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SCIENTIFIC BASIS FOR RISK ASSESSMENT AND MANAGEMENT OF URANIUM MILL TAILINGS

Uranium Mill Tailings Study Panel
Board on Radioactive Waste Management
Commission on Physical Sciences, Mathematics, and Resources
National Research Council

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CONTENTS

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TABLES AND FIGURES

TABLES

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FIGURES

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Executive Summary

Uranium mill tailings are the finely ground sand-like material that is left after uranium is extracted from ore. As of early 1983, approximately 200 million metric tons of uranium mill tailings covering 1300 ha at 51 sites had accumulated in the United States. Most of these sites are in the arid Southwest.

Uranium mill tailings present health and environmental concerns because of the residual radioactivity that they contain and because of a variety of other potential pollutants, such as chlorides, sulfates, and heavy metals. The milling process makes the radioactive and nonradioactive materials contained in the ore when it was mined much more mobile and also adds several potential contaminants to the tailings material during the milling process. Exposure routes of concern are release of the gas ^{222}Rn, airborne dust, and surface and groundwater contamination. In addition, the use of tailings as construction material or fill can lead to dangerously high levels of radon in associated buildings.

At the request of the Department of Energy, a National Research Council study panel, convened by the Board on Radioactive Waste Management, has examined the scientific basis for risk assessment and management of uranium mill tailings and issued this final report containing a number of recommendations. Chapter 1 provides a brief introduction to the problem. Chapter 2 examines the processes of uranium extraction and the mechanisms by which radionuclides and toxic chemicals contained in the ore can enter the environment. Chapter 3 is largely devoted to a review of the evidence on health risks associated with radon and its decay products. Chapter 4 provides a consideration of conventional and

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possible new technical alternatives for tailings management. Finally, Chapter 5 explores a number of issues of comparative risk, provides a brief history of uranium mill tailings regulation, and concludes with a discussion of choices that must be made in mill tailing risk management.

Several major themes recur throughout the report:

- To ensure the effectiveness of control measures over the long term, it will be necessary to maintain a low level but ongoing program of surveillance of uranium mill tailings piles with provisions for any necessary corrective actions.
- Because of the variability of physical, chemical, radiological, and demographic factors at each site, risk-management strategies must be site specific.
- The health risks posed by exposure to radon from uranium mill tailings piles are trivial for the average U.S. citizen, range from small to modest for most persons who live close to piles, but in special circumstances could be significant for a few individuals who live in close proximity to certain uncontrolled piles.

The principal recommendations of the panel are set forth below.

A major focus of regulation and risk management is control of radon emitted from tailings piles. The relationship between the concentration of radionuclides in a tailings pile and the flux of radon that leaves the pile is complex. The relationship between the flux of radon leaving a pile and the concentration of radon in air at points surrounding a pile is also complex. There is considerable pile-to-pile variability in both of these relationships.

> *Recommendation*: In developing or revising a risk-management strategy to control the potential risk of radon released from uranium mill tailings piles, the panel recommends a site-specific approach because of the great variability in radon flux at different piles. (page 71)

While the basic factors that give rise to the complexity and variability in the relationships between the concentration of radionuclides in a pile, the flux of radon leaving a pile, and the concentration of radon in the air around the pile are understood, models that attempt to relate pile concentrations to radon flux or radon flux to air concentrations give rise to estimates that may be in error by as much as several orders of magnitude.

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> *Recommendation*: In developing or revising a radon risk-management strategy for tailings piles, undue reliance should not be placed on general models of radon emission and dispersion. (page 71)

Despite the existence of simple, inexpensive, and reliable passive integrating radon monitors, only a modest and incomplete set of measurements of radon in the vicinity of controlled and uncontrolled piles has been collected.

The existing set of field measurements around uncovered tailings piles shows that, at some piles, average radon concentrations fall to background levels at a distance of less than a kilometer, and at all measured piles they fall to background within a few kilometers.

> *Recommendation*: There is a clear need for a systematic program of measurements of radon concentrations around piles before and after the implementation of radon control strategies. These measurements should include an adequate determination of background levels and the generation of concentration isopleths out to background. (page 74)

Brief periods of time (e.g., hours to days) spent outdoors near uranium mill tailings piles pose no significant lung-cancer risk. Persons of average lifestyle living in close proximity to uncontrolled uranium mill tailings piles may, depending on site-specific circumstances, experience a significant increase in total lifetime radon lung-cancer risk. If a person were to live right at the edge of a few piles that involve particularly unfavorable exposure conditions, their lung-cancer risk could be substantially higher than the average U.S. population lung-cancer risk. However, persons living at distances greater than a kilometer from most uncontrolled uranium mill tailings piles, and perhaps somewhat closer to some piles, will experience no significant increase in lifetime radon lung cancer risk due to exposure to radon from the pile.

> *Recommendation*: People who are not occupationally involved in uranium milling should not live or spend a large fraction of their time in close proximity to uncontrolled uranium mill tailings piles. If for political, economic, or other reasons it is not feasible to preclude people from living or spending a significant fraction of their time in

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close proximity to an uncontrolled mill tailings pile, then by almost any risk management decision criterion, steps should be taken to control radon emissions from that pile. (page 110)

Radon flux from a cover of less than a few meters of earth cannot now be predicted, with precision, either immediately after construction of the cover or in the long term when various physical changes, such as change in water content, have taken place in the cover.

Recommendation: The design of covers for the purpose of limiting radon release should be validated on a site-specific basis by measurements of radon concentrations in the vicinity of the pile. Such measurements could lead to a reduction in the minimum thickness of earth cover needed to limit radon release to acceptable levels. (page 153)

Long-term irregular settlement of tailings piles may cause mechanical disruption of any type of cover (earth, asphalt, stabilized tailings, plastic sheeting, or some combination) with consequent changes in rate of radon emission and rainfall infiltration. Chemical deterioration of cover materials may also change their effectiveness over time.

Recommendation: Periodic inspection and maintenance are essential to ensure that covers of any type will continue to function effectively over the long term. (page 153)

Because it contains radionuclides, airborne dust from uranium mill tailings piles presents a small, but real, health risk to persons who might inhale such dust. In addition, windblown dust may also present a local health risk from increased external gamma radiation around some piles. Based on available data, the increment of radionuclides from pile dust is indistinguishable from natural background at distances commonly less than a kilometer and never more than a few kilometers; however, the data, particularly data on background levels, are sparse. Strategies to reduce radon emanation appear likely to control the problem of windblown dust.

Recommendation: Piles should be controlled in a manner that prevents pile material from being blown away as dust. (page 110)

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Even over extended periods of time, the spatial extent of groundwater contamination from most uranium mill tailings is likely to be limited to dimensions of between several hundred and several thousand meters. However, uncontrolled uranium mill tailings piles hold the potential to produce significant local contamination of groundwaters and surface waters. The pattern and nature of this contamination display considerable complexity and a large amount of pile-to-pile variability.

> *Recommendation*: The specific strategies adopted to control groundwater and surface-water contamination from uranium mill tailings piles should be selected and implemented on a site-specific basis. (page 74)

Liners that are installed under tailings piles are almost certain in time to fail because of construction flaws, mechanical movements caused by the weight of the tailings, chemical changes associated with the leachate, and other factors.

> *Recommendation*: If liners are used, and local groundwater protection is deemed essential, they should be backed up by a drainage system for collecting leachate that may leak through the liner and by a system to monitor the effectiveness of the liner/drainage system. A contingency plan should be developed for treating any contamination that does enter the groundwater. (page 153)

In the case of existing piles, leachate from the tailings may reach groundwater.

> *Recommendation*: Groundwater monitoring should be undertaken to evaluate the need for remedial action. (page 153)

When surface-water and groundwater problems arise from uranium mill tailings piles, nonradioactive contaminants may pose the principal water-quality problem. Such water contamination may have many features in common with contamination from mill tailings produced in recovery of some other metals and with that emanating from solid-waste disposal sites.

> *Recommendation*: It is recommended that surface-water and groundwater contamination related to uranium mill tailings not be considered as a separate water-quality

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problem but rather in the context of the broader issue of water contamination emanating from man-made accumulations in general. (page 75)

While in special circumstances, tailings may pose health risks through a variety of secondary pathways, such as the food chain, in general, it does not appear that such risks are significant. However, if tailings are misused by man as construction material and backfill for occupied buildings, they can pose a serious health risk.

Recommendation: Successful protection against inappropriate use of tailing materials by man requires a low-level ongoing program of societal recordkeeping (e.g., maps), land-use control, and occasional on-site inspection. (page 75)

Based on a consideration of the geological evidence, the stability of tailings accumulations in surface or near-surface piles cannot be assured over time periods of more than thousands of years (and in some cases hundreds of years) without the option of active human intervention.

Recommendation: Plans for the management of mill tailings piles should recognize and explicitly incorporate plans for a continued low-level program of active monitoring and the option for active human intervention should it become necessary. (page 152)

Protection of some piles against catastrophic floods will not be possible even with heroic measures. However, the risk posed by piles under such circumstances is small and inconsequential compared with other impacts of such a catastrophic event, because tailings would almost certainly be mixed with and diluted in a much larger volume of sediments.

Recommendation: While protection of piles against local periodic floods is feasible and appropriate, protection against large, truly catastrophic floods is neither feasible nor warranted from the perspective of the risk presented. (page 152)

Solidification of tailings by any of several techniques may be a useful option in selected special circumstances, but because of high costs, uncertainties as to long-term stability, and problems

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presented for possible future reprocessing, it is not an attractive general solution.

Modifications and alternatives to traditional processing technologies hold potential for possible reprocessing of old ore and improving the future milling of new ore.

Both in reprocessing existing tailings and in applying various new or modified processes in new extraction, most of the thorium and much of the radium could be removed. These processes would produce relatively small volumes of thorium- and radium-bearing concentrates, the disposal of which might pose significant technical and political problems, although it is conceivable that a market for these materials could develop in the future. New process technologies may also be more efficient than those currently in use. For these reasons, reprocessing of tailings and the development of new process technologies should be evaluated both as a strategy for hazard management and in economic terms.

> *Recommendation*: A program of research on the process technologies that might be applied in reprocessing should be considered. (page 154)

> *Recommendation*: An expanded program of research on technologies that might be used to modify or replace existing processes in order to reduce or eliminate radiological hazards in tailings management should be undertaken. (page 155)

Viewed in the perspective of the wide variety of risks that face U.S. society, simple order-of-magnitude arguments and comparisons suggest that the health risks posed by exposure to radon from uranium mill tailings piles are trivial for the average U.S. citizen and range from small to modest for most persons who live close to uncontrolled piles. However, if persons were to live right at the edge of a few uncontrolled piles that involve particularly unfavorable exposure conditions, their risk could be significant.*

The controls now being implemented for inactive piles will be more than sufficient to manage the risks that they may pose for the next few decades. However, further risk analysis is needed to

* For this case, the incremental lung-cancer risk per 70-year lifetime is of the order of 0.06, or on an annual basis roughly 8×10^{-4}. For purposes of comparison, this is roughly two-thirds of the individual lung-cancer risk of smoking, or three times the per capita risk of motor vehicle accidents.

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understand how best to manage the risks of piles now classified as inactive and to understand the requirement for ongoing observation and corrective intervention on piles now being controlled. The panel makes a number of recommendations for improving future risk assessments, observing that those undertaken to date in the United States have not made adequate use of modern techniques for the characterization and treatment of the substantial scientific uncertainties that are involved.

The panel's findings carry a number of implications for the ongoing regulation of risk from uranium mill tailings.

> *Recommendation*: If regulation is to be based on the science of the problem, generic models must be validated for actual conditions. (page 182)
>
> *Recommendation*: In future regulatory reviews of uranium mill tailing risk management, the U.S. Environmental Protection Agency and the U.S. Nuclear Regulatory Commission should
> - Be explicit about the decision criteria that they chose to apply;
> - Address the need for ongoing observation and corrective intervention that the science of the problem indicates will be necessary;
> - Work to separate and identify the "judgment calls" explicitly, to identify their implications clearly, and to justify the choice compared with other possible alternatives. (pages 182-183)
>
> *Recommendation*: The U.S. Environmental Protection Agency should strive to achieve greater internal consistency in its approach to this problem and greater consistency between the way in which it deals with risks from tailings and the way in which it has dealt with other similar risks. (page 183)

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1
Introduction

This chapter sets forth the charge that the panel followed, briefly discusses the organization of the report, and provides background information on the circumstances that led to the accumulation of uranium mill tailings and their regulation.

THE CHARGE

A proposal for a study of the health hazards related to uranium mill tailings and the technical management alternatives to comply with standards, both now and in the future, was prepared by the Board on Radioactive Waste Management of the Commission on Physical Sciences, Mathematics, and Resources of the National Research Council and submitted to the Department of Energy (DOE). The proposal was accepted, and the Board assembled a panel of experts in chemical engineering, environmental science, geochemistry, geology, geotechnical engineering, hydrology, public policy, radiobiology, radiochemistry, radiological health, and risk analysis and charged it to perform the study.

The Uranium Mill Tailings Study Panel undertook to

- Review the scientific evidence concerning the potential hazards from uranium mill tailings;
- Evaluate management alternatives for control of uranium mill tailings;
- Compare safety standards and criteria governing uranium mill tailings with those for similar forms of radioactive waste;

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- Consider the health implications of options for managing the risk from uranium mill tailings;
- Assess the sufficiency of the database and current research and development programs;
- Recommend any needed research and development.

ORGANIZATION

The panel operated under a number of self-imposed constraints in meeting its charge. It did not

- Conduct original research, but did survey relevant literature, interview experts in the field, and perform simple analyses;
- Consider whether the level of protection of public health and the environment provided by current numerical standards is appropriate—this is a judgment to be made within the political system;
- Consider, in depth, issues related to groundwater pollution;
- Investigate hazards from mining and milling, consider nondomestic uranium mill tailings, or evaluate either detailed management systems or remedial action programs for any specific uranium mill tailings sites.

The panel did not limit its deliberations to the requirements of current legislation or regulations.

The panel met 11 times in full session, from March 1984 to June 1986. It conducted field inspections of representative uranium mill tailing piles in the vicinity of Grants, New Mexico. Individual panel members visited other facilities covering a range of geological settings in the United States (Appendix A). The panel called numerous experts from government, academia, and industry to present data and review their experimental programs (Appendix B).

At the conclusion of its review and evaluation, the panel reported on its findings and conclusions; these are intended to provide a scientific basis for the risk-assessment and management alternatives of uranium mill tailings. It also recommended a series of research and development efforts to reduce uncertainties, lower risk, and improve processing technology.

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The report contains an Executive Summary and five chapters. The Executive Summary provides an overview of the report including the panel's major recommendations and conclusions.

Chapters 2 through 4 contain the scientific basis for the report. Chapter 2, Exposure Processes: The Transport of Radionuclides and Toxic Chemicals from Tailings to the Environment, describes the movement of radionuclides and chemical pollutants from uranium ore through processing to the tailings piles and hence to the environment. Chapter 3, Potential Biological Hazards, discusses possible pathways of the radionuclides and chemical pollutants contained in the tailings and their potential biological effects on man. Chapter 4, Technical Alternatives for Tailings Management, considers management strategies that may be used for existing piles and options that might be developed to reduce the toxicity of tailings produced at future mills and improve the safety and integrity of new piles.

Chapter 5 opens with a discussion of comparative risk and issues of risk perception. It then reviews the history of regulation of uranium mill tailings, the development of standards, and the perceived inconsistencies among these standards. The balance of the chapter examines a number of choices that must be made in adopting a risk-management strategy for uranium mill tailings, drawing on material presented in preceding chapters.

THE PROBLEM

Uranium mill tailings are the residual waste products created by the processing of ore to extract uranium. After uranium ore is mined, it is usually ground to the consistency of fine sand. Uranium is recovered by an acid-, or less frequently an alkaline-, leach process followed by ion exchange or solvent extraction to yield ammonium diuranate—$(NH_4)_2U_2O_7$—a first-stage yellowcake. At least 90 percent of the uranium is removed from the ore during processing. The residue, called tailings, is a slurry of relatively coarse sands and fine particles and contains more than 90 percent of the radionuclide species produced in the uranium decay chain and more than 99 percent of the mass of the original ore. In addition, the tailings contain varying amounts of the chemicals employed in the extraction process and potentially toxic elements that were part of the rock containing the ore (see Chapter 2). Generally uranium mill tailings are deposited above ground in

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rectangular piles with central ponds containing the slimes and liquid surrounded by a solid, predominantly sand, border. In the western semiarid mining districts, most of the moisture evaporates, although some water sinks through the pile and may enter the underlying groundwater. This continuous process leaves growing mounds of fine ground material, topped by evaporating ponds, underlain by a damp core.

As of early 1983, approximately 200 million metric tons* of uranium mill tailings had accumulated (U.S. Environmental Protection Agency, 1983b). These tailings piles encompass an area of about 1300 ha at more than 50 sites, chiefly located in Colorado, New Mexico, and Wyoming, with some in Arizona, Idaho, North Dakota, Oregon, South Dakota, Utah, and Washington. Only one site is in the east, at Canonsburg, Pennsylvania.

The concentration of radioactivity in the tailings is low compared with that in other types of radioactive wastes and consists of naturally occurring isotopes. However, the radionuclides and toxic metals in, or adsorbed on, the particles and dissolved in the aqueous phase of the tailings are now more available for dispersal through hydrologic and atmospheric processes than in the original underground ore.

Potential sources of hazards to public health and the environment from mill tailings piles are the following:

- Gaseous ^{222}Rn,† being produced continuously from the decay of its parent ^{226}Ra, is emitted from the piles and decays sequentially into daughter radionuclides ^{218}Po, ^{214}Pb, ^{214}Bi, and ^{214}Po that may, when inhaled and deposited, produce lung cancer. Since ^{230}Th, the grandparent source of ^{222}Rn, has a half-life of about 80,000 years, the radon emission source strength of tailings will ultimately decline at a rate of about 1 percent per thousand years. ^{230}Th will continue to be generated by the decay of the small fraction of ^{238}U that is present in the piles; however, the precursor half-lives are long and the generation rate will be relatively low.
- Radioactive particulates from the surface of the piles may be transported by wind, thus increasing background whole-body radiation for persons located near the piles. Such persons may

* This report utilizes the International System of Units (SI) with cgs or English units in parentheses when that would make the report easier to use.

† Although there are several radon isotopes, the term "radon" means ^{222}Rn throughout this report.

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receive internal radiation exposure by inhalation or ingestion of these particles.

- Radionuclides, heavy metals, or other toxic materials may contaminate surface water and groundwater through runoff or seepage from the piles and then eventually be ingested with food or drinking water. Tailings may be dispersed over a wider area by erosion or flooding.
- Tailings may be used in site grading and building construction, which could expose people to increased whole-body radiation and radon daughter inhalation.

There is considerable disagreement among public policy makers about the significance of the risks to public health and the environment posed by uranium mill tailings. At one extreme, some contend that tailings are the most serious radioactive waste problem of all because of the large volume of the material and the long-lived radionuclides present in them. At the other extreme, some hold that because the level of radioactivity emitted by the piles is relatively small compared with that in background concentrations (even near the piles) and because most of the piles are located at sites away from major population centers, tailings represent a negligible radiological hazard to public health. Public perception of the scope of the problem is complicated by the fact that tailings involve ionizing radiation, which makes the issue of their management politically and ideologically symbolic. The perceptions that radioactive hazards provoke are different than those elicited by other types of risk (Slovic et al., 1980).

This report seeks to examine the scientific foundation for the assessment of hazards posed by uranium mill tailings and thus provide a basis for resolution of some of these differences.

DEVELOPMENT OF THE URANIUM INDUSTRY

As shown in Figure 1-1, uranium mining and milling developed very rapidly in the 1950s. By 1955, the United States was the world's largest producer of uranium ore. Growth was initially stimulated by the U.S. government to assure a reliable domestic supply of uranium for weapons. This impetus, later strengthened by a developing commercial nuclear power industry, led to annual U.S. production of 8600 tons of uranium oxide by 1957—a tenfold increase in only five years. There were 24 ore-processing mills in operation in 1962.

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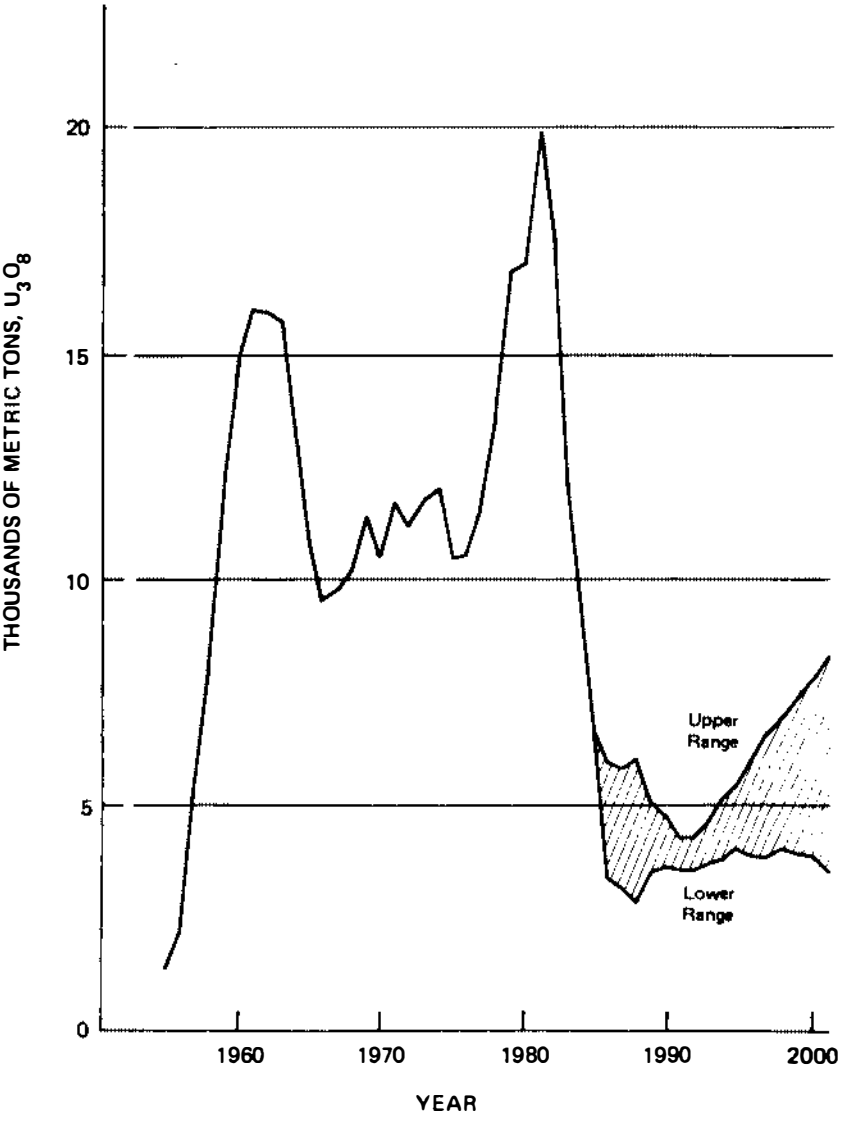

FIGURE 1-1 Historical and projected U.S. uranium production (free-market conditions, low-demand case).
SOURCE: After U.S. Department of Energy (1985).

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Commercial demand was strong in the 1970s owing to domestic growth in nuclear power. The years 1978, 1979, and 1980 each set records for output, with 21 processing mills operating near capacity to process the ore that was steadily declining in grade (U.S. Department of Energy, 1983). Beginning in 1981, annual production dropped; by the end of 1983 there were only 13 conventional uranium mills operating in the United States, and by 1985 only 2 mills had plans for continued operation.

DOE projects that the annual uranium production in the United States in the next 15 years may be about one half to one third of the annual production during the past 15 years. Inasmuch as the generation of tailings is directly tied to uranium production, future tailings generation will be reduced by similar fractions. Consequently, the major concern of this report is the risk from and management of uranium mill tailings piles at existing sites.

The panel also considers it possible that, in the near term, uranium production in the United States will drop below the DOE projection (Figure 1-1) but that it may increase significantly in the longer term as the result of increased use of nuclear power worldwide. Therefore, this report has also given attention to the need for research and development aimed at developing new technology that might economically reduce or eliminate the problems associated with tailings that may be produced and stored in the future.

REGULATION OF URANIUM MILL TAILINGS

Significant events related to the evolution of the regulation of uranium mill tailings are summarized in Table 1-1. Although the federal government was, for many years, the principal customer for uranium, the U.S. Atomic Energy Commission (AEC), the agency originally charged with both regulating and encouraging nuclear development, was never actively concerned with regulating uranium mill tailings at commercial mills.

After the AEC was split in 1975, its successor as the nuclear regulator, the Nuclear Regulatory Commission (U.S. NRC), declared that acceptable tailings management had to be in place before they would issue an operating license for a mill. Requirements included long-term stabilization of mill tailings and groundwater protection.

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TABLE 1-1 Significant Uranium Mill Tailings Related Events

1948	Federal government uranium procurement program produces 34,000 metric tons of mill tailings.
1951	Epidemiological study of Colorado Plateau uranium miners begins.
1957	Public Health Service (PHS) recommends limiting miner exposure to 1.7 Sv/y (12 WLM/y; 170 rem/y).*
1959-1960	National Council on Radiation Protection and Measurements (NCRP), International Commission on Radiological Protection (ICRP), and Federal Radiation Council (FRC) recommend limiting general population whole-body dose to 0.005 Sv/y (0.5 rem/y) and 0.015 Sv/y (1.5 rem/y) to lung.
1962	PHS issues waste guides for uranium milling industry and warns about environmental pollution.
1966	AEC, PHS, and U.S. Department of Transportation (DOT) recommend that abandoned tailing piles be regulated by states.
1967	Department of Labor, Mine Safety and Health Administration (MSHA) limits miners to 0.6 Sv/y (4 WLM/y; 60 rem/y) (reaffirmed in 1972).
1969	FRC guidance limits miners to 0.6 Sv/y (4 WLM/y; 60 rem/y) effective 1971.
1972	Surgeon General issues guidelines for Grand Junction, Colorado, buildings constructed with mill tailings mandating remedial action at 0.05 WL (0.35 Sv/y; equivalent to 2.5 WLM or 35 rem/y) above background.
1972	Congress acts to clean up Grand Junction tailings used in construction (PL 92-314).
1975	AEC becomes U.S. NRC and Energy Research and Development Administration (ERDA).
1977	DOE supplants ERDA.
4/78	Colorado issues regulations on stabilization and control of mill tailings.
11/78	Uranium Mill Tailing Radiation Control Act (UMTRCA) passed (PL 95-604).
9/80	U.S. NRC issues generic Environmental Impact Statement (EIS) on uranium milling (NUREG-0706).
10/80	U.S. NRC promulgates interim regulations.
10/81	New Mexico establishes stabilization and control regulations for mill tailings.

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TABLE 1-1 (continued)

4/82	Congress suspends U.S. NRC long-term care criteria pending EPA issuance of final standards.
6/82	DOE report to Congress urges relaxation of standards.
8/82	House Armed Service Committee Hearings on Management of Commingled Uranium Mill Tailings.
1/83	EPA issues standards for cleanup of uranium mill tailings at inactive sites via PL 95-604 authorization.
7/83	New Mexico and Colorado Governors criticize EPA inactive site standards as nonpermanent solutions and urge more stringent standards.
9/83	EPA issues Final Environmental Impact Statement for Standards for the Control of Byproduct Materials from Uranium Ore Processing (40 CFR Part 192).
10/83	EPA issues standards for cleanup for uranium mill tailings at commercial sites (40 CFR Part 192).
11/84	U.S. NRC publishes regulations conforming to EPA standard for public comment.
1/85	Mine Safety and Health Administration, U.S. Department of Labor, undertakes re-examination of limit of 0.6 Sv/y (4 WLM/y; 60 Rem/y) to miners.

*Appendix C defines and compares alternative terms for expressing key radiation units.

In response to increasing scientific and public concern about possible hazards of uranium mill tailings, Congress passed the Uranium Mill Tailings Radiation Control Act of 1978 (UMTRCA). This Act established two programs to protect public health and safety and the environment from risks posed by uranium mill tailings: one for sites designated inactive (that is, those not under license and at which all milling has stopped) and one for active sites (still licensed by the U.S. NRC or the state in which they are located but not necessarily actively producing uranium). This dual approach stems from the difficulty in assigning financial responsibility to existing organizations for corrective actions at those sites that have long been inactive.

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The U.S. Environmental Protection Agency (EPA) issued its Standards for Remedial Actions at Inactive Uranium Processing Sites in January 1983 (U.S. Environmental Protection Agency, 1983a). With the standards in place, DOE began remedial action, under a March 1990 deadline for completion. The mill tailings standards contained in 40 CFR 192.12 concern the control of tailings piles, the cleanup of buildings and land, and a procedure for exceptions; they may be summarized as follows (Turi, 1986):

> *Control.* Tailings piles must be controlled for up to 1,000 years to the extent reasonably achievable, but in every case, for at least 200 years. Radon emissions from the disposal site are limited to 20 pCi/m^{2}s, or 0.5 pCi/L in air outside the disposal site. Water must meet existing state and federal standards. Site-specific measures are to be used where needed.
>
> *Cleanup of buildings.* Indoor radon decay products shall not exceed 0.03 WL and, to the extent practicable, shall be limited to 0.02 WL. Indoor gamma radiation shall not exceed 20 micro R/hr.
>
> *Cleanup of land.* The 15 cm surface layer should not exceed 5 pCi/g nor should any 15 cm layer below the surface layer exceed 15 pCi/g.
>
> *Exceptions.* Supplemental standards may be applied on generic or site specific bases where health and safety would be endangered, or where costs clearly outweigh benefits. Exceptions are also provided to avoid cleanup of small amounts of tailings and inaccessible tailings posing minimal hazards.

The goal of the regulatory regime imposed by the UMTRCA at active sites is to assure full stabilization and control of the radiological and nonradiological hazards of the tailings so as to avoid future need for remedial action programs of the scale currently necessary at inactive sites. The standards for active sites became final in October 1983 (U.S. Environmental Protection Agency, 1983b). Following promulgation of these standards by the EPA, the U.S. NRC moved to bring its own regulations into compliance with them (U.S. Nuclear Regulatory Commission, 1984). Tailings at the active sites, now approximately 175 million metric tons, are managed by the mill owners and operators under licenses issued by the U.S. NRC or by Agreement States (Groelsema, 1982).

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These regulatory actions have by no means settled the question of the proper goals for uranium mill tailings management. On July 8, 1983, the Governors of New Mexico and Colorado expressed their concern over what they considered to be an inadequate level of protection and regulation provided by EPA active and inactive standards (Lamm et al., 1983). Environmental groups have unsuccessfully brought lawsuits challenging the standards both for inactive sites and for active sites, claiming that they are inadequate to protect the public health and safety and the environment in the uranium development areas. The states of Colorado and Wyoming joined the suit seeking more stringent requirements. Uranium companies and industry trade groups have brought suits claiming that both standards are overly strict and too costly.

Several basic points of contention about the EPA and U.S. NRC standards that have been raised repeatedly by the various constituencies are summarized below. Whether or not some of these issues have been, or are, the subject of litigation, they all remain part of the public policy debate.

- The standards are not consistent with other public health standards (including some recently promulgated by EPA).
- The actual operational standards should be performance standards rather than, or possibly in addition to, design standards—a point on which many in DOE, industry, and environmental groups agree, although for greatly varying reasons.
- The release limits based on assumed radon flux are considered to be either too strict or too lenient.
- The standards are not based on a proper assessment of risk. Some believe that too much emphasis is placed on the number of people who would be affected rather than on the risk to the maximally exposed individual. They fear that a focus on population risk would set a dangerous precedent, making underpopulated areas into "risk dumps." Others believe that the risk has been overestimated, placing too much emphasis on the maximally exposed individual and taking too little notice of the overall level of risk.
- In approaching the control of radon from piles, too little attention has been paid to the relative contribution of radon from background outdoor sources and from indoor sources. In addition, too little attention has been paid to the variability in the contribution of different sources.

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- The emphasis on control of radon emissions may lead to inadequate consideration of other potential sources of hazard associated with uranium mill tailings piles.
- Technology requirements are inappropriate. Some industry and DOE spokesmen have stated that the investment required to meet the standards is disproportionate to the reduction in risk to public health. Some environmentalists say that it is not enough for the standards to be based on current technology but rather that they should be stringent enough to force the industry to develop new methods to control possible environmental hazards economically as well as effectively.

These controversies illustrate the difficulties in making public policy decisions when there is genuine disagreement on both the effectiveness of various solutions to control hazards that will persist for tens of thousands of years and on the levels of acceptable risk. While it is not in the province of this panel to indicate what regulatory or policy decisions should be made, the panel has examined alternative frameworks for assessing risk and risk-management strategies, recognizing that regulatory philosophies are not fixed once for all time. Rather, they should undergo periodic review and perhaps revision as our knowledge, capabilities, and social priorities change. In his congressional testimony on the new EPA mill tailing standards in October of 1983, EPA Administrator William D. Ruckelshaus made this point quite explicitly (Ruckelshaus, 1983):

> Let me conclude with some views on the future of this standard. I don't think any of the difficulties I've just described bar us from continuing our efforts to grapple with the issues posed by this standard. In fact, the act specifically authorises the Administrator to periodically revise the standards set under it. . . .
>
> Information on the technical and economic aspects of alternative levels of control may become available within the next several years as the Department of Energy proceeds with the disposal program for inactive tailing piles. We may find that experience improves our ability to design disposal systems, or, alternatively we may conclude that we have grossly underestimated practical problems and costs. As we continue to think about these questions we may perfectly well develop new approaches to the regulation of toxic pollutants in general that would change our approach to regulating mill tailings. We may also develop new approaches as mill tailings regulations are actually implemented and we have a chance to learn from experience how they operate. If either of these things happen, then I do think it would be proper, and probably

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even necessary, to take steps to amend the regulations that we are discussing today.

It is in the interests of both society as a whole and the uranium industry to clarify the issues, improve the database, and invest in research to lower the cost for adequate protection. These matters are considered in the body of this report.

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2
Exposure Processes: The Transport of Radionuclides and Toxic Chemicals from Tailings to the Environment

INTRODUCTION

Uranium mill tailings are the source of a variety of environmental loadings including radon gas and its daughter products, airborne particulates, and a variety of surface-water and groundwater contaminants. Most of these contaminants originate in the ore, but their mobility is substantially increased by the process of extraction and milling.

This chapter begins with a review of the mineralogy of uranium ores in the western United States. It then briefly describes the two principal milling processes, acid leach and alkaline leach, that have been used in the production of the existing tailings. These introductions are followed by discussions of the physical and chemical nature of tailings and their evolution over time.

The remainder of the chapter is devoted to a fairly extended discussion of the transport of contaminants from the pile to the environment. It begins with a discussion of radon, then discusses groundwater contamination, and concludes with a brief discussion of physical transport of tailings material away from the pile.

NATURE OF URANIUM ORES

Most uranium production in the United States has been from sandstone-type deposits that are widespread on the Colorado Plateau and adjoining regions in New Mexico, Colorado, Utah, Wyoming, and Texas. Vein and replacement deposits are not discussed further in this report. For further detail on both sandstone and nonsandstone types of deposits, see Brookins et al. (1981).

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TABLE 2-1 Averages and Ranges of Critical Minor Elements and Radionuclides in Sandstone-Type Ores

Element	Average	Range	Number of Samples or Analyses
Sandstone Ore, Grants Mineral Belt, parts per million			
Mo	22	0.3-15,000	326
Se	60	0.4-11,000	374
V	634	30-15,000	326
As	38	8-330	18

SOURCE: Spirakis et al. (1981).

