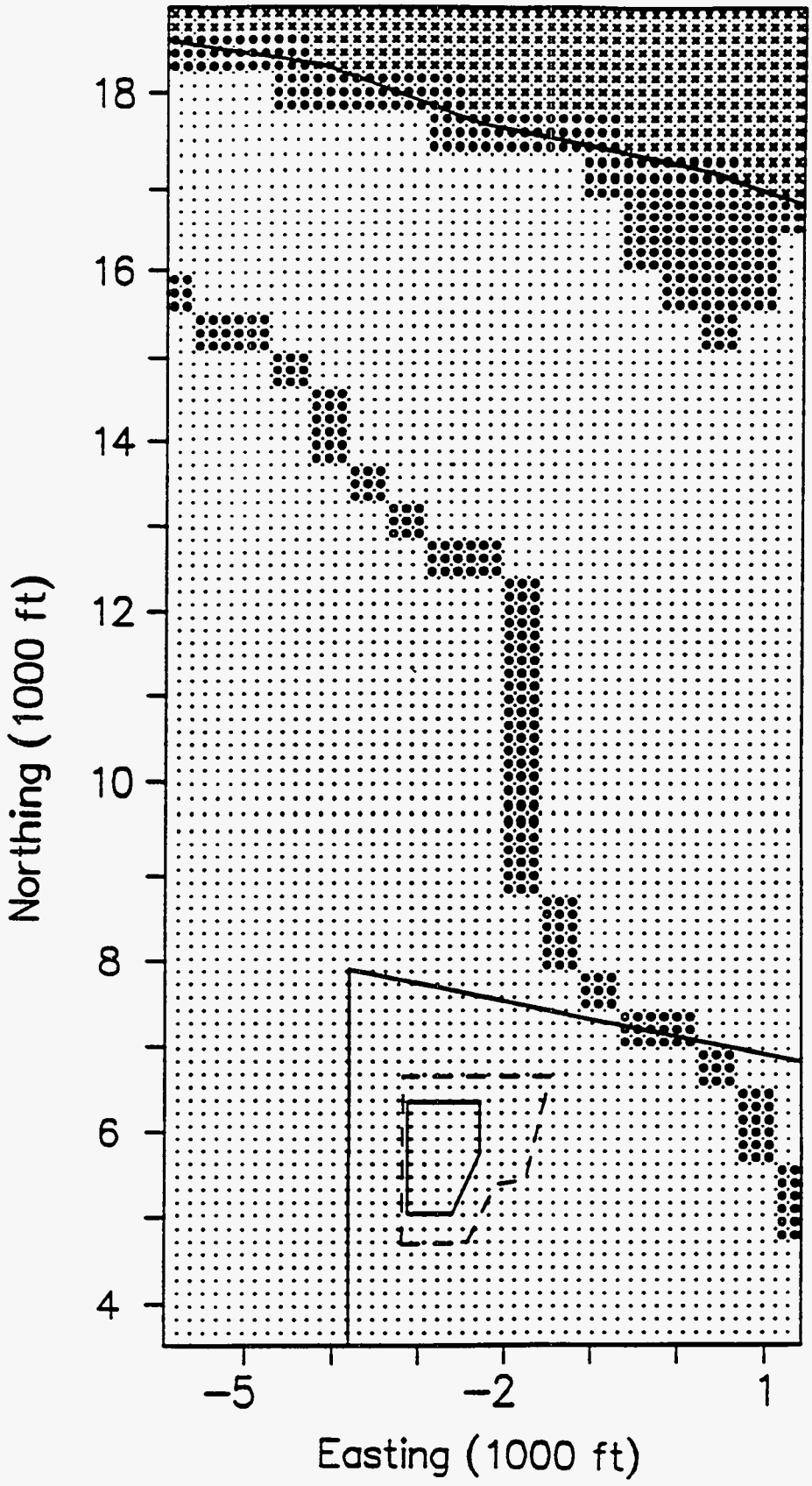


Fig. 3-6. Model grid used for transport calculations.
(1000 ft = 305 m)

Table 3-2. Distribution coefficients, K_d, of radionuclides in wastes and soils and corresponding retardation factors in groundwater for transport calculations

Element	K_d (mL/g) In waste	K_d (mL/g) In soils	Retardation factor
Tc	0.15	0.15	1.9
Th	15,000	15,000	90,000
U	27[a]	45	270
Np	3	3	19
Pu	450	450	2,700
Am	3,000[b]	3,000[b]	18,000

[a]K_d in waste for U was estimated from Paducah Gaseous Diffusion Plant (PGDP) experimental results (Story 1993). For other radionuclides, K_d values were assumed to be the same as in soils.
[b]Values for Am were estimated from the work of Sheppard and Thibault (1990). Others are estimates obtained from PGDP Environmental Restoration Program.

As described in Sect. 3.2.4, exposures of inadvertent intruders are assumed to occur according to the agriculture-homesteader scenario. The results of the dose analysis for the agriculture scenario for inadvertent intruders are given in the form of annual affective dose equivalents per unit concentration of radionuclides in the disposal facility. These scenario dose conversion factors are estimated as the sum of the pathway dose conversion factors for the different exposure pathways for the agriculture scenario listed in Sect. 3.2.4.

The dose per unit concentration of a radionuclide in the disposal facility for the agriculture scenario is estimated using simple, multiplicative-chain models for the different exposure pathways. The models and databases used in the dose analysis are the same as those used in a performance assessment for low-level radioactive waste disposal at the Savannah River site (MMES et al. 1994). Similar results are given in a performance assessment for the low-level radioactive waste disposal at the Oak Ridge site (ORNL 1994); however, the results for the Savannah River site include updated dosimetry data for external exposure pathways and, thus, are preferable for use in this analysis. The previous dose analyses for inadvertent intruders at the Oak Ridge and Savannah River sites reasonably can be applied to the PGDP because environmental conditions are similar at the different humid sites and activities of inadvertent intruders in the agriculture scenario should be essentially the same at all three sites. The agriculture scenario dose conversion factors for the radionuclides of concern to waste disposals at the PGDP, as well as the source of these data, are presented in Table 4-4 of Sect. 4.3.

4. RESULTS

This section presents the results of the calculations of the performance of the proposed solid waste landfill. The calculated concentration limits for each radionuclide for groundwater transport and direct intrusion are presented. The calculated results are interpreted to provide the operating limit for the proposed solid waste landfill.

4.1 SOURCE TERMS

As shown in Fig. 3-2, the contaminant flux to groundwater per unit landfill area for a given radionuclide during the early times is controlled by the effects of the liner and cover and their degradation. The flux of contaminants, or flux density, reaches a peak value and then decreases with time. Table 4-1 lists the calculated peak flux densities and their times of occurrence for the elements of concern. These results are for the first part of the landfill; results for other parts of the landfill are similar but with delay times of 20, 40, and 60 years. As mentioned in Sect. 3.1, the fluxes from all parts were superimposed as the total source term in the transport calculations.

Table 4-1. Calculated peak flux densities to groundwater and times of occurrence for Part 1 of the simulated landfill

Element	K_d in waste (mL/g)	Peak flux density (a.u./d/cm^2)	Occurrence time (year)
Tc	0.15	13,000	75
Th	15,000	1.3	189
U	27	690	127
Np	3	5,100	105
Pu	450	44	155
Am	3,000	6.6	173

Note: The initial concentration in wastes was assumed to be 10^6 in arbitrary units (a.u.) per g (density = 0.6 g/cm^3). Changing the initial concentration will change the calculated flux density and concentrations in groundwater proportionally and will not change the dilution factor presented in Sect. 4.2. The use of the value of 10^6 is mainly for the convenience of numerical calculation.

As can be seen from Table 4-1, the time for the flux density to peak is after the institutional control period because the liner and cover are assumed to start to degrade at the end of the institutional control period. The weak K_d-dependence of the peak occurrence time is explained by noting that radionuclides with larger K_d values tend to remain in the waste for longer times. The peak flux density itself is very sensitive to the value of K_d, which determines the partitioning of contamination between the solid waste and the water. The overall result is that for a radionuclide with a high K_d value in waste, the peak flux to groundwater is small and the concentration limit determined from groundwater transport is less restrictive.

4.2 GROUNDWATER

Figure 4-1 shows the uranium concentration contours in the RGA after 4750 years when the calculated concentration in the groundwater peaks near the plant boundary (see Fig. 3-2 for source model results). At this time and location, the U concentration had dropped from 10^6 in arbitrary units (a.u.) per gram of emplaced waste to 350 a.u./mL, giving a dilution factor of $10^6/350 = 2500$ mL/g. This result and results for other radionuclides considered are listed in Table 4-2. The time-dependent maximum concentrations at the plant boundary are shown in Figure 4-2. As can be seen from the figure, the larger the K_d value of a radionuclide in soil, the longer the transport time required for the concentration to peak at the plant boundary. The K_d dependence of the peak concentration in the groundwater mostly comes from the K_d dependence of the peak flux density (Table 4-1). As mentioned in Sect. 4.1, this dependence is evident because K_d defines the partitioning of contamination between soil or waste and water. The results shown in Fig. 4-2 and Table 4-2 are also presented in Fig. 4-3. The results for thorium were extrapolated from those of other radionuclides, because of the extraordinarily long computational time for thorium transport.

Table 4-2. Calculated peak concentrations and occurrence times at the plant boundary and the corresponding dilution factors for various radionuclides

Element	K_d in soil (mL/g)	Peak concentration (a.u./cm^3)	Time (year)	Dilution factor (g/g per g/mL)
Tc	0.15	7,880	125	127
Th[a]	15,000	0.922	1,590,000	1,080,000
U	45	347	4,740	2,880
Np	3	3,800	480	263
Pu[b]	450	29.2	53,400	34,300
Am[b]	3,000	4.38	355,000	229,000

Note: The initial concentrations in wastes are 10^6 in arbitrary units (a.u.) per gram.
[a]Values for Th were extrapolated from those of other radionuclides (see Fig. 4-3).
[b]Radioactive decay was not included in the calculations (see Table 2-3 for half-lives).

4.3 DOSE ANALYSIS

As described in Sect. 3.3.3, the results of the dose analysis for off-site individuals are given in the form of annual effective dose equivalents from the drinking water pathway per unit concentration of radionuclides in water. The pathway dose conversion factors for the radionuclides of concern to waste disposals at PGDP are obtained from recent performance assessments for low-level radioactive waste disposal facilities (ORNL 1994; MMES et al. 1994) and are given in Table 4-3. The values for ^{232}Th and ^{238}U include contributions from radiologically significant short-lived decay products which are assumed to be in activity

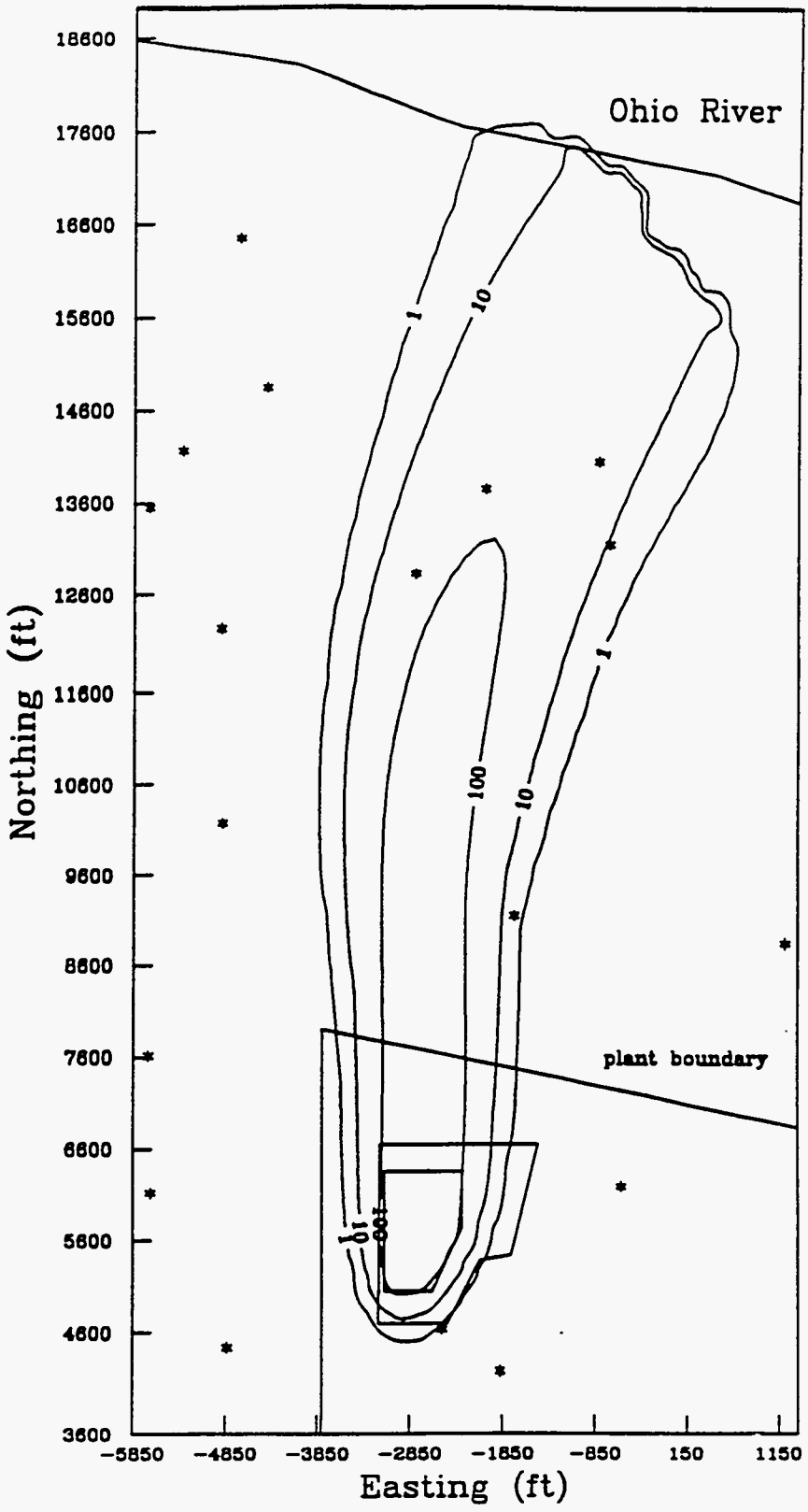


Fig. 4-1. Concentration contours of uranium in the RGA after 4750 years, with initial concentration in waste being 10^6 in arbitrary units. Asterisks indicate locations of monitoring wells. (1000 ft = 305 m)