The typical feed to a uranium mill is a dark impure sandstone containing 0.1 to 0.2 percent U_3O_8, which in unoxidized ore occurs as disseminated pitchblende (U_3O_8), coffinite [$U(SiO_4)_1 - X(OH)_4X$], and, in some ores, possibly as a urano-organic phase. Vanadium, in some deposits, is more abundant than uranium. Selenium and molybdenum commonly are present in significant but much lesser amounts (see Table 2-1 for compositional data on typical sandstone-type ores). Vanadium occurs as roscoelite (vanadium-bearing mica), vanadiferous chlorite, or, more rarely, the oxides montroseite [VO(OH)] or haggite ($V_2O_3 \cdot V_2O_4 \cdot 3H_2O$); selenium occurs as a substitution for sulfur in pyrite or occasionally as native selenium (Se) or ferroselite ($FeSe_2$); and molybdenum occurs as jordisite (amorphous MoS_2). All may also be present as constituents of clay minerals. Pyrite (FeS_2), or more rarely marcasite (also FeS_2), is ubiquitous in minor amounts in unoxidized ore. Calcite ($CaCO_3$) is a common accessory mineral and, if abundant, may qualify the ore for treatment by an alkaline-leach process rather than by the more common acid leach.

The matrix to the ore minerals typically consists of quartz grains, rock fragments, and a variety of interstitial clay minerals. This material, generally making up 99 percent or more of the initial feed to the mill, will become (after grinding) the principal

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solid constituent of the discharged effluent. The clay minerals, including vanadiferous chlorite, illite, smectite, and mixed-layer clays, are of considerable importance, since they are the common carriers of vanadium and may also trap uranium and thorium from solution. The uranium may be incorporated into the clay minerals as surface-adsorbed phases, minute pockets of coffinite or pitchblende, or organic-sorbed species. Most of the surface uranium is removed during the milling process. In the tailings, the clay minerals are concentrated in the slime fraction, where they continue to act as adsorbents for elements such as thorium and radium that are not removed during the milling process.

The ores contain the uranium isotopes ^{238}U (99.3 percent) and ^{235}U (0.7 percent) essentially in equilibrium with their radionuclide daughters. The half-life and energies of the radionuclides of the ^{238}U decay chain are given in Table 2-2. The ^{235}U series will be ignored in the rest of this report, as it contributes negligible hazards compared with the ^{238}U chain. The radionuclides of major concern with regard to uranium mill tailings are the long-lived isotopes ^{230}Th with a half-life of 80,000 years and ^{226}Ra with a half-life of 1600 years; the noble gas nuclide ^{222}Rn with a half-life of 3.83 days, which is produced from the decay of ^{226}Ra; and the various decay products or "daughters" of ^{222}Rn that are discussed further in later chapters. Typical averages and ranges of the minor chemical elements and radionuclides of the sandstone-type ores are shown in Table 2-1.

In the weathering process these elements and radionuclides are released to the groundwater, thereby contributing to background levels. In addition, radon formed in natural soils can migrate through fissures and, depending on the depth and length of path, may be released to the atmosphere. As a result, the concentrations of radon in surface air in the ore districts of the Colorado Plateau are generally higher than average continental air (Table 2-3).

URANIUM MILLING

Ore is milled in order to separate the target minerals from the gangue. The first step generally involves crushing in several stages. This action releases further radon and produces some radioactive dust but essentially does not change the chemical and radionuclide concentration. The crushing may be followed by any one of several techniques for the mechanical separation of ore material from

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TABLE 2-2 Physical Constants for the ^{238}U Decay Series

Nuclide	Half-life	Type of Decay	Effective Energy per Decay, MeV
^{238}U	4.51×10^{9} y	Alpha	4.268
^{234}Th	24.1 d	Beta	0.060
^{234}Pa	1.17 m	Beta	0.868
^{234}U	2.47×10^{5} y	Alpha	4.856
^{230}Th	8.0×10^{4} y	Alpha	4.767
^{226}Ra	1602 y	Alpha	4.869
^{222}Rn	3.821 d	Alpha	5.587
^{218}Po	3.05 m	Alpha	6.11
^{214}Pb	26.8 m	Beta	0.60
^{214}Bi	19.7 m	Beta	2.349
^{214}Po	1.64×10^{-4} s	Alpha	7.835
^{210}Pb	21 y	Beta	0.047
^{210}Bi	5.01 d	Beta	0.444
^{210}Po	138.4 d	Alpha	5.408
^{206}Pb	Stable	--	--

SOURCE: Benedict et al. (1981).

gangue (Brookins et al., 1981), following which acid- or alkaline-leaching processes are used to extract the uranium (U.S. Nuclear Regulatory Commission, 1980). In brief, the uranium in the ore is oxidized to soluble U(VI) in the presence of transporting ligands (i.e., sulfate or carbonate) and the solutions passed through ion exchangers until the uranium content in solution is maximized. Final stripping, drying, washing, removal of impurities, and roasting yield the yellowcake product U_3O_8. Flow diagrams for both the acid and alkaline processes are shown in Figures 2-1 and 2-2. Further details on the operation of conventional milling processes

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TABLE 2-3 Selected Outdoor Average Background Radon Concentrations in Air

Location	Concentration	
	Bq/m^3	pCi/L
World Sites[a]		
South Pole	0.02	0.0005
Indian Ocean	0.07	0.002
South Pacific	0.07	0.002
North Atlantic	0.2	0.006
Japan	2	0.056
Germany	3	0.070
London	3	0.090
Russia	6	0.170
France	9	0.250
U.S. Sites[b]		
New York City	6	0.170
New York Rural	8	0.210
New Mexico	9	0.240
Ohio	18	0.480
Western Uranium Milling Sites[c]		
Monticello, Utah	13	0.340
Salt Lake City, Utah	14	0.380
Durango, Colorado	19	0.510
Grand Junction, Colorado	31	0.830
Northern Hemisphere Estimate[c]	4	0.100

[a]Reported in U.S. Congress, 1982.
[b]Shearer and Sill, 1969.
[c]Reported in National Council on Radiation Protection and Measurements, 1975.

are provided in Appendix D. Typically, in conventional mills, 90-95 percent of the uranium is extracted, but nearly all of the other radionuclides, particularly the ^{230}Th and ^{226}Ra, are included in the tailings. Chapter 4 discusses process alternatives that may make it possible to remove these long-lived nuclides economically in the milling process.

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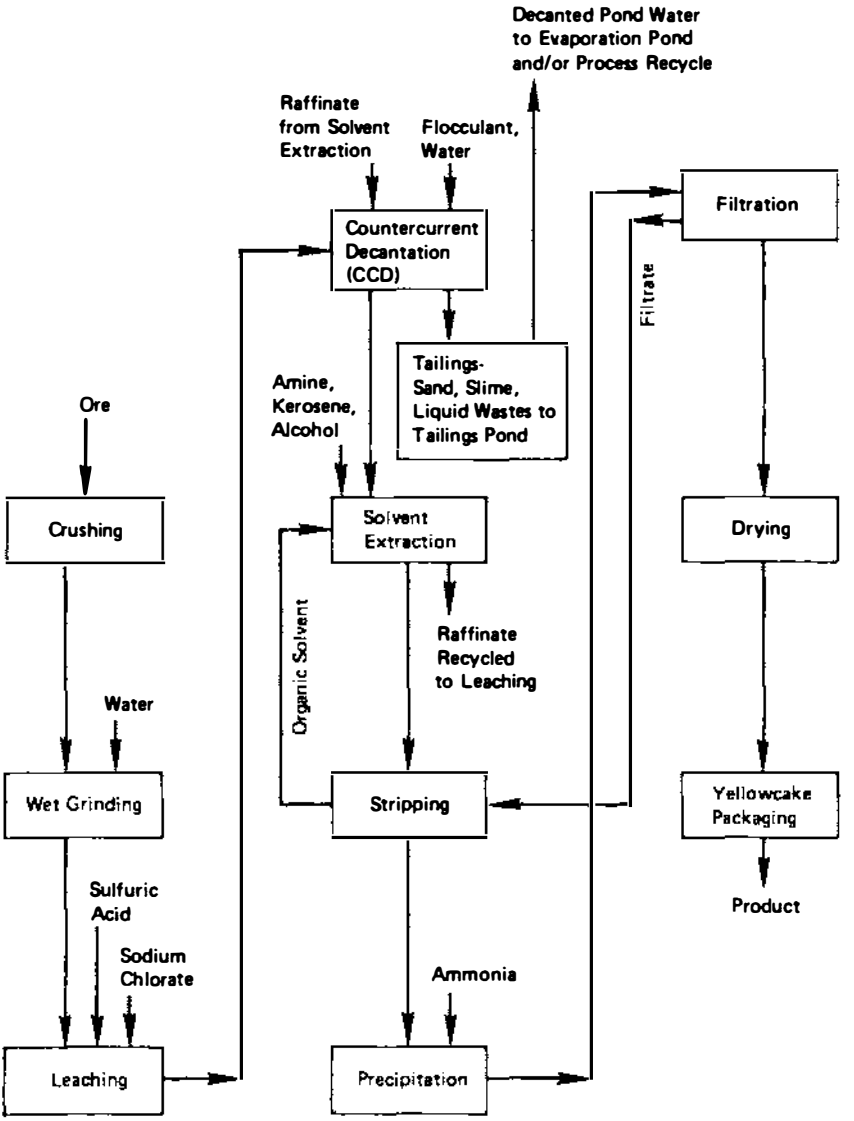

FIGURE 2-1 Flow diagram for acid-leach process.
SOURCE: U.S. NRC (1980).

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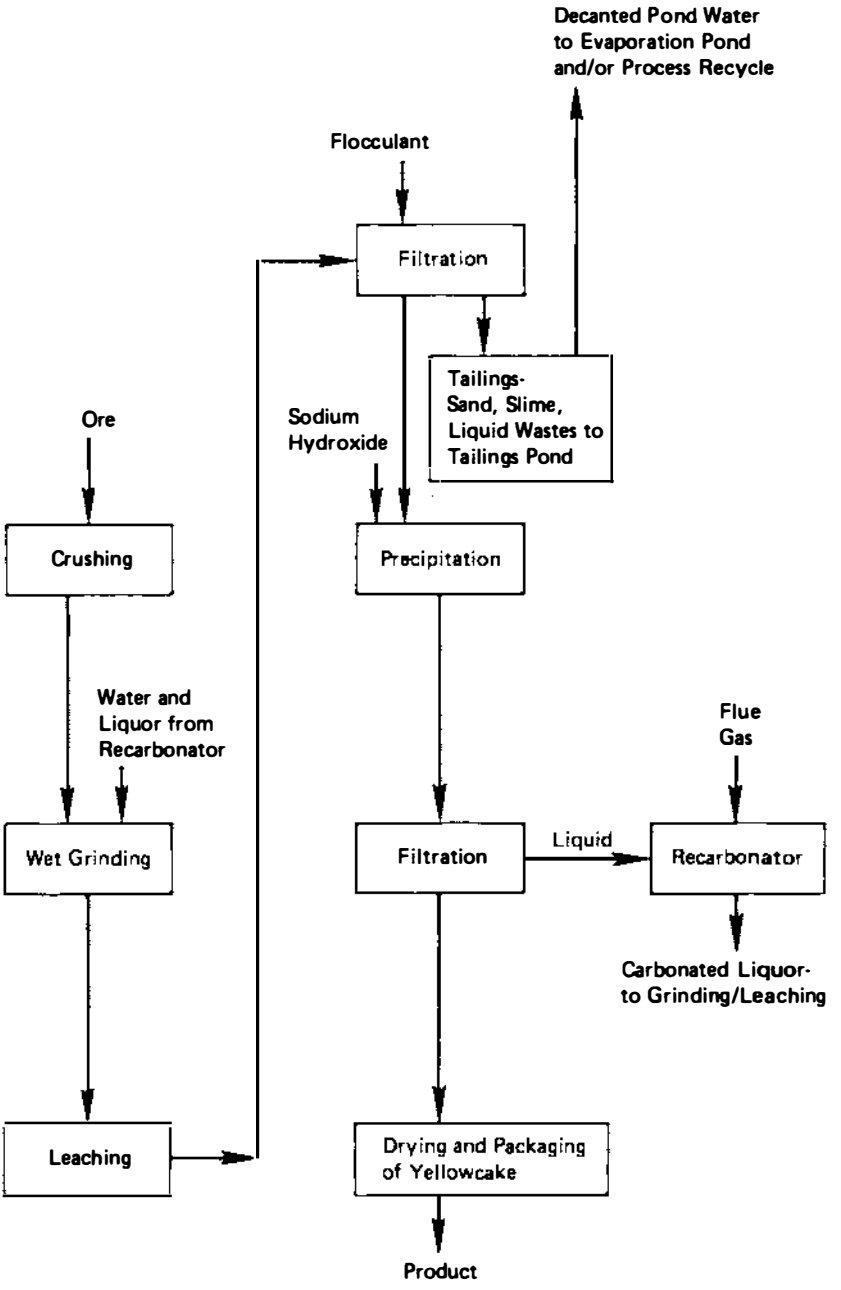

FIGURE 2-2 Flow diagram for alkaline-leach process. SOURCE: U.S. NRC (1980).

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Emissions and effluents from the milling process itself are summarized in Table 2-4 for a typical 1800 metric ton/day mill. Potential hazards from these emissions, most of which are small, can be controlled or eliminated using known technology.

URANIUM MILL TAILINGS

At the time of discharge from the mill, the tailings consist of a slurry (density about 1.6 g/cm^3) generally containing about 40 percent solids (30-40 percent slimes, 60-70 percent sand) and 60 percent water. This is pumped through either steel or plastic pipes to a tailings pond, which is typically a basin bounded by sand embankments. A generic, model mill that has been considered by the U.S. Nuclear Regulatory Commission (1980) generates about 2000 tons/day of dry tailings. On average, about 30 percent of the tailings liquid is recycled for use in uranium milling. The net consumption of water is about 1400 tons/day, most of which is lost ultimately by evaporation from tailings ponds. In some cases, fluids may have to be transferred to additional evaporation ponds or, in extreme situations, disposed of by injection into deep wells (Lynn and Arlin, 1962). It might be possible to design mills to recycle water from the piles, but this is not generally done.

Initial Physical and Chemical Composition

The sand fraction of tailings consists of solids greater than 75 μm in diameter composed of quartz, feldspars, rock fragments, minor amounts of calcite, and other minerals. Some of these grains are coated with clay minerals as well. Slimes consist of solids smaller than 75 μm in diameter, composed of finely ground rock-forming minerals, unleached ore minerals, organic material, clay minerals, and calcite. The liquids contain ore-generated chemicals and process reagents (see Table 2-5 for the output of a typical mill at Church Rock, New Mexico). The liquid phase of an acid-leach mill has a pH of 1.2 to 2.0, contains Na^+, NH_4^+, $SO_4^=$, Cl^-, and $PO_4^=$, 0.001 to 0.01 percent U, 700 to 280,000 Bq of $^{226}Ra/m^3$ (20 to 7500 pCi of $^{226}Ra/L$), and 70,000 to 800,000 Bq of $^{230}Th/m^3$ (2000 to 22,000 pCi of Th^{230}/L). For an alkaline-leach process, the liquid phase of the mill effluent has a pH of 10-11 and contains $CO_3^=$ and HCO_3^-, about 7000 Bq of $^{226}Ra/m^3$ (200 pCi of $^{226}Ra/L$), and essentially no ^{230}Th (insoluble).

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TABLE 2-4 Emissions and Effluents from a Typical 1800 Metric Ton/Day Mill

Emission/Effluent	Source	Rate	Hazard
Ore dust	Ore storage pads and crushing and grinding	6.3 kg/d	Radionuclide ingestion/inhalation
U_3O_8 (yellowcake)	Drying and packaging	1.4 kg/d	Radionuclide ingestion/inhalation
Organics (92 percent kerosene)	Solvent extraction system	70.0 kg/d	Toxic chemicals
SO_2, H_2SO_4 fumes	Acid-leach tank venting	1.0 kg/d	Lung damage
SO_2	Fuel oil combustion	22.0 kg/d	Lung damage
NO_2	Fuel oil combustion	5.0 kg/d	Lung damage
Sewage	Washrooms, showers, etc.	30,000 L/d	Bacterial insult
^{230}Th, ^{226}Ra	Ore crushing and grinding	33 MBq/y (0.90 mCi/y)	Radionuclide inhalation/ingestion
^{230}Th	Yellowcake drying and packaging	27 MBq/y (0.73 mCi/y)	Radionuclide inhalation/ingestion
^{222}Ra	Ore hauling, crushing, and grinding; emission from storage pad	2.5×10^{12} Bq/y (68 Ci/y)	Lung cancer

SOURCE: U.S. NRC (1980).

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TABLE 2-5 Breakdown of Uranium Mill Output to Tailings Pond at Church Rock, New Mexico

Point of Release	Solids, kg/min (lb/min)	Water, L/min (gal/min)	H_2SO_4 kg/min (lb/min)	NaCl, kg/min (lb/min)	Polyacrylomide, kg/min (lb/min)	Kerosene, L/min (gal/min)	Amine, kg/min (lb/min)	Isodecanol, kg/min (lb/min)	NH_3, kg/min (lb/min
Leaching thickener	2500 (5600)	1670 (440)	7 (15)	1 (2)	0.2 (0.4)				
Extraction		980 (260)				15 (0.4)	0.3 (0.6)	0.1 (0.2)	
Sand filter dryer		87 (23)							3.3 (7.3)
Total	2500 (5600)	2737 (723)	7 (15)	1 (2)	0.2 (0.4)	1.5 (0.4)	0.3 (0.6)	0.1 (0.2)	3.3 (7.3)

SOURCE: Longmire (1983).

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TABLE 2-6 Elemental Composition of Uranium Mill Tailings Composite from Two Mines

Element	Shiprock, N.M., Mine Sands	Shiprock, N.M., Mine Slimes	Durango, Colo., Mine Sands	Durango, Colo., Mine Slimes	Vicinity Average Soil
Si, %	42	25	41	27	--
Al, %	1.3	4.0	2.5	6.3	--
Ca, %	1.3	7.7	1.3	4.2	--
Fe, %	0.3	1.2	1.4	4.1	--
K, %	0.8	1.7	0.8	0.9	--
Mg, %	0.1	0.5	0.5	1.1	--
Na, %	0.1	0.2	1.2	3.8	--
Ba, μg/g	820	1030	1700	22000	--
U, μg/g	30	140	630	770	--
Cr, μg/g	20	43	220	550	--
Mn, μg/g	31	86	290	900	--
Ti, μg/g	830	2600	1100	1700	--
V, μg/g	1000	4130	2180	3790	--
Zn, μg/g	100	180	340	1780	--
Co, μg/g	1.2	5.3	24	110	--
Cu, μg/g	20	46	110	730	--
Ni, μg/g	16	49	19	130	--
"Toxics"					
As, μg/g	30	110	72	320	6
Cd, μg/g	2.1	2.2	2.9	21	0.4
Mo, μg/g	26	96	11	8	1.2
Pb, μg/g	120	230	160	200	35
Se, μg/g	59	200	23	29	0.4
Li, μg/g	33	49	36	470	0.3
^{226}Ra, Bq/g (pCi/g)	15 (396)	75 (2020)	27 (719)	67 (1810)	(1)

SOURCE: Dreesen et al. (1983, 1984).

The elemental chemical composition of mill-tailing samples taken in the top 1 m of sand and slime fraction from two mines of the Colorado plateau (i.e., Shiprock, New Mexico, and Durango, Colorado) is shown in Table 2-6. Also given are the concentrations of "toxic" stable elements found in sand in the vicinity.

Also, toxic elements in tailings are a factor of 5 to 100 higher than in background soil. Generally, the heavy metals inside the

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mineral grains of the sand are unavailable to ambient water. The amount of these metals and radionuclides chemisorbed on the surfaces is minor compared with that on the slimes owing to the low surface area on the sand grains. The ^{238}U in these sands is only 30-40 $\mu g/g$ compared with 4000-10,000 $\mu g/g$ in the ores.

Generally about 85 percent of the radioactivity is contained in the slimes (see Table 4-1, later in this report, for specific examples). The most important radionuclides for the long-term hazard are ^{230}Th and ^{226}Ra.

Mineralogy of Thorium and Radium in Piles

The distribution of ^{230}Th and ^{226}Ra in tailings has been investigated in a preliminary way by Stieff (1984) employing a novel microautoradiographic technique. By using 40 samples from drill cores from four tailings piles at the Monticello Mill, Utah, it was found that each of the long-lived radionuclides was distributed in several mineral phases. The ^{230}Th was most commonly (70-80 percent) chemisorbed on surfaces. When these atoms decay to ^{226}Ra, this product would be relatively available for leaching, and the ^{226}Ra daughter, ^{222}Ra, could diffuse along grain boundaries. The second most abundant location of ^{230}Th appears to be in hydrous ferric oxide, which over time would be expected to crystallize into a goethite ($Fe_2O_3 \cdot H_2O$) and probably at these low concentrations would tightly bind the ^{230}Th and its progeny. The third host seems to be in minute grains of original ore, which represents a very small fraction of the ^{230}Th in the pile. ^{226}Ra on the other hand is found primarily in very small grains of (Ba, Ra)SO_4, which is insoluble and has relatively low emanation of ^{222}Rn. A secondary site is in what appears to be radium colloids, which could release radon easily; both the radon and the colloids could be hazardous if inhaled. Some ^{226}Ra is present at a site that remains to be identified. Finally, a few percent of the ^{226}Ra occurs on colloidal films. These results demonstrate the complex mineral environment of the long-lived radionuclides and the potential for diffusion and recrystallization into more stable forms with time. Certainly, compaction and dehydration will cause change. It has already been shown (Brookins, 1981) that the $CaSO_4$ crystallizes out of solution to form crusts over ponds and on top of slimes, reducing the radon flux significantly. To predict adequately the

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internal mineral transformations, the long-term emanation potential, and the leachability of critical elements from the piles, it will be necessary to investigate the mineral assemblages, identifying in which places the radioactive material occurs, and the distribution of the critical radionuclides as a function of location in the pile over time.

TRANSPORT OF CONTAMINANTS FROM THE PILE

Radon

Radon Flux

The release of radon from mill tailings is commonly treated mathematically as due to diffusional loss only. This is an oversimplification as radon loss is affected by, and in some cases may be solely dependent on, convection and other nondiffusive factors.

Radon may enter the pore space of a tailings pile by various processes (Krishnaswami et al., 1982): (1) dissolution of a mineral containing radon, (2) in situ radioactive decay of a dissolved parent nuclide already in the pore space, and (3) direct recoil across the solid-liquid or solid-air boundary. During its birth by radioactive decay in a mineral phase, the amount of radon entering the pore space per unit time is related to the concentration and distribution of ^{226}Ra in the various mineral phases in the pile. Arguments presented by Kigoshi (1971), Krishnaswami et al. (1982), and Rama and Moore (1984) indicate that radon and other isotopes in the uranium-thorium series are largely introduced into the pore space as a result of alpha recoil.

From the local pore space, the radon may diffuse through the intergranular fluids and then into the air. The rate of emission from the surface is largely determined by the air-to-water ratio affecting the permeability and porosity of the pile. The diffusion coefficient for radon in water is 4 to 5 orders of magnitude smaller than in air, thus it is not surprising that the emanation of radon from wet tailings is insignificant compared with that from dry tailings. Furthermore, water in the intergranular spaces largely restricts the pumping action that results from changes in atmospheric pressure.

Radon concentration in air is generally described in terms of Becquerels per cubic meter (Bq/m^3), picocuries per liter (pCi/L)

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or Working Levels (WL). The rate of emanation, or flux, of radon from a surface is defined as the rate at which ^{222}Rn atoms enter the atmosphere across a unit area of that surface per unit time (e.g., atoms/cm^2-s, Bq of ^{222}Rn/m^2-s, pCi of ^{222}Rn/m^2-s).

Overall, for any given substrate and meteorological conditions, the radon flux will be proportional to the rate of its production from ^{226}Ra. The flux is highly variable in space and time: in space because of differences in ^{226}Ra concentrations and porosity and permeability over very short distances, in time because of continually varying meteorological conditions and the degree of water saturation. Furthermore, it is difficult to make an accurate measurement of this flux—even at a specific time on a specific small area—without great care and special equipment (Shearer and Sill, 1969).

Despite the measurement problems, several studies (Kovach, 1945; Kraner et al. 1964; Shearer and Sill, 1969; Clements and Wilkening, 1974) have examined the fluctuations in radon flux and radon concentration in air at a given location produced by changes in atmospheric factors. These studies show a substantial, but far from perfect, correlation between changes in atmospheric pressure and radon flux (Figure 2-3).

The Environmental Protection Agency (EPA), in developing its regulatory approach to radon control from tailings piles (U.S. Environmental Protection Agency, 1983c) assumed an average radon emission rate per unit area, or flux, of 1 pCi of ^{222}Rn/m^2-s per pCi of ^{226}Ra per gram of tailings in all piles. (This relationship also holds if activities are expressed in Becquerels.)

An evaluation of the EPA assumption is provided in Table 2-7, using data reported by EPA for inactive processing sites (U.S. Environmental Protection Agency, 1982). The results are plotted in Figure 2-4. It is clear that, for most piles, the assumption (dashed curve in Figure 2-4) commonly is in error by a factor of 10 or more and generally overstates the flux. Further it is clear that pile-to-pile variation precludes the adoption of any simple model of radon flux. Ignoring such pile-specific details as pile structure, water content, porosity, and gas permeability as a function of depth can lead to very large errors. While data are not available, ignoring the effects of compaction and the probable decrease of porosity over time can probably also lead to significant errors in estimates of long-term radon flux.

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TABLE 2-7 Radioactivity in Inactive Uranium Mill Tailings Piles

Location	Average ^{226}Ra Concentration, Bq (pCi) ^{226}Ra per g of tailings	Estimated ^{222}Rn Flux Based on the EPA Assumption. Emission Activity Rate, Bq (pCi) $^{222}Rn/m^2$-s	Measured ^{222}Rn Flux. Emission Activity Rate, Bq (pCi) $^{222}Rn/m^2$-s
Monument Valley, Arizona	2 (50)	2 (50)	0.5-1 (14-29)
Tuba City, Arizona	34 (920)	34 (920)	0.4-15 (11-400)
Durango, Colorado	26 (700)	26 (700)	1.3-11 (35-310)
Grand Junction, Colorado	29 (780)	29 (780)	0.9-24 (25-660)
Gunnison, Colorado	16 (420)	16 (420)	18 (480)
Maybell, Colorado	10 (270)	10 (270)	3-3.7 (75-100)
New Rifle, Colorado	32 (870)	32 (870)	2.6-52 (70-1400)
Old Rifle, Colorado	37 (1000)	37 (1000)	7.8-48 (210-1300)
Slick Rock (North Continent), Colorado	29 (780)	29 (780)	0.1-9.2 (4-250)

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Slick Rock (Union Carbide), Colorado	25 (690)	25 (690)	0.2-0.9 (6-24)
Lowman, Idaho	20 (530)	20 (530)	1.8-5.5 (50-150)
Ambrosia Lake, New Mexico	24 (640)	24 (640)	1.5-11 (40-300)
Shiprock, New Mexico	26 (700)	26 (700)	2.0-5.9 (53-160)
Belfield, North Dakota	--	--	0.05-2.3 (1.3-63)
Bowman, North Dakota	--	--	1.8-3.5 (48-94)
Lakeview, Oregon	16 (420)	16 (420)	6.9-26 (187-710)
Canonsburg, Pennsylvania	--	--	6.8-11 (185-296)
Falls City, Texas	17 (450)	17 (450)	0.1-2.9 (3-78)
Green River, Utah	30 (810)	30 (810)	1.2-4.8 (32-128)
Mexican Hat, Utah	29 (784)	29 (784)	0.6-59 (16-1600)
Salt Lake City, Utah	33 (900)	33 (900)	0.04-0.7 (1-20)
Converse County, Wyoming	13 (340)	13 (340)	7-106 (190-2860)
Riverton, Wyoming	21 (560)	21 (560)	1.8-3.0 (50-80)

SOURCE: After U.S. Environmental Protection Agency (1982).

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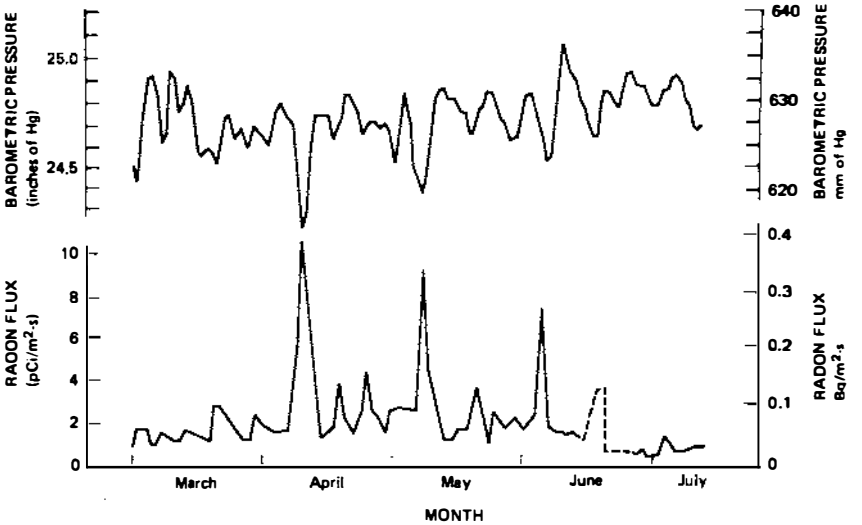

FIGURE 2-3 Example of a comparison of radon flux with barometric pressure.
SOURCE: Kisieleski et al. (1980).

The data of Table 2-7 are not characterized as to moisture content. Thus these data may change as a function of, for example, dewatering with time.

Transport and Dispersion from Piles

Once the radon is emitted from the surface of the pile, it is available for transport and diffusion over long distances, subject to its radioactive decay and to meteorological processes. Like the relationship between flux and the concentration of radionuclides in a pile, the relationship between the flux of radon and the concentration of radon and associated decay products at sites in the vicinity of a pile can be complex and display great pile-to-pile variability.

Various general models have been developed in an attempt to predict the temporal and spatial distribution of the radon after emission. Models such as AIRDOS (Moore et al., 1979), which was used by the EPA in assessing the dispersion of radon from uranium mill tailings piles are based on standard Gaussian plume dispersion concepts (Turner, 1970; Pasquill, 1974; Liu et al., 1982). These models are based on the fact that, over a rather wide range of averaging times, the downwind spatial distribution of

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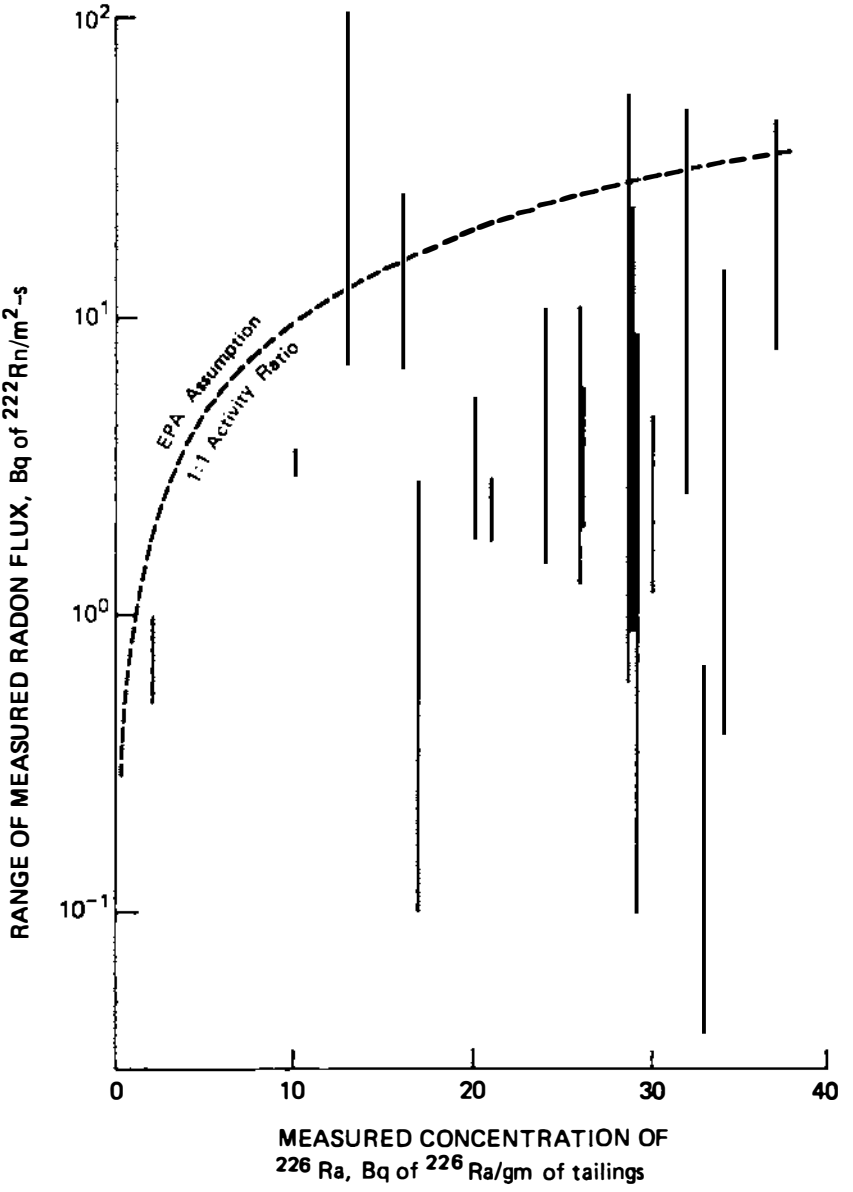

FIGURE 2-4 Comparison of the reported range of measured radon flux from inactive uranium mill tailings piles with the EPA flux assumption. SOURCE: Data are reported in Table 2-7.

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the concentration of material released into a uniform wind field in an atmosphere of specified stability from a point source located on or above a flat surface is well described in both the horizontal and vertical direction by a Gaussian distribution. The width of these distributions at any downwind distance depends on a number of factors including averaging time, some appropriate measure of atmospheric stability, wind speed, and direction. Corrections based on geometrical optics are easily made to incorporate the influence of reflection and vertical containment of material by the ground and by an atmospheric temperature inversion. Nonpoint sources can be synthesized through the superposition of multiple point sources and a number of models that handle the details of such superposition. The problem of rough terrain has been more demanding. Several attempts have been made to modify Gaussian plume models to handle such situations, the best-known example being VALLEY (Burt, 1977).

A limited number of validation studies of the performance of Gaussian plume models under a variety of different operating conditions have been performed (see Appendix of Hillyer et al., 1979, for a summary). These models provide fair performance over moderate distances and in simple flat terrain. In general, performance is better over longer averaging times. There is, however, fairly wide recognition that the performance of Gaussian plume dispersion models can be substantially in error in environments that involve distributed sources whose emissions are transported over complex topography. COMPLEX I and II are two models developed by EPA to provide improved performance over that achieved by VALLEY in complex terrain situations. A set of field evaluations of COMPLEX I and II, using a sulfur hexafluoride tracer, yielded decidedly mixed results. On an event-by-event basis, the correlation between observed and modeled 1-hour concentrations ranged between 0 and 0.3. Performance was better for estimates of the cumulative distributions of 1- to 3-hour concentrations, but even here predicted concentrations differed from observed concentrations by factors of 2 or 3, with predicted concentrations systematically larger than those observed (Gutfreund et al., 1983). The Electric Power Research Institute has for a number of years supported a research program designed both to perform field validation studies and to try to devise more sophisticated physically based models that can outperform simple Gaussian plume models (Reynolds et al., 1984). While some progress has been made

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toward this latter objective, performance of even advanced dispersion models in complex terrain still displays serious inadequacies.

The preceding discussion carries a number of implications for the assessment of radon dispersion from uranium mill tailings piles. Many piles are located in topographically complex areas, and most are themselves topographically complex. Meteorological studies performed by Momeni et al. in the vicinity of the Anaconda and Homestake piles (Momeni et al., 1979; Momeni and Kisieleski, 1980), both of which are of reasonably simple shape and constructed on fairly flat terrain, suggest that even for these piles the resulting flow patterns are fairly complex. For more peculiarly shaped piles located in more irregular terrain, local estimates of the distribution of atmospheric radon concentrations based on the use of Gaussian plume dispersion models are likely to be in error by as much as several orders of magnitude.