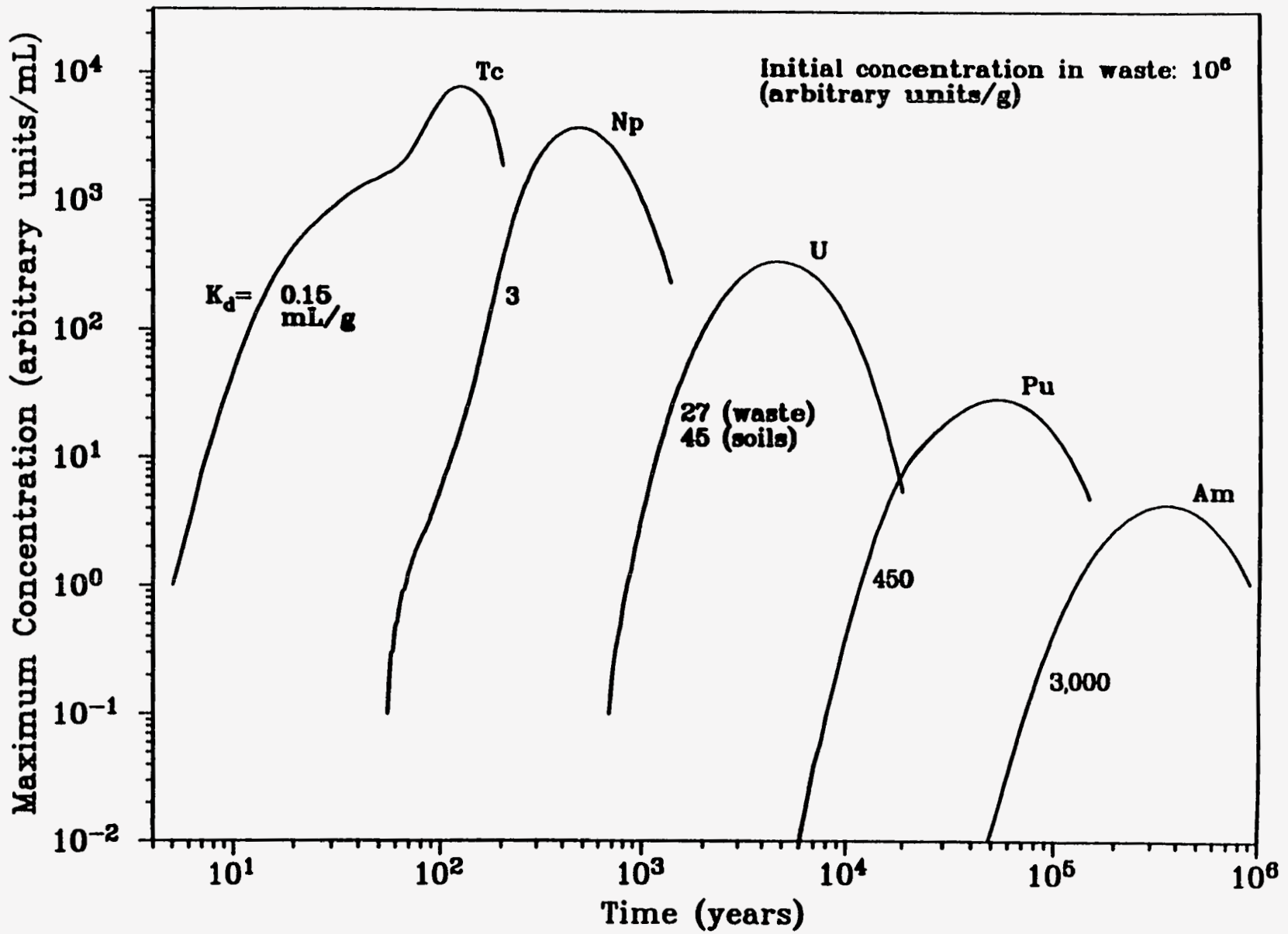


Fig. 4-2. Graphs for calculated maximum concentration versus time at the Paducah Gaseous Diffusion Plant boundary for radionuclides with various K_d values.

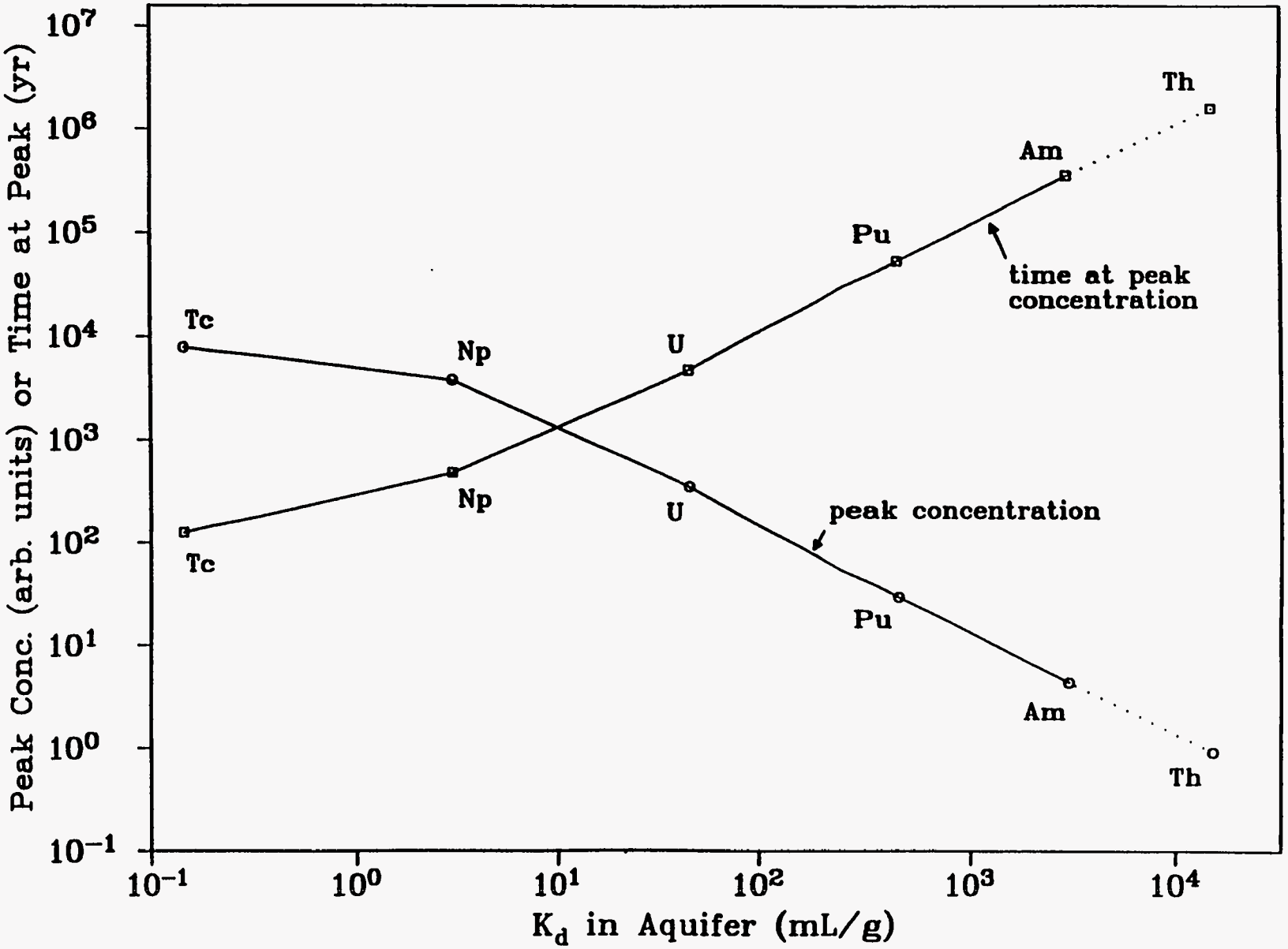


Fig. 4-3. Calculated peak concentration and its occurrence time vs. K_d graphs for locations at the Paducah Gaseous Diffusion Plant boundary. Data are from Table 4-2 and values for thorium were extrapolated from those of other radionuclides.

equilibrium with the parent radionuclide. No other radionuclides in this table have radiologically significant short-lived decay products for the drinking water pathway.

Table 4-3. Annual effective dose equivalents for drinking water pathway per unit concentration of radionuclides in water

Radionuclide	Annual effective dose equivalent (mrem/yr per pCi/mL)
^{99}Tc	9.5×10^{-1}
^{230}Th	3.9×10^{2}
$^{232}Th^{a}$	3.4×10^{3}
^{234}U	1.9×10^{2}
^{235}U	1.8×10^{2}
^{236}U	1.8×10^{2}
$^{238}U^{b}$	1.8×10^{2}
^{237}Np	2.8×10^{3}
^{238}Pu	2.8×10^{3}
^{239}Pu	3.1×10^{3}
^{241}Am	3.3×10^{3}

Source: Table G.7 of ORNL (1994) or Table A.4-6 of MMES et al. (1994).

[a]Value assumes that radiologically significant short-lived decay products ^{228}Ra, ^{228}Th, and ^{224}Ra are in activity equilibrium with parent radionuclide.

[b]Value assumes that radiologically significant short-lived decay product Th-234 is in activity equilibrium with parent radionuclide.

Similarly, the results of the dose analysis for inadvertent intruders, according to the agriculture scenario, are given in the form of annual effective dose equivalents per unit concentration of radionuclides in the disposal facility. The agriculture scenario dose conversion factors for the radionuclides of concern to waste disposals at the PGDP are obtained from the recent performance assessment for a low-level radioactive waste disposal facility at the Savannah River site (MMES et al. 1994) and are given in Table 4-4. The values for ^{232}Th, ^{235}U, ^{238}U, and ^{237}Np include contributions from radiologically significant short-lived decay products which are assumed to be in activity equilibrium with the parent radionuclide. No other radionuclides in this table have radiologically significant short-lived decay products for the agriculture scenario for inadvertent intruders.

To determine the concentration limits in waste based on the groundwater analysis, the dose limit of 4 mrem/yr, the dilution factors in Table 4-2, and the pathway dose conversion factors for the drinking water pathway in Table 4-3 are used. For example, for ^{238}U the resulting concentration limit is given by

$$\frac{4(mrem/yr)}{180(mrem/yr\ per\ pCi/mL)} \times 2900(g/g\ per\ g/mL) = 64\ pCi/g \quad . \qquad (6)$$

This and similar results for other radionuclides are listed in the third column of Table 4-5.

Table 4-4. Annual effective dose equivalents per unit concentration of radionuclides in disposal facility for agriculture scenario for inadvertent intruders

Radionuclide	Annual effective dose equivalent (mrem/yr per pCi/cm^3)
^{99}Tc	1.1×10^{-2}
^{230}Th	1.1×10^{-2}
$^{232}Th^a$	3.6
^{234}U	1.1×10^{-2}
$^{235}U^b$	1.8×10^{-1}
^{236}U	1.0×10^{-2}
$^{238}U^c$	3.9×10^{-2}
$^{237}Np^d$	5.0×10^{-1}
^{238}Pu	3.4×10^{-2}
^{239}Pu	4.0×10^{-2}
^{241}Am	5.6×10^{-2}

Source: Table A.4-14 of MMES et al. (1994).

[a]Value assumes that radiologically significant short-lived decay products ^{228}Ra, ^{228}Ac, ^{228}Th, ^{224}Ra, ^{212}Pb, ^{212}Bi, and ^{208}Tl are in activity equilibrium with parent radionuclide.

[b]Value assumes that radiologically significant short-lived decay product ^{231}Th is in activity equilibrium with parent radionuclide.

[c]Value assumes that radiologically significant short-lived decay products ^{234}Th, ^{234m}Pa, and ^{234}Pa are in activity equilibrium with parent radionuclide.

[d]Value assumes that radiologically significant short-lived decay product ^{233}Pa is in activity equilibrium with parent radionuclide.

As noted in Table 4-3, contributions from all radiologically significant short-lived decay products are taken into account. To determine the concentration limits based on the intruder-exposure analysis, the dose limit of 4 mrem/yr and the scenario dose conversion factors for inadvertent intruders in Table 4-4 are used. For example, for ^{238}U the concentration limit is given by

$$\frac{4(mrem/yr)}{4.0\times10^{-2}(mrem/yr\ per\ pCi/cm^3)} = 100\ pCi/cm^3 \quad . \tag{7}$$

This concentration limit, in pCi/cm^3, is for exhumed waste after the institutional control period (ORNL 1994). Assuming the density of disposed waste remains 0.6 g/cm^3 over time, this limit is multiplied by a factor of 1/0.6 cm^3/g to convert the result to that for disposed waste in pCi/g. The result for ^{238}U and similar results for other radionuclides are also listed in the second column Table 4-5. As noted in Table 4-4, contributions from all radiologically significant short-lived decay products are taken into account.

In obtaining the dilution factors in Table 4-2 from the analysis for the groundwater transport pathway, radioactive decay during transport to the off-site receptor location at the plant boundary was not considered. As described below, radioactive decay was taken into account for ^{238}Pu, ^{239}Pu, and ^{241}Am in developing concentration limits for groundwater in Table 4-5.

Table 4-5. Concentration limits in pCi/g determined from the groundwater-contamination analysis at the plant boundary and the intruder-exposure analysis at the landfill

Nuclide	Intruder-exposure	Groundwater
^{99}Tc	610	560[a]
^{230}Th	610[a]	11000
^{232}Th	1.9[a]	1300
^{234}U	610	61[a]
^{235}U	37[a]	64
^{236}U	670	64[a]
^{238}U	170	64[a]
^{237}Np	13	0.38[a]
^{238}Pu	200[a]	170,000[b]
^{239}Pu	170[a]	200
^{241}Am	120[a]	1900[b]

[a]More restrictive limit.
[b]Limit based on long-lived decay products ^{234}U for ^{238}Pu and ^{237}Np for ^{241}Am.