The exposure estimates developed by the EPA (U.S. Environmental Protection Agency, 1983a, 1983b) and the U.S. NRC (U.S. Nuclear Regulatory Commission, 1980) are not for individual real piles but rather are for a composite "model pile" that is assumed to be symmetrical in shape and located in flat terrain. In addition, the EPA estimates are for annual average exposure. It is difficult to estimate the potential error in the resulting exposure estimates, and the EPA analyses do not provide any significant discussion of this matter. Integrated measurement of radon concentration can now be made fairly readily, but there has not yet been sufficient environmental monitoring of background radon concentrations in the vicinity of piles to allow much empirical evaluation of the EPA exposure estimates. Further, since for all persons, except those living right at the edge of the pile, outdoor airborne radon concentrations are dominated by background radon, monitoring of the region next to the pile without an understanding of the variations in background that result from shifting wind directions and other atmospheric variations would not be sufficient to allow model validation.

EPA's model pile exposure calculations were presumably undertaken to obtain an order-of-magnitude estimate, but there are three problems with the calculation as actually performed and reported (U.S. Environmental Protection Agency, 1983a, 1983b). First, it is not made sufficiently clear that the model yields only an order-of-magnitude result, at best, and is poorly calibrated. Second, the individual elements of an order-of-magnitude calculation

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TABLE 2-8 Data on Radon Emission Rates and Concentrations at Inactive Sites Prior to Remedial Action

Location	Flux from Tailings, Bq (pCi)m^2-s	Concentration over Tailings, Bq/m^3(pCi/L)
Baggs	3.1 (84)	
Belfield	1.7 (46) (max.)	
Bowman	1.5 (41) (max.)	
Durango	1.3-11.5 (35-310)	
Falls City[a]	0.1-2.9 (3-78)	70 (1.9)
Grand Junction[b]	0.9-24 (25-656)	2200 (59) (max. reported)
Green River[b]	1.2-4.8 (32-130)	
Gunnison Site[a]		133 (3.6) (max. reported)
Lakeview[a]	22.6-26.3 (610-710)	
Lowman	1.8-5.5 (50-150)	
Maybell[b]	2.8-3.7 (75-99)	540 (14.6)
Mexican Hat	0.6-59 (16-1600)	3220-10,400 (87-282)
Monument Valley	0.5-1.1 (14-29)	250 (6.8)
Naturita[b]	28-94 (760-2540)	
New Rifle	2.6-52 (70-1400)	1080 (29)
North Continent[b]	0.15-9 (4-246)	
Old Rifle[b]	7.8-48 (210-1300)	5200 (180)
Phillips/United	1.5-11 (40-300)	
Riverton[a]	1.9-3.0 (51-81)	244 (6.6) (max.)
Salt Lake City (Vitro Chemical)	3.7-33 (100-900)	7.4-500 (0.2-13.4)
Shiprock[a]	2.0-5.8 (53-157)	474 (12.8)
Spook	7.0-106 (190-2860)	630 (17) (max.)
Tuba City	0.4-15 (11-406)	810 (22)
Union Carbide[b]	0.2-0.9 (6-24)	

SOURCE: From data presented in U.S. Department of Energy (1981).

[a]Covered with various thicknesses of material.
[b]Covered with up to 6 in. of material, various states of erosion.

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Other Concentrations, Bq/m^3 (pCi/L)
0.074 (0.002) at 2 km
Background reported as 37 (1) measured at 0.5 km
96 (2.6) at 0.5 km
85 (2.3) at 5.4 km, 33 (0.9) at 0.2 km, background 55 (1.5)
37 (1.0) at 1 to 8 km
22 to 26 (0.6 to 0.7), distances not reported
Background averages 44 (1.2)
Background as high at 110 (3) owing to large amount of ore in region
7.4 (2) at 0.6 km, background 22 (0.6) at 2 km
Background 22 (0.6) at 0.6 km
Maximum 560 (15) at 0.3 km distance, background of 74 (2) at 5 km
Background at 1.6 km
Background at 1.6 km
1040 (28) at 0.2 km, background reported at 110-180 (3 to 5)
Background 41 (1.1) at 1.6 km
Background 22 (0.6) at 0.8 km
37 (1) at site boundary
Background at 3 km
74 (2) at 0.2 km, background 26 (0.7) at 5 km
140 (3.8) at 0.3 km

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should not be any more complex than is necessary for the overall accuracy of the calculation. There is some risk that the use of the fairly complex AIRDOS models may introduce a false sense of accuracy. Third, there is an implicit prior assumption that the appropriate way to undertake the exposure assessment problem is with a model, or generic, pile rather than on an individual pile-by-pile basis.

Measured Radon Concentrations

While reliable models of radon flux and dispersion are not available and measurements of radon flux are difficult, measurements of radon concentration in air, especially time-average concentrations, are simple, reliable, and inexpensive.

Unfortunately, there have not been sufficient systematic measurements of radon concentrations made around most piles. Nevertheless, some measurements have been made at 25 inactive sites prior to remedial action. The results are summarized in Table 2-8.

Extensive atmospheric sampling has been conducted at four sites in the western United States: (1) Grand Junction (operating mill, uncovered tailings), (2) Durango (inactive mill, uncovered tailings), (3) Monticello (inactive mill, covered tailings), and (4) Salt Lake City (operating mill with only vanadium recovery currently being carried out, uncovered tailings) (Shearer and Sill, 1969). The results are summarized in Table 2-9.

In discussing their results, Shearer and Sill (1969) conclude that they could find no correlation between radon concentration and distance from the pile owing to the influence of atmospheric and site-geometry aspects on the readings. They suggest that the pile stabilization carried out at the Monticello site may be responsible for the lower radon concentrations observed both on and off this pile but that this cannot be confirmed since no measurements were made prior to stabilization.

While it may be complex, some relationship should exist between the average integrated radon flux from the tailings pile and the radon air concentration both above a pile and at the downwind edge of a pile. Such a relationship is not discernable in the data presented thus far, largely because of limited concentration measurements at the inactive sites (Table 2-8) or a lack of flux measurements in the study by Shearer and Sill (1969) (Table 2-9). While the data are not for a tailings pile, researchers at Argonne

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TABLE 2-9 Summary of On-Pile and Off-Pile Stations at Four Study Areas

Area	Number of Stations	Number of Samples	Concentration of ^{222}Rn, Bq/m^3 (pCi/L) Average	Range
Grand Junction				
On-pile	5	85	290 (7.8)	40-1000 (1-28)
Near-pile	4	68	70 (1.9)	18-170 (0.5-5)
Other	16	252	30 (0.8)	5-160 (0.1-4)
Durango				
On-pile	2	30	590 (16.0)	140-1300 (4-34)
Station 54[a]	1	10	52 (1.4)	16-85 (0.4-2)
Other	5	72	18 (0.5)	3-50 (0.1-1)
Monticello				
On-pile	4	63	130 (3.5)	33-440 (0.9-12)
Off-pile	8	131	12 (0.3)	1-50 (less than 0.1-1)
Salt Lake City				
On-pile	2	31	270 (7.2)	60-810 (2-22)
Off-pile	10	150	14 (0.4)	2-50 (0.1-1)

[a]About 600 m SE of pile.

SOURCE: Shearer and Sill (1969).

National Laboratory (Kisieleski et al., 1980) have demonstrated such a relationship at the St. Anthony open-pit mine in the Grants, New Mexico, mineral belt (Figure 2-5). Over this 24-h monitoring period, the air concentration varied by a factor of about 4 while the flux varied by a factor of about 2.

In a detailed study over the course of one year, Momeni and Zeilen (1982) measured radon concentrations in air over the Anaconda mill pile and observed diurnal variation by factors of 5-10 and radon daughter variation by factors of 10-50. Data are as follows:

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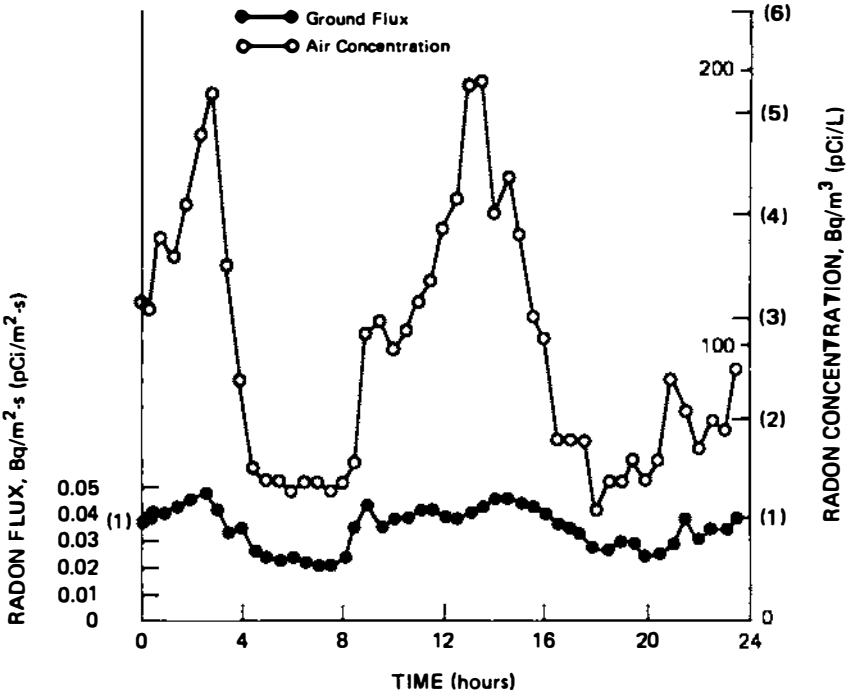

FIGURE 2-5 Comparison of ground flux and radon air concentration at the St. Anthony open-pit mine.
SOURCE: After Kisieleski et al. (1980).

Station/distance (km)	*Period of Sampling*	*pCi/L*	*Bq/m³*
102/1	7-12/77	0.82 ± 0.15	30 ± 8
	1-6/78	0.27 ± 0.10	8 ± 4
103/0.5	7-12/77	1.92 ± 0.31	70 ± 11
	1-6/78	0.79 ± 0.15	30 ± 8
104/1.5	7-12/77	0.51 ± 0.12	18 ± 4
	1-6/78	0.58 ± 0.14	22 ± 4
701, 702/25	12/77-11/78	0.3 av.	11 av.

Here again, at the Anaconda site, it is demonstrated that radon concentrations approach background at 1-2 km from the pile.

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Shearer and Sill (1969) took 48-h continuous samples once every three weeks out to 3.2 km from the Salt Lake City tailings pile. The results averaged over one year, Figure 2-6, show that background, about 13 Bq/m^3 (0.3-0.4 pCi/L), is reached within 1.6 km except in one case downwind, where it may not have been reached until about 2 km.

One of the most comprehensive studies has been done at Canonsburg, Pennyslvania (Rudy et al., 1985), where radon concentration in air has been measured continuously at 40-50 locations surrounding the inactive site of the former Vitro Manufacturing Plant from 1/80 to 7/83 at distances up to 5 km from the edge of the pile. The most important results were as follows:

- The background in this area is about 10 Bq/m^3 (0.2-0.3 pCi/L).
- The maximum on-site concentration was 110 Bq/m^3 (3.0 pCi/L) (11 months' average).
- The maximum off-site (300 m) concentration was 33 Bq/m^3 (0.9 pCi/L) (39 months' average).
- The average annual concentration downwind (east) at 0.5 km was essentially at background.

The monthly variation at a background site and for a downwind site at 300 m is shown in Figure 2-7. A contour map out to background levels is shown in Figure 2-8. These values can be compared with the Department of Energy (DOE) limit of 110 Bq/m^3 (3.0 pCi/L) at the fenceline and the EPA standard of 18 Bq/m^3 (0.5 pCi/L) over background at any off-site location. Thus in this case *without* remedial action the EPA standard (i.e., total of about 28 Bq/m^3 or 0.8 pCi/L) could have been met if the fenceline was at about 0.4 km downwind and about 0.1 km upwind of the tailings pile.

The Monsanto Research Corporation has done detailed monitoring at five sites as a part of the UMTRA and FUSRAP programs (Canonsburg, Pennyslvania; Salt Lake City, Utah; Shiprock, New Mexico; Middlesex, New Jersey; and Lewiston, New York). They conclude that the average radon concentration in air at the pile edge or fenceline of inactive piles without treatment is about an order of magnitude above background, but that it generally drops to background within 2 km. After the proposed remedial action program, the concentration of radon in air at the fenceline should be less than 18 Bq/m^3 (0.5 pCi/L) over background (Rudy et al.,

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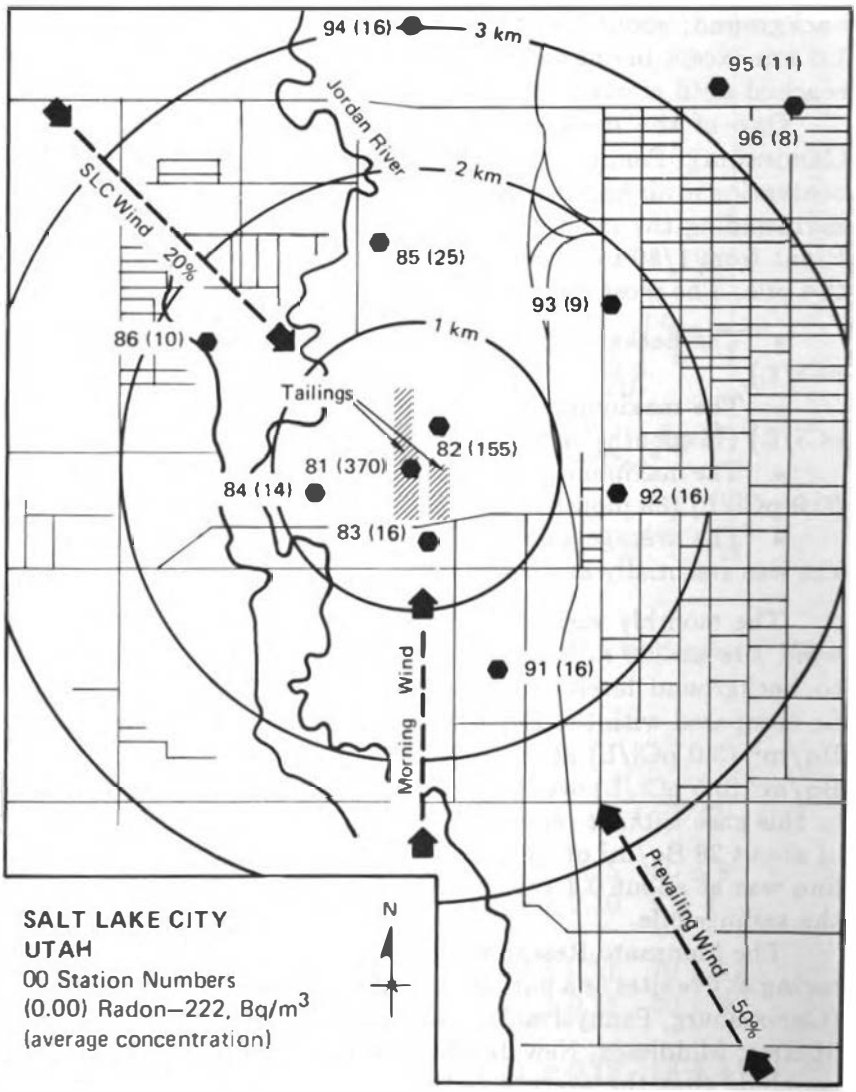

FIGURE 2-6 Radon from tailings at Salt Lake City, Utah.
SOURCE: After Shearer and Sill (1969).

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1984). The data from Table 2-8 suggest that, for most inactive tailings piles, background levels of radon are reached in about 0.3 to 5 km.

In mining districts such as at Grants, New Mexico, air from mine vents is an important source of radon. The highest ambient air concentration near a vent from the Ambrosia Lake Mine was over 220 Bq/m^3 (6 pCi/L), a value above that commonly found at the edge of the tailings piles. Therefore, the general background in the Grants-Milan area has a complex origin from natural soils, ore outcrops, many mine vents, and several tailing piles and ranges from 15 to 30 Bq/m^3 (0.4-0.8 pCi/L) annual average downwind from the district, whereas the background values upwind lie in the range of 7 to 15 Bq/m^3 (0.2-0.4 pCi/L).

Degree of Equilibrium of Radon and Its Daughters

An understanding of the degree of equilibration of radon with its daughters after emission from the piles is necessary to assess the potential health effects. ^{222}Rn has negligible biological effect as it remains in the gas phase, has a short residence time in the lung, and thus delivers relatively little dose to lung tissue. On the other hand, radon in full equilibrium with its daughters, which are present as charged ions on aerosol particles or as unattached ions, can lead to significant exposures.

Very few measurements have been made of the equilibrium of radon on or near piles. Values can be expected to vary considerably depending upon environmental conditions (see Chapter 3 for further discussion).

Monitoring

Measurement techniques for radon and its daughters include scintillator-photomultipliers, thermoluminescent dosimeters, and solid-state electron devices (U.S. Energy Research and Development Administration, 1977). These are active devices generally requiring mechanical or electrical components and cannot measure a long time-integrated concentration. Grab samples are inadequate for such information, and integrating devices such as the above systems are too specialized and costly to be used for long-term measurements at numerous locations.

A simple, low-cost, passive and time-integrating system has been developed (Alter and Fleischer, 1981) and fully calibrated

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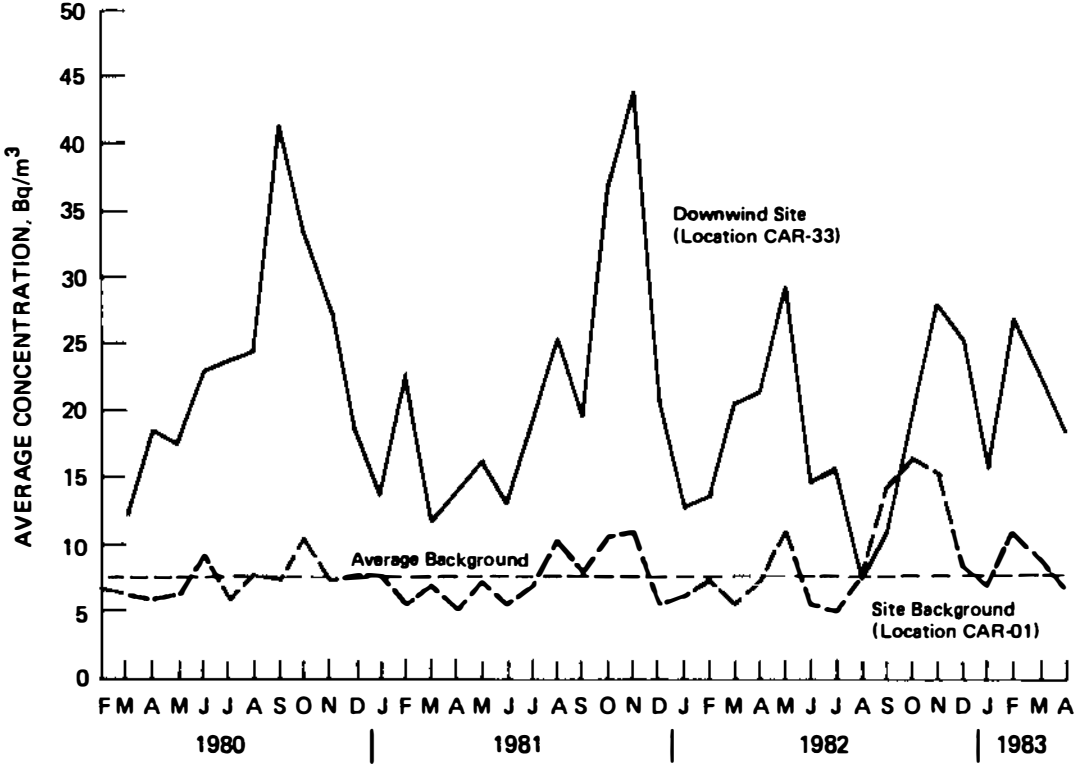

FIGURE 2-7 Monthly radon concentrations, Canonsburg Industrial Park.
SOURCE: Rudy et al. (1985).

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against radon standards and the active methods. This system consists of exposure of a piece of plastic in a cup covered with a filter across its mouth to remove radon daughter products from the outside air so that only radon is measured. Inside the cup, the alpha particles from the radon, and its equilibration daughters formed inside the device, bombard the plastic and cause stable radiation-damage tracks that are subsequently displayed by etching and counted under a microscope. Inasmuch as exposure times can extend over a year or more, the system is ideal for low-level environmental measurement of the average concentration of radon in air. Thus a low-cost, straightforward method exists to establish the current average annual radon flux in air from a pile in all directions out to background levels, to redetermine the pattern after any remedial action, and to monitor this pattern with time, presumably at geometrically decreasing intervals and progressively fewer stations. The accuracy of this system is about ± 20 percent at background levels (for a year-long measurement interval), which lies within the annual variation in natural radon background.

This passive system has been applied at a number of uranium mill tailings sites but, unfortunately, for only a few stations in most cases. One of the more comprehensive studies was around the Shirley Basin Mine/Mill Complex, Wyoming. Figure 2-9 shows the results for stations out to about 6 km. Note that, within the uncertainty, background has been reached at about 2 km.

Implications for Radon Exposure Control Strategy

The preceding technical discussions carry a number of implications for regulatory approaches undertaken with the objective of providing protection from radon exposure for persons living in the vicinity of uranium mill tailings piles. There is considerable site-to-site variability in the flux of radon out of piles and in the distribution of radon around piles given some known flux. The current level of understanding is such that efforts to model flux based on measured concentrations of radionuclides in piles and efforts to estimate airborne concentrations in the vicinity of piles are both liable to be subject to errors of up to several orders of magnitude. For any given site, the combined errors from adopting a modeling approach would thus be so large as to make the modeling results meaningless.

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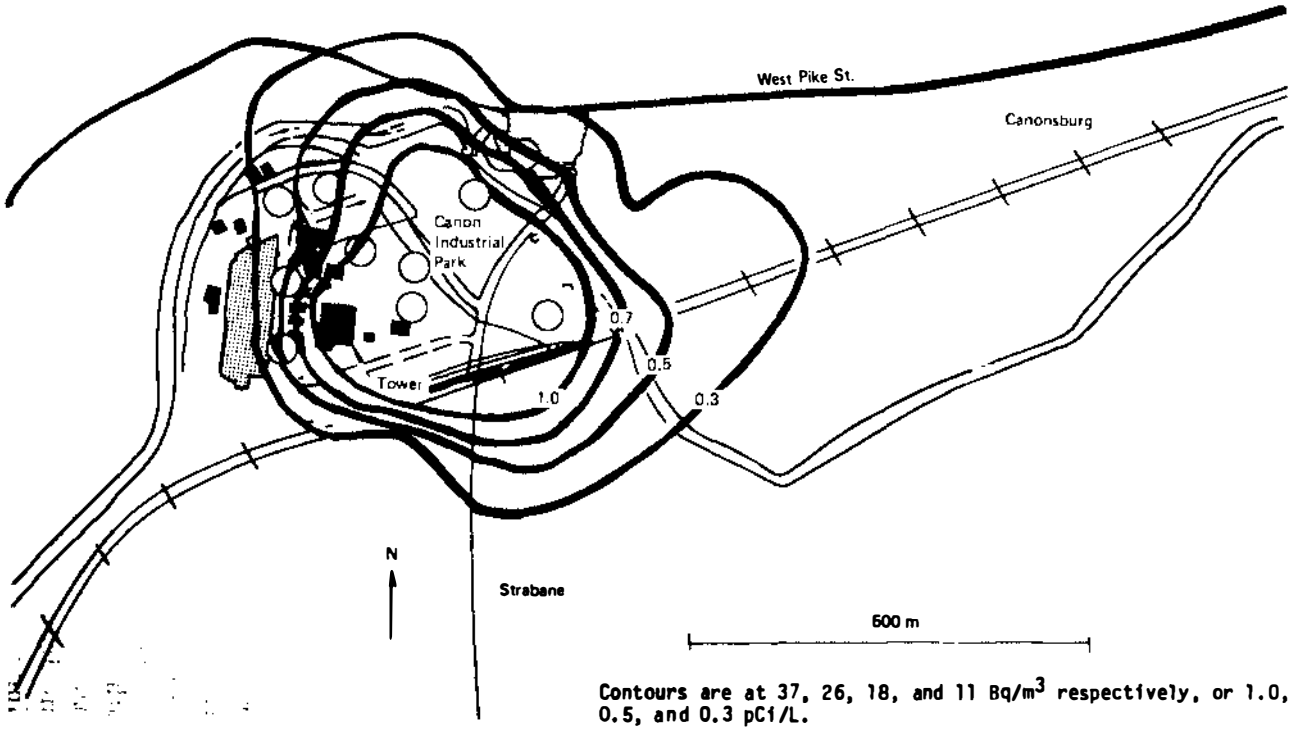

FIGURE 2-8 Contours of equal radon concentration, Canonsburg.
SOURCE: After Rudy et al. (1985). (Note: data based on 1-3 years of continuous measurements.)

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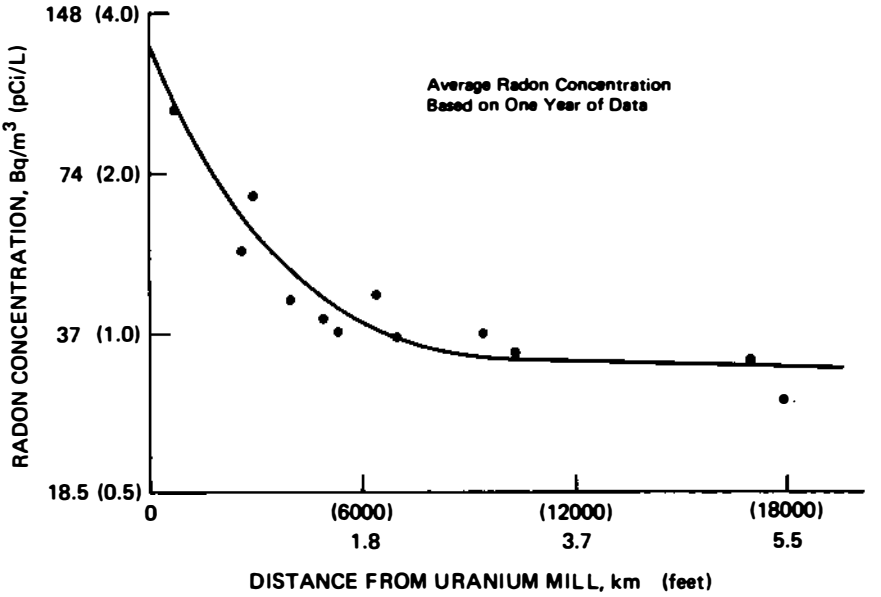

FIGURE 2-9 Variation of atmosphere radon concentration with distance, Shirley Basin Mine/Mill Complex.
SOURCE: Gingrich et al. (1982).

In contrast, integrated measurements of actual airborne concentrations of radon are simple, reliable, and inexpensive. An array of passive radon monitors surrounding a pile can directly yield the actual annual average concentration of radon in air from which appropriate isopleths can be drawn. With background levels deduced from a set of remotely located monitors, one could then directly estimate both the incremental concentration due to the pile and the effectiveness of the control strategy. If measurements indicated a continuing radon problem once controls had been adopted, additional control efforts could be undertaken. The set of data from a number of controlled piles would become an engineering base for the improved design of control techniques for future piles. At some future time, such data might also prove valuable in validating and calibrating improved air-dispersion models.

Together, the preceding considerations suggest that a site-specific exposure management strategy based on direct measurement is far more compatable with available technical understanding and modeling and measurement capabilities than a model "generic pile" approach of the type that EPA has employed in

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developing its current approach to radon emission control (U.S. Environmental Protection Agency, 1983a, 1983b).

Chemicals in the Saturated Zone of Groundwater

The nonradioactive elements potentially available for groundwater contamination are, in some respects, similar to those from metal mining activities in general, as well as those from solid waste disposal. Release from tailings water may occur to the groundwater by the leaching of radionuclides and toxic and other chemicals in the solid tailings and by seepage of tailings pond liquids. The contaminants present will depend mainly on the ore source and the type of processing. Extensive hydrogeologic studies of individual tailings sites indicate that groundwater mounds form under the solid and liquid disposal areas, with hydraulic gradients directed radially from the sites (Williams and Associates, 1983; Williams and Osiensky, 1983). Thus, the contaminants can move to a limited extent in all directions from the source, but the major component of movement is, of course, in the direction of groundwater flow.

Early in the history of the uranium milling industry, little or no concern had been given to the transport of constituents by groundwater flow. Consequently, little monitoring was available before 1977 when the U.S. NRC implemented monitoring programs. These efforts were accelerated in response to the Uranium Mill Tailings Act of 1979. Since the implementation of this program, some data have been compiled for existing and inactive sites. Included in this inventory are the concentrations of elements found in the tailings, in the tailings ponds liquids, and in the adjacent groundwater. In addition, extensive laboratory measurements on column and batch experiments have been performed to assess the mobility of contaminants under various environmental controls.

A plume of contaminants should behave, to a first approximation, as if it were passing through a site-specific chromatographic column. The site-specific features should reflect those processes going on within the source and the subsurface. Within the source itself, the important factors include the height, decay, or buildup of the prevalent groundwater mound, the proportions and kinds of constituents in the source tailings or pond water, and the type of leaching operation. The acid leach will mobilize more of the constituents than an alkaline leach will. In addition, chemical

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reactions or exchange processes that lead to immobilization of constituents are possible within the source itself. This may be an important immobilization process in alkaline-type extractions but may not be nearly so effective in acid-leach operations.

Contaminants produced in the source are emitted into the groundwater flow, generally producing elongated plumes in the direction of flow. In the vicinity of the source, groundwater mounding generally extends the zone of contamination to some length transverse to, or even counter to, the main direction of groundwater flow. The seven active tailings sites discussed by Williams and Osiensky (1983) and the additional five sites reported by Williams and Associates (1983) are all characterized by groundwater mounds and elongated plumes of contaminants in the main flow direction. Figure 2-10 is given here as one typical example of a rather simple plume at the Cotter Corporation mill site in Colorado.

Once the contaminants enter the groundwater flow, the site-specific features that control the migration and concentration levels include the ambient pH of the native groundwater, the buffering capacity available to neutralize the acid leach fluids, the recharge of meteoric water available to dilute the contaminant plume, the types and amounts of organic materials in the subsurface that can provide reaction sites or exchangeable substances, and the hydraulic conductivity distribution and its effect on groundwater velocity and hydrodynamic dispersion. In general, these features may be broadly classified and discussed under (1) source material, (2) physical processes, and (3) chemical processes.

The average concentrations of elements found in seepage from 19 inactive mill tailings have been reported by the EPA (U.S. Environmental Protection Agency, 1983c). These include arsenic, barium, cadmium, chromium, copper, iron, lead, mercury, selenium, silver, vanadium, zinc, and radium. The range in concentrations for each of these species varies significantly. Arsenic, for example, ranges from less than 1 ppm to over 250 ppm. Other reported ranges are from under 1 to 391 ppm for selenium, 1 to 2030 ppm for chromium, 3 to 1080 ppm for copper, and less than 3 to 3060 ppm for lead. Uranium reportedly varies from 50 to 480 ppm and vanadium from 80 to 3990 ppm. Pond waters at the base of tailings piles have been discussed by Williams and Osiensky (1983) and are generally characterized by total dissolved solids in the range

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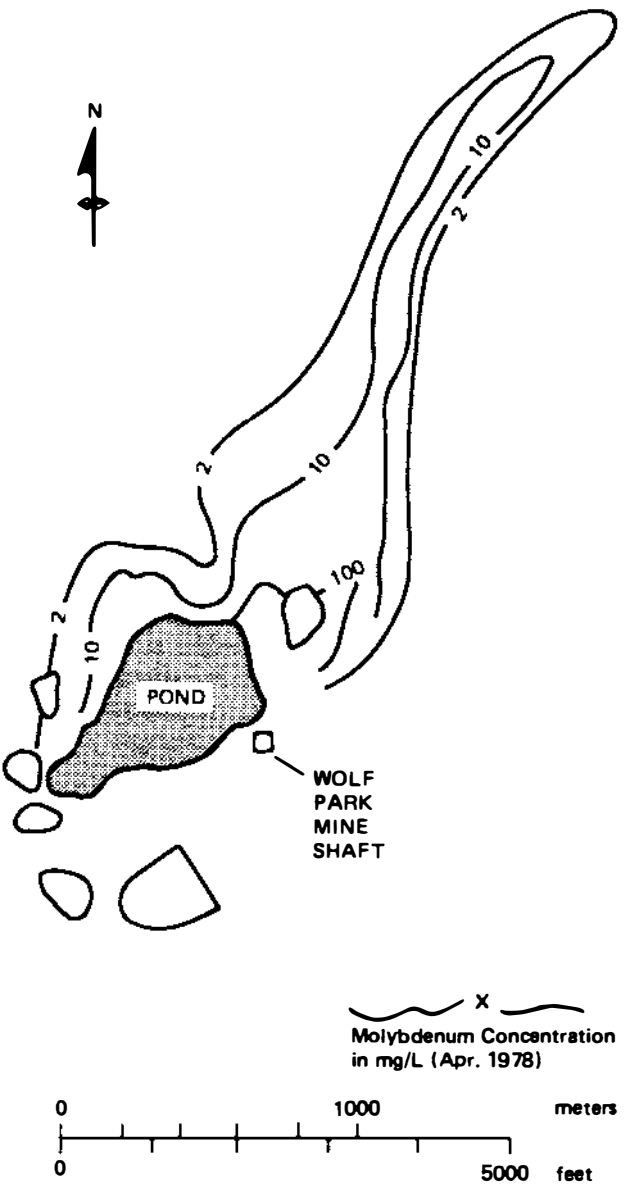

FIGURE 2-10 Distribution of molybdenum in groundwater, April 1978, Cotter Corporation, Colorado.
SOURCE: After Williams and Osiensky (1983).

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12,000 to 30,000 ppm. The major dissolved species in pond waters include sulfate, iron, aluminum, sodium, chloride, manganese, calcium, arsenic, and selenium. The dissolved radionuclides of interest include ^{226}Ra, ^{230}Th, and ^{238}U. The pH of the water in tailings ponds can average 1.8 for mills that utilize acid extraction and about 10.2 for mills that use an alkaline extraction process.

The dominant physical processes in contaminant transport include advection, or groundwater flow, and dispersion. These processes are influenced largely by the geology of the system.

Most uranium mills constructed in the United States between the late 1950s and the middle 1970s are located in fluvial depositional environments. These environments include fluvial plains associated with meandering and/or braided streams, as well as some alluvial fan material. The net result of this type of geologic setting is interbedded layers of sand, silt, and clay, with lenses of sand and gravel. The lenses are the remnants of buried stream channel deposits that are long and sinuous with equally sinuous remnant tributary deposits. Frequently, these buried channel deposits are interpreted as discontinuous between test holes and in cross section because of their sinuousity. Hydrologically, this gives rise to complex sinuous and sometimes intermingling high-conductivity pathways for fluid movement, transecting otherwise lower-conductivity pathways. Some fluid pathways are thus favored, resulting in a fingering pattern of the seepage plume. The hydraulic conductivity distribution in fluvial deposits as determined by Williams and Osiensky (1983) for several tailings sites demonstrates a significant variation in values for any given site, thereby supporting the contention of a heterogeneous geologic environment.