For ^{238}Pu and ^{241}Am, the relatively short half-lives (Table 2-3) in conjunction with the groundwater travel time and large retardation factors in groundwater transport (Table 3-2) indicate that decay of these radionuclides would essentially be complete during the transport time

to the off-site receptor location at the plant boundary. Hence, the dose from these radionuclides would be essentially zero for any initial concentrations in disposed waste. However, ^{238}Pu and ^{241}Am decay to the longer-lived radionuclides ^{234}U and ^{237}Np, respectively. Therefore, the concentration limits for the parent radionuclides are obtained from the limits for the respective decay products in the following way. If the half-life of the decay product is much longer than the half-life of the parent, which is the case for the two radionuclides of concern, the maximum activity of the decay product is simply the initial activity of the parent multiplied by the ratio of the half-lives of the parent and the decay product. Therefore, the concentration limit for the parent radionuclide is simply the concentration limit for the longer-lived decay product multiplied by the ratio of the half-lives of the decay product and the parent, the ratio being considerably greater than unity. This approach was used to obtain the concentration limits for ^{238}Pu and ^{241}Am in Table 4-5.

The effect of decay in the calculation of the concentration limits for direct intrusion need not be considered for ^{238}Pu and ^{241}Am because the time of direct intrusion into the waste is shortly after the wastes have been disposed of in the solid waste landfill. While decay products from these radionuclides may eventually be present in the landfill, their concentrations will always be so low that the dose from the decay products will be much less than the peak dose from the parent radionuclides.

In obtaining the groundwater concentration limit for ^{239}Pu, only decay of the parent radionuclide need be taken into account because the concentrations of the longer-lived decay product ^{235}U in waste would be much less than those of the parent and the parent reaches the site boundary before radioactive decay is complete. With 53,000 years calculated as the travel time (see Table 4-2), the activity of ^{239}Pu will have decreased by a factor of 4.6. As a result, the concentration limit for ^{239}Pu in groundwater can be increased by this factor.

Several assumptions have been incorporated into this analysis that need to be considered in the interpretation of results. Most importantly, all of the wastes are considered to have the radionuclide of interest at the concentration limit as a uniform average across the landfill. Each radionuclide is considered separately, so that the effect of mixtures has not been considered in developing the concentration limit for each radionuclide. Nearly as important is neglecting the dilution of contamination from daily soil cover of the wastes with uncontaminated soils, and the assumption that the waste density remains constant over time. Other assumptions have been identified in the preceding discussions as they were invoked in the analysis. The effect of these assumptions along with available regulatory guidance provides the means for interpreting the results and developing the operating limit for the proposed solid waste landfill, which is discussed in the next session.

4.4 INTERPRETATION OF RESULTS

The calculated concentration limits presented in Sect. 4.3 provide a quantitative basis for establishing an operating limit for radioactivity in waste materials in the solid waste landfill that is defensible and protective of public health and the environment. The simple application of the calculated results in Table 4-5, however, does not address the assumptions built into the calculations, the uncertainties in the calculations, the expected waste compositions, and the existing regulatory guidance that is critical to establishing an operating limit that is meaningful. In

addition, requiring the concentration of all isotopes of concern to be measured for all wastes sent to the landfill is not practical. This section examines the model results and regulatory guidance, and proposes an operating limit for the proposed solid waste landfill. The consequences of the proposed operating limit are evaluated by examining its application to wastes being considered for disposal in the landfill and comparing the results to the calculated concentration limits.

4.4.1 Model Results

As noted above, the calculated concentration limits are not sufficient to provide the operating limits without examining the underlying assumptions in the model. Foremost amongst the inherent assumptions is the averaging of the concentration of each radionuclide uniformly across the landfill. This averaging does not consider the daily cover of wastes with uncontaminated soil, the uncontaminated soils used to construct the final cover (although the hydraulic properties of the final cover are included in the calculations), or the presence of any native soils between adjacent phases of the overall landfill. The presence of uncontaminated soils in the landfill will tend to reduce concentrations of radionuclides both in soils within the landfill and in leachate. Existing plans for landfill operations suggest that the addition of uncontaminated soils will reduce the overall concentration of radionuclides by at least a factor of two within each phase of the landfill. With the consideration of the dilution of wastes by uncontaminated soils alone, the effect of the addition of uncontaminated soils would increase the operating limits by a factor of two or more.

The reduction in the average radionuclide concentrations in wastes by uncontaminated soils is offset to some extent by the assumption that the density of the wastes will remain constant over time. Wastes are assumed to be disposed of at a density of 0.6 g/cm^3. During normal compaction operations in the landfill, the density will be increased to a density of approximately 1.0 g/cm^3. Therefore, compaction of wastes would tend to increase local concentrations of radionuclides by a factor of about two. Once water is introduced into the wastes and leachate begins to form, the two assumptions of soil compaction and dilution of wastes with uncontaminated soils will tend to offset each other.

The calculated concentration limits do not conserve mass between the direct intrusion limit and the groundwater limit. For the calculation of the groundwater limit, assumptions were made that tended to increase the mobility of radionuclides and allow for the highest release of radionuclides that could be considered reasonable for the proposed solid waste landfill. This approach yields a conservative limit for groundwater protection. For direct intrusion, on the other hand, no radionuclides are assumed to have been transported away from the landfill. This leads to a conservative concentration limit for direct intrusion. However, radionuclides cannot be transported quickly and remain in waste at the same time, leading to a lack of conservation of mass. In reality, some radionuclides will be transported from the wastes, reducing the potential for exposures of inadvertent intruders. Similarly, because the groundwater analysis is conservative in its formulation, some radionuclides will be transported to groundwater at concentrations that reasonably are expected to be less than those incorporated in the calculations, reducing the potential for exposures of off-site individuals. The approach taken in this analysis provides conservatism to the calculated concentration limits.

Each radionuclide was analyzed independently in the modeling of the facility. The possibility of mixtures of radionuclides in wastes was not considered. Strictly applying the

calculated concentration limits to mixtures of radionuclides requires the combination of the concentration limits for the radionuclides considered. A simple way to address mixtures is to take the measured concentration of each radionuclide present in waste and divide it by its limit. The resulting ratio for each radionuclide present in waste is then summed over all of the radionuclides present. If the resulting sum is less than one, then the mixture of radionuclides can be considered to be less than the concentration limit for the mixture. In reality, this approach leads to a conservative application of the limits for groundwater, because each radionuclide is transported at different rates. Consequently, the radionuclide mixture will be present in groundwater in ratios less than the ratios in disposed waste. The application of the sum-of-fractions rule for mixtures of radionuclides to direct intrusion, by comparison, is a reasonable approach that does not introduce additional conservatism, because exposures of individuals to radioactivity all occur at the same time. In defining the operating limit for the proposed solid waste landfill, the sum-of-fractions rule for mixtures is used with the understanding that it introduces additional conservatism for groundwater limited isotopes.