Dispersion is generally considered to be caused by velocity fluctuations and tortuous pathways through which mass is carried by groundwater flow in a porous medium. For heterogeneous formations such as those described above, the concentration patterns that evolve depend more significantly on the contrasts in hydraulic conductivity. Given such contrasts in conductivity, extensive protrusions of water high in concentrates will occur within the high conductivity zones, with lower concentrations in the adjacent lower permeability material. The attenuating effect or dispersion associated with such mixing patterns is shown in Figure 2-11. Chloride, which is chemically conservative, emerges from the source at about

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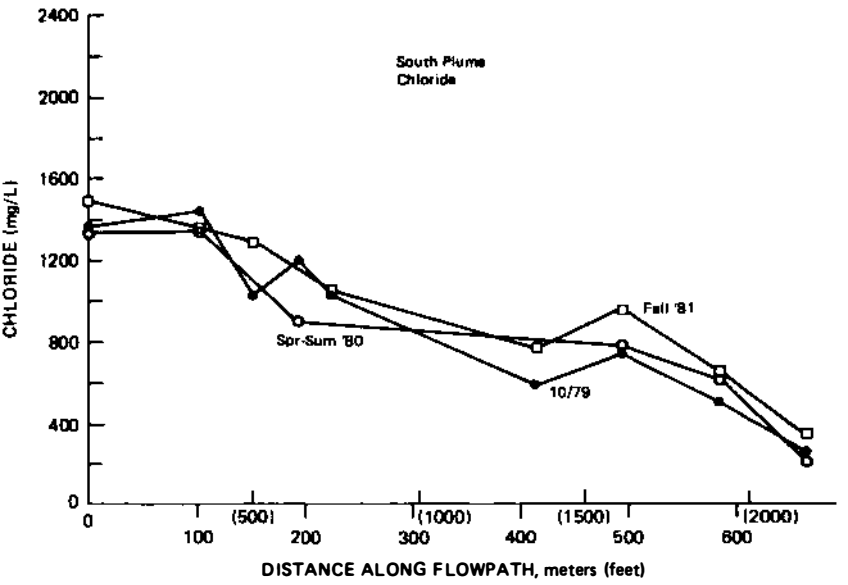

FIGURE 2-11 Chloride in plume. Shirley Basin Mine and Mill, Pathfinder Mining Corporation, Wyoming.
SOURCE: After Williams and Associates (1983).

1400 ppm and diminishes in concentration to about 400 ppm over a flow distance of about 670 m.

Unfortunately, no quantitative data have been collected on dispersivity, largely because it is an illusive measurement even under the most homogeneous of conditions. However, a review of the mixing patterns for several contaminant plumes presented by Williams and Associates (1983) and Williams and Osiensky (1983) leaves little doubt that, at least qualitatively, the dispersion is as described above.

Table 2-10 presents a summary of experimental results of Pacific Northwest Laboratory (1983), where the mobilization potential of various anions, trace metals, and radionuclides has been determined. Almost all of the constituents cited by the EPA (U.S. Environmental Protection Agency, 1983a) as being important in groundwater contamination are given in this table. The immobilization potential was determined from the interaction of a particular constituent and clay material at various hydrogen ion concentrations (pH). These data, coupled with the studies of Markos and

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Bush (1981), Longmire et al. (1981, 1984), Erickson and Sherwood (1982), and Brookins et al. (1982), suggest the following:

1. Arsenic, chromium, lead, silver, thorium, barium, and vanadium are immobilized between pH 3.5 and 8.5. In part this immobilization is due to fixation by clay minerals, but where conditions are favorable for sulfur reduction [i.e., transformation of S(VI) to S(-II) so that pyrite forms] then arsenic, lead, silver, and other chalcopyrite elements are fixed in the sulfides over a wide range of pH.

2. At pH above 4.5, cadmium, manganese, iron, nickel, and zinc are either fixed by clay minerals, precipitated out as oxyhydroxide phases, or fixed in sulfides. At pH above 4.5, uranium is both fixed by clay minerals and precipitated out as oxyhydroxide phases.

3. Predictions of molybdenum and selenium mobility at high Eh and over a wide range of pH by Longmire et al. (1981) are consistent with the data of Pacific Northwest Laboratory (1983). Selenium, though, may be attenuated more than molybdenum owing, probably, to formation of native selenium at Eh values where molybdenum is still mobile.

4. Pacific Northwest Laboratory (1983) suggests that the data for radium are too few to allow quantitative comment on its immobility or mobility, yet since barium is fixed (see above) and since $RaSO_4$ is less soluble than $BaSO_4$, it is likely that radium, too, is even more immobile than barium. This applies not only to clay mineral fixation but also to sulfate mineral coprecipitation and carbonate mineral coprecipitation.

Along with cation exchange, other probable controlling mechanisms discussed by Pacific Northwest Laboratories include precipitation of hydrous oxides, formation of element-specific low-solubility compounds, and anion exchange on protonated hydrous oxide surfaces (Longmire et al., 1981; Markos and Bush, 1981).

The fate of most elements of concern mobilized by waters in mill tailings can be readily explained by use of activity diagrams coupled with Eh-pH diagrams (see Longmire et al., 1981, 1984; Brookins et al., 1982; Longmire, 1983; and Brookins, 1984).

The Fate of Contaminants in Groundwater Flow

Given the experimental and empirical relationships cited above, along with the qualitative arguments on dispersion in groundwater

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TABLE 2-10 Summary of Laboratory Studies of Migration Potential

Element or Constituent	Largely Immobilized at pH Values between 3.5 and 8.5	Largely Immobilized at Near-Neutral pH Values (4.5-8.5) but Mobile at Low pH Values (pH below 4.5)	Mobile	Insufficient Data
Cl			X	
NO_3			X	
Al		X		
SO_4			X	
V	X			
Cr	X			
Mn		X		
Fe		X		
Co		X		
Ni		X		
Cu		X		
Zn		X		
As	X			
Se			X	
Mo			X	
Ag	X			
Cd		X		
Ba	X			
Pb	X			
Ra				X
Th	X			
U		X		

SOURCE: After Pacific Northwest Laboratory (1983).

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flow, certain expectations may be entertained regarding the fate of contaminants at uranium mill tailings sites. Normally, because concentration attenuation can occur in response to both dispersion and chemical reactions, it is not generally possible to separate out the relative influence of each. However, most uranium mill tailings have one other feature in common that permits some assessment of the attenuating capacity of these processes. This common feature is a prevalent pH halo or zoning for those sites using acid-leach operations, which constitutes the vast majority of both active and inactive sites.

Figure 2-12 shows a typical halo for the Federal American Partners Mill site in Wyoming. As noted in the diagram, the pH varies from less than 2.5 up to approximately the background level of 7.4 over a distance of about 1000 m in the direction of flow. For most of the constituents cited in Table 2-10, the lower pH zones are expected to correspond with zones of greatest chemical mobility so that the relative abundance of constituents in the direction of increasing pH provides an opportunity to assess chemical retardation in the field. Dispersion will also result in attenuation of the concentration in the flow direction. Its relative effect can be ascertained, provided some conservative species such as chloride are included in the sampling program.

The data in Table 2-11 represent average concentrations from monitoring points with the stated pH ranges for the Petrotomics Uranium Mill, Wyoming (Williams and Associates, 1983). The mill is an acid-leach facility and has been in operation since 1962. Background data on trace and radioactive elements unfortunately are not available.

The data of Table 2-11 are arranged in accordance with the pH zoning characteristic of acid-leach tailings. It is noted that in the pH range 2.2 to 2.8, which is expected to be in the near vicinity of the source, virtually all of the constituents sampled take on their maximum values, many of which exceed EPA Drinking Water Standards. Chloride decreases from 423 ppm to 118 ppm over the given pH range, which is the result of dispersion. The following constituents become immobilized in the pH range 3.5 to 4.4: arsenic, cadmium, lithium, cobalt, and silver. In the range 4.5 to 7.4, these constituents are joined by aluminum, chromium, iron, lead, vanadium, and uranium. In general, this behavior is in accord with the laboratory data of Table 2-10. However, selenium and molybdenum are not present in these particular waters.

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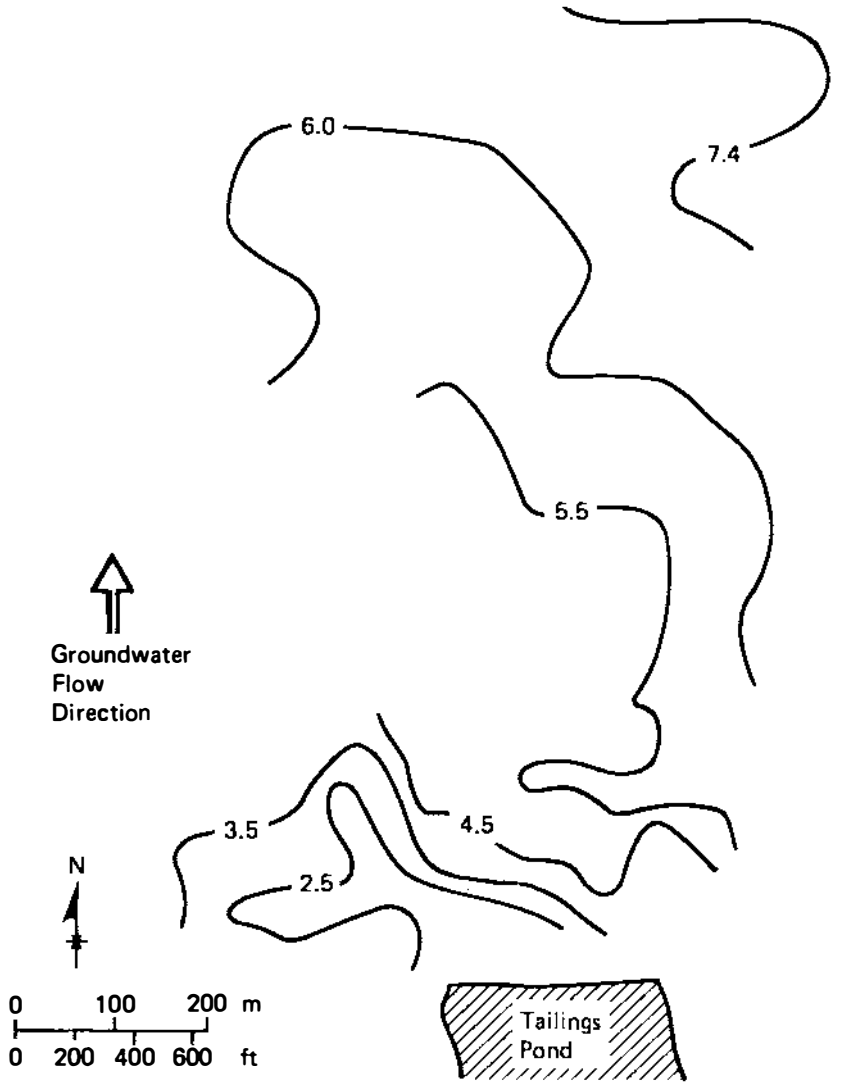

FIGURE 2-12 Typical pH halo of zoning at Federal American Partners Mill in Wyoming.
SOURCE: Data used are from Williams and Osiensky (1983).

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Table 2-11 Chemical Data by pH Zones at Petrotomics Uranium Mill, Wyoming

	pH					Drinking Water Standards*
	2.2-2.8	3.5-4.4	4.5-5.8	6.3-7.4	Background	
			mg/L			
Na	471	544	144	67	39	-
K	26	35	13	11	9	-
Ca	470	476	430	244	39	-
Mg	1,000	771	188	67	13	-
SO_4	19,775	10,921	1,623	770	101	-
Cl	423	379	372	118	13	-
HCO_3	0.	0.	2.5	99	68	-
F	0.05	1.42	1.2	0.2	-	-
NO_3	1.75	14.4	3.35	1.73	-	-
NH_3	355	145	0.2	0.6	-	-
TDS	23,812	13,672	2,790	1,344	238	-
Al	1,599	598	323	0.05	-	-
As	0.3	0.002	0.002	0.0015	-	0.05

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TABLE 2-11 (continued)

	pH					Drinking Water Standards*
	2.2-2.8	3.5-4.4	4.5-5.8	6.3-7.4	Background	
	mg/L					
Ba	0.02	0.02	0.02	0.02	-	1.00
B	1.0	1.0	1.0	1.0	-	-
Cd	0.126	0.072	0.07	0.0065	-	0.01
Cr	5.8	0.61	0.03	0.025	-	0.05
Cu	2.85	0.6	0.08	0.355	-	-
Fe	2,215	997	0.04	0.09	-	-
Pb	0.38	0.28	0.1	0.065	-	0.05
Mn	84	67	6.94	3.64	-	-
Hg	0.001	0.001	0.001	0.001	-	0.002
Mo	0.025	0.05	0.05	0.05	-	-
Ni	4.1	3	0.32	0.13	-	-
Se	0.002	0.002	0.0015	0.0015	-	0.01
V	37	16	0.05	0.05	-	-

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Zn	15	4.9	0.51	0.15	-	-
U	23	6.3	0.134	0.028	-	-
CN	0.005	Trace	0.0025	0.01	-	-
La	0.84	Trace	Trace	0.035	-	-
Co	0.89	Trace	0.015	0.05	-	-
Be	0.08	Trace	0.00125	0.0025	-	-
NO_2	0.08	Trace	0.0025	0.005	-	-
Ag	0.04	Trace	0.0025	0.005	-	-
			Bq/m^3 (pCi/L)			
^{226}Ra	172 (4.46)	179 (4.83)	666 (18)	90 (2.43)	-	185 (5)
^{230}Th	NA	5,513 (149)	466 (12.6)	Trace	-	-
^{210}Pb	NA	NA	703 (19)	NA	-	-
^{210}Po	NA	NA	(0)	NA	-	-

*Drinking water data are from the Safe Drinking Water Act (42 U.S.C. 300 f et seq.).
SOURCE: After Williams and Associates (1983).

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Regarding the inorganics in this environment, there is no apparent sink for calcium in the waters at low pH, nor does any appreciable bicarbonate form below a pH of 5.8. The decrease in sulfate and iron may be suggestive of precipitation of some iron sulfide in the higher pH zones, along with perhaps some calcium and magnesium sulfate. The marked decrease in total dissolved solids reflects largely the decrease in iron and sulfate.

For acid-leach tailings, precipitation of gypsum will effectively remove much of the sulfate, and precipitation of jarosite will remove iron and sulfate (Brookins et al., 1982; Longmire, 1983). At higher pH values, the activity of dissolved carbon dioxide becomes important and calcium carbonate will coprecipitate some of the other metals (e.g., barium, radium, lead, and perhaps cadmium, copper, and zinc), especially in the absence of sulfide.

Results from two other acid-leach sites are shown in Tables 2-12 and 2-13. Table 2-12 presents data from the Pathfinder Mines Corporation Mill site, Wyoming, which has been in operation since 1958. Background water quality is not known, but sulfates in excess of 800 ppm are considered by Williams to be the result of contamination. The decrease in chloride from 925 ppm to 471 ppm is again suggestive of dispersion. The decrease in sulfate is again quite marked over the given pH range, and this is suspected to be the result of precipitation with some constituent (iron?) that was not sampled in this limited sampling program. As expected, thorium also is becoming attenuated at higher pH, as is arsenic.

Table 2-13 summarizes the results from the seepage plume at the Federal American Partners Mill site, the pH distribution for which has already been given in Figure 2-12. This limited sampling program clearly demonstrates the site-specific nature of contaminant migration. Of interest here is the large quantities of uranium (and to a lesser extent thorium) in the groundwaters and the persistance of substantial quantities of these substances in the 6 to 7.9 pH range. In addition, the sulfate is not diminished to the same extent as demonstrated at the previous mine sites.

Although these three examples are not necessarily representative of all cases, some generalizations follow: (1) most heavy metals are immobilized by ion exchange or precipitation reactions that appear to be pH controlled near the pile; (2) chloride and, in most cases, sulfate appear to persist and generally occupy the furthermost points from the pile. For alkaline extraction processes, chloride and bicarbonate are expected to be key constituents.

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TABLE 2-12 Chemical Data by pH Zones at Pathfinder Mines Corporation, Wyoming

	pH		
	2.4-2.9	3.5-4.0	6.3-6.8
As	0.69 mg/L	0.35	0.0035
Se	0.39 mg/L	0.235	0.052
Cl	925 mg/L	915	471
SO_4	18,125 mg/L	11,400	2,941
U	45 mg/L	34	0.6
^{226}Ra	87 Bq/m^3 (2.34 pCi/L)	200 (6)	67 (1.8)
^{230}Th	466,000 Bq/m^3 (12,600 pCi/L)	30,700 (830)	52 (1.4)
^{210}Pb	22,600 Bq/m^3 (612 pCi/L)	6,840 (185)	92 (2.5)

SOURCE: Williams and Osiensky (1983).

TABLE 2-13 Chemical Data by pH Zones at Federal American Partners Mill Site

	pH			
	2.2-3.4	3.5-5.4	4.5-6.0	6.0-7.9
U, pCi/L	18,636	3,635	2,453	1,500
^{230}Th, pCi/L	9,209	78	129	162
Cl, mg/L	432	358	233	163
SO_4, mg/L	9,330	4,598	3,396	3,341
TDS, mg/L	14,445	7,477	7,906	5,487

SOURCE: After Williams and Osiensky (1983).

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However, even in these operations, sulfate may be quite dominant if some sulfuric acid is used in the extraction process. Normally, it is expected that the radionuclides uranium and vanadium will be immobilized, but the data of Table 2-13 indicate that they remain in substantial quantities throughout the plume relative to drinking-water standards. Beyond these generalizations, certain site-specific considerations may play an important role. Boron, selenium, arsenic, and nitrate can occur as complex anions only slightly attenuated by adsorption (Williams and Osiensky, 1983). Nitrate may be important if ammonia is present in the tailings. Ammonia can be oxidized to nitrite if oxidizing conditions persist under the tailings ponds, which may likely be the case before a groundwater mound is established. Last, the quantities and kinds of constituents in the source tailings and ponds can vary significantly, and this will be reflected in the seepage plume.

Movement of Solid Tailing Materials

Solid tailings materials can be dispersed from piles by natural processes that involve gravity, wind, or water. Tailings can also be dispersed through human activities. Water may transport solids from a tailings pile in several ways: (1) erosion by surface runoff, (2) piping that leads to internal erosion, see Appendix E, p. 210, (3) solifluction of unstable slopes, (4) erosion at the base by streams that, over time, have shifted course, or (5) washout by massive flooding. Because these erosion processes are also of great importance in the deterioriation of covers that are designed to prevent windblown dust and to impede the release of radon, we will defer a discussion of erosion to Chapter 4.

Gravity-induced movement is a special case involving simple failure of the outer slope of a pile or its covering and in dry inactive piles is unlikely to move tailings more than several tens of meters. However, the resulting slide may make both the remaining pile and the slumped material more susceptible to loss from water and wind.

Windblown Particulates

Because tailings consist of finely ground material, they can be picked up and blown by the wind. In moderate to high winds, dust and sand can be seen blowing off uncontrolled piles. Once

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suspended, finer respirable materials may stay airborne for significant times and distances. Figure 2-13 shows the rapid decrease of the windblown contaminant from the Vitro pile as measured by decreases in the levels of external gamma radiation. Note that background is reached by 1 km. This is typical of all piles that have been measured. Breslin and Glauberman (1970) measured ^{226}Ra in airborne dust at several mill sites and concluded that concentrations are reduced by a factor of 10 at a distance of 1 km from the piles.

Uranium, ^{230}Th, and ^{226}Ra were measured in the air-filter samples taken in the vicinity of the Anaconda Mill Pile, New Mexico (Table 2-14), by two methods: continuous monthly and shorter-term Hi-Vol samples. Samples 101 and 102 were taken from essentially the same site. At site 104 both sampling methods were used. Within the very wide variations, the methods agree—although the continuous monitors probably give a more accurate monthly average.

Stations 104 and 100 downwind from and nearest to the mill show evidence of yellowcake by the high uranium values compared with those of ^{230}Th and ^{226}Ra. The highest ^{230}Th and ^{226}Ra values (Station 103) were taken 0.5 km east of the pile in the downwind direction. Samples from Station 105 at 2 km was supposed to represent background for this area, and indeed the measurements fall within the range of the upwind samples from Stations 101 and 102.

Airborne particulates from uncontrolled tailings piles would appear, at most, to present a small problem. Almost any management program undertaken to reduce radon emissions from a pile will completely eliminate airborne particulates.

Transport by Man

Uranium mill tailings have roughly the consistency of beach sand. Before the risks became generally understood and appropriate safeguards were put in place, a number of piles were used as sources of sand for construction materials and backfill around foundations. The result was elevated levels of radon in indoor air. In some cases elevated levels of external gamma radiation were also a problem. Once the risks were recognized, a major remedial action program was undertaken to remove the contaminants and make the buildings safe.

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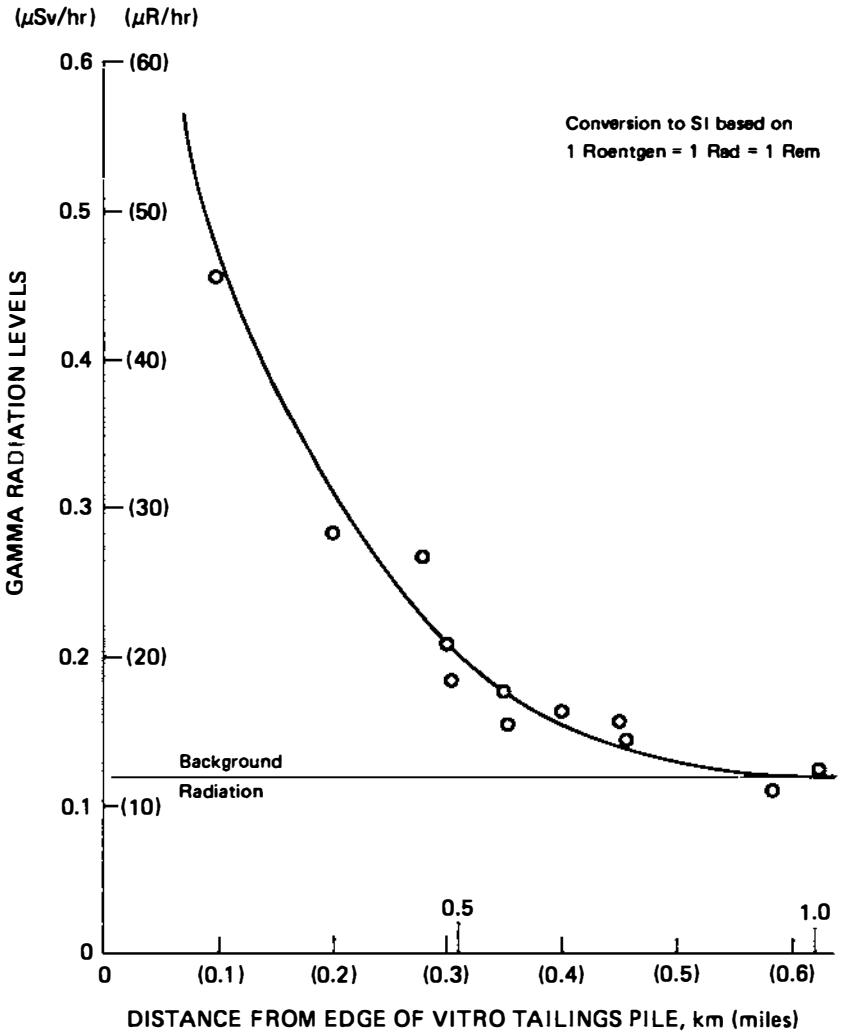

FIGURE 2-13 External gamma radiation levels as a function of distance from the Vitro tailings pile, Salt Lake City, Utah.
SOURCE: Ford, Bacon & Davis Utah, Inc. (1981).

It is clearly important to assure that such inappropriate use of tailings is avoided in the future. Earth covers will impede casual use but probably not serious excavation. At feasible economic levels, the only assured way to guard against possible future human misuse is through adequate social recordkeeping (for example,

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marking tailings piles clearly on topographical and land-use maps), land-use controls (for example, posting of land and zoning), and through a program of low-level but occasional site inspection.

CONCLUSIONS AND RECOMMENDATIONS

On the basis of the preceding discussions, we reach the following conclusions and make the following recommendations:

1. Major chemical differences exist in the composition of the effluents from acid-leach and alkaline-leach processes, but, for either process, most of the radioactivity in the ore, including the key radionuclides ^{226}Ra and ^{230}Th, is contained in the effluent and ends in the tailings pile.

2. The relationship between the concentration of radionuclides in a tailings pile and the flux of radon that leaves the pile is complex. The relationship between the flux of radon leaving a pile and the concentration of radon in air at points surrounding a pile is also complex. There is considerable pile-to-pile variability in both of these relationships.

> *Recommendation*: In developing or revising a risk-management strategy to control the potential risk of radon released from uranium mill tailings piles, the panel recommends a site-specific approach because of the great variability in radon flux at different piles.

3. While the basic factors that give rise to the complexity and variability in the relationships between the concentration of radionuclides in a pile, the flux of radon leaving a pile, and the concentration of radon in the air around the pile are understood, models that attempt to relate pile concentrations to radon flux or radon flux to air concentrations give rise to estimates that may be in error by as much as several orders of magnitude.

> *Recommendation*: In developing or revising a radon risk-management strategy for tailings piles, undue reliance should not be placed on general models of radon emission and dispersion.

4. The existing set of field measurements around uncovered tailings piles shows that at some piles average radon concentrations fall to background levels at a distance of less than a kilometer,

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Table 2-14 Radionuclides on Aerosols in the Vicinity of the Anaconda Mill Pile

Station	Method	Direction	Distance, km	^{226}Ra, Bq/m^3 (pCi/L) Average	Range
100	Hi Vol.	SSW Pile	1.5	67 (1.8)	18-180 (0.5-5)
	1-2 days	SW Mill	1		
101	Hi Vol.	NNW Pile	1	74 (2.0)	4-260 (0.0-7.1)
	1-2 days				
102	Continuous, composited monthly	NNW Pile	1	120 (3.3)	11-590 (0.3-16)
103	Continuous, 3-7 days, composited monthly	E Pile	0.5	270 (7.3)	55-670 (1.5-18)
104	Continuous, 3-7 days, composited monthly	S Pile S Mill	1.5 1	48 (1.3)	15-130 (0.4-3.4)
	Hi Vol.			41 (1.1)	4-140 (0.1-3.9)
105	Hi Vol. 1-2 days	NE Pile	2	30 (0.8)	7-48 (0.2-1.3)
Background	--	Argonne National Laboratory, Ill.			
Background	--	New York State (several sites)			

SOURCE: After Momeni and Kisieleski (1980).

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^{230}Th, Bq/m^3 (pCi/L)		^{238}U, Bq/m^3 (pCi/L)	
Average	Range	Average	Range
74 (2)	18-140 (0.5-3.7)	340 (9.2)	33-810 (0.9-22)
170 (4.6)	4-890 (0.1-24)	81 (2.2)	22-190 (0.6-5.1)
48 (1.3)	11-140 (0.3-3.9)	81 (2.2)	11-240 (0.3-6.4)
200 (5.5)	37-440 (1-12)	67 (1.8)	11-110 (0.3-2.9)
44 (1.2)	15-100 (0.4-2.7)	326 (8.8)	30-1200 (0.8-32)
78 (2.1)	22-190 (0.6-5.1)	410 (11)	26-1600 (0.7-43)
63 (1.7)	18-110 (0.5-2.9)	45 (1.2)	15-92 (0.4-2.5)
2 (0.05)		4 (0.1)	
		15 (0.4)	

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and at all measured piles they fall to background within a few kilometers.

5. At distances of a kilometer or more from tailings piles the equilibrium of radon with its daughters is roughly the same as for radon in background air.

6. Despite the existence of simple, inexpensive, and reliable passive integrating radon monitors, only a modest and incomplete set of measurements of radon in the vicinity of controlled and uncontrolled piles has been collected.

> *Recommendation*: There is a clear need for a systematic program of measurements of radon concentrations around piles before and after the implementation of radon control strategies. These measurements should include an adequate determination of background levels and the generation of concentration isopleths out to background.

In addition to being useful for regulatory and risk-management purposes, the data from such a measurement program might also prove useful in the validation and calibration of air dispersion models.

7. Uncontrolled uranium mill tailings piles hold the potential to produce significant local contamination of groundwaters and surface waters. The pattern and nature of this contamination displays considerable complexity and a large amount of pile-to-pile variability.

> *Recommendation*: The specific strategies adopted to control groundwater- and surface-water contamination from uranium mill tailings piles should be selected and implemented on a site-specific basis.

8. Despite the variability, there are a few features common to most piles: (1) most heavy metals are immobilized by ion exchange or precipitation reactions that appear to be pH controlled near the pile; (2) chloride and, in most cases, sulfate appear to persist and generally occupy the furthermost points from the pile. For alkaline extraction processes, chloride and bicarbonate are expected to be key constituents; however, even in these operations sulfate may be dominant if some sulfuric acid is used in the extraction process. Normally, uranium and vanadium can be expected to be immobilized, but some data suggest that this may not always occur.

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9. For most uranium piles, the spatial extent of groundwater contamination from uranium mill tailings is likely to be limited to dimensions of between several hundred and several thousand meters.

10. When surface-water and groundwater problems arise from uranium mill tailings piles, nonradioactive contaminants may pose the principal water-quality problem. Such water contamination may have many features in common with contamination from mill tailings produced in recovery of some other metals and with that emanating from solid-waste disposal sites.

> *Recommendation*: It is recommended that surface-water and groundwater contamination related to uranium mill tailings not be considered as a separate water-quality problem but rather in the context of the broader issue of water contamination emanating from man-made accumulations in general.

11. There is evidence that, over time, important pile parameters (solubility, speciation, element distribution, pore space, permeability, and others) will change. These changes could significantly affect (probably decrease) the environmental loadings and risks posed by tailings. The available data are not sufficient to allow an adequate description of these changes.

> *Recommendation*: A systematic program of research is required to examine and describe the microscopic and bulk changes that occur in tailing material over periods of many years.

12. Windblown dust may present a local health risk from inhalation or increased external gamma radiation around some piles. Based on available data, the increment of radionuclides from pile dust is indistinguishable from natural background at distances of about a kilometer; however, the data, particularly data on background levels, are sparse. Strategies to reduce radon emanation appear likely to control the problem of windblown dust.

13. Piles can be dispersed by a number of natural and human processes. Over the long term, erosion will dominate these processes for most piles. Detailed conclusions and recommendations about erosion can be found at the end of Chapter 4.

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14. If tailings are misused by man as construction material and backfill for occupied buildings, they can pose a serious health risk.

Recommendation: Successful protection against inappropriate use of tailing materials by man requires a low-level ongoing program of societal recordkeeping (e.g., maps), land-use control, and occasional site inspection.

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3
Potential Biological Hazards

INTRODUCTION

This chapter discusses the potential health risks arising from the dispersal of radioactive and chemical pollutants from uranium mill tailings. The two risks of primary concern are exposure to radon daughters and exposure to contaminated water. Since many features of the water contamination that could result from uranium mill tailings may not be unique, but may be similar to those produced in recovery of some other metals and by effluents from some solid waste disposal sites, the primary focus of this chapter is an examination of the risks arising from the inhalation of the short-lived daughters of radon and subsequent irradiation of the tracheobronchial region of the lung. An excess rate of lung cancer has been observed in various groups of underground miners occupationally exposed to elevated levels of radon, to various animals chronically exposed to radon, as well as to persons who received radiation doses to the lung in other ways, including patients who received x-ray treatment for ankylosing spondylitis and Japanese atomic bomb survivors (National Research Council, 1980).

In addition to radon and water contamination, the chapter briefly considers other potential radiological risks arising from sources such as airborne dust, external gamma radiation, and food contamination.

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ENVIRONMENTAL PATHWAYS TO HUMANS

As a result of the decay of ^{226}Ra, uranium mill tailings release the noble gas ^{222}Rn to the air. The tailings also can be a source of suspended particulates that carry a variety of radionuclides including ^{230}Th, ^{226}Ra, ^{210}Pb-^{210}Bi-^{210}Po, and the short-lived daughters of ^{222}Rn (^{218}Po and ^{214}Pb-^{214}Bi-^{214}Po). These latter radionuclides are also formed in the air from the decay of their ^{222}Rn parent and frequently then become attached to existing particulates.

In addition to the airborne route, materials can be removed from tailings piles by water through erosion or by the leaching of contaminants into surface and groundwaters from the pile and from eroded pile material. Finally, there are a number of gamma-emitting radionuclides in the uranium decay chain that can cause elevated levels of external gamma exposure to people in close proximity to tailings.

The above are primary transfer pathways from the tailings pile to humans. Exposure through ingestion of intermediate accumulators, such as crops and livestock, is an example of a secondary pathway to humans. The various primary and major secondary environmental pathways by which environmental loadings from mill tailings may reach people are illustrated in Figure 3-1 and described in Table 3-1.

While in most circumstances only routes T_1, T_2, T_3, E_1, and E_3 in Figure 3-1 pose significant risks, others can on occasion become important as a result of special circumstances at specific sites. For example, the collapse of an impoundment dam at a tailings pond near Church Rock, New Mexico, resulted in soil contamination miles downstream in the Rio Puerco. After cleanup, the level of remaining contaminants is reported to be low and appears to pose little risk, but continued study of a few pathways such as S_{11} (soil ingestion by livestock) seems prudent (see discussion in Chapter 4).

EXPOSURE VIA AIRBORNE RADON

Of the various known and potential risks to human health posed by uranium mill tailings, radon (and its daughters) has clearly received, and is likely to continue to receive, the major share of attention by both members of the public and government risk managers for the following reasons:

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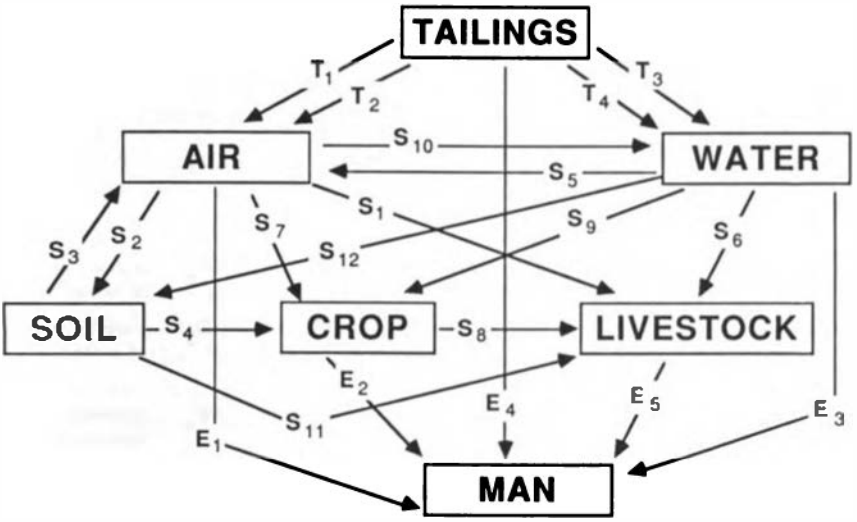

FIGURE 3-1 Major environmental transport pathways from uranium mill tailings to man. T, Primary transfer route from pile to and through the environment. S, Secondary transfer route through the environment. E, Routes by which people can be exposed.

- Clear evidence from studies of former and current uranium miners and from animal studies indicates that chronic exposure to elevated levels of radon can give rise to an increased incidence of lung cancer.
- There has been particular concern by the public about risks that involve radioactive materials and exposure to ionizing radiation (Slovic et al., 1980).
- Use of mill tailings in building materials and foundations in homes and other structures at several locations such as Grand Junction, Colorado, produced conditions (i.e., increased levels of indoor radon concentration) that were clearly hazardous.
- There has been growing public awareness and concern that the naturally occurring high levels of radium in some geographical locations can lead to hazardous radon levels in homes.

Radon-induced lung cancer from inhaled alpha-emitting radon daughters occurs primarily in the more proximal airways of the tracheobronchial region and is generally believed to result from the

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TABLE 3-1 Major Environmental Transport Pathways from Uranium Mill Tailings to Man

Primary Transfers		Secondary Transfers		Exposure Routes	
T_1,	Rn-222 emanation and decay to short-lived daughters	S_1,	inhalation by livestock	E_1,	inhalation
T_2,	resuspension of particulates	S_2,	deposition from air to soil	E_2,	ingestion of crops
T_3,	leaching into surface waters and groundwaters	SS,	resuspension of particulates from soil	E_3,	ingestion of water
T_4,	runoff and erosion	S_4,	crop root uptake	E_4,	external radiation exposure
		S_5,	evaporation	E_5,	ingestion of animal products
		S_6,	ingestion of water by livestock		
		S_7,	foliar deposition		
		S_8,	ingestion of crops by livestock		
		S_9,	water transport to crops		
		S_{10},	fallout from air to surface water		
		S_{11},	ingestion of soil by livestock		
		S_{12},	irrigation		

irradiation of the layer of basal cells near the basement membrane of the bronchial epithelium (Harley and Pasternak, 1982).

In comparison with the alpha doses from the short-lived daughter products ^{218}Po and ^{214}Po, the alpha dose from the 3.82-day half-life ^{222}Rn and the beta radiation from the decay of ^{214}Pb and ^{214}Bi is insignificant and can be ignored. The degree of equilibrium between radon and its daughter products is determined by a number of factors, including the length of time since the radon was released from the pile, the nature and concentration of aerosols

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in the air, and the nature and characteristics of the surfaces on which ions and particles may be lost by deposition. A typical activity ratio of decay products in outdoor air is 1/0.9/0.7/0.7 for $^{222}Rn/^{218}Po/^{214}Pb/^{214}Po$, respectively (National Council on Radiation Protection and Measurements, 1984a). A typical ratio for indoor air is 1/0.5/0.3/0.2. The concentration of the daughter products is typically slightly lower in indoor air because of the greater opportunity for deposition on surfaces.

Since it is the concentration of the daughter products and not radon itself that is of primary importance in determining dose, a unit called the "working level" (WL) was introduced and is often used in radon health studies as an index of the concentration of radioactivity. The WL is defined as any combination of radon daughters in a liter of air that collectively gives rise to 1.3 x 10^5 MeV of potential alpha energy. This is equivalent to the concentration of radon daughters in equilibrium with 3700 Bq of radon per m^3 of air (100 pCi/L). The working-level month (WLM) is defined as exposure to 1 WL for one working month or 170 h. Continuous exposure (i.e., 730 h per month) at a concentration of 1 WL results in an accumulation of approximately 50 WLM/year.

The "equilibrium factor" is the ratio of the potential alpha dose for the daughter products in the air of interest to the potential alpha dose for air in which the daughters are in equilibrium with radon. Thus the equilibrium activity ratios for indoor and outdoor air cited above correspond to equilibrium factors of 0.27 and 0.7, respectively. Measurements by George and Breslin (1980, as quoted in National Council on Radiation Protection and Measurements, 1984a) yield indoor equilibrium factors of about 0.5 and an outdoor factor of about 0.89. Measurements by McGregor et al. (1980, as quoted in National Council on Radiation Protection and Measurements, 1984a) yield an indoor equilibrium factor of 0.6. In calculations performed in a later section of this chapter, we have chosen to represent this variability with a range of equilibrium factors for indoor air of 0.3 to 0.6 and in outdoor air away from piles of 0.5 to 0.8. The contribution to dose from ^{214}Po is greater than from ^{218}Po owing to its higher energy and greater range in irradiating the basel cells of the bronchi.

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Factors Determining Dose and Risk

A number of factors enter into determining the dose that an individual exposed to radon daughters receives, and thus the resulting increased cancer risk. These factors include the following:

1. The concentration of radon over time to which the person is exposed.
2. The degree of equilibrium between ^{222}Rn and its short-lived daughters.
3. The aerosol composition and size distribution. Composition and size are not unrelated since composition can have a significant effect on aerodynamic diameter. Hygroscopic particles can double or triple in size when they encounter the 100 percent humidity of the respiratory tract. Particle size controls both the location and probability of deposition of particles in the lungs and respiratory tract.
4. The concentration of unattached free ions, which influences the probability that radon decay products will attach to aerosols.
5. The fraction of the ^{218}Po not adhering to aerosols. This is important because of the affinity of the unattached ^{218}Po ions to adhere to the inner surfaces of the airways in the upper bronchial tree. The unattached fraction is inversely related to the dust concentrations and the particle size.
6. The type (nose versus mouth), depth, and frequency of breathing. Mouth breathing results in a greater fraction of unattached free ions reaching the upper tracheobronchial tree than does nose breathing, with resulting higher dose. Breathing rates and depth control the amount of radioactivity inhaled and deposited in the airways.
7. Airway clearance rates. This is important because it can determine how much material is removed from the respiratory system before the full decay process has occurred. Clearance rates depend on a variety of factors such as age, health status, particulate loading levels, work history, and smoking habits.
8. Factors such as the sex and age of an individual, personal health status, and personal habits such as smoking also influence risk.

An excellent and detailed discussion of these and other factors that influence dose, and thus risk, can be found in Chapters 4, 5, and 6 of the report of the National Council on Radiation Protection

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and Measurements (1975) and in a later report of the Council (National Council on Radiation Protection and Measurements, 1984b).

The effect of smoking is more complex than one might think. Cigarette smoke can increase both mucus production and the bronchial epithelium thickness (International Commission on Radiological Protection, 1974). A thicker bronchial epithelium may result in lower dose because of the greater distance that the alpha particles must travel to reach the basal-cell layer. On the other hand, the elimination of the 7-μm cilia layer by the smoke irritant permits the ^{218}Po and ^{214}Po ions to migrate to the surface of the bronchial epithelium with a resultant increased dose. Further, cigarette smoke can itself produce lung cancer and is a co-carcinogen that encourages the more rapid development of lung cancer via additive and multiplicative interactions with other agents mechanisms (National Council on Radiation Protection and Measurements, 1984a). The degree to which the effects of cigarette smoking and the presence of radon daughters interact is not clear. There are animal and epidemiological studies that provide evidence that points to both increased and decreased risk among smoking subjects.

Epidemiological Evidence

While there have not been conclusive epidemiological studies that correlate an increased incidence of lung cancer in the general population with increased exposure to radon daughters (Letourneau and Wigle, 1980; Letourneau et al., 1983), considerable work has been undertaken in studying various groups of adult males engaged in underground mining (National Research Council, 1980; National Council on Radiation Protection and Measurements, 1984a; Muller et al., 1984). Causal relationships between lung cancer and high-LET radon daughters have been established in epidemiological investigations of uranium miners in the United States, Czechoslovakia, and Canada as well as in iron miners in Sweden and Great Britain, Swedish lead and zinc miners, and fluorspar miners in Newfoundland (National Research Council, 1980; National Council on Radiation Protection and Measurements, 1984a; Muller et al., 1984).

An increased incidence of lung cancer has also been observed from external low-LET ionizing radiation exposure in Japanese

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atomic bomb survivors and British patients treated with x-ray therapy for ankylosing spondylitis (National Research Council, 1980).

Although the cumulative exposure of approximately 100 WLM had been considered the minimum to be valid in demonstrating an excess of neoplasms (National Council on Radiation Protection and Measurements, 1984a), the cause-effect relationship is generally presumed to be a linear process for high-LET radiation without the presence of a threshold that enables one to extrapolate and predict lung cancer effects at cumulative exposures of less than 100 WLM. There is considerable controversy (National Research Council, 1980) over whether such a linear assumption overestimates or underestimates the risk. In the case of radon, data are available at exposure levels quite close to those of concern, so that the amount of extrapolation required is relatively small.

A summary of the results of five of the radon miner studies is provided in Table 3-2.

There are a number of problems involved in extrapolating from the results of the miner studies to the general population. They parallel the factors listed at the beginning of this section and include uncertainty about the measurements of radon concentration; the employment and exposure histories of miners; changes over time in mine ventilation and in the particulate loadings of mine air and the size and physical and chemical composition of the particulates that can influence the degree of equilibrium and the fraction of attached daughters; the presence of other contaminants such as ore dust and diesel fuel or exhaust; differences in breathing patterns between miners and nonminers; differences in lungs due to a variety of factors including occupational histories, age, and sex; the role of smoking; and the confounding effect of pollutants such as NO_x in homes.

Since the majority of the miners in the epidemiological studies are alive and some will develop lung cancer in the future, it has been necessary to develop models of risk projection to take this into account. An absolute risk model, developed by the Committee on the Biological Effects of Ionizing Radiation (BEIR I) (National Research Council, 1972), defined absolute risk as the number of excess cancer cases per unit of population per unit of time and per unit of radiation dose. BEIR I also adopted a relative risk in which the excess cancer risk for the interval after the latent period was

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TABLE 3-2 Summary of a Number of Studies That Correlate Incidence of Bronchial Cancer in Miners with Radon Daughter Concentrations[a]

United States Uranium Miners

A prospective epidemiological cohort study of about 4000 underground uranium miners from the Colorado Plateau region was begun in 1951. While followup of the miners continues, data complete through 1974 (Archer et al., 1976) provide an average absolute risk of 3.52 cases of lung cancer per 10^6 person years per WLM (National Research Council, 1980). For exposures of less than 360 WLM, the absolute risk estimate is 6 cases of lung cancer per 10^6 person years per WLM, and the relative risk is 0.8 percent per WLM (National Research Council, 1980).

Czechoslovakian Uranium Miners

A cohort of 2200 miners hired between 1948 and 1952 (Thomas and McNeil, nd) received an average cumulative exposure of 300 WLM, which is much lower than that received by U.S. miners before 1960 (National Research Council, 1980). Since only radon concentrations were measured, estimates of the daughter concentrations had to be made (National Council on Radiation Protection and Measurements, 1984a). BEIR III (National Research Council, 1980) calculated the total excess risk for these miners to be 19 cases of lung cancer per 10^6 person years per WLM.

Canadian Uranium Miners

About 10,000 miners who worked in the Ontario uranium mines between 1954 and 1963 received a mean cumulative exposure of 36 WLM (Thomas and McNeil, nd). These miners appear to have an excess of lung cancer in the lowest dose group (1 to 30 WLM; mean 10.9 WLM) at a cumulative dose to the bronchi of 0.04 to 0.09 Gy (4 to 9 rads) (National Research Council, 1980).

Newfoundland Fluorspar Miners

DeVilliers and Wigle (nd) have studied the 2400 fluorspar (CaF_2) miners who received mean cumulative exposures of 204 WLM from radon emanating from mine water (Thomas and McNeill, nd). The absolute risk was 17.7 deaths per 10^6 person years per WLM with a relative risk of 8.0 percent per WLM, yielding a doubling dose of 12.5 WLM (National Research Council, 1980). The latent period of approximately 23 years did not appear to vary as a function of age when mining began (National Research Council, 1980).

Swedish Metal Miners

Estimates of risk coefficients for Swedish iron, zinc, and lead miners range from 7 to 35 lung cancers per 10^6 years per WLM (National Council on Radiation Protection and Measurements, 1984a). The preliminary results suggest that the excess risk for smokers may not be markedly different than that for nonsmokers (National Research Council, 1980). After cancer is induced, smokers died an average of 11 years earlier than nonsmokers. The irritating effect of cigarette smoke to increase mucus production and thickness in the tracheobronchial tree has been hypothesized as increasing the distance that the alpha particles of the ^{218}Po and ^{214}Po deposited in the mucous layer must travel to reach the basal cells of the bronchial epithelium. Smoking, however, can reduce the length of the cilia or denude the cilia, resulting in a decreased distance to the target cells.

[a] A recent Office of Science and Technology Policy report (1986) provides a compilation of later data.

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expressed as a multiple of the natural age-specific cases risk for that population. While each model generates different predictions of risk during the course of the studies, both models should eventually produce the same results for risks at the conclusion of the lifetime studies of the miners (National Research Council, 1980).

Controlled Animal Dose-Response Studies

In addition to the epidemiological studies on man, various studies have been conducted of the biological response in animals from radiation exposure of inhaled radon daughters. Before the 1950s, the focus was primarily on life-span shortening from exposure to radon. In 1953, Shapiro and Bale demonstrated that the radiation dose was a function of the extent of attachment of the radon daughters to dust particles and further postulated that the dose was primarily from the two short-lived alpha-emitting polonium radioisotopes ^{218}Po and ^{214}Po. Desrosiers et al. (1978) later determined the radiation dose to the various conducting and respiratory regions of the Syrian Golden Hamster lung for three radon daughter aerosols of differing degrees of attachment and particle size. The hamster studies show greater radiation insult to the respiratory portion of the lung in contrast to the conductive portions—the opposite of calculations for man. Little et al. (1975) demonstrated an excess of lung neoplasms in hamsters exposed to 0.15 Gy (15 rads) of alpha radiation from polonium daughters (National Research Council, 1980). The three major studies on lung pathology with a large number of animals have been done by the University of Rochester (mice, rats, and dogs); Centre d'Etudes Nucleaires (CEN), Cogema, France (rats); and Battelle Pacific Northwest Laboratory (hamsters, rats, and dogs). Principal characteristics are summarized in Table 3-3.

University of Rochester: Morken and others studied lung pathology with inhaled radon daughters in the mouse, dog, and rat. Radon concentrations were 18 TBq/m^3 (0.5 Ci/L) for the mouse and dog and 37 T Bq/m^3 (1.0 Ci/L) for the rat. Particle sizes were less than 1 μm, with a median of 0.2 μm. There were 20 percent unattached free ions. While the exposures of all animals were quite high, the absence of bronchial or pulmonary carcinoma and their lifespan-shortening effect may have been precluded from developing by the early sacrifice of the animals (National Council on Radiation Protection and Measurements, 1984a).

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TABLE 3-3 Radon Daughter Inhalation Studies in Animals

Institution	Animal	Number	Total (WLM) Exposure	Effect Studied
University of Rochester	Mice	2000	14,000-72,000	Life-span shortening, lung lesions, adenomas, and carcinomas
University of Rochester	Dogs	80	200-10,000	Life-span shortening, lung lesions, adenomas, and carcinomas
University of Rochester	Rats	100	26,000	Life-span shortening, lung lesions, adenomas, and carcinomas
Cogema, France	Rats	3000	20,000 21,000	Life-span shortening, pathological changes, carcinomas
Pacific Northwest Laboratory	Hamsters	800	20-15,000	Life-span shortening, hematological, renal, pulmonary carcinoma
Pacific Northwest Laboratory	Rats	4000	-	Pulmonary fibrosis from ore dust
Pacific Northwest Laboratory	Dogs	80	-	Pulmonary fibrosis from ore dust

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Centre d'Etudes Nucleaires: The purpose of these studies was to correlate the induction of tumors in rats with exposure to radon daughters (Chameaud et al., 1977). After exposures of 500 to 14,000 WLM, various types of pulmonary cancers were observed in rats (National Council on Radiation Protection and Measurements, 1984a). Exposures as low as 65 WLM, with daughter concentrations of less than 5 percent of equilibrium, had an observed lung cancer incidence of 370 cases per million population per WLM (National Council on Radiation Protection and Measurements, 1984a). Chameaud et al. also exposed rats to both cigarette smoke and radon daughters to determine any synergistic effects. Inhalation of the smoke occurred after the radon daughter exposure. The production of tumors was greater with the dual insult than that observed in a group of rats exposed to radon daughters alone (National Council on Radiation Protection and Measurements, 1984a).

Pacific Northwest Laboratory: Both Cross et al. (1978) and Stuart (1978) evaluated life-span shortening and pulmonary effects from different combinations of radon daughters, fumes from diesel exhausts, and uranium ore dust in Syrian Golden hamsters, beagle dogs, and rats. These studies appear more relevant to uranium mining environments than to populations near uranium mill tailings. The results of the animal experiments suggest that the rate of exposure as well as the total exposure influence the incidence of lung cancer. In considering the applicability of the induction of pulmonary cancer by alpha-emitting radionuclides to animal studies to man, it is noted in BEIR III (National Research Council, 1980) that tumor origin in rodents and dogs is commonly bronchoalveolar, whereas the tumor site in man nearly always arises from epithelium in proximal regions of the bronchial tree. The significance of the site difference is unresolved. Interestingly, while the site of lung cancer is different, the similarity of the magnitude of exposures causing lung cancer in both human and animal studies can be seen in Figure 3-2; for perspective, some non-cancer-related risks are quantified in Chapter 5, Table 5-1. Further caveats noted by the National Council on Radiation Protection and Measurements (1984a) on the extrapolation of animal data to man relate to the presence and role of ore dusts and the rate of radon daughter exposure.

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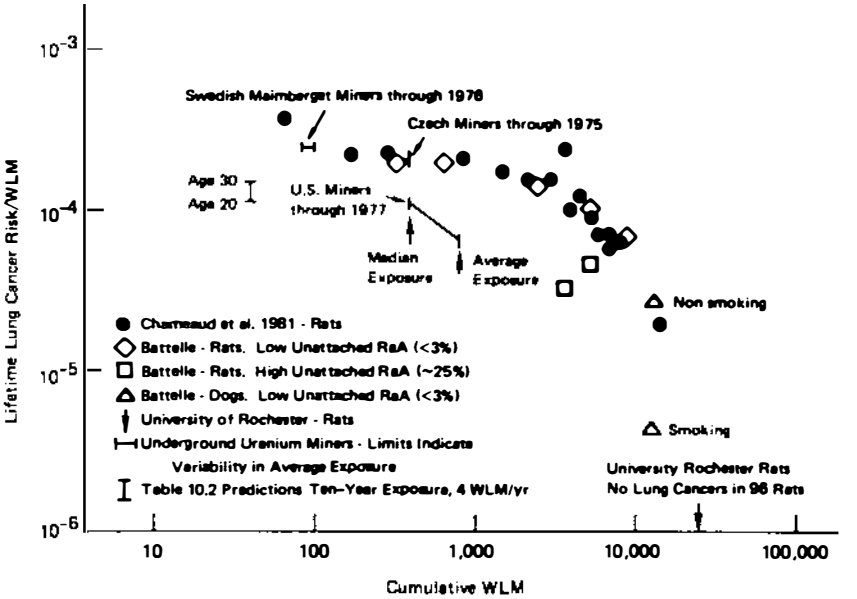

FIGURE 3-2 Lifetime lung cancer risk in humans and animals as a function of cumulative radon daughter exposures (National Council on Radiation Protection and Measurements, 1984a).

TABLE 3-4 Summary of Results of Studies of Lung Cancer Risk in Humans and Animals from Radon Daughter Exposure

Lifetime Risk Coefficient (Lung Cancers/Person, WLM)	Annual Risk Coefficient (Lung Cancers/Year-Person, WLM)	Reference
(2 to 4.5) x 10^{-4}	-	UNSCEAR (1977)
7.3 x 10^{-4}	(6 to 47) x 10^{-6}	NRC (1980)
(1.5 to 4.5) x 10^{-4}	-	ICRP (1981)
1.5 x 10^{-4}	10 x 10^{-6}	NCRP (1984a)

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TABLE 3-5 Estimated Lifetime Lung Cancer Risk in Terms of WLM per Year as a

Exposure Duration	Age at First Exposure				
	1	10	20	30	40
1 Year	6.4×10^{-5}	9.1×10^{-5}	1.3×10^{-4}	1.8×10^{-4}	2.1×10^{-4}
5 Years	3.4×10^{-4}	5.0×10^{-4}	6.9×10^{-4}	9.8×10^{-4}	1.0×10^{-3}
10 Years	7.7×10^{-4}	1.1×10^{-3}	1.5×10^{-3}	2.1×10^{-3}	2.0×10^{-3}
30 Years	3.4×10^{-3}	4.8×10^{-3}	5.5×10^{-3}	5.5×10^{-3}	4.2×10^{-3}
Life	9.1×10^{-3}	9.1×10^{-3}	7.7×10^{-3}	7.7×10^{-3}	4.5×10^{-3}

[a]For radon daughters measured under environmental rather than underground mining conditions.
[b]For a population with age characteristics equal to that in the whole United States in 1975.

SOURCE: National Council on Radiation Protection and Measurements (1984a).

TABLE 3-6 Estimated Lifetime Lung Cancer Risk[a] Under Environmental Conditions[b]

Exposure Duration	Age at First Exposure				
	1	10	20	30	40
1 Year	2.5×10^{-8}	3.6×10^{-8}	5.0×10^{-8}	7.1×10^{-8}	8.3×10^{-8}
5 Years	1.3×10^{-7}	1.9×10^{-7}	2.7×10^{-7}	3.8×10^{-7}	4.0×10^{-7}
10 Years	2.9×10^{-7}	4.2×10^{-7}	5.8×10^{-7}	8.1×10^{-7}	7.5×10^{-7}
30 Years	1.3×10^{-6}	1.8×10^{-6}	2.1×10^{-6}	2.1×10^{-6}	1.6×10^{-6}
Life	3.6×10^{-6}	3.5×10^{-6}	3.0×10^{-6}	2.5×10^{-6}	1.7×10^{-6}

[a]Activity expressed as pCi of $^{222}Rn/m^3$; to obtain MBq/m^3 multiply by 0.037.
[b]Radon-to-radon daughter ratio Rn/RaA/RaB/RaC equal to 1/0.9/0.7/0.7; ratio of unattached RaA to Rn is equal to 0.07.
[c]For a population with age characteristics equal to that in the whole United States in 1975.

SOURCE: National Council on Radiation Protection and Measurements (1984a).

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Function of Age and Duration of Exposure[a]

50	60	70	Lung Cancers in a Population of 10^4 Persons[b]
1.7×10^{-4}	1.3×10^{-4}	7.0×10^{-5}	13
8.4×10^{-4}	5.5×10^{-4}	2.8×10^{-4}	66
1.4×10^{-3}	9.1×10^{-4}	3.8×10^{-4}	130
2.5×10^{-3}	1.3×10^{-3}	3.8×10^{-4}	380
2.7×10^{-3}	1.3×10^{-3}	3.8×10^{-4}	560

per pCi $^{222}Rn/m^3$ as a Function of Age and Duration of Exposure

50	60	70	Lung Cancers in a Population of 10^5 Persons[c]
6.7×10^{-8}	4.8×10^{-8}	2.7×10^{-8}	0.0051
3.1×10^{-7}	2.1×10^{-7}	1.1×10^{-7}	0.026
5.6×10^{-7}	3.6×10^{-7}	1.5×10^{-7}	0.051
1.0×10^{-6}	4.8×10^{-7}	1.5×10^{-7}	0.14
1.0×10^{-6}	4.8×10^{-7}	1.5×10^{-7}	0.21

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Radon Cancer Risk

A variety of groups have made estimates of annual and lifetime lung cancer risk coefficients for radon, as indicated in Table 3-4.

Using a model that incorporates duration of exposure, a latency period, and some reasonable assumptions about cell repair, Harley, Cross and Stewart (National Council on Radiation Protection and Measurements, 1984a) have constructed estimates of individual lifetime lung cancer risk under environmental conditions as a function of exposure duration and age of first exposure. Their results are displayed in Tables 3-5 and 3-6.

Despite the various uncertainties, there is remarkable agreement between the cancer risk estimates generated by the miner studies and the animal studies when they are compared for equivalent dosages. Figure 3-2 displays a comparison developed by Harley, Cross, and Stewart (National Council on Radiation Protection and Measurements, 1984a).* It is clear from these results, and in particular from Figure 3-2, that the lifetime risk of lung cancer per WLM arising from chronic exposure to the levels of radon of concern for ambient and pile-related exposures is of the order of 2×10^{-4}, with an associated uncertainty of approximately a factor of 3.

Environmental Radon

Radon arises from a number of sources including soils, rock, and various building materials. Since homes are often closed, and radon that seeps in through foundations or out of building materials accumulates and remains in the building long enough for decay to take place, indoor concentrations frequently exceed outdoor concentrations.

Table 3-7 provides an estimate of the amount of radon released from different sources in the United States, as well as the percentage of total dose from each source to the lung due to radon and daughters as estimated by Travis et al. (1979).

---

* Note that at exposures above several thousand WLM the lifetime lung cancer risk drops off. Presumably this is because the dose levels are so high that cells are being killed before they have a chance to undergo transformation and become cancerous.

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TABLE 3-7 Radon Sources in the United States[a]

Source	Amount Released PBq/y (Ci/y)	Approximate Percentage of Total U.S. Dose to Lung from Rn + Daughters (%)
Natural		
Soil	4400 (120,000,000)	40
Evapotranspiration	330 (8,800,000)	3
Technology Enhanced		
Building interiors	1.0 (28,000)	55
Tillage of soil	110 (3,100,000)	1
Natural gas (industrial use)	0.41 (11,000)	1
Phosphate mines	2.0 (53,000)	0.02
Coal mines	0.52 (14,000)	0.005
Phosphate fertilizers	1.8 (48,000)	0.02
Geothermal power	0.021 (580)	0.2
Uranium		
Mining	7.4 (200,000)	0.07
Milling (active sites)	5.6 (150,000)	0.03

[a]Volcanic eruptions, not included in this summary, could be a significant source. Certain granites and gneisses are also known to produce high radon levels, as, e.g., in the Reading Prong in Pennsylvania.

SOURCE: After Travis et al. (1979).

Outdoor

There has not been a sufficiently systematic program of ambient measurement of radon levels in the southwestern United States. Measured background levels reported above in Chapter 2 range from 7 Bq/m^3 (0.2 pCi/L) to 44 Bq/m^3 (1.2 pCi/L). Roger Nelson (Jacobs Engineering Company, personal communication, 1986) has completed one of the larger sets of measurements of both background and pile-related radon. These data have not yet been published and thus must be treated as preliminary. Based on track-etch measurements, he estimates the average annual outdoor background radon concentration across the Southwest to be about 18 Bq/m^2 (0.5 pCi/L) with a site-to-site variability of approximately a factor of 2.

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A variety of other outdoor radon measurements from various locations around the world that range from 0.004 to 31 Bq/m^3 (0.0001 to 0.85 pCi/L) are summarized in the report of the National Council on Radiation Protection and Measurements (1975).

Because of the ubiquitous nature of radon in the environment, the average U.S. exposure (about 0.2 WLM/yr; National Council on Radiation Protection and Measurements, 1984b) is closer to the permissible occupational limit of 4 WLM/yr than general population limits are to their corresponding occupational limits for other sources of radiation exposure.

As the data in Chapter 2 make clear, the concentration of radon over piles varies considerably with site-specific factors. In the unpublished measurements referenced above, Nelson has found that the average annual concentration at the edge of six piles located in wide open spaces is about 56 Bq/m^3 (1.5 pCi/L) with an associated variability of about a factor of 2. He told the panel that radon concentrations reach background levels at roughly 0.5 km from these piles. At the opposite extreme, five piles located in areas of high relief, such as canyons, involve average fence-line values of roughly 370 Bq/m^3 (10 pCi/L), again with a variability of about a factor of 2. Levels of 37-74 Bq/m^3 (1-2 pCi/L) are observed 1-3 km from these piles. He told the panel that the remaining 13 piles lie in intermediate settings and involve average radon concentrations that fall between these extremes.

Indoor

There have been a variety of studies of indoor radon concentrations in the United States and a number of other countries.

Swedjemark (1983) reports results from integrated indoor radon measurements made during the autumn of 1982 in a representative sample of 500 Swedish homes. A countrywide average indoor concentration of 105 Bq/m^3 (2.8 pCi/L) was obtained. Average concentrations in single detached houses are reported to be 123 Bq/m^3 (3.3 pCi/L) and for apartment houses to be 85 Bq/m^3 (2.3 pCi/L). Figures 3-3 and 3-4 reproduce the actual distributions in the measurements for these two cases. Averages in different regions of the country are reported to vary by about a factor of 2. Significant differences are also noted between different construction materials, but these differences appear, in general, to be less important that the radium content of the underlying soil.

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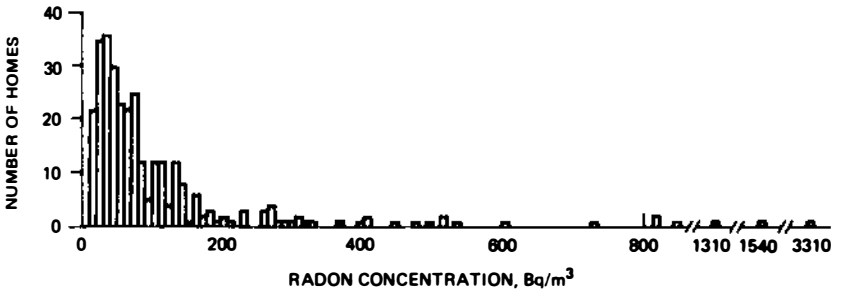

FIGURE 3-3 Distribution of radon concentration in Swedish detached houses as measured by Swedjemark during the autumn of 1982.
SOURCE: George et al. (1983).

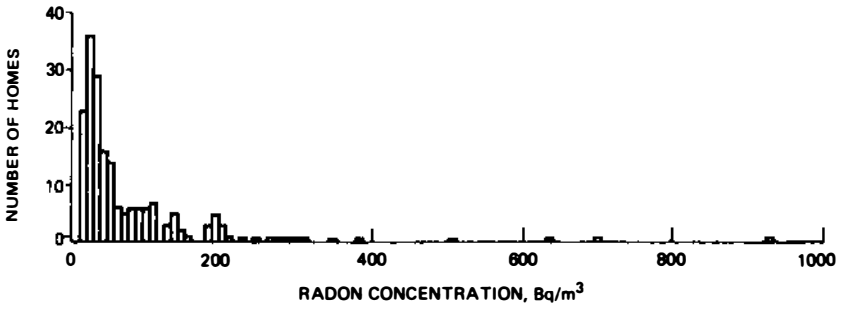

FIGURE 3-4 Distribution of radon concentration in Swedish apartment houses as measured by Swedjemark during the autumn of 1982.
SOURCE: George et al. (1983).

Bouville (1982) reports grab sample measurements in 100 homes in the Massif Central region of France. The arithmetic mean of the measurements was 80 Bq/m^3 (2 pCi/L) and ranged from 4 to 850 Bq/m^3 (0.05 to 23 pCi/L). No seasonal information was reported.

The most comprehensive study of radon in single-family U.S. homes has been done by Nero and his colleagues (Nero and Nazaroff, 1984; Nero et al., 1984; Nero, 1986). Figure 3-5 displays the resulting annual average concentration distribution deduced from a compilation of data from 1377 homes across 38 different locations. This distribution is well represented by a log-normal distribution with a geometric mean of 33 Bq/m^3 (0.9 pCi/L) and a geometric standard deviation of 2.8. This distribution is used to

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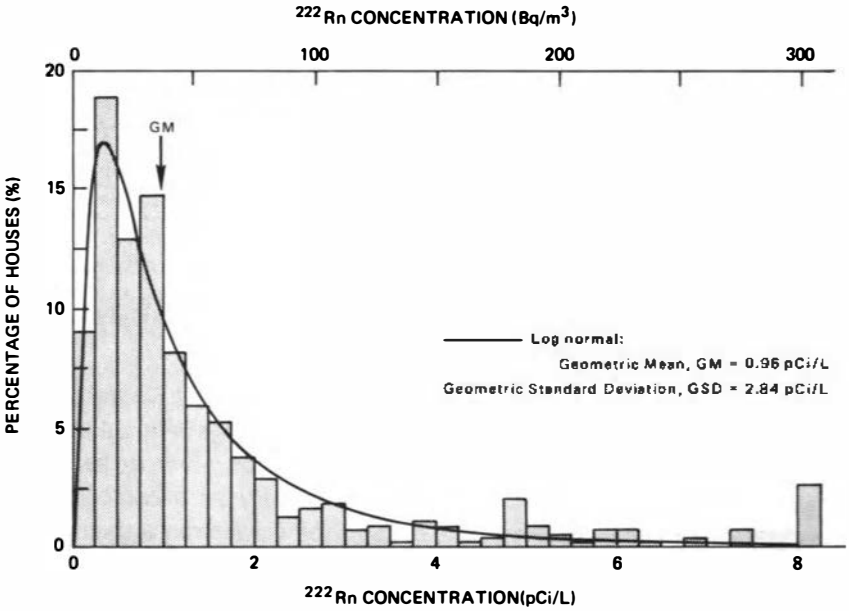

FIGURE 3-5 Distribution of U.S. single residence indoor radon concentrations.
SOURCE: Nero et al. (1984).

characterize indoor concentrations in the approximate calculation performed in the section that follows.

Absolute and Relative Radon Exposures and Risks

As discussed in Chapter 2, given the great pile-to-pile variability, both the available air dispersion models and the body of radon measurements made in the vicinity of piles are inadequate to allow a good assessment of the distribution of radon concentrations within a kilometer of an arbitrary pile. With some appropriate measurements, these problems can be made manageable.

In the absence of such measurements, the panel did not attempt to provide a general assessment of the radon health risk posed by tailings piles; rather, it performed a number of much simpler calculations in order to obtain an order-of-magnitude estimate of the risk for a single individual living right at the edge of several different classes of piles. This was done in order to develop some appreciation of the approximate range of lifetime

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lung cancer risks that may be involved. The results should *not* be treated as more than very crude case-specific estimates.

The total effective radon exposure received by someone living at the edge of a uranium mill tailings pile can be represented as the sum of their exposure to indoor and outdoor air; thus:

$$\text{Impact\_twp} = \text{Impact\_inwp} + \text{Impact\_outwp},$$

where Impact_twp is the total health impact (in increased lifetime cancer risk) from all radon exposures, Impact_inwp is the impact from indoor exposures received both at home and away from home, and Impact_outwp is the impact from outdoor exposures received both at home and away from home.

$$\begin{aligned}\text{Impact\_inwp} = {} & \text{FT\_in\_home} * (\text{C\_inwp} + \text{C\_pile}) * \text{F\_in} * \text{Rpd} \\ & + \text{FT\_in\_nhome} * (\text{C\_inwp}) * \text{F\_in} * \text{R\_pcl},\end{aligned}$$

where FT_in_home and FT_in_nhome refer to the fraction of time spent indoors at home and at locations away from home; C_inwp is the concentration of indoor radon at locations away from pile; C_pile is the edge of the pile concentration arising from the pile; F_in is the indoor equilibrium factor; and R_pcl is the exposure risk factor.

Similarly for the impact arising from outdoor exposure:

$$\begin{aligned}\text{Impact\_outwp} = {} & [\text{Ft\_out\_home} * (\text{C\_back} * \text{F\_out} + \text{C\_pile} * \text{F\_pile}) \\ & + \text{Ft\_out\_nhome} * (\text{C\_back} * \text{F\_out})] * \text{R\_pcl},\end{aligned}$$

where the notation parallels that used above, C_back is the average background outdoor radon concentration, and F_pile is the equilibrium factor of the pile radon at the edge of the pile.

Since this model is intended *only* to produce order-of-magnitude estimates, several simplifying assumptions have been made. Indoor concentrations in structures away from the pile have all been treated as equal, although there is considerable structure-to-structure variability. Indoor radon concentrations in the house at the edge of the pile are approximated as the sum of the indoor concentration occurring in other structures (which is assumed to include the influence of outdoor background concentration) and the radon from the pile. The actual relationship is certainly more

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complicated, but in the absence of data on houses next to piles, this estimate is adequate.

On the basis of the discussion in Chapter 2 and the estimate by Nelson provided in the personal communication cited above, the panel estimates the average concentration of radon in background air in the Southwest to be 18 Bq/m^3 (0.5 pCi/L) with an associated variability of about a factor of 2. To incorporate the wide range of concentrations associated with piles in different topographical settings, the panel modeled both a low-radon pile, in which C_pile is taken as 37 Bq/m^3 (1 pCi/L), and a high-radon pile, in which case C_pile is taken to be a factor of 10 higher at 370 Bq/m^3 (10 pCi/L), both with a variability of about a factor of 2.

Based on the work of Nero et al. (1984) discussed above, the panel estimates the indoor concentration of indoor radon to be 33 Bq/m^3 (0.9 pCi/L) with an uncertainty of about a factor of 2.8. If the person spends a significant fraction of time in a structure other than a conventional detached house (e.g., in office buildings or apartment houses), this number is likely to be an overestimate. Hummon and his colleagues (1985) have used U.S. census data to estimate the fraction of time spent indoors, outdoors at home, and outdoors away from home for the U.S. population as a whole. They inform us that studies of regional variation in these fractions show remarkably little variation across regions. On the basis of these data, we estimate Ft_out_home as 0.022, Ft_out_nhome as 0.013, and Ft_in as the remaining time.