The groundwater model used in this analysis includes many uncertainties, but the model has been developed and applied to represent the current state of knowledge of the geohydrology of the PGDP. Several uncertainties incorporated into the model are appropriate to note in the development of the operating limit for the proposed solid waste landfill.

A significant uncertainty in the groundwater model is the filling rate and overall site utilization of the landfill, which depends on waste generation rates at the PGDP and the manpower dedicated to landfill operations. The filling rate of one part of the landfill every 20 years, with each part being roughly comparable to one phase of operations, and about one-half of the overall landfill area being utilized, has been used in the calculation of the concentration limits. The rate of landfill utilization will have no effect on calculated direct intrusion concentration limits, but would have a limited effect on calculated groundwater concentration limits. More rapid filling would tend to increase the quantity of radionuclides available for transport at any time and reduce the dilution of leachate in the underlying aquifer. For this model application, increases in the filling rate would have little, if any, effect on the concentration of radionuclides in leachate, because the leachate is generated in the landfill independent of the volume of waste and transported through the overlying strata to the RGA with limited interaction during vertical transport. This lack of sensitivity to increases in the filling rate is the result of the projected disposal rate being sufficiently high to ensure maximum concentrations in leachate. Most of the reduction in the concentrations of radionuclides in the disposed waste, as they are transported to the site boundary, occurs in the landfill, the strata beneath the landfill, and the RGA directly beneath the landfill. Concentrations of radionuclides in the waste in the landfill, as compared to the concentrations in the RGA beneath the landfill, are reduced by factors ranging from 83 to 166,000, depending on the radionuclide. From the location beneath the landfill to the site boundary in the RGA, the concentrations of radionuclides are further reduced by about a factor of two. Since the primary mechanisms for reducing the concentrations of radionuclides in waste by groundwater transport are in the generation of leachate and the transport through the strata beneath the landfill, and these mechanisms are not sensitive to the filling rate of the landfill, little effect on the operating limits would be associated with changes in the landfill filling rates or overall site utilization.

Modeling uncertainties are also associated with the choice of distribution coefficients (K_ds) for each radionuclide in waste and in groundwater transport. Sorption has been modeled as a

linear process in leachate generation and transport in groundwater. Low concentrations of many radionuclides tend to have higher K_ds, especially for uranium (Seeley and Kelmers 1984). The estimated K_d values provided by the PGDP Environmental Restoration Program are generally lower than the corresponding values quoted in the literature (Sheppard and Thibault 1990). Since lower K_d values result in lower dilution factors, which yield lower concentration limits, the calculated groundwater limits are probably conservative. Conservative choices for K_d values also lead to more rapid transport of contamination. Consequently, the calculated time for maximum concentrations of contaminants is likely to be earlier than actually expected. The early arrival time reduces the potential decay of radionuclides in groundwater transport and suggests, in particular, that the groundwater limit for ^{239}Pu is conservative.

4.4.2 Regulatory Guidance

As discussed in Sect. 1, regulations for identifying the quantities of radionuclides in wastes that can be managed as non-radioactive wastes have not been promulgated. Managing wastes at the PGDP is made difficult by this lack of regulatory guidance because of the naturally occurring isotopes of uranium and thorium that are present in many types of waste materials independent of their origin. Administrative controls are in place to segregate process wastes derived from uranium enrichment from wastes not associated with uranium enrichment, but the possibility remains that waste materials incidentally contaminated by the uranium enrichment process could be included in wastes sent to the solid waste landfill for disposal. More stringent administrative controls would reduce the waste volumes disposed of at the solid waste landfill without necessarily reducing the possibility that wastes incidentally contaminated by the uranium enrichment process could be included in wastes sent to the solid waste landfill for disposal. Consequently, an operating limit for waste disposal at the solid waste landfill that can be used to discriminate non-radioactive waste from radioactive waste is important for protecting public health and the environment.

Existing regulatory guidance concerning the disposal of uranium-contaminated wastes is available in the U.S. Nuclear Regulatory Commission Branch Technical Position Paper on the Disposal or On-site Storage of Thorium or Uranium Wastes from Past Operations (NRC 1981). This paper is the current basis for regulating the disposal of large volumes of waste (e.g., contaminated soil and debris) containing low-levels of uranium and thorium generated at commercial nuclear fuel processing facilities. Wastes are evaluated on a case-by-case basis with consideration of the types and quantities of material to be disposed of, the packaging of the waste, the disposal site, access restrictions, radiation safety procedures, record keeping, and local disposal restrictions. Four disposal options are identified, with the first option referring to the disposal of wastes with acceptably low concentrations of uranium and thorium that no restrictions on burial method are specified. The maximum allowable concentrations allowed for this option are 1) natural thorium—10 pCi/g; 2) natural uranium ore—10 pCi/g; 3) depleted uranium—35 pCi/g; and 4) enriched uranium—30 pCi/g. Since PGDP waste materials can most directly be associated with enriched uranium, the application of the Branch Technical Position Paper to the proposed solid waste landfill would suggest that a limit of 30 pCi/g would be acceptable for the unrestricted disposal of uranium wastes generated at the PGDP. Importantly, the concentration limit in the Branch Technical Position Paper of 30 pCi/g for enriched uranium is less than the calculated concentrations limits for uranium waste disposal in the proposed solid waste landfill at the PGDP given in Table 4-5.

The EPA (1994) and NRC (1994) are considering standards for cleanup of residual radioactivity at contaminated sites which include a dose limit of 15 mrem/yr in contaminated soil to permit unrestricted use. This dose limit is substantially higher than the dose limit of 4 mrem/yr used in this analysis for calculating the operating limits for the proposed solid waste landfill. The EPA is also considering a proposed rule for management of low-level radioactive wastes that limits the potential exposure of off-site members of the public to a dose of 15 mrem/yr from all pathways. Again, this dose limit is greater than the dose limit of 4 mrem/yr used in this analysis for calculating the concentration limits for the proposed solid waste landfill. Finally, the EPA has promulgated a dose limit of 4 mrem/yr for the protection of drinking water from community drinking water supplies after treatment. This dose limit is equivalent to the dose limit used in this study to limit the potential contamination of groundwater without treatment.

The Commonwealth of Kentucky utilizes a compliance point of 75 m (250 ft) from the edge of the landfill for meeting groundwater contamination standards in the regulation of solid waste landfills. The concentration limits obtained in this analysis were determined using the PGDP site boundary as the compliance point, taking credit for DOE ownership and management of the PGDP site. The PGDP site boundary is 370 m (1200 ft) north and 210 m (700 ft) west of the landfill boundary. The groundwater model of the landfill performance shows that any contaminant plume originating from the proposed landfill will travel north and not reach the boundary to the west of the landfill. In establishing the operating limit for the landfill, if the Commonwealth of Kentucky were to reject the use of the PGDP reservation boundary for compliance with the groundwater dose limit, the calculated concentration limits would need to be reduced by 25% from the limits presented in Table 4-5.