The panel evaluated the model with the fraction of time spent away from home at 35 percent and at 0 percent.

As noted above, equilibrium factors vary considerably depending on a variety of environmental factors. For purposes of this order-of-magnitude calculation, the panel has allowed the equilibrium factor to range uniformly across the interval from 0.3 to 0.6 for indoor air, from 0.5 to 0.8 for background outdoor air, and from 0.005 to 0.1 for pile radon at the edge of the pile. We have been unable to identify published measurements of the equilibrium factor for radon over or at the edge of piles. M. Momeni* informs us that unpublished measurements he made while working at Argonne National Laboratory yielded values well under

* M. Momeni, Department of Nuclear Safety, State of Illinois, 1986 personal communication to G. Morgan.

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0.1 at all times and much lower than this in the afternoon when vigorous mixing is occurring. This is consistent with what one would expect on the basis of theory (Evans, 1955). Sensitivity studies show that even if the equilibrium factor at the edge of the pile were to be as high as 0.5, the effect on the simulation results reported below would be small.

All the equilibrium factors that the panel has assumed are clearly approximations since different physical situations will give rise to different degrees of equilibrium. In a more precise estimate, this variation would have to be measured carefully and incorporated, but these ranges are sufficient to provide the kind of order-of-magnitude estimates that the panel needs.

Using these estimates in a stochastic simulation together with a lifetime risk coefficient of 0.0002 cancers per WLM with an associated uncertainty of a factor of 2.5, the panel produces *order-of-magnitude* estimates of lifetime cancer risk for an average individual living away from all piles, living at the edge of our low-radon pile, and living at the edge of our high-radon pile as illustrated in Figure 3-6. A listing of the Demos model used is provided in Appendix F. The resulting mean value of the lifetime radon-induced lung cancer risk for this average person living away from all piles is estimated as 0.008. The estimated mean value of lifetime radon-induced cancer risk for a person living near a low-radon pile is 0.013 and near a high-radon pile is 0.056—an increase of 0.005 and 0.048, respectively. To an order of magnitude, the 0.008 figure is consistent with the actual U.S. lifetime incidence of lung cancer from all sources, which we estimate as between 0.04 and 0.05.

Clearly one factor that can substantially influence such estimates is the fraction of time spent indoors. Figure 3-7 shows the results obtained when the estimate is repeated under the assumption that persons spend *all* their life outdoors and, in the case of the two pile curves, all their life at the edge of the respective pile. The previous order-of-magnitude mean estimates of lifetime lung cancer incidence drop to 0.004, 0.005, and 0.012 for the no pile, low-radon pile, and high-radon pile cases, respectively. One can clearly deduce from these estimates that brief periods of time spent outdoors around piles pose no significant radon lung cancer risk.

A similar calculation was also performed in which the model subject was assumed to spend all indoor time at home (as opposed

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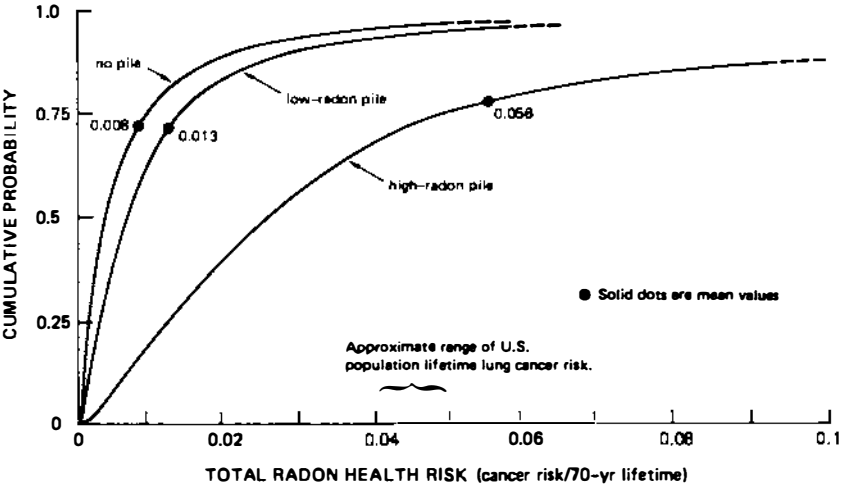

FIGURE 3-6 Cumulative probability distributions for order-of-magnitude estimate of cancer risk. Estimates are of 70-year lifetime cancer risk for a person living in the southwestern United States but away from uranium mill tailings piles (curve labeled no pile) and at the edge of a low-radon and high-radon pile. Solid dots are mean values. Details of the estimate are provided in the text.

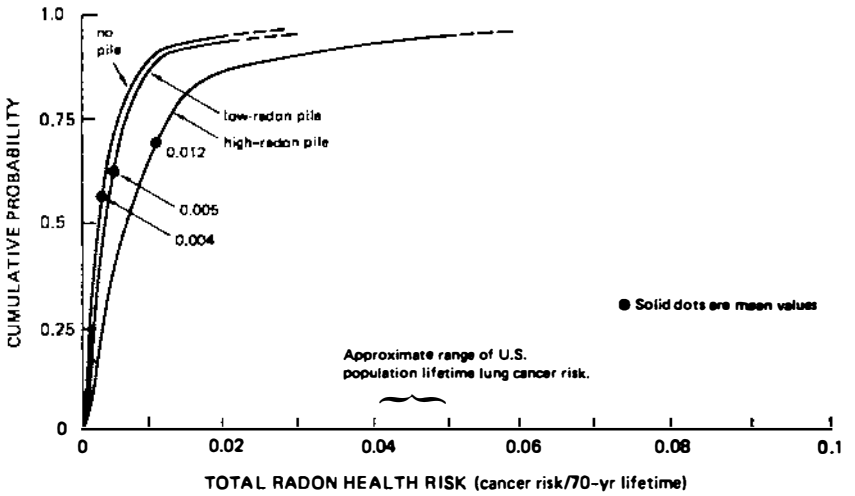

FIGURE 3-7 Evaluation of the model shown in Figure 3-6 under the assumption that exposed persons spend all their time outdoors. See text for similar results for all indoor time spent at home.

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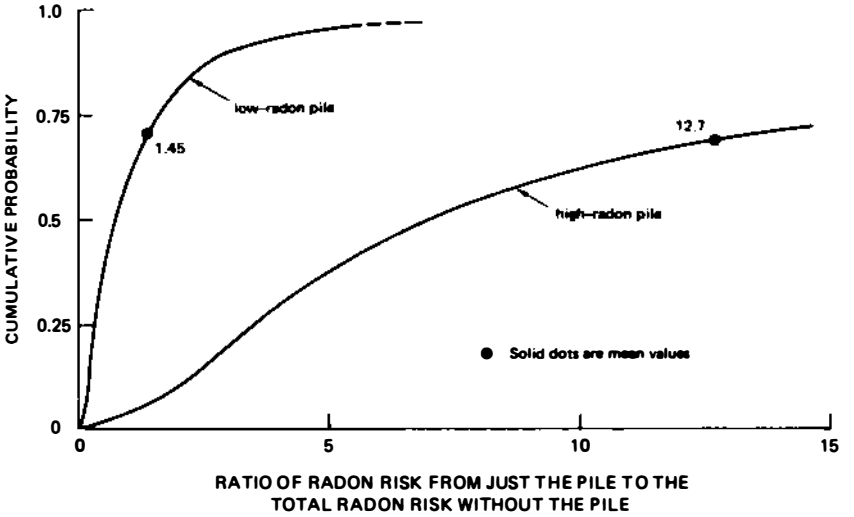

FIGURE 3-8 Ratio of lifetime cancer risk from just the radon from the pile to the total risk experienced by a person not living near a pile. Results are for base-case activity patterns for a person living right next to a high- and low-radon pile in the panel's order-of-magnitude estimate.

to 35 percent of indoor time away from home). In this case the resulting order-of-magnitude mean lifetime lung cancer incidence values for the low-radon and high-radon piles are 0.016 and 0.084, respectively.

Finally, Figure 3-8 displays the ratio of the lifetime lung cancer risk from just the radon from the pile to the risk from all sources of radon for a person who does not live near a pile. The relatively greater impact apparent in this display as compared with the impact indicated in Figure 3-6 results from the difference between the ratio of the mean and the mean of the ratio for distributions that are significantly asymmetric.

While it is clear that radon from uncontrolled uranium mill tailings piles poses no significant excess individual risk of lung cancer at distances beyond 1 km from most piles, and in some cases at even lesser distances, it is also clear from the preceding order-of-magnitude calculations that under some circumstances, particularly for a person living very close to specific uncontrolled piles, the excess lung cancer risk imposed could be substantial—approximately doubling, for such a person, their average lifetime lung cancer risk. One simple strategy for controlling the risk in

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these circumstances is a land-use regulation that excludes people from spending substantial time in close proximity to piles. Where this is not feasible, and lung cancer risk is significant, control of radon emissions is clearly required.

OTHER POTENTIAL HAZARDS

While the bulk of the concern with uranium mill tailings piles has focused on radon, there are several other potential hazards to health that deserve consideration.

Inhaled Dust

The potential health risk from suspended particulate material from the tailings piles is probably not so great as that of ^{222}Rn daughters, since the radiation dose to the lung is smaller and the dose would be delivered to the pulmonary region of the lung rather than to the bronchial epithelium. However, long-lived, windblown particulate material might translocate to the skeleton where ^{210}Pb will proceed to decay to its ^{210}Po progeny. This latter nuclide would continue to be produced by ^{210}Pb and would emit high-LET alpha particles, possibly causing the increased risk of carcinoma at its deposition site. This risk is further increased because of the chronic nature of the irradiation.

The whole-body radiation dose equivalent from gamma-rays emanating from the pile is of less significance, since it is considerably smaller than the dose equivalent to the bronchial epithelium from ^{222}Rn daughters and to bone surfaces from the long-lived ^{222}Rn daughter products. Furthermore, the nature of the population distribution about the uranium mill tailings piles also tends to reduce the risk of detrimental health effects from external whole-body radiation.

External Gamma Radiation

Biological risks due to gamma-emitting constituents in tailings piles are negligible when compared with those posed by radon. Measurements of gamma exposure over tailings piles range from 11 to 730 μR/h, and decrease to 7 to 97 μR/h over the area adjacent

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to the piles (Kisieleski, 1981).* Considering that normal background gamma measurements are typically in the 7 to 14 μR/h range and considering the very small number of people residing directly adjacent to the tailings piles, gamma radiation emitted from tailings does not appear to pose a significant biological risk.

Ingestion of Contaminated Food

Toxic chemicals or radionuclides in mill tailings can contaminate the human food chain. It is possible that radon daughters such as ^{210}Pb, with an atmospheric residence time of 3-10 days, may be deposited on soils on which crops are grown. A plant might then incorporate the radionuclide into the edible portion, leading to ingestion by farm animals or humans.

Holtzman et al. (1979) report measurements of the concentration of radionuclides in a variety of wild and domestic plants and animals raised in the vicinity of several uranium mill tailings piles near Grants, New Mexico. Samples collected included wild rabbits, cows and cow droppings, garden vegetables, and grass. A number of the measurements appear to show increased concentrations of pile-related radionuclides. However, in several cases the very small numbers of samples collected, or the absence of locally collected control samples, make interpretation of the data difficult.

Concentrations of ^{226}Ra in muscle of rabbits captured near the edge of piles ranged from 0.04 to 0.6 Bq (0.95 to 16.5 pCi)/kg (wet tissue) versus concentrations that ranged from 0.01 to 0.02 Bq (0.32 to 0.48 pCi)/kg for background samples collected off-site. Somewhat smaller elevations of ^{226}Ra, ^{210}Po, and ^{210}Pb were observed in most other tissues. The authors conclude that "Because of the wide variability and few samples the means are not necessarily statistically different . . . . However, the fact that all or most of the levels in the exposed animals are greater than those in the controls is a strong indication of significant differences" (Holtzman et al., 1979).

Four cows were studied, two from on-site and two controls. Concentrations in soft tissues appear not to differ greatly, except that the ^{210}Pb and ^{210}Po levels in the livers of the on-site animals

* We should note the existence of one exceptional measurement of 250 μR/h.

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appear to be elevated by roughly an order of magnitude. Measurements of cow droppings showed increases in ^{226}Ra concentrations of about a factor of 50 and in ^{210}Pb concentrations of about a factor of 8.

No background measurements of locally grown vegetables were collected. However, the concentrations of ^{226}Ra and ^{210}Pb in vegetables grown in soil mixed with tailings were elevated by approximately a factor of 5 over the concentrations in commercially available fresh and canned vegetables in New York City, San Francisco, and elsewhere that have been reported by other authors.

The authors estimate doses of 0.75 mSv (75 mrem)/yr from contaminated vegetables and 0.25 mSv (25 mrem)/yr from contaminated meat to the critical organ, bone, for a person who ate only vegetables grown on the edge of the pile and animals raised on or near the pile. However, for the people living near the piles studied, this represents a substantial overestimate. Only 50 percent of the homes had gardens, and locally grown meat was probably not a significant factor in the diet. The authors estimate that, in the case studied, the average population exposure through the food supply is less than 0.001 mSv (0.1 mrem)/yr to the skeleton. The population in this study apparently consisted of Caucasians. The impact could be somewhat higher for American Indian or other groups that rely more on locally grown food.

Even a rather modest soil cover on the piles, enough to hold down the dust, should effectively eliminate the increased concentrations of radionuclides reported in this study.

Measurements of tissue from ten cattle raised near uranium mines and mills in Ambrosia Lake, New Mexico, showed higher concentrations of the ^{238}U decay chain radionuclides than 10 control animals from the Crownpoint area. The first group of 5 animals had elevated concentrations of ^{226}Ra and ^{210}Po in muscle, liver, and kidney tissue. Vegetation in the area had higher levels of ^{230}Th and ^{226}Ra than did controls. The second group had elevated concentrations compared with controls of uranium and ^{230}Th in liver and all radionuclides except ^{210}Po in kidney. While the evidence indicates these increases resulted from the uranium mining and milling in the area, it is not possible to determine the relative fraction of the animals intake from the ingestion of deposited windblown particulates from the piles, consumption of mine dewatering effluent discharged on the ground (which was the

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sole source of water for one group), ingestion of soil or vegetation grown in soil that had been flooded for years with untreated mine effluent, the inhalation of radon released from the piles or mine vents, or the ingestion of the decay products of radon (Lapham et al., 1986).

Ingestion of Contaminated Water

As indicated in Chapter 2, material leached from mill tailings can contaminate local surface and groundwaters. The problem is likely to be quite localized. In most (although apparently not all) cases, radionuclides will be immobilized. This means that, for the most part, as discussed in Chapter 2, any threats to health posed by water contamination from piles will be confined to the vicinity of the pile and will not be significantly different from the threats posed by a large volume of other kinds of tailings throughout the West.

In most cases the principal offenders, other than chlorides and sulfates, appear to be a variety of such metals as selenium and molybdenum. Since these same problems arise for a wide variety of other tailings, the panel will not dwell on them here; the recommendation made in Chapter 2 for more general study of the groundwater problems associated with a wide variety of different types of tailings piles is, however, reiterated.

CONCLUSIONS AND RECOMMENDATIONS

On the basis of the preceding discussion, the panel reaches the following conclusions and makes the following recommendations:

1. Brief periods of time (e.g., hours to days) spent outdoors near uranium mill tailings piles pose no significant lung cancer risk. Persons of average lifestyle living in close proximity to uncontrolled uranium mill tailings piles may, depending on site-specific circumstances, experience a significant increase in total lifetime radon lung cancer risk. If a person were to live right at the edge of a few piles that involve particularly unfavorable exposure conditions, their lung cancer risk could be substantially higher than the average U.S. population lung cancer risk. However, persons living at distances greater than a kilometer from most uncontrolled uranium mill tailings piles, and perhaps somewhat closer to some piles, will experience no significant increase in

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lifetime radon lung cancer risk due to exposure to radon from the pile.

> *Recommendation*: People who are not occupationally involved in uranium milling should not live or spend a large fraction of their time in close proximity to uncontrolled uranium mill tailings piles. If for political, economic, or other reasons it is not feasible to preclude people from living or spending a significant fraction of their time in close proximity to an uncontrolled mill tailings pile, then by almost any risk-management decision criterion, steps should be taken to control radon emissions from that pile.

2. While the size of populations living near most uranium mill tailings piles is quite small, future populations near piles could increase significantly. This could affect estimates of total population risk.

3. Because it contains radionuclides (Stieff, 1984), airborne dust from uranium mill tailings piles presents a small, but real, health risk to persons who might inhale such dust (Breslin and Glauberman, 1970).

> *Recommendation*: Piles should be controlled in a manner that prevents pile material from being blown away as dust.

4. While, in special circumstances, tailings may pose health risks through a variety of secondary pathways—such as via the food chain—in general, it does not appear that such risks are significant.

5. Conclusions and recommendations in Chapter 2 that groundwater and surface-water problems from piles be considered in the context of a broader set of contamination sources, including some other kinds of mill tailings and some solid-waste disposal sites, apply to the consideration of health effects from such contamination.

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4
Technical Alternatives for Tailings Management

INTRODUCTION

The objectives in managing the risk from uranium mill tailings can be stated broadly as follows:

1. Reduction to acceptable levels of the hazard to human health posed by radon emission from tailings accumulations;
2. Prevention of significant physical dispersion of hazardous tailings by natural processes such as mass movement, wind transfer, and water erosion;
3. Prevention or minimization of contamination of surface water and groundwater at or adjacent to disposal sites;
4. Prevention of misuse of hazardous tailings or of hazardous disposal sites; and
5. Minimization of the need for long-term maintenance and institutional controls.

Figure 4-1 displays the principal choices that are available to meet these objectives for existing tailings. The choice to be made first is between reprocessing tailings or leaving them as is for at least a while. If the decision is not to reprocess, the pile can be moved or left in place, with various options related to liners and covers. A further choice is between a one-time control option and

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Sections of this chapter are based on or quote directly from reports prepared for the panel by Stephen G. Wells, Department of Geology, University of New Mexico, and by Inderjit Nirdosh, Department of Chemical Engineering, Lakeland University, Ontario, Canada.

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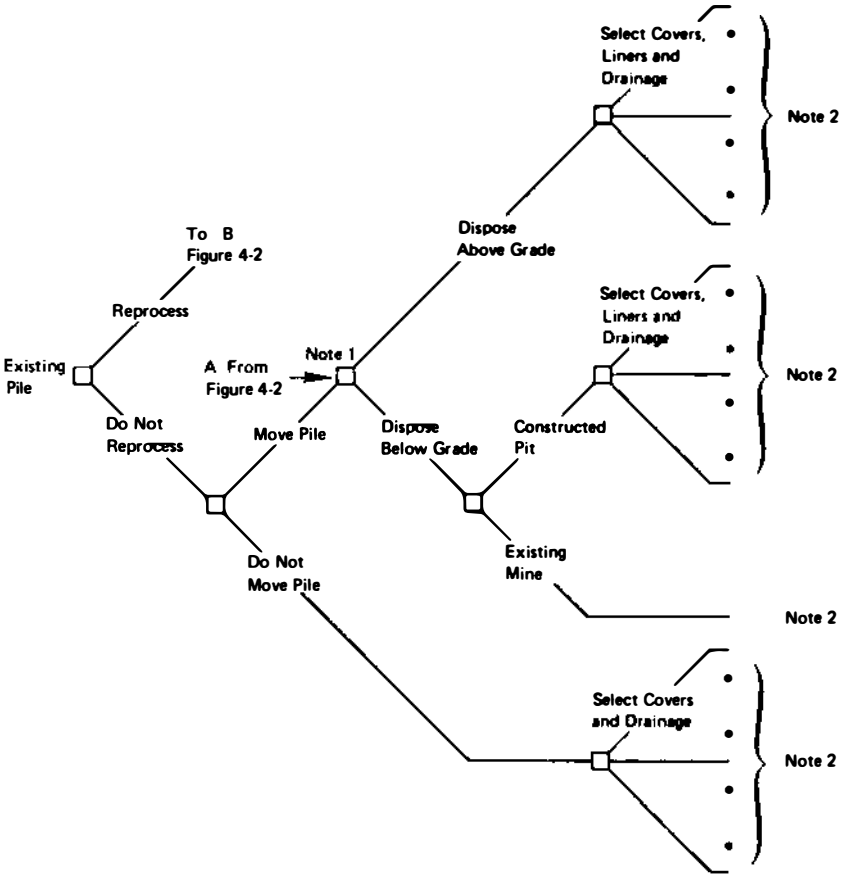

FIGURE 4-1 Options for handling existing tailings piles.

a strategy of continued attention. Figure 4-2 displays a similar set of choices available for new uranium mining.

In this chapter the various options for tailings management are reviewed, beginning with an examination of the strategies now in general use and their likely long-term effectiveness. The panel appraises a number of less-conventional strategies that involve modifications of the characteristics of the tailings and explores the possibility of changing substantially the chemical composition of newly generated or reprocessed tailings through modifications to the milling process. The chapter concludes with a brief discussion of the option of eliminating tailings production through in-situ extraction (solution mining).

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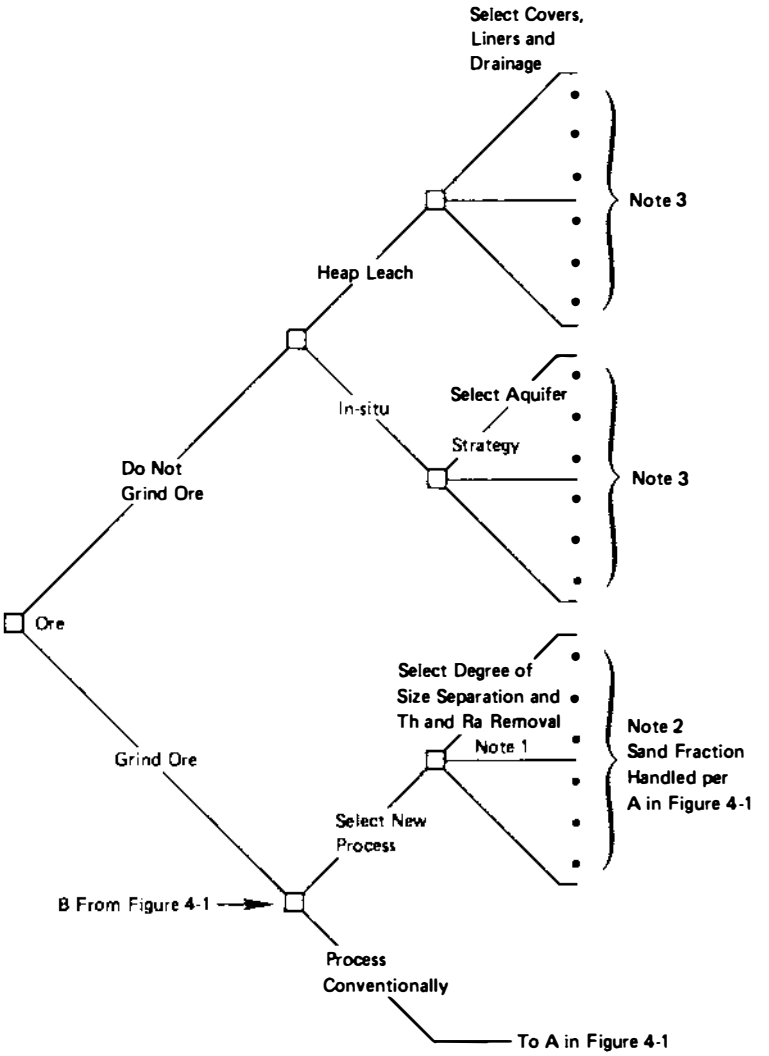

FIGURE 4-2 Options for handling new ore and reprocessed tailings.

REVIEW OF CONVENTIONAL MANAGEMENT STRATEGIES

Present strategies for managing the risks posed by uranium tailings approach the objectives outlined above by containment schemes that are designed to isolate and enclose the tailings, rather than

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by treatment to reduce those toxic or radioactive elements that give rise to low-level but long-lasting potential hazards to human health. In this section the various disposal methods now employed in managing uranium mill tailings in the United States are categorized and described in summary fashion. The effectiveness of these procedures is appraised, but only in relation to the general objectives outlined above and not in terms of specific regulations and stipulations now in force.

Most uranium mining and processing activity in the United States is located in arid or semiarid regions* (the Colorado Plateau and adjoining regions) that have low population density. These regions have similarities in age of bedrock units, type of structural features, nature of deposits, tectonic history, and general geomorphic history. However, there are significant local differences in the regions, particularly in erosional rates and processes. For example, Wells and Jercinovic (1983) have shown that within the limited area of one 1:24000-scale quadrangle (about 200 km^2 or 80 mi^2), the rate of erosion varies enormously, depending on topographic form and type of material. Measured sediment yields per unit area range from nearly zero to 20,000 $cm^3/m^2/yr$. In areas where infiltration dominates, erosion may be small; in areas where runoff dominates, erosion may be large (see Figure E-4 and Table E-1 in Appendix E). These differences make it evident that a management strategy for a uranium mill tailings pile must be site specific and may require combinations of the various management strategies discussed below.

Methods of tailings disposal now being used or recommended can be grouped broadly into above-grade and below-grade disposal.

Above-Grade Disposal

The bulk of existing mill tailings is located in above-grade storage sites, most commonly on flat or low-sloping surfaces adjacent to an operating or abandoned mill. During the active stage, when slurried material is being added continually to the tailing pile, the storage site comprises a ponded area bounded by natural

* This is in sharp contrast to the situation in neighboring Canada, where most tailings are located on the Canadian Shield in regions with considerable amounts of precipitation and many streams and lakes.

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features, man-made embankments, or some combination of both. The embankments commonly are constructed of locally derived surface materials or of a separated coarser fraction of the tailings themselves. The storage piles range from a few to a few hundred hectares in surface area and are as high as 30 m. The pond in the central portion of the pile loses water by evaporation and, unless restrained by a liner, by downward seepage into the underlying natural foundation materials. The slimes, which have a high water content when they are placed, drain slowly and are compressible; i.e., they will decrease in volume and the surface of the slimes will settle. Since this process is time dependent, the surface of an active pile will continue to subside long after tailings placement has ceased.

After completion and closure of a disposal site, an above-grade tailings pile, even one optimally located, will be subject to natural forces that will tend to remove or reduce this man-made anomaly. Mass movement, sheet wash, gullying, and wind transport all work toward physical dispersal of the tailings. Elements in solution, including radionuclides and associated toxic constituents, will migrate toward surface or groundwater systems.

In conventional practice, these dispersive forces are countered in three ways:

1. By shaping the pile to reduce gradients and thus reduce rates of erosion,
2. By installing a relatively impervious undercourse prior to filling, and,
3. By installing a cover or stabilizing the surface after closure.

An impervious underlay (or liner) generally may be placed before the tailings are deposited. Liners may consist of compacted clay; a sheet of material such as polyvinyl chloride (PVC), chlorinated polyethylene (CPE), or synthetic rubber (Hypalon); or both clay and sheet material (U.S. Environmental Protection Agency, 1983). Other materials that have been tested (U.S. Environmental Protection Agency, 1983) include asphalt concrete, asphalt rubber, and catalytic airblown asphalt, any of which can inhibit seepage. It is generally accepted, however, that no engineered barrier is likely to remain effective over the long term; its main purpose is to prevent seepage during the active life of tailings emplacement and, after closure, to restrict seepage.

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Surface stabilization can assume varying degrees of sophistication, ranging from treatment with a surface spray to inhibit wind erosion to complexly layered covers topped with riprap or vegetation. These design elements are discussed in various public documents (see, e.g., Voorhees et al., 1983; Baker et al., 1984) and are reviewed briefly later in this chapter.

Site location doubtless is the most important factor in long-term stability of an above-grade tailings pile, particularly with respect to erosion by running water, no matter how thorough the preaccumulation and postaccumulation procedures. In the past, most sites appear to have been selected mainly on the basis of accessibility and proximity to the operating mill and availability of adequate acreage. Many are located on or adjacent to valley bottoms or flood plains with a clear record of erosional instability (see, for example, Table E-2 in Appendix E, which summarizes the location of major piles in northwestern New Mexico). In the future, greater care in site selection is clearly warranted, although it must be recognized that in some districts choices are limited.

An ideal site for above-grade tailings disposal would be on an extensive flat or low-dipping surface formed during a previous climatic cycle, underlain by a few or hundreds of meters (tens or hundreds of feet) of moderately permeable unsaturated materials that would trap toxic elements from seepage water. Properly located and designed, such a site could be relatively free from floods, gullying, and headwall erosion; the only essential corrective measures would be those needed to control erosion by wind. However, few existing sites approach this ideal. Most require extensive treatment to assure physical stability of the pile and to prevent groundwater contamination, even on a time scale measured in a few tens or hundreds of years.

Below-Grade Disposal

In simple terms, below-grade disposal means burying the tailings below the general topographic level of the area, either in a trench or pit dug for that purpose or in a pre-existing open-pit or underground mine. The range of disposal schemes possible is almost unlimited, particularly if the prime elements of location and design are combined with other options, such as size separation of the tailings, dewatering, and asphalt or cement fixation (for a few illustrative examples, see U.S. Nuclear Regulatory Commission,

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1980). For purposes of the present review, the panel grouped the approaches into two categories: disposal in a surface excavation, either one produced earlier by open-pit mining or one dug for this particular purpose, and disposal in underground mine workings.

Two major factors immediately emerge in consideration of a surface excavation: stability of the site itself and the relation of the site to the static water table. An open-pit mining operation will have been in part or wholly in bedrock, and some reasonable stability against erosive forces generally can be assumed. This assumption cannot be made for a site at which the trench or pit is excavated in sediments, unless prior consideration has been given to the nature and age of the surface being entered. As with above-grade disposal, the ideal site is one located on a surface of some extent produced during an earlier geomorphic or climatic cycle; flood plains and valley bottoms of recent development generally are not desirable locations, unless major revision of the drainage system is undertaken.

In the ideal situation, the excavation will be entirely above the level of permanent groundwater, in which case the need for an effective liner is principally during the filling stage, when a surface pond exists; thereafter, if the liner fails, seepage water will traverse reactive materials that will tend to trap toxic constituents. If, however, the excavation intersects the water table, more intensive steps will be required if entry of toxic fluids into the groundwater is to be prevented: placement of thicker, more durable, underlays and possibly installation of peripheral pumping stations. In either case, it is desirable after closure to construct a cover designed to inhibit access of surface waters, as well as to prevent wind dispersal of the tailings and to reduce radon escape.

Backfill of worked-out stopes and other abandoned workings is not an uncommon engineering practice in many operating underground metal mines, where it serves to stabilize the ground and to prevent catastrophic roof failure, and it has been used in some coal-mining districts as a means of waste disposal. The material used, however, is generally innocuous in character—typically surface-derived sands or mine waste that pose little or no chemical or radioactive hazard. In a purely physical sense, untreated uranium mill tailings could serve much the same purpose as these materials, but introduction of the slurry into the permeable, water-saturated zones from which ore has been mined can result in contamination of the groundwater system, which cannot be effectively sealed off.

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A typical slurry contains appreciable amounts of sulfate and chloride, in addition to metals that have survived the milling process. These latter may include uranium, vanadium, molybdenum, selenium, and arsenic. The metals may in part precipitate in the tailings or in the immediate surroundings of the backfill (Longmire et al., 1984), and some sulfate may precipitate as gypsum. Chloride, however, is likely to remain mobile and to enter the local groundwater together with that fraction of sulfate and metals that remains in solution. An additional problem, at least in an active mine, may be created by radon emission from the tailings backfill, since this is additive to the existing flux in the mine workings. These various potential problems may prove manageable in specific situations, but in general it appears that uncontrolled disposal of untreated tailings in underground cavities may present unacceptable hazards. The degree to which negative environmental consequences can be countered by engineering design is not yet fully known, but in at least one uranium mining district research has been directed toward that end. As reported by Patrick Longmire, New Mexico EID (presentation to the Uranium Mill Tailings Panel, December 3, 1984), some success has been achieved in the mines of the Grants Minerals Belt. The design of the disposal system is shown diagrammatically in Figure 4-3. Several critical features are to be noted: (a) only the sand-sized fraction of a normal tailings discharge is used; (b) the separated sand fraction is reslurried using available surface and mine discharge water; (c) the slurry is piped into an unlined but bulkheaded cavity, where it mixes with existing groundwater; and (d) the excess mixed water is decanted and returned to the surface for recycling. The chemical reactions that take place in the deposited tailings are complex; they involve oxidation-reduction, adsorption, authigenic mineral formation, and cementation. The impact on the regional groundwater is not entirely clear, in part because of complexities introduced earlier by the mining operations themselves. Increases in sodium and sulfate contents are to be expected and in fact have been measured along with some increase in dissolved uranium. These effects are thought to be of short-term duration, but the *long*-term impact has yet to be determined.

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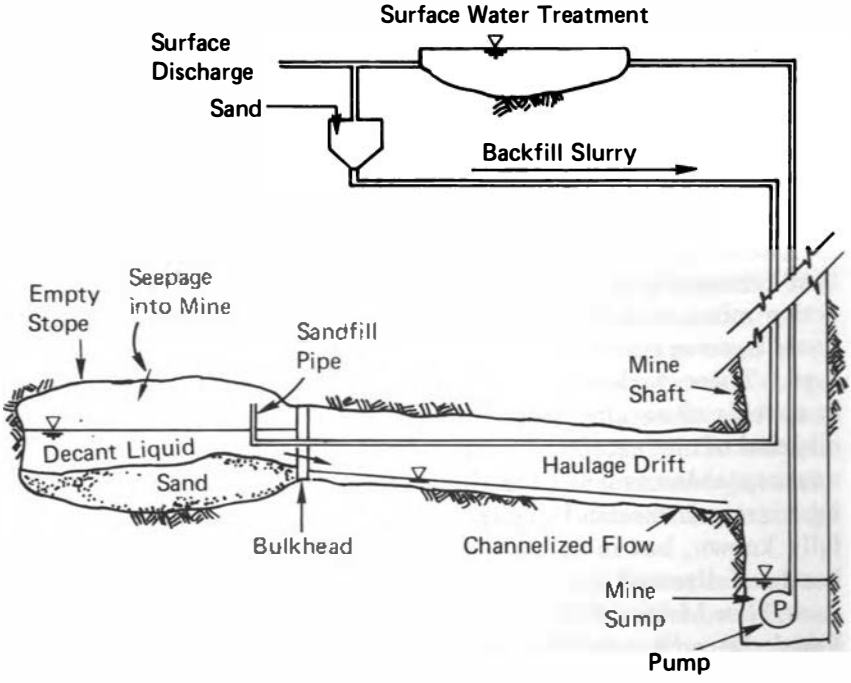

FIGURE 4-3 Generalized design of a backfill system in the Grants Minerals Belt, New Mexico.
SOURCE: Written material accompanying presentation by Patrick Longmire to the panel, December 3, 1984.

Cover Design

The engineered cover for a tailings pile serves four primary purposes:

1. To reduce erosion of the tailings either by rain that falls on the pile or by surface flows of water (sheet wash or streams) along the perimeter of the pile;
2. To reduce erosion of the tailings by wind;
3. To reduce radon emission from the pile; and
4. To reduce infiltration of rainwater that may leach contaminants from the pile and carry them into the groundwater and surface water.

The design of the cover to achieve these purposes requires that the outer layer of the cover be coarse enough, or be somehow stabilized, to be resistant to wind and water erosion. It also requires

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that there be some portion of the cover that, by a combination of thickness and low permeability, is adequate to reduce rainfall infiltration and radon exhalation to the desired levels.

Resistance to erosion by rainfall, surface-water flows, and wind can be achieved by the following:

1. Placing riprap (gravel, cobbles, or boulders) of sufficient size that the individual pieces cannot be moved by water flowing at the velocities predicted for the design flood conditions; or
2. Stabilizing the surface of the tailings by the use of cement, asphalt, or chemical agents.

The design of riprap to prevent erosion by rainfall can be done using standard engineering procedures that have been proven effective.

The design of riprap to prevent erosion by surface flows of water along the perimeter of the pile can also be done using standard engineering procedures, based on selected design values of precipitation and calculated values of runoff volumes and velocities.

Whether erosion resistance is provided by riprap or stabilizing agents (as discussed in the next paragraph) catastrophic flooding and erosion may affect the entire landscape at geologically infrequent intervals. This phenomenon is discussed in the section on Long-Term Effectiveness of Conventional Management Strategies. Because of the possibility of such catastrophic flooding and erosion, it is not possible to design a tailings pile cover with 100 percent certainty that it will never be eroded.

Stabilization of the surface of the tailings pile with cement, asphalt, or chemicals is subject to significant uncertainty with respect to the long-term stability of the stabilizing agent under the influence of weather, chemical interactions with the materials in the tailings, or mechanical cracking of the stabilized surface caused by consolidation of the tailings and settlement of the surface of the tailings pile.

The cover on a tailings pile does not provide protection against headwall erosion and gullying by streams in the vicinity of the pile, nor can the headwall erosion and gullying be predicted. Therefore, surveillance is necessary to identify situations in which future maintenance will be required to arrest headwall erosion and gullying before it reaches the pile.

Radon release and rainwater infiltration can be reduced by placing a sufficiently thick cover of earth (about 1 to 3 meters

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thick) over the surface of the pile (U.S. Environmental Protection Agency, 1982). The effectiveness of such a barrier is dependent on its not being eroded and is therefore subject to the same uncertainty as the erosion protection itself. In addition, even if the earth cover is not eroded, there is the possibility that it may be cracked owing to irregular settlement of the surface as the underlying tailings slowly consolidate.

Radon flux from a cover of less than a few meters of earth cannot now be predicted with precision, either immediately after construction of the cover or in the long term when various physical changes, such as change in water content, have taken place in the cover.

Radon release and rainwater infiltration can also be reduced by artificial barriers such as layers of asphalt or plastic sheeting. The artificial barriers may or may not be stable under long-term exposure to atmospheric agents and chemical constituents of the tailings. They may also be subject to mechanical failure as the result of irregular settlement of the surface as the underlying tailings slowly consolidate.

Washouts

If material is eroded and transported away from a pile, and spread over a considerable area downstream from the pile, or if water from the pond on top of a pile is released, the impact may be significantly less than if the water and material are concentrated in one place.

Two failures have been documented, one at Church Rock, New Mexico, and one at Green River, Utah.

Millard et al. (1983) report that the Church Rock spill in July 1979 was the "largest single release of radioactive waste in the United States" and that "Most of the 1100 tons of solid material released was caught by a small emergency catchment dam." They conclude after the cleanup of some of the sediments that were deposited downstream of the pile, based on studies made both before and after the spill, that "although the spill was potentially hazardous, its short-term and long-term impacts on people and the environment were quite limited." Specifically, the following impacts were assessed on the basis of studies that were made:

1. No long-term effects of the spill on surface-water quality was evident at the time.

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2. Groundwater contamination is slight and confined to limited areas.

3. The hazard associated with inhalation of contaminated river sediments suspended in the air as *dust* is negligible *for local* residents.

4. Native vegetation contained concentrations of radioactivity that fell within the range of background values.

5. Long-term exposure to residual materials left in the streambed is of concern.

Miller (1985) has found no evidence of contaminants from the Church Rock spill in flood-plain sediments downstream from the site.

Lapham et al. (1986) have, however, reported elevated levels of some radionuclides in tissues from a sample of seven cattle at Church Rock when compared with a group of ten controls from Crown Point, New Mexico.

R. Beverly, AMC/Umetco, in a presentation to the panel at Grants, New Mexico, on May 21, 1984, described briefly the washout of a tailings pile at Green River, Utah.

The Church Rock and Green River washouts, and others that have occurred or that occur in the future, should be evaluated for quantitative data and insights that they provide for the assessment of risks involved with washouts and erosion of tailings piles.

Relative Merits of Above-Grade and Constructed Below-Grade Sites

The management alternative found acceptable by the Environmental Protection Agency (EPA) and the U.S. Nuclear Regulatory Commission (U.S. NRC) calls for a "passive" engineered treatment, designed to inhibit radon emission and to remain effective for 1000 years (U.S. Environmental Protection Agency, 1983; U.S. Nuclear Regulatory Commission, 1980). The requirements are in addition to those specified for groundwater protection in the Solid Waste Disposal Act. Though not explicitly stated, these design elements evidently are applicable to existing or active above-grade sites. For new impoundments, below-grade siting is strongly urged. The technical advantages of below-grade disposal, implied in this EPA recommendation for new impoundments, are not entirely clear-cut. Certainly some added immediate protection against

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dispersal by wind and surface wash is gained by burial. The gain, however, is somewhat transitory considering the long-lived nature of the radioactive hazard (many times the arbitrary 1000-year specification); in the long term, the forces of erosion are likely to prevail—just as for above-grade repositories. With respect to protection against groundwater contamination, a newly constructed below-grade site could be less effective than one perched on the surface, at least in the long term. Both will (now and in the future) presumably have a requirement for a liner, but this will be effective, at best, only during the active period and for a short time thereafter; ultimately, it almost surely will fail. After failure, seepage from the pile will work its way to water courses or to the static water table, with movement of toxic constituents inhibited only by reaction with clays and other constituents of the material being traversed. Other factors being equal, an above-grade site will be more advantageous than a below-grade site in the same locality because of somewhat lengthier travel paths, which provide greater possibilities for adsorption, reaction, and precipitation of toxic elements from the traversing fluids. Further, the basinlike design of a below-grade repository, plus the restraining influence of even a partly failed liner, creates an artificial trap for surface water; tailings are likely to remain in a state of nearly permanent saturation, with interstitial fluids fully charged with toxic constituents. It must be granted, however, that comparisons of this sort are difficult to quantify, given the natural variations within even local areas. About all that can be concluded with reasonable certainty is that below-grade siting has few, if any, intrinsic long-term advantages over properly located above-grade locations.

Moving Piles

Moving specific uranium mill tailings piles has been proposed and, in the case of the Vitro pile in Salt Lake City, has actually been done. Moving an existing pile is clearly expensive and involves risks from dust, transportation, and other hazards. Moving does allow the installation of a liner, something otherwise not possible with an existing pile. The panel found only limited analysis of the risks associated with moving piles (U.S. Department of Energy, 1984) and on this basis cannot make definitive recommendations. Better risk analyses of moving piles are clearly needed. The panel concludes that it makes sense to move a pile, if at all, only in very

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special circumstances—such as if it is located in the path of an expanding urban area.

LONG-TERM EFFECTIVENESS OF CONVENTIONAL MANAGEMENT STRATEGIES

The 80,000-year half-life of ^{230}Th, the principal long-term precursor of radon in the uranium mill tailings, is very long, not only in comparison with recorded human history but also in comparison with the times in which substantial changes in the landscape occur owing to natural processes of erosion. The effectiveness of tailings management strategies should be judged, therefore, by the stability of the tailings on the time scale of these natural processes, as well as on the time interval of the stream gauge and meteorological record (of the order of 100 years in the western United States) or arbitrary intermediate time intervals, such as 1000 years.

Stability of a tailings pile, or its lifetime against natural processes of erosion, is strongly dependent on the topography and geology of the site (see Appendix E). Owing to the requirement for water in mill operations, many of the older tailings deposits on the Colorado Plateau are located along the Colorado River or its principal tributaries. In some cases (e.g., tailings at Grand Junction, Colorado; Naturita, Colorado; Moab, Utah) the tailings are on a mainstream flood plain. At other sites, tailings are located along valley walls, on the flanks or tops of mesas, on flood plains of minor drainage systems, or in the headwater reaches of minor drainages. The substrates beneath the various tailings piles range from easily eroded alluvium, colluvium, and shale to relatively resistant bedrock, which most commonly consists of sandstone. The majority of tailings piles are located on relatively easily eroded substrates. Partly because of the diversity of the sites, the tailings are subject to a variety of natural processes of erosion and dispersal. This is the case for the tailings in their present state and will also be true for tailings that have been graded and covered.

Natural Processes That Affect the Integrity of Tailings Piles

Fairly extensive records of stream flow exist for the western United States for the past 90 years; these records provide a statistical basis for estimating the most probable highest flow over a time period

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of approximately a century. Considerable uncertainty, however, attends the extrapolation of this record to longer time periods. On the one hand, there is no guarantee that the climate of the past 100 years is representative of climate over longer periods of time. Indeed, there is strong evidence of significant climatic fluctuation over the past 1000 years (National Research Council, 1975). On the other hand, it is not clear how the most probable maximum flow over a millennium, for example, is related to the maximum flow over a century (see Appendix E). The most direct way to approach this problem is to examine the geologic evidence of large floods.

The highest discharge *recorded* by stream gauges in the Grand Canyon section of the Colorado River was 3600 cubic meters per second (cms) [127,000 cubic feet per second (cfs)] in 1927 (U.S. Geological Survey, 1980). The maximum discharge known from floodmarks is about 8500 cms (300,000 cfs) (flood of 1884). In the confined inner gorge of the Grand Canyon, this flow corresponds to a rise in water level of about 12 m (40 ft). From a study of the dimensions of the river channel at about 50 Holocene debris fans deposited by tributaries in the Grand Canyon, Kieffer (1985) concludes that the flow of the Colorado required to clear the channel to the observed widths is approximately 11,000 cms (about 400,000 cfs). This discharge corresponds to a rise in water level of about 15 m (50 feet) in the confined inner gorge of the Grand Canyon. It is not known when the last mainstream floods, responsible for the present channel configuration, occurred. From the observed frequency of historic changes in the debris fans, however, it appears likely that discharges on the Colorado equal to or greater than 11,000 cms (400,000 cfs) have occurred on the order of once per 1000 years.

Discharges vastly greater than 11,000 cms may occur on the 100,000-year time scale. Driftwood dated at about 60,000 to 100,000 years is preserved in a cave in Marble Canyon, a short distance upstream from the Grand Canyon at a height of 44 m (145 ft) above the river. The presence of driftwood at this level suggests that a discharge of the order of 100,000 cms may have occurred at the time that the wood was deposited.* While large

* It has been suggested (Hereford, 1984) that the high water level was caused by damming of the canyon by a rockfall downstream. A uranium-trend age reported by Machette et al. (1986) indicates that the rockfall is

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precipitation events alone can give rise to large-scale flooding, a variety of other processes such as large landslides into the canyon or the overtopping and subsequent erosion of silted-up dam sites may contribute significantly to the largest possible events.

Tailings piles at Moab and Green River, Utah, and Grand Junction, Colorado (and possibly Rifle, Colorado), will be subject to erosion by major floods on the Colorado River in a time period of 1000 years. Other tailings piles on major tributaries at Naturita and Durango, Colorado, also will be subject to major floods in this time span. While these tailings sites are in valleys more open than the Grand Canyon, major rises in water level will nevertheless occur at all these sites. Some or most of the tailings at several of these sites probably will be entrained in the bedload of the river and dispersed downstream in the next 1000 years.

Whether or not the tailings have been covered and whether or not they have been protected by riprap, the history of changes in the canyons of the Colorado near Moab in the past 50,000 years (Richmond, 1962) indicates that, without continued and occasionally heroic protective efforts, probably all tailings piles along the Colorado will be eroded away in a time much less than the 80,000-year half-life of ^{230}Th.

If this erosion occurs in a single big event that would mix the tailings with a much larger load of sediment, the tailings are unlikely to pose a significant threat to man or the environment. Given the magnitude of other damage to natural and human systems that would accompany such an event, it is clearly unreasonable to worry about protecting tailings piles under such circumstances.

Local Geomorphic Changes

Most uranium mill tailings piles are located in the valleys of small tributary drainage systems (see representative list for New Mexico in Appendix E). Long-term stability of these tailings against erosion depends on the specific geologic processes that modify the landscape at each of these sites. The principal geologic hazards that threaten the integrity of the existing tailings piles are

1. Large discharge local floods produced by extremely intense rainstorms (100- to 10,000-year local floods) and

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much older than the driftwood, but a remnant rockfall dam may nevertheless have contributed to the high water level at the cave.

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2. The incremental effects of
   - drainage network extension (channel headcutting),
   - piping and gullying on the tailings pile,
   - bank erosion and meander growth on local drainage systems,
   - drainage network rejuvenation (channel incision), and
   - vertical downwasting of slopes.

A description of these various erosional processes is given in Appendix E.

A few of the extant uranium mill tailings piles are adjacent to local major drainages. These tailings will be subject to erosion by rare very large discharge events. Prediction of the probable frequency of these events is subject to considerable uncertainty (see Figures E-7 and E-8 in Appendix E), but discharges of the order of 300-600 cms (10,000 to 20,000 cfs) are likely 100-year events in some of the ephemeral arroyos on the Colorado Plateau (Table E-3). Observed discharges of this magnitude produce profound changes in the channels of the ephemeral streams of this region (e.g., Cooley et al., 1977; Patton and Baker, 1977) and are likely to produce lateral erosion of tailings on their banks. Covering of the tailings by 3 m of unconsolidated material or even by riprap would provide negligible protection against these floods. Boulders more than 3 m across can be transported in floods of this magnitude.

Tailings that are remote from major drainages will be affected by the incremental results of other erosional processes. Tailings on valley floors are at risk from drainage network extension, bank erosion and meandering of the existing drainages, downcutting (incision) of local drainage channels and the cumulative direct effect of piping, gullying, and vertical downwasting on the tailings piles. Observed rates of these processes are illustrated in Figure 4-4. Drainage extension and lateral migration of channels can proceed at rates on the order of 10 m/yr. Hence, some tailings piles located in valley floors but not now adjacent to channels will be at risk to lateral erosion of channel banks within a few centuries. Most tributary valley floors investigated on the Colorado Plateau have been repeatedly incised and regraded in the course of the past 10,000 to 20,000 years (e.g., Hack, 1942).

Even tailings piles located on the most stable interfluve surfaces are subject to severe degradation over time spans that are short relative to the half-life of ^{230}Th. Sheetwash, gullying, and

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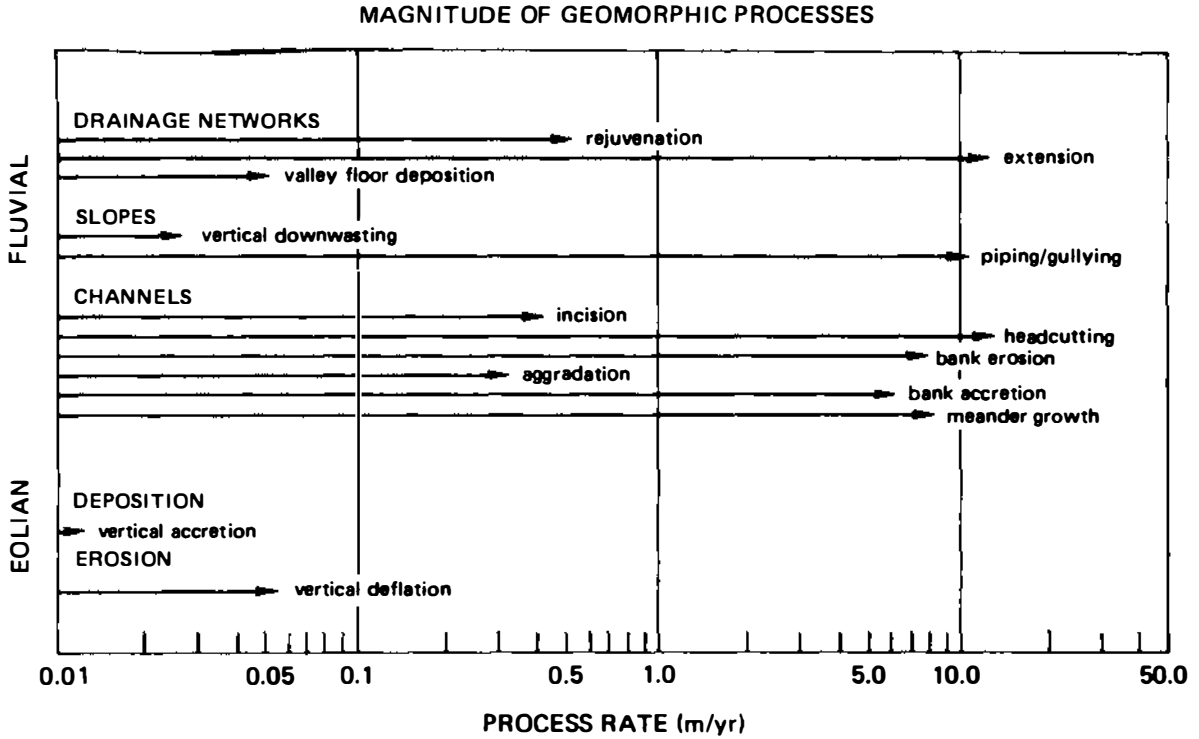

FIGURE 4-4 Magnitudes of geomorphic processes operating over historic time scale in northwestern New Mexico. SOURCE: Appendix E.

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small-scale local fluvial erosion ultimately take their toll. A useful natural analog to tailings piles produced by processing sandstone-type uranium ores is provided by the rim deposit of ejecta surrounding Meteor Crater, Arizona. The crater is accurately dated as being 49,000 plus or minus 3000 years old (Sutton, 1985). It is located on a stable surface of low relief remote from the principal drainages. Initially, a layer of crushed sandstone up to several tens of meters thick at the rim crest extended to distances of at least 0.5 km in all directions from the rim crest. This layer of crushed rock, which was deposited in less than a minute, is roughly comparable in physical characteristics with a tailings pile graded to slopes of generally less than 5 degrees. The crushed sandstone was covered by a layer of mixed debris several meters thick containing coarse strong rock fragments. The crushed sandstone deposit has now been entirely stripped from about 90 percent of the crater rim by local processes of erosion (Shoemaker, 1963; Shoemaker and Kieffer, 1974). Average denudation of the rim crest is estimated to be about 20 m. Relatively resistant bedrock in the vicinity of the crater has been denuded, on average, by about 5 m since the crater was formed. A layer of riprap, therefore, would not have prevented erosion of the natural pile of crushed sandstone.

Climate Change

Rates of erosion are a sensitive function of climate, particularly of rainfall. The climate of the Colorado Plateau and the adjacent area has changed dramatically during the past 50,000 years. Periods of time during which the annual precipitation was about twice that of the present have alternated with dry periods in which the climate was about similar to that of the present. Stripping of the crushed sandstone layer of the rim of Meteor Crater occurred chiefly during two wet periods in the southwestern United States that correspond to two periods of advance of continental glaciers in the mid-continent region. As judged from the thickness of sediment deposited on the crater floor, the average rate of erosion during the wet periods was more than four times higher than during the last dry period (Shoemaker and Kieffer, 1974). Large oscillations in the climate can be expected to occur in the future with about the same frequency components as have been observed

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during the past several hundred thousand years (Imbrie and Imbrie, 1980). During wet periods, the rates of local geomorphic change may be several times greater than they are at present.

Summary

In a time span 20 times as great as that of recorded human history, the rate of radon production in existing uranium mill tailings piles will decay by only a factor of about 2. Over this long period of time the tailings piles are unstable against the natural processes of erosion, even when located in the most favorable sites and regardless of whether they have been covered by several meters of earth or protected by riprap. The integrity of covered tailings piles at unfavorable sites (on or near the banks of rivers and ephemeral tributaries) is at risk over the time span of a few centuries.

The current strategy, covering the tailings with 3 m of earth and then leaving them, does not fully address the long-term hazard of radon exposure. Active monitoring and repair of the cover will be required to maintain the integrity of the piles.

During this long period, protection of piles against the largest likely floods will be difficult, and at some sites, impossible.

Because of the very long lifetime of the radionuclides and radon production of the mill tailings piles, consideration is given in the sections that follow to possible alternative means of hazard reduction.

ALTERNATIVE OR SUPPLEMENTARY OPTIONS OTHER THAN REPROCESSING OR MAJOR CHANGES IN EXTRACTION PROCEDURES

Before considering significant modifications in milling and extraction practice for newly mined material or reprocessed tailings—topics that will be taken up in subsequent sections of this chapter—a number of supplementary or alternative procedures for treatment of tailings and mill effluents warrant brief review and appraisal. Taken individually, none of these is likely to be a major factor in tailings management; in certain circumstances, or in specific combinations, they can be of considerable value in reducing the hazards associated with commercial recovery of uranium from ores.

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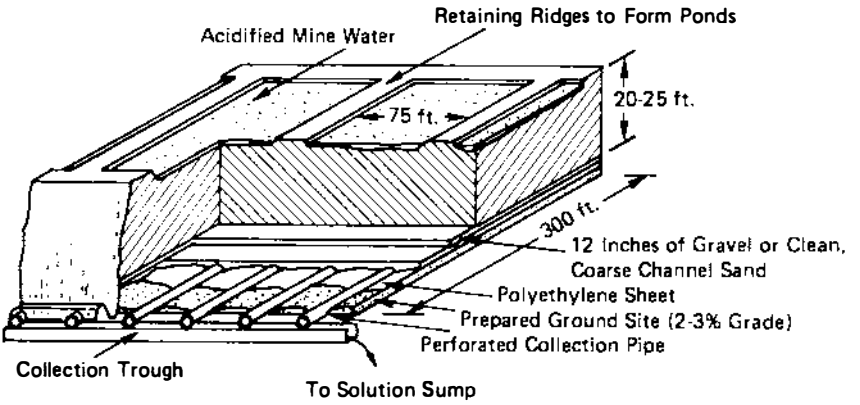

FIGURE 4-5 Schematic diagram of a typical heap leach pile.
SOURCE: U.S. Nuclear Regulatory Commission (1980).

Heap Leaching

The production of uranium by the leaching of mined, but uncrushed, ore is a common practice on a small scale in many districts. It is particularly useful as an extraction technique on low-grade ores or on ores mined in small quantity at sites distant from regular milling facilities. In a typical operation, the site is prepared by leveling the site to an even, low-sloping surface, which is then covered with a plastic sheet overlain by 30 cm (a foot) or so of coarse gravel. The ore is stacked on this base in a rectangular plan (as shown in Figure 4-5), shaped to provide solution reservoirs on the upper surface, and trenched at the basal perimeter to form a collecting trough. The choice of leaching solution depends on the local situation: commonly, mine water acidified with H_2SO_4 is used, but, if the pile is adjacent to an operating mill, acid solutions may be drawn from the milling circuit, circulated through the pile, and returned to the circuit. In either case, uranium is recovered from the enriched solutions by standard extraction techniques.

Heap leaching is a useful adjunct to the overall uranium production scheme, but it is likely to remain of minor quantitative importance. In terms of environmental hazard assessment, the absence of finely ground tailings is of positive value, but many of the problems associated with tailings remain—disposal of spent solutions, avoidance of surface and groundwater contamination, and disposal of residues.

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Bacterial Leaching

Bacterial leaching of uranium ores has been discussed by Brierley (1978). It is considered a potentially economic method for pyritic ores. In particular, Brierley points out that it has been used at Agnew Lake Mines in Ontario, Canada, for in-situ recovery of uranium. Bacterial leaching has also been studied by Brierley for uranium ore from Grants, New Mexico, where he concludes that lack of energy sources, slow development, and low activity of the bacteria in the ore all limit the potential effectiveness of the method. While the studies are limited, they do indicate that established bacterial leaching techniques do not appear to be effective for low-pyrite uranium ores.

Separation of Tailings into Sand and Slime Fractions

Tailings can be readily separated at the time of slurry discharge into size fractions by the use of hydrocyclones. In itself, this physical partitioning can be advantageous in providing a coarser fraction suitable for embankments in the disposal area, and this is a common practice. A more important potential, however, lies in the chemical differences between the fractions; size separation could serve as an essential preliminary step toward more intensive systems of tailings management.

It is well known that more than 90 percent of the radionuclide species originally present in uranium ore passes through the milling circuit and is discharged with the tailings. The tailings will contain essentially all of the initial ^{230}Th and ^{226}Ra, as well as an unextracted 2 to 10 percent of the initial uranium. These elements, moreover, are selectively concentrated in the finer fraction, as illustrated in Table 4-1 by data on radium distribution from six western mining operations. In each case, about 85 percent of the initial radium content of the tailings is contained in the slime fraction. Data on thorium and remnant uranium are limited, but indications are that distributions are similar to that of radium. In some operations, it is known that most of the thorium is in solution at the time of discharge. This striking difference in radionuclide content of readily separable fractions can be taken advantage of in several ways, ranging from selective reprocessing to recover radium, thorium, and uranium to selective thermal or nonthermal treatment of the lower-bulk slime fraction. These options—each of

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TABLE 4-1 Distribution of Radium between Sands and Slimes in Tailings from Six Western Mining Operations

Operation	Sands (greater than 75 μm)		Slimes (less than 75 μm)		Concentration Factor (Sands and Slimes)
	Weight %	Percentage of Total ^{226}Ra	Weight %	Percentage of Total ^{226}Ra	
Ambrosia Lake[a]	80	15	20	85	23
Rio Algom	54	15	46	85	6.6
Denison	49	16	51	84	4.8
Wyoming	69	12	31	88	16
Anaconda	73	12	27	88	17
Beaver Lodge	48	16	52	84	4

SOURCE: Levins et al. (1982).
[a]Separated at 150 mesh.

which would require considerable further research before practical applications can be made—are discussed in the following sections. Selective treatment of the slime fraction would, of course, leave the problem of disposing of the sand fraction. This material, however, presents a relatively low hazard potential. It still would require planned disposal and protection, but at less rigorous levels than that called for in present tailings control measures. Much of it probably could be returned to the mines as backfill.

Thermal Stabilization

Experimental work, such as that at the Los Alamos National Laboratory, has shown that the level of "emanating" ^{226}Ra (that is, the amount of ^{226}Ra that yields ^{222}Rn to the atmosphere) can be reduced drastically by sintering. As shown in Table 4-2, nearly complete reduction is achieved at sintering temperatures of 1100°C to 1200°C.

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TABLE 4-2 Percentage of Reduction in Emanating ^{226}Ra at Sintering Temperatures from 500°C to 1200°C

Sintering Temperature (°C)	Shiprock Sands[a] (%)	Shiprock Fines[a] (%)	Durango Sands[a] (%)	Durango Fines[a] (%)
500	15	16	48	61
600	29	27	64	68
700	44	37	76	80
800	63	58	87	88
900	83	84	92	91
1000	92	96.1	95.5	92
1100	96.4	98.8	99.0	99.8
1200	97.7	99.2	99.5	99.8

SOURCE: Dreeson et al. (1982).

[a]Initial emanating ^{226}Ra: Shiprock sands, 39 pCi/g; Shiprock fines, 214 pCi/g; Durango sands, 140 pCi/g; Durango fines, 473 pCi/g.

It is evident that the sintering procedure is capable of immobilizing the radionuclides and reducing the radon hazard to insignificant levels. The energy requirement for such treatment, however, is very high, and questions remain concerning long-term stability of the sinter and engineering design of the processing facility. If this strategy were adopted, acceptable disposal procedures for the sintered product would have to be developed, and potential misuse would have to be guarded against, since gamma radiation will require some attenuating cover, and the sinter, at least under some conditions, may not prove wholly stable. Finally, sintering of the tailings effectively forecloses the reprocessing option, in which toxic radionuclides would be extracted along with economically valuable uranium. For all these reasons, sintering does not appear to be a particularly promising option for dealing with the great

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bulk of existing uranium mill tailings, or even with quantitatively lower tonnages of the more radioactive slime fraction.

Nonthermal Stabilization

Tailings, whether separated by size fraction or in an untreated state, can be immobilized by mixing with a consolidating agent, such as Portland cement or asphalt. The tailings are prepared for treatment by neutralization and drying. The latter is a difficult task for the slime fraction. Relatively large quantities of the mixant are required to assure a satisfactory product: a ratio of about 1:5 for cement to tailings and about 1:3 for asphalt to tailings (U.S. Nuclear Regulatory Commission, 1980). Consolidation in a hardened matrix has a number of desirable aspects: immediate physical stability is assured, resistance to erosion is enhanced, radon emission is strongly inhibited by gross reduction of pore space, and the potential for leaching and groundwater contamination is greatly reduced because of relative impermeability. The long-term stability of the product and the economic feasibility are, however, yet to be demonstrated, and, as with thermal stabilization, such processing effectively eliminates the possibility of reprocessing for radionuclide recovery. Further, the stabilizing agent itself would have to be evaluated as to its potential for breaking down over time and releasing contaminants to groundwater at the storage site. A variant of the nonthermal stabilization option is that of in-situ grouting of the tailings pile. In this procedure the tailings would be perforated at intervals and injected with a grouting agent, such as Portland cement, sodium silicate, or an organic compound (Tamura and Boegly, 1983). Injection of a grouting agent is not likely to prove practical in the finer-grained portions of a tailings pile, but it might have applications in stabilizing sand portions of some piles in which the sands and slimes have been separated. In some cases grout "curtains" may be formed in the foundation soils at the perimeter of a pile to create a barrier against seepage of leachate from the tailings pile into the regional groundwater system. Such a barrier would be effective only if it extended down to an underlying impervious layer and if it were itself essentially impervious. Experience with grout curtains that have been used to prevent seepage of water through the foundations of dams has shown that they are frequently ineffective (Casagrande, 1961),

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presumably because of gaps in the grout curtain, i.e., zones of soil in which the voids were not completely filled by grout.

Effluent Treatment

One of the major problems in tailings management is reduction of contaminant levels in seepages or reduction in volume of the seepage itself. Two general procedures warrant appraisal: chemical treatment of the effluent and dewatering to yield a thickened discharge.

As decribed in Chapter 2, the discharge from an acid-leach mill typically has a pH in the range of 1.2 to 2.0. Under such conditions much of the ^{230}Th, a precursor to ^{226}Ra, is in solution; discharges in the Grants, New Mexico, area are reported to have thorium concentrations of 5.5-6.3 MBq/m^3 (150,000 to 170,000 pCi/L) (Landa, 1980). The concentration of radium in the liquid fraction at the time of discharge is not so well known; but in the deposited tailings, the distribution is found to be complex.

Precipitation of the thorium and radium that are in solution along with a host of heavy metals is readily accomplished by neutralization generally through the application of lime (CaO), hydrated lime [$Ca(OH)_2$], or limestone ($CaCO_3$). Lime neutralization is an effective way of immobilizing many potentially contaminating elements that might otherwise escape in seepage water. It is to be recommended particularly when a potential for overflow or leakage into water courses exists.

It has been noted (U.S. Nuclear Regulatory Commission, 1980) that although in clear sulfuric acid solutions addition of barium chloride ($BaCl_2$) will remove 90-99 percent of radium initially in solution, this treatment is not effective in precipitating radium in discharge tailings slurries. Possibly some combination of filtering and barium chloride treatment could be developed to trap radium before disposal in the tailings pond. Radium can also be removed from clear solutions or dilute slurries by ion exchange, using organic resins, but the process is complex and not generally considered practical.

Seepage problems can, to some degree, be reduced by partial dewatering of tailings, to yield a thickened discharge. Several advantages are claimed for a disposal method in which water content is reduced to about half of the usual level (Robinsky, 1982): the need for confining embankments is reduced or eliminated;

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the choice of location for disposal is less critical, and the site is more readily reclaimed; the formation of a slime-filled pond is avoided; and the overall porosity of the tailings accumulation is reduced by the fact that sands and slimes are not separated and radon is, therefore, inhibited. None of these advantages has been demonstrated, as yet, by actual operation. Further research as to technical feasibility may, however, be warranted—particularly if dewatering is being considered as a component of some more radical method of tailings management, such as reprocessing.

APPRAISAL OF POTENTIAL FOR HAZARD REDUCTION BY CHANGES IN MILLING PRACTICE OR REPROCESSING OF TAILINGS

As previously noted, the bulk of the radioactivity initially exhibited by uranium ore persists through the milling process and becomes a property of the discharged tailings. Aside from an unextracted 5 to 10 percent of the uranium, the radionuclides chiefly responsible for the long-lived radioactivity of the tailings and that are precursors of radon and its toxic daughter products, are ^{230}Th and ^{236}Ra, with half-lives of 8×10^4 yr and 1600 yr, respectively. It is evident, therefore, that much of the radiological hazard associated with tailings piles could be eliminated if ^{230}Th and ^{226}Ra were to be extracted at the milling stage or by subsequent reprocessing of the tailings. The disposal of the thorium and radium might pose significant technical and political problems. It is conceivable, although quite unlikely, that a market for these materials could develop in the future. Whether this can be done at a cost competitive with other means of tailings management is an issue yet to be fully explored. In this section, possible alternative schemes are discussed for extraction of these various radionuclides, and a preliminary appraisal is made of their present or future practicability.

Extraction of Uranium and Other Marketable Metals by Conventional Reprocessing of Tailings

The most extensive investigations of recovery possibilities have been those conducted by Ford, Bacon & Davis Utah Inc. (hereafter FBD) on behalf of the Department of Energy as part of an engineering assessment of inactive tailings piles in the United

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States. These studies, although covering a large number of districts, are limited in scope in that they are confined to appraisal of uranium and vanadium recovery only and make use of conventional techniques—either treatment in a standard mill or heap leaching. The results have only indirect bearing on the problem of hazard control, but they are useful in evaluating the overall economics of a tailings reprocessing.

The starting point of the FBD studies is, simply, the uranium and vanadium assay of the tailings under consideration. Values for three selected localities are as follows, in weight percent:

Site	U_3O_8	V_2O_5	Reference
Shiprock, N.M.	0.012	0.030	Ford, Bacon & Davis Utah (1981b)
Maybell, Colo.	0.015	0.012	Ford, Bacon & Davis Utah (1981c)
Durango, Colo.	0.048	0.039	Ford, Bacon & Davis Utah (1981a)

The Shiprock and Maybell assays are fairly typical for tailings from most U.S. deposits; those for Durango are probably extreme values. Percentage recovery of uranium and vanadium varies strongly with tenor (ore grade) and with method of treatment: in mill treatment, it ranges from 35 percent recovery of U_3O_8 at 0.01 percent grade to 60 percent or more at 0.05 percent grade. Heap leach recovery is from 20 to 35 percent for the same grades, and vanadium recovery (not always measured) by either technique generally is about 40 percent. In terms of amount recoverable per ton of tailings, some illustrative examples are as follows, given in pounds of U_3O_8 and V_2O_5 recovered per ton of tailings processed:

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Site and Process	U_3O_8	V_2O_5	Reference
Shiprock, mill treatment	0.086	0.12	Ford, Bacon & Davis Utah (1981b)
Shiprock, heap leach	0.072	Not available	Ford, Bacon & Davis Utah (1981b)
Maybell, mill treatment	0.11	Not available	Ford, Bacon & Davis Utah (1981c)
Maybell, heap leach	0.07	Not available	Ford, Bacon & Davis Utah (1981c)
Durango, mill treatment	0.77	About 3	Ford, Bacon & Davis Utah (1981a)
Durango, heap leach	0.58	About 3	Ford, Bacon & Davis Utah (1981a)

The value of the recovered product, assuming nominal prices of $55 per kg ($25 per pound) for U_3O_8 and $7 per kg ($3 per pound) for V_2O_5, would range from approximately $2.50 per ton of tailings from districts such as Shiprock and Maybell to $20 or more for Durango. Studies have been made elsewhere on recovery possibilities for a wider suite of elements contained in tailings. As reported by Dreesen et al. (1982), using then-current prices, recoverable values obtained by concentrated sulfuric acid leach range from about $6 per ton for Shiprock (mostly vanadium and molybdenum) to $40 per ton for Durango tailings, which contain significant amounts of cobalt, manganese, copper, and zinc in addition to uranium and vanadium. These results, though preliminary in nature and limited in scope, do suggest that the economics of reprocessing for removal of the deleterious elements radium and thorium could be significantly improved if the procedures could be expanded to permit recovery of associated marketable metals.