4.4.3 Proposed Operating Limit

In consideration of the calculated results from the evaluation of the performance of the proposed solid waste landfill and the existing regulatory guidance for the management of solid wastes containing small quantities of radioactive materials, an operating limit of 30 pCi/g for uranium in wastes is proposed for the PGDP solid waste landfill. All wastes would be considered to have the potential to be contaminated by uranium enrichment operations and would be certified to have less than the proposed operating limit prior to disposal in the solid waste landfill. This proposed operating limit would require concurrence with the Commonwealth of Kentucky prior to implementation. The proposed certification procedures also would require concurrence by the Commonwealth of Kentucky. The proposed operating limit takes into account the NRC Branch Technical Position Paper, noting that the NRC limit for uranium waste disposal without restrictions is less than the calculated concentration limits for the solid waste landfill at the PGDP.

Additionally, as indicated in Sect. 4.4.4, the proposed operating limit for uranium should be sufficient to address the presence of other radionuclides that may be present in wastes generated at the PGDP, based on the known characteristics of wastes and the likelihood that other radionuclides, such as ^{238}Pu, ^{239}Pu, ^{241}Am, ^{230}Th, ^{232}Th, ^{237}Np, and ^{99}Tc, may be present. To provide additional assurance that the performance objectives are not exceeded, the proposed operating limit should be applied to each waste package disposed of in the PGDP solid waste landfill. By imposing the operating limit on individual waste packages, a significant level of conservatism is introduced, because the calculated concentration limits in Table 4-5 were determined using average concentrations of radionuclides uniformly distributed across the entire

landfill. By imposing the average concentration limit across the landfill on each package disposed of in the landfill, the average concentration in the landfill is certain to be significantly reduced, based on the known characteristics of the waste materials at the PGDP. In the next section, information on existing wastes at the PGDP is presented in support of the proposed operating limit.

4.4.4 Application of the Proposed Operating Limit to PGDP Wastes

Waste materials at the PGDP are not routinely characterized for all of the radionuclides considered in this analysis, because of the high analytical costs associated with determining trace quantities of ^{99}Tc, ^{237}Np, Pu, and Th. Similarly, characterization of uranium in wastes typically reports only ^{235}U and ^{238}U. However, a few waste streams that could be considered for disposal at the PGDP solid waste landfill have been characterized completely.

Thirty-four pallets returned from SEG Inc. were fully characterized. Of these pallets, 11 had uranium concentrations greater than the proposed operating limit of 30 pCi/g and would not be suitable for disposal in the PGDP solid waste landfill. Of the remaining 23 pallets having uranium concentrations less than 30 pCi/g, one pallet had a detected concentration of ^{237}Np of 7.7 pCi/g, which exceeds the calculated operating limit of 0.38 pCi/g in Table 4-5. The amounts of all other radionuclides for this pallet were less than the calculated concentration limits in Table 4-5. The remaining 22 pallets had concentrations less than the Table 4-5 limits for all isotopes. When the sum-of-fractions mixture rule was applied to the 23 pallets with quantities of uranium less than the 30 pCi/g operating limit, two pallets had a sum-of-fractions greater than one. One of these had the ^{237}Np concentration of 7.7 pCi/g, and the other pallet had a ^{237}Np concentration of 0.35 pCi/g that resulted in the sum-of-fractions for the mixture exceeding one when other isotopes in small concentrations were considered. In the second case, however, the data for the ^{237}Np concentrations were not definitive, because the limit of detection for ^{237}Np was 0.2 pCi/g with an error of plus or minus 0.1 pCi/g. Additionally, the reading of 7.7 pCi/g is extremely high and very atypical of any other measured values of ^{237}Np at the PGDP, as shown in activity spectra presented in Tables 4-6 and 4-7. Possible sources of error include the sample size or cross contamination with wastes generated by sources other than the PGDP. A profile of the activity in these data for the 11 pallets with concentrations of uranium greater than 30 pCi/g is presented in Table 4-6, where the concentrations have been normalized to the concentration of ^{238}U. Pallets with concentrations of U less than 30 pCi/g were not used for Table 4-6 because the effect of limits of detection for isotopes such as ^{237}Np are falsely exaggerated by normalizing the data to ^{238}U.

Twenty-six selenium rectifiers, circuit boards, and respirator cartridges were fully characterized for all the isotopes considered in this analysis, except for ^{230}Th and ^{232}Th. None of these samples had uranium concentrations greater than the proposed operating limit of 30 pCi/g, and none of these samples had concentrations of other radionuclides greater than the calculated concentration limits presented in Table 4-5. Applying the sum-of-fractions mixture rule to these samples resulted in one sample having a sum greater than one. This sample exceeded one because of a reading of 0.29 pCi/g for ^{237}Np. Again, the limit of detection for ^{237}Np was 0.2 pCi/g with an error of plus or minus 0.1 pCi/g. The contributions of the other trace isotopes were sufficient to cause the sum-of-fractions to exceed one for this one sample.

Finally, contaminated soils, concrete, and asphalt at the PGDP also have been analyzed by ORISE (1992). While the soils investigated all had concentrations of uranium in excess of the proposed operating limit of 30 pCi/g, the data provide insight in the spectra of contamination at PGDP. Fifteen soil samples with concentrations of uranium exceeding the proposed operating limit were normalized to ^{238}U and are presented in Table 4-7. As can be seen from this table, the typical spectra of isotopes other than uranium is extremely small. Two exceptions appear in Table 4-7 where the normalized concentrations of ^{99}Tc are high. These particular samples would exceed the calculated concentration limits in Table 4-5 if the uranium concentrations approached the proposed operating limit of 30 pCi/g for uranium.

The application of the proposed operating limit of 30 pCi/g for uranium to wastes typical of the PGDP shows that nearly all the samples for which data are available are less than the calculated concentration limits presented in Table 4-5. The exceptions associated with ^{237}Np are suspect due to apparent detection errors in the equipment used for analysis and apparent statistical variations in the concentrations in wastes. Table 4-7 indicates that there is a possibility that exceedances of Table 4-5 concentration limits could occur for some waste packages. However, application of the limits to individual waste packages would result in averages within the landfill to be reduced such that the average concentration within the landfill could reasonably be expected to be less than the average concentration limits in Table 4-5. Therefore, the application of the proposed operating limit of 30 pCi/g of uranium to wastes at the PGDP solid waste landfill can be reasonably expected to result in waste disposals consistent with the calculated concentration limits presented in Table 4-5. Conservatism incorporated into the calculation of the concentration limits and the averaging of waste packages within the landfill provides reasonable assurance that the health and safety of the public can be assured by the adoption of the proposed operating limit.

Table 4-6. Activity profile for SEG pallets with total U > 30 pCi/g

^{238}U	^{234}U	^{235}U	^{237}Np	^{238}Pu	^{239}Pu	^{241}Am	^{99}Tc	^{230}Th	^{232}Th
1.0	0.44	--	--	--	0.0006	--	0.023	0.024	0.0026
1.0	0.16	0.019	0.0046	--	0.00008	--	0.058	0.0005	--
1.0	0.14	0.015	--	--	0.0002	--	0.022	--	--
1.0	0.12	0.016	0.00095	--	--	--	0.030	0.0006	--
1.0	0.90	0.041	--	0.00006	0.0011	--	0.081	0.035	0.0011
1.0	0.37	0.045	--	0.00013	--	--	0.19	0.0085	0.0008
1.0	0.25	0.029	0.0057	--	0.00008	--	0.44	0.10	--
1.0	0.16	0.018	--	--	0.00013	--	0.036	0.00077	--
1.0	0.17	0.017	--	0.000007	0.000045	--	0.0025	0.0085	--
1.0	0.30	0.028	--	--	--	--	--	--	--
1.0	0.16	0.014	--	0.00029	--	--	0.015	0.0028	--
					Average values				
1.0	0.29	0.024	0.0038	0.00012	0.00032	--	0.09	0.02	0.015

Table 4-7. Activity profile for soils, concrete, and asphalt with total U > 30 pCi/g

^{238}U	^{234}U	^{235}U	^{237}Np	^{238}Pu	^{239}Pu	^{241}Am	^{99}Tc
1.0	0.78	0.010	--	--	0.000026	0.00004	0.0032
1.0	0.78	0.030	0.024	0.00011	0.0079	0.0024	0.51
1.0	0.74	0.052	--	0.0038	0.15	0.00012	110
1.0	0.74	0.059	0.023	--	0.46	0.079	23000
1.0	0.77	0.044	0.0061	0.00050	0.033	0.0052	0.24
1.0	0.77	0.050	0.0099	--	0.0032	0.0010	0.24
1.0	0.76	0.048	0.00003	--	0.000027	0.00024	0.0094
1.0	0.78	0.028	0.0060	0.00050	0.0098	0.0023	1.2
1.0	0.72	0.054	--	--	0.00037	0.00041	0.037
1.0	0.74	0.058	0.00053	--	0.0024	0.00081	0.026
1.0	0.77	0.043	0.0013	--	0.0025	0.00059	0.060
1.0	0.76	0.062	0.0038	0.0044	0.037	0.0063	7.7
1.0	0.76	0.044	0.0032	0.0013	0.27	0.0089	0.044
1.0	0.83	0.017	0.00066	--	0.0098	0.0019	0.27
1.0	0.82	0.047	0.00053	--	0.0023	0.0036	0.074
				Average values			
1.0	0.77	0.043	0.0066	0.0018	0.066	0.0075	1600

5. PERFORMANCE EVALUATION

The performance of the proposed solid waste landfill at PGDP has been analyzed to determine an operating limit protective of public health and the environment for the disposal of wastes containing small amounts of radioactive material. Performance objectives that limited dose to an off-site individual to 4 mrem/yr from the consumption of contaminated groundwater and limited dose to an inadvertent intruder to 4 mrem/yr from all exposure pathways involving direct intrusion into the disposal facility were used to define acceptable disposals of slightly contaminated materials. A site-specific analysis of the long-term performance of the proposed disposal facility was prepared to calculate concentrations of radioactivity in wastes that would not exceed these performance objectives. Interpretation of existing and proposed regulations related to wastes typical of those generated at PGDP suggested a proposed operating limit of 30 pCi/g for uranium for identifying wastes suitable for disposal in the solid waste landfill. Examination of PGDP wastes with known waste characteristics suggested that wastes considered suitable for disposal by the application of the proposed operating limit would be consistent with the limitations imposed by the maximum concentrations determined from the analysis of landfill performance. Consequently, the proposed operating limit provides reasonable assurance that the public health and environment will be protected for wastes disposed of in the proposed solid waste landfill.

The analysis was performed using an analytical model of the performance of the wastes disposed of in the landfill considering periods of time for waste operations, institutional control, and post-institutional control. During operations, wastes were assumed to be emplaced at a uniform rate in successive parts of the landfill, with each part closed after filling. A 30-year period of institutional control was considered during which the installed leachate collection system and final cover over the waste were assumed to remain functional. During the post-institutional control period, the leachate collection system was assumed to fail and the final cover to degrade. Leachate was generated in the wastes using a sorption model and transported through the environment using a calibrated numerical model (FTWORK) of the hydrogeology of the PGDP site. Concentrations of radionuclides in groundwater at the site boundary that would result in a dose of 4 mrem/yr from the consumption of 2 L/d of contaminated water were used to identify the maximum allowable concentrations in waste for each radionuclide of concern at PGDP. Additionally, maximum allowable concentrations of radionuclides in waste that would result in a dose of 4 mrem/yr from direct intrusion into waste after the end of institutional control were calculated. The lowest maximum allowable concentration from these two calculations was considered to determine acceptable waste disposals for each radionuclide of concern.

The proposed operating limit of 30 pCi/g of uranium in wastes was derived from the NRC Branch Technical Position Paper on the disposal of thorium and uranium wastes from commercial operations (NRC 1981). Consideration was given to other regulatory positions related to the management of uranium-contaminated wastes by the EPA and NRC in selecting the proposed operating limit, but these would have led to a less restrictive operating limit than the limit derived from the NRC Branch Technical Position Paper. The proposed operating limit is intended to be applied to each waste package with the presence of any other radionuclides in wastes to be addressed by the use of the operating limit. By using the uranium-based operating limit of 30 pCi/g, the presence of extremely small quantities of other radionuclides in waste is considered to be effective in assuring the combined concentrations of radionuclides in waste will be less than

the calculated concentration limits. Critical to the acceptance of the proposed operating limit is the determination of how protective the proposed limit is to public health and the environment when applied to the wastes to be disposed of at the proposed PGDP solid waste landfill.

The suitability of the proposed operating limit to environmentally sensitive wastes from PGDP was evaluated by examining known wastes potentially suitable for disposal in the solid waste landfill for which a complete characterization was available. For the 60 samples of potential wastes, 49 samples of wastes would have remained suitable for disposal in the solid waste landfill after the application of the proposed operating limit. The characterization of these 49 samples of waste was compared with the maximum allowable concentrations determined by the analysis of the solid waste landfill. One sample exceeded the maximum allowable concentrations calculated for the solid waste landfill. The sum-of-fractions rule for mixtures of radionuclides was also applied to the 49 samples of wastes, using the maximum allowable concentrations as limits for each radionuclide present in the wastes. A total of three samples exceeded the disposal limit associated with sum-of-fractions rule for mixtures. All of the apparent high concentrations of trace radionuclides were associated with neptunium, where the detection limit of the instrumentation used to characterize the wastes was close to the maximum allowable concentration determined by the analysis. Profiles of wastes and contaminated soils showed that the presence of neptunium in wastes may have been exaggerated, but the presence of technetium in wastes could be associated with some wastes exceeding the maximum allowable concentrations for some waste packages. However, averaging of all wastes in the landfill, which was assumed in the determination of the maximum allowable concentrations, is expected to account for the occasional waste package which would pass the proposed operating limit, but may exceed the maximum allowable concentration limits for some isotopes (e.g., ^{99}Tc) calculated for the landfill.

In summary, the operating limit of 30 pCi/g for uranium in each waste package is proposed for wastes considered to be acceptable for disposal at the PGDP solid waste landfill. The operating limit is supported by existing regulatory guidance and can be applied to waste disposal operations at PGDP. The application of the proposed operating limit of 30 pCi/g for uranium gives acceptable concentrations for known radionuclides of concern in wastes generated at PGDP. This site-specific analysis of the long-term performance of the proposed solid waste landfill at PGDP provides reasonable assurance that the application of the operating limit will be protective of public health and the environment.

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