Future Operations

In this section, two possibilities are discussed:

1. The treatment of existing tailing piles and modifications to existing mills to produce benign, or at least significantly less hazardous, tailings and

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2. New mill processes to extract uranium, radium, and thorium so that benign, or significantly less hazardous, tailings are produced.

These processing options hold the potential to essentially eliminate the hazards arising from the radionuclides in the tailings.

Background on the Form of Radium and Thorium in Tailings

Considerable work has been done on the form of ^{226}Ra in acid-leach mill tailings, while somewhat less is known about carbonate-leach tailings. Landa (1980, 1982b) and Nirdosh et al. (1984) have inferred, on the basis of chemical investigations, that radium in the tailings is present primarily as sulfate coprecipitates, e.g., barium-radium sulfates and jarosite—SEM (scanning electron microscope) studies are desirable to validate this inference. Seeley (1977) and Ryon et al. (1977), among others, have suggested that coprecipitates with other alkaline earths, such as calcium and strontium, may also be important. In any case, these coprecipitates appear to form fine particles termed "radiocolloids" and "radiobarites" by Stieff (1984). Stieff's work, using nuclear emulsion techniques, indicates that these radium-alkaline earth coprecipitates are difficult to dissolve but are fine enough (the radiocolloids are too small to resolve at 1000 times magnification) to remain suspended once they become windborne until they are redeposited by turbulent fluctuations or washed out by rain. Radium is also, to some degree, adsorbed on silica particles, on metal hydroxides, on clays, and in the pores of larger solid particles (Nirdosh et al., 1984). Studies by Landa (1980, 1982b) and Stieff (1984) generally support these findings. In particular, Stieff has found patches of intense alpha activity on quartz grains. The active material appears to be in the form of fine red particles deposited on a portion of the grain.

These results may be explained by recalling that ^{226}Ra probably goes into solution during leaching but rapidly precipitates because the alkaline-earth sulfates are quite insoluble. Seeley (1977) also suggests such a mechanism by showing that the distribution of ^{226}Ra as a function of particle size changes significantly from the ore to the tailings.

With regard to ^{230}Th, Landa (1980, 1982b) reports that a significant amount (up to 90 percent) is dissolved and remains in

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solution during acid leaching. Data supplied by Kennedy (1985) and Cleveland (1984) support this hypothesis. The nuclear emulsion studies by Stieff also indicate that ^{230}Th is adsorbed on the surfaces of mineral phases such as quartz and on aggregates of fine-grained materials that could be mixtures of clays and iron hydroxides. Unlike the coprecipitates of ^{226}Ra, which form fine, intensely active particles, ^{230}Th is much more widely dispersed—consistent with the finding that it is mostly in solution in the mill effluent and is subsequently adsorbed or precipitates rather uniformly.

With this background, strategies for extracting ^{226}Ra and ^{230}Th are now discussed. The basis of all methods is to convert the radium and thorium to soluble salts, e.g., chlorides, nitrates, or soluble complexes, and then to partition these from uranium by selective precipitation, adsorption, ion exchange, or solvent extraction.

Reprocessing of Existing Tailings Piles

Radium Removal

The removal of radium from uranium mill tailings has been investigated by Borrowman and Brooks (1975), Seeley (1977), Itzkovitch and Ritcey (1979), McLaughlin (1979), Silver and Andersen (1979), Haque et al. (1980), Landa (1980, 1982b), Ryan and Levins (1980), and Yagnik et al. (1981). Aqueous solutions of inorganic salts (mainly chlorides and nitrates), inorganic acids, and organic chelating agents have been tried.

The highest levels of radium removal from acid-leach tailings have been obtained with concentrated inorganic acids and with organic chelating agents. For example, Seeley (1977) indicates that 3M HNO_3 removes 94% ^{226}Ra at room temperature when contacted with 25% sands for 1 h, and 72% ^{226}Ra when contacted with 25% slimes for 1 h. Ryon et al. (1977) report 85-98% ^{226}Ra removal in a two-stage leach with 3M HNO_3 at 33% solids (sands plus slime) over 5 h at 70°C. ^{230}Th removal was 92-99% during these experiments. Borrowman and Brooks (1975) obtained 92-95% ^{226}Ra removal when processing acid-leach tailings with 1.5-3.0M HCl. ^{230}Th removal in HCl is likely as well. The processes for tailings leaching with concentrated inorganic acids would be similar to conventional sulfuric acid leach processes, but

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capital costs and reagent costs would be somewhat higher. Possible groundwater contamination with nitrates or chlorides would require careful control of effluent streams.

Borrowman and Brooks (1975) and Yagnik et al. (1981) had relatively good results with ethylenediaminetetraacetic acid (EDTA), but EDTA consumption was high because of large amounts of calcium in the tailings arising out of the Canadian requirement for neutralization. Nixon et al. (1983) treated the tailings stream before neutralization and reduced EDTA consumption greatly. This led to the reducing-complexing process, proposed by Nirdosh et al. (1985), that is based on the following:

- An organic complexing agent is used to dissolve radium-alkaline earth coprecipitates and keep the radium in solution in the form of a soluble Ra-ligand complex, e.g., Ra-EDTA;
- An inorganic alkali metal cation that forms weak complexes with the organic agent is used to adsorb preferentially on the silica sites and mask them from adsorption of Ra^{+2};
- A reducing agent is added to help dissolve metal hydroxides, e.g., with ferric hydroxides and jarosites that would otherwise resist dissolution. Dissolution of these metal hydroxides also eliminates radium adsorption sites.

Most of the radium is associated with the slime fraction in the tailings (Nirdosh et al., 1984); hence, it may only be necessary to treat the slimes provided that the sands have acceptably low radium content.

While small-scale experiments by Nirdosh et al. (1984, 1985) have demonstrated the feasibility of the process, pilot-scale studies would be required before implementation. Cost factors, particularly with regard to EDTA consumption, would also have to be evaluated carefully. Present estimates of EDTA recovery are 92 percent per cycle.

With regard to processing of tailings at inactive sites, it may be possible to use mills that have been shut down to provide many of the components necessary for the filtration and washing steps whether inorganic acid-based or EDTA-based reprocessing is selected. This would significantly reduce the capital cost of such an undertaking.

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While removal of the ^{226}Ra during processing will temporarily eliminate the source of radon, secular equilibrium with the longer-lived ^{230}Th parent will result in radium ingrowth (as well as radon) to 36 percent of the ^{230}Th activity in 1000 years, 88 percent in 5000 years, and 97.7 percent in 10,000 years. Hence, removal of the radium alone will not solve the radon emanation problem.

Thorium Removal

Studies by Landa (1980, 1982b) and Stieff (1984) indicate that ^{230}Th is widely distributed in the tailings; there is need to identify in or on which phases the ^{230}Th occurs. Landa also indicates a large difficult-to-extract fraction (about 70 percent). Nirdosh (1985) has conjectured that the low extraction could be due to two factors:

- Formation of thorium sulfate into an insoluble form when samples were dried at 110°C before leaching and
- Possible precipitation of the ^{230}Th as hydroxide during the course of sequential leaching—the hydroxide is very difficult to dissolve even in the presence of a reducing agent.

Therefore, the feasibility of removing ^{230}Th from tailings remains an open question. Research is required to determine whether the reducing-complexing treatment, which appears attractive for ^{226}Ra removal, will extract the precipitated ^{230}Th. It is known that concentrated HNO_3-based processes would stabilize ^{230}Th, but this is not certain for concentrated HCl-based processes.

Modifications to Existing Acid-Leach Plants

The EDTA-based process discussed for removal of radium from tailings can also be used as circuit add-ons in existing acid-leach plants. An advantage in doing this would be that the leached solids could always be washed, maintaining a pH of 1.5-2.0 before treatment with the complexing agent. This would prevent precipitation of difficult-to-dissolve metal hydroxides, which may provide adsorption sites for radium. Some process simplification may therefore be possible.

As mentioned previously, Landa (1982b) and others agree that up to 90 percent of the ^{230}Th is dissolved during sulfuric acid leaching. The ^{230}Th is discharged with the solid tailings and may then precipitate, be adsorbed, or react with the solids in

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the tailings piles. Once the ^{230}Th is so stabilized, it appears to be difficult to extract. However, ^{230}Th may easily be removed from solution either by ion exchange or solvent extraction in a thorium recovery circuit. The technology to accomplish this is well developed (see White, 1960; Law, 1966; Vermeulen, 1966; Sethi and Rai, 1970; and Ritcey and Lucas, 1972). Therefore, it would be relatively straightforward to remove this long-lived precursor of ^{226}Ra by modifications to existing acid-leach mills. ^{230}Th does not appear to go into solution during carbonate leaching, and carbonate leaching mill modifications for its extraction do not appear feasible.

Another possibility, which is just beginning to receive serious consideration in Canada, is to add an inorganic salt such as $FeCl_3$, KCl, or $CaCl_2$ to the tailings slurry immediately after leaching (and before solid-liquid separation).

New Mill Processes to Produce "Safer" Tailings

As discussed above, existing processes for extracting uranium would require substantial modifications to produce tailings with acceptably low levels of ^{226}Ra and ^{230}Th.

These considerations have resulted in work being done on alternate uranium milling processes, which co-extract the uranium, ^{226}Ra, and ^{230}Th. Partitioning of the extract can then be done by solvent extraction or ion exchange, provided the radionuclides are in solution. The alternative leachants that have been studied have included the following:

1. Nitric acid: Ryon et al. (1977).
2. Hydrochloric acid: Borrowman and Brooks (1975), Ryon et al. (1977), Haque et al. (1980).
3. High-temperature chlorination: Cable and Schlundt (1918), Skeaff and Laliberte (1980).
4. Ferric chloride: Sawyer and Handley (1959).
5. Ferric chloride: Nirdosh et al. (1983).
6. Ferric nitrate: Nirdosh et al. (1985).

Of these, the first four processes are difficult to commercialize because of the high concentrations of the reagents and their corrosiveness. The fifth process appears to be more feasible because of the lower ferric chloride content but it is still quite corrosive. Both the fifth and sixth processes would have to recycle Cl^- or

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NO_3^- because of environmental concerns. The same concern, of course, applies to processes 1 through 4 above; i.e., the nitric acid-based processes and the ferric chloride-based process (Sawyer and Handley, 1959). Typical environmental regulations allow 10 ppm for the discharge of NO_3^- and 750 ppm for Cl^-. Clearly, the ferric nitrate-based processes would have to have more sophisticated recycle requirements.

The ferric chloride and ferric nitrate processes are similar. If sulfide is present in the ore (not a feature of most U.S. ores) this is removed by flotation, so as to prevent co-precipitation of $RaSO_4$ during later leaching owing to production of SO_4^{2-} by oxidation of the sulfide. The ore, whether treated by flotation or not, is then treated with either a dilute solution (0.02M) of ferric nitrate acidified with 0.02M nitric acid or of ferric chloride (0.1M) acidified with 0.1M hydrochloric acid. The tailings obtained from this operation are nonsulfidic with source strengths of about 1 Bq (30 pCi)/g of ^{226}Ra.

The dissolved uranium and radium may be separated by several partitioning processes such as solvent extraction or surface adsorption (e.g., on MnO_2). A very large fraction of the leach liquor is recycled to give economies in reagent costs and to keep within environmental regulations of NO_3 and Cl^- discharge. In general, the ferric nitrate process is preferable because it is relatively noncorrosive and gives 97 percent uranium and 93 percent radium extraction. Typical leaching times for both processes are about a day (24 h) at 75°C with a solid-to-liquid ratio of 1 to 2 mL/g. Note that no results on ^{230}Th extraction are available in the work reported on $FeCl_3$ and $Fe(NO_3)_3$ leaching (Nirdosh et al., 1983, 1985). It appears likely that ^{230}Th would extract and remain dissolved, but this would have to be verified in further work.

Disposal of Extracted Radionuclides and Residues

Reprocessing of tailings or modification of milling practice can be expected to yield (1) large volumes of residual tailings with relatively low radionuclide content and (2) small-to-moderate volumes of more strongly radioactive material. If the reprocessing option is to be considered seriously, procedures will have to be developed for the management of these very different fractions.

At present, laboratory-scale extraction of radium and thorium can reduce radioactivity to a level of under 1 Bq (20 pCi)/g in

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residual tailings (e.g., Baird et al., 1982; Ryon et al., 1982). At production scale, however, a remnant radioactivity of perhaps about 2 Bq (40-50 pCi)/g is a more realistic target. Material of this radioactivity is not innocuous, but the concentration of radionuclides is only about a tenth of that of untreated tailings. As with a hydrocyclone-separated fraction of normal tailings, this material can be returned as backfill to open pit or underground working or surface stored with a modest amount of protective control.

The strongly radioactive fraction of a reprocessing operation represents a more complex problem, the dimensions of which will vary greatly with the nature of the process and of the products—neither of which can be specified at present. Uranium (together with other marketable metals) presumably will be recovered as separate fractions and simply transferred to the commercial market. Thorium (^{230}Th) and radium (^{226}Ra) conceivably could also be recovered in relatively concentrated form, but neither can be considered as a potential resource. Common thorium (^{232}Th) is marketed in minor quantities for use (as ThO_2) in gas lantern mantles and as an alloying element and, after conversion to fissionable ^{233}U by neutron bombardment, has been tested as a nuclear fuel. No commercial uses, however, are known for radioactive ^{230}Th. Radium, which once commanded a fantastic price—$135,000 per gram in 1918 and briefly nearly $180,000 per gram (Landa, 1982a)—is no longer marketable; what minor needs remain are readily met from stocks on hand. At present, therefore, these two metals, if recovered as fairly pure compounds, would require special consideration; the evident alternatives are to stockpile them in secure storage on the assumption that a use ultimately will develop or to classify them as a nuclear waste and dispose of them accordingly. The quantities involved are difficult to estimate, but for radium some hint of the magnitude is provided by calculations based on the radioactivity of average uranium ore, about 10 Bq (280 pCi)/g. Assuming secular equilibrium (known not to be an entirely valid assumption), each million metric tons of ore would contain 280 g of Ra, nearly all of which will be in the tailings. In fact, however, rather than separation in purified form, radium and probably thorium also would emerge as constituents of a resin or a complex sludge. The nature and volume of this material, in which the radionuclides might be only a small fraction, would depend on the separation process. It is likely, however, that the material

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would fall into the loosely defined category of intermediate-level waste, requiring special handling and disposal. Institutional responsibilities for the disposal of such materials are diffuse and pose potential problems.

Preliminary calculations have been done by Nirdosh (1985) to estimate typical masses of intermediate-level waste that would need to be handled. If the well-established process for a precipitating radium form solution as barium-radium sulfate is used, the concentration factors are typically 1:25,000, i.e., starting with 1000 kg of ore with 20 Bq (500 pCi)/g of ^{226}Ra, a sludge of 0.04 kg would be formed. Further concentration by a factor of 50 is considered feasible by Nirdosh if the sludge is redissolved and the ^{226}Ra preferentially precipitated. This would reduce the sludge volume to 0.0008 kg/1000 kg of ore containing 20 Bq (500 pCi)/g of radium.

IN-SITU EXTRACTION OF URANIUM (SOLUTION MINING)

Much of the problem of future tailings management would be eliminated if uranium were extracted without fine grinding of ore transferred to the surface. This option, in-situ extraction or solution mining, is discussed in some detail in the Final Generic Environmental Impact Statement issued by the U.S. NRC (U.S. Nuclear Regulatory Commission, 1980).

Extraction of uranium from unmined ore is feasible if certain specific conditions are met: (1) the ore minerals occur in a unit of permeable rock that is generally horizontal and underlain (and preferably overlain also) by relatively impermeable strata; (2) the ore body is below the static water table; (3) the uranium is leachable by the proposed reagents. Other significant factors are the direction and volume of groundwater flow, depth and form of the ore-bearing horizon, and presence or absence of cross-cutting fissures.

The layout of a solution mining operation is shown diagrammatically in Figure 4-6. The leaching solution (lixiviant) may be any of a variety of compounds, such as ammonium carbonate, sodium carbonate, or sulfuric acid, that are capable of forming stable soluble complexes with hexavalent uranium. The lixiviant is forced into the ore-bearing horizon from a series of spaced injection wells and returned via centrally located production wells.

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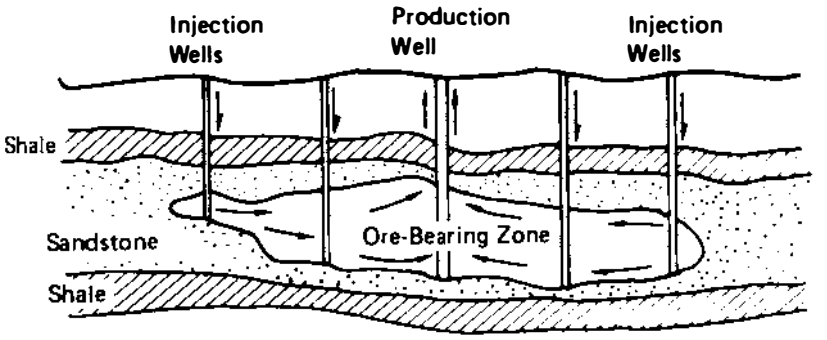

FIGURE 4-6 Simplified diagrammatic representation of a solution mining operation. Directions of fluid flow indicated by arrows.

Uranium is recovered from the pregnant solution by standard techniques of ion exchange, elution, and precipitation as U_3O_8.

In 1978, the last year for which significant data are readily available, solution mining was being used at 25 locations, mostly in Wyoming and Texas. The strict requirements of structure, stratigraphic confinement, rock permeability, and location with respect to the groundwater table effectively limit application of solution mining to an unknown but probably modest fraction of known ore deposits—only to certain deposits of the sandstone type in the United States. Nevertheless, where feasible and carefully controlled, this is an effective means of exploiting ore deposits of both standard and substandard grade with relatively minor environmental impacts. In contrast with conventional mining, in which virtually all ^{226}Ra and other radionuclides associated with uranium end up in tailings, in leach mining most will remain unextracted. In a specific case, it is estimated that less than 5 percent of the radium in the ore body would be brought to the surface (U.S. Nuclear Regulatory Commission, 1978). A potential in leach mining for groundwater contamination does exist, but at least during the production phase this is strongly inhibited by the creation of a pressure sump; groundwater flow is inward, toward the pumped production wells.

The major environmental problems are the safe disposal of spent fluids, which may contain high levels of toxic elements, and the restoration of the aquifer after the operation is completed. In recent operations the liquid wastes generally are transferred to

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lined evaporation ponds (U.S. Environmental Protection Agency, 1983). After the ore is exhausted, the aquifer is restored by water flooding, with care being taken not to disperse remnant lixiviant and mobilized toxic metals to the broader groundwater system. Chemical reactants may be added to the injected waters to induce reducing rather than oxidizing conditions. The method suggested by Deutsch et al. (1985) involves addition of reductants such as sodium sulfide that promote precipitation of oxidized metals, notably uranium, arsenic, selenium, and molybdenum.

CONCLUSIONS AND RECOMMENDATIONS

On the basis of the preceding discussion the panel draws the following conclusions and makes the following recommendations:

1. Based on a consideration of the geological evidence, the stability of tailings accumulations in surface or near-surface piles cannot be assured over time periods of more than thousands of years (and in some cases hundreds of years) without the option of active human intervention.

> *Recommendation*: Plans for the management of mill tailings piles should recognize and explicitly incorporate plans for a continued low-level program of active monitoring and the option for active human intervention should it become necessary.

2. Protection of some piles against catastrophic floods will not be possible even with heroic measures. However, the risk posed by piles under such circumstances is small, and inconsequential compared with other impacts of such a catastrophic event, because tailings would almost certainly be mixed with and diluted in a much larger volume of sediments.

> *Recommendation*: While protection of piles against local periodic floods is feasible and appropriate, protection against large, truly catastrophic floods is neither feasible nor warranted from the perspective of the risk presented.

3. Radon flux from a cover of less than a few meters of earth cannot now be predicted with precision either immediately after construction of the cover or in the long term when various physical changes, such as change in water content, have taken place in the cover.

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Recommendation: The design of covers for the purpose of limiting radon release should be validated on a site-specific basis by measurements of radon concentrations in the vicinity of the pile. Such measurements could lead to a reduction in the minimum thickness of earth cover needed to limit radon release to acceptable levels.

4. Long-term irregular settlement of tailings piles may cause mechanical disruption of any type of cover (earth, asphalt, stabilized tailings, plastic sheeting, or some combination), with consequent changes in the rate of radon emission and rainfall infiltration. Chemical deterioration of cover materials may also change their effectiveness over time.

Recommendation: Periodic inspection and maintenance are essential to ensure that covers of any type will continue to function effectively over the long term.

5. There is enormous variability in the geological conditions of mill tailings sites. Any tailings management strategy that is effectively going to control radon emissions and groundwater contamination from piles must deal with at least some of this variability in a site-specific way.

Recommendation: Background water-rock-soil chemistry and groundwater hydrology should be better characterized at most piles. Monitoring systems should be installed at most piles to test for possible transport of hazardous materials.

6. For existing piles, leachate from the tailings may reach local groundwater.

Recommendation: Groundwater monitoring should be undertaken to evaluate the need for remedial action.

7. Liners that are installed under tailings piles are almost certain in time to fail because of construction flaws, mechanical movements caused by the weight of the tailings, chemical changes associated with the leachate, and other factors.

Recommendation: If liners are used, and local groundwater protection is deemed essential, they should be backed up by a drainage system for collecting leachate that may leak through the liner and by a system to monitor the

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effectiveness of the liner/drainage system. A contingency plan should be developed for treating any contamination that does enter the groundwater.

8. Radionuclides are concentrated in the micrometer and submicrometer size fraction of the tailings.

Recommendation: Surface stabilization, which is technically feasible and can be accomplished with modest treatment, is necessary to prevent wind dispersal. Covers designed to control radon emissions typically will also solve this problem.

9. Solidification of tailings by any of several techniques may be a useful option in selected special circumstances, but because of the high costs due to the large volume of tailings, uncertainties as to long-term stability, and problems presented for possible future reprocessing it is not an attractive general solution.

10. Both for purposes of the design of adequate management strategies and as a basis for decisions about possible future reprocessing, a far better chemical and mineralogical characterization of the piles is required than now exists.

Recommendation: A pilot program of coring and detailed characterization is indicated for selected piles.

11. Reprocessing of tailings should be considered both as a strategy for hazard management and in terms of the recovery of economically useful materials, such as uranium. Processes to remove ^{226}Ra from tailings to levels of 1 or 2 Bq (20-50 pCi)/g are technologically feasible. We do not yet have an adequate understanding of the extent to which ^{230}Th can be recovered by some of these processes.

Recommendation: A program of research on the process technologies that might be applied in reprocessing should be considered.

12. The following conclusions relate to modifications of existing and future processes:

For operating and future conventional acid-leach mills the technology to remove ^{230}Th, which is in solution, is well developed. However, once the ^{230}Th is deposited

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in tailings piles, it appears to form compounds that are difficult to extract.

- Extraction of radium without substantial modifications to acid-leach processes may be possible by adding reagents that form soluble complexes of radium at low pH.
- Two new processes for coextraction of uranium and radium using either ferric chloride or ferric nitrate have been developed at a bench scale.

Recommendation: An expanded program of research on technologies that might be used to modify or replace existing processes in order to reduce or eliminate radiological hazards in tailings management should be undertaken.

13. Both in reprocessing existing tailings and in applying various new or modified processes in new extraction, most of the thorium and much of the radium could be removed. These processes would produce relatively small volumes of thorium- and radium-bearing concentrates, the disposal of which might pose significant technical and political problems, although it is conceivable that a market for these materials could develop in the future.

14. At the time of processing, or during subsequent reprocessing or relocation of tailings, it is possible to separate the slime fraction of the tailings, which contains the bulk of the environmentally hazardous materials, from the less radioactive sand fraction. Separation, now a common practice for purposes of embankment construction, also is desirable from the point of view of possible subsequent recovery of valued constituents, which are concentrated in the slime fractions and are thus more readily available for extraction. If little or no human intervention is anticipated, however, separation may have undesirable environmental impacts, since it gives rise to large quantities of materials that are more hazardous than well-mixed tailings.

15. Where the geological conditions allow, in-situ extraction is a proven technology that is likely to become increasingly important in the future. Advantages of this method include the absence of tailings and of associated radon and other wastes and the absence of dust-control problems. An important part of this technology is the cleanup of the aquifer once extraction has been completed.

Recommendation: Further research is needed on the problems of aquifer cleanup after in-situ mining.

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5
Problems in the Assessment and Management of Risks Posed by Uranium Mill Tailing Piles

This chapter first considers how some of the risks posed by uranium mill tailings piles compare with other risks that society and its individuals face regularly. It then briefly reviews the history of uranium mill tailings risk-management activities undertaken by both the federal government and the states. With this background completed, it considers some of the basic philosophical choices that should be made to select a decision framework for use in uranium mill tailings risk management and examines the extent to which scientific evidence is available, and has been effectively used, in choosing and implementing such a framework. It evaluates the current regulatory decision against some alternatives. On the basis of this analysis the chapter closes with a number of specific conclusions and recommendations.

COMPARATIVE RISKS

Most problems in risk assessment are complicated by a variety of scientific uncertainties. Modern techniques of quantitative risk assessment provide a number of tools that allow a risk analyst to characterize and analyze these uncertainties (Benjamin and Cornell, 1970; Howard, 1976; Keeney, 1982; Morgan, 1984; Morgan et al., 1984). Chambers and his colleagues (SENES, 1985) have begun to use such techniques in analyzing the potential risks of uranium mill tailings in Canada. Because Canadian piles are in a particularly wet environment, often on or in lakes, and because the tailings have somewhat different chemical and physical characteristics, the results of these Canadian assessments are not applicable

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to a U.S. setting. In the United States, Rogers and Nielson (1985) have done some uncertainty analysis on pile cover designs, but none of the general risk assessments of uranium mill tailings that have been undertaken in the United States have yet made adequate use of modern techniques for the characterization and treatment of uncertainty. In the context of other nuclear wastes, the U.S. Environmental Protection Agency (EPA) has recommended the adoption of quantitative techniques for characterizing and analyzing uncertainty (40 CFR Part 191).

As we will elaborate later in this chapter, the risk analysis that EPA performed in support of the current standards (U.S. Environmental Protection Agency, 1983a, 1983b) suffers from at least three shortcomings: (1) it adopts specific model formulations without adequately comparing their appropriateness with possible alternative model forms and then uses single-value estimates in those models rather than a range or full probability distribution, (2) it provides little discussion of the uncertainties and sensitivities of the resulting assessments of health impacts, and (3) it focuses primarily on radioactive exposures and pays insufficient attention to assessing risks from contaminated groundwater.

With the resources available to this study, it has not been possible to perform a probabilistic health risk assessment for uranium mill tailings piles. Chapter 3 contains a simple exposure risk calculation, which suggests that a person living next to a uranium mill tailings pile may face a lifetime lung-cancer risk of between 4×10^{-4} and 4.6×10^{-2}, depending on the pile characteristics and the person's lifestyle. Because of the limited amount of previous analysis of the potential hazards posed by groundwater contamination from piles, the panel has been unable to perform even simple calculations for this latter class of hazards. It strongly believes, however, that a full modern assessment of both the air and water hazards is needed in order to assess properly the risks posed by uranium mill tailings piles. How does the range of radon lung-cancer risks estimated in Chapter 3 compare with a variety of other risks that exist in our society? The first thing that one can do is to make an actuarial comparison; that is, ask how uranium mill tailing pile radon risks compare with the average expected mortality imposed by a variety of other risks.

Crouch and Wilson (1982) have provided estimates of the annual average mortality risk for a variety of common risks, a number of which are summarized in Table 5-1. The radon health

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risk estimates from Chapter 3 can be approximately compared with these by dividing by a 70-year lifetime.

While such actuarial comparisons are useful, there is considerable evidence that when people make judgments about risks, and in particular when they make judgments about the need for risk management, they consider many factors in addition to expected mortality. That is, they use multiattribute criteria. Slovic et al. (1980, 1985) performed a series of studies in which people were asked to evaluate a large number of known and hypothetical risks in terms of a number of attributes, such as whether the risk can be controlled by individuals, whether the benefits and risks involved are equitably distributed, whether the risk is voluntary, and whether the effect is to be delayed or immediate. In these studies, subjects were asked to evaluate each risk against each attribute on seven-point scales (e.g., Controllable 1-2-3-4-5-6-7 Uncontrollable). A factor analysis performed on the results of the evaluations yielded three groups of attributes that displayed high correlation among the members of each group and low correlation across groups. The lower portion of Figure 5-1 identifies a group of eleven attributes that were highly correlated which Slovic et al., have termed collectively the "dread risk" factor (horizontal axis), and a group of five attributes that were highly correlated which they named the "unknown risk" factor. A third factor, involving the number of people exposed, is not shown. Geometric mean results from one such study, in which subjects were asked to evaluate 90 risks in terms of 18 different attributes, are displayed in the upper part of Figure 5-1. Various replications of this study have yielded consistent results.

Additional studies by these same investigators (Slovic et al., 1985) have yielded similar results and have revealed that the location of a risk in this factor space is highly correlated with the degree to which people believe that strong risk-management activity is called for. This is illustrated in Figure 5-2. Since, in these studies nuclear power, weapons, and all waste-related issues tend to fall well over on the right-hand side of this factor space, one can reasonably speculate that many people will feel that a higher level of risk management effort is appropriate for mill tailings than is appropriate for at least some of the other risks that have comparable actuarial levels in Tables 5-1 and 5-2.

While people apparently adopt a multiattribute approach in making risk judgments, most of the great disparity that can be

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TABLE 5-1 Examples of Annual Average U.S. Mortality Risk from a Variety of Activities and Occupations[a]

Risk	Annual Average Risk	Uncertainty
All risks from smoking	3.0×10^{-3}	Factor of 3
Cancer risk from smoking	1.2×10^{-3}	Factor of 3
Work in agricultural jobs	6×10^{-4}	9 percent
Motor vehicle accidents	2.4×10^{-4}	10 percent
Home accidents	1.1×10^{-4}	5 percent
Work in service and government jobs	1.1×10^{-4}	8 percent
Work in manufacturing jobs	8.2×10^{-5}	8 percent
Work in trade-related jobs	5.3×10^{-5}	15 percent
Fires	2.8×10^{-5}	5 percent
Firearms	1.0×10^{-5}	8 percent
Electrocution	5.3×10^{-6}	5 percent
Lightning	5.3×10^{-6}	5 percent
Tornadoes	6×10^{-7}	Factor of 2
Chapter 3 order of magnitude estimates by panel of lung cancer risk		
Radon, w/o pile	1×10^{-4}	Factor of 3
UMT increment to a person living right next to the high-radon pile with base case activity	8×10^{-4}	Factor of 3
UMT increment to a person living right next to the low-radon pile with base case activity	7×10^{-5}	Factor of 3

[a]Note that while these are reported in terms of annual average risk, the actual risk faced by an individual is often not uniform across a lifetime. For example, the risk of cancer increases with age. By dividing by a 70-year lifetime, the risks estimated in Chapter 3 can be roughly compared (bottom of table). This table contains risks with a variety of attributes, including occupational-nonoccupational and voluntary-involuntary. Note also that the number of people exposed to the various risks varies significantly from risk to risk.

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TABLE 5-1 (continued)

SOURCE: After Crouch and Wilson, 1982.

Disclaimer: Panel member Robert H. Neill believes that these risks should not be combined in the same table even for purposes of comparison. He argues that society has adopted levels of acceptability for federally mandated risks to populations from involuntary radiation exposure that are different from those involving occupational risks to adults compensated for their services. Similarly, society grants the individual the right to voluntarily accept the considerable risks of cigarette smoking. Mr. Neill argues that while the data in Table 5-1 are factual, the comparison and ranking will invite some readers to view this as a yardstick of acceptability.

observed in the rates at which society invests to manage risks across different programs and risk categories probably arises from other sources. Table 5-2 summarizes the results of one study of the average cost of mortality prevention in a set of federal risk-management programs (Graham and Vaupel, 1981). Most of the wide disparity displayed probably results from the different political histories of the programs, and the associated effects of the finite agendas of public agencies, and not primarily from the effects of applying multiattribute risk judgments.

In summary, viewed in the perspective of the wide variety of risks that face U.S. society, the lung-cancer risk posed to most people living near uranium mill tailings ranges from small to modest, but it is clearly large enough for most Americans to consider it worth societal attention. In special circumstances for a few individuals living near a few uncontrolled piles the risk can become significant. Simply for comparison, without making judgments about acceptability, it is useful to note that in all cases the actuarial risk appears to be less than the risk from smoking. By virtually any measure, the risk for people living at distances beyond several kilometers from a pile is trivial.

PREVIOUS RISK-MANAGEMENT ACTIVITIES

Before 1970, permit requirements for a uranium mill consisted primarily of obtaining an AEC "source material" license, basically a construction and operating permit with a requirement to maintain a radiological safety program (Nordhausen, 1979). The Atomic Energy Commission (AEC), and subsequently the U.S.

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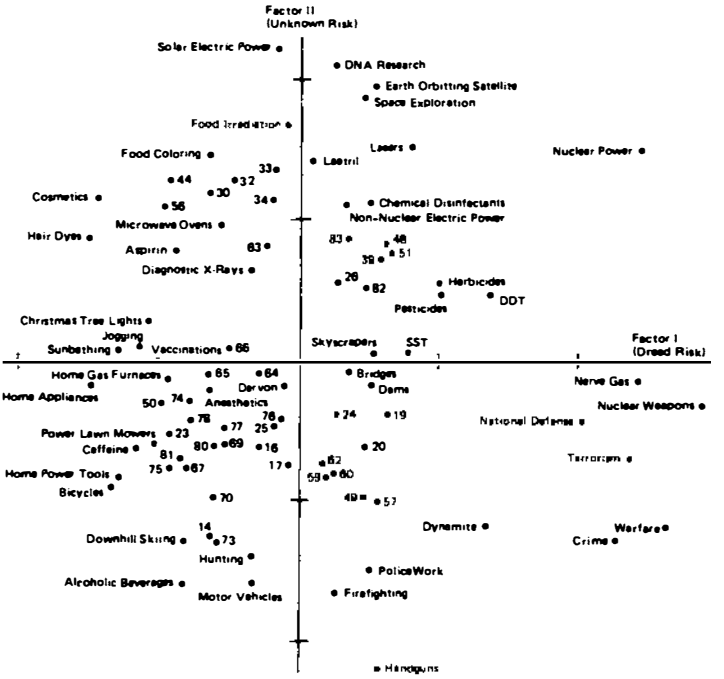

1 Home Gas Furnaces	19 Jumbo Jets	37 Space Exploration	55 Aspirin	75 Recreational Boating
2 Home Appliances	20 Commercial Aviation	38 Lasers	56 Marijuana	76 Roller Coasters
3 Home Power Tools	21 Anesthetics	39 Asbestos	57 Heroin	77 Scuba Diving
4 Microwave Ovens	22 Vaccinations	40 Police Work	58 Laetril	78 Skateboards
5 Power Lawn Mowers	23 Pregnancy, Childbirth	41 Firefighting	59 Amphetamines	79 Sunbathing
6 Handguns	24 Open-Heart Surgery	42 Christmas Tree Lights	60 Barbiturates	80 Surfing
7 Terrorism	25 Surgery	43 Cosmetics	61 Darvon	81 Swimming Pools
8 Crime	26 Radiation Therapy	44 Fluorescent Lights	62 Morphine	82 Fossil Elec. Power
9 Nerve Gas	27 Diagnostic X Rays	45 Hair Dyes	63 Oral Contraceptives	83 Hydroelectric Power
10 Nuclear Weapons	28 Alcoholic Beverages	46 Chem. Disinfectants	64 Valium	84 Solar Electric Power
11 National Defense	29 Caffeine	47 DNA Research	65 Antibiotics	85 Non-Nuc. Elec. Power
12 Warfare	30 Water Fluoridation	48 Liquid Natural Gas	66 Downhill Skiing	86 Nuclear Power
13 Bicycles	31 Food Coloring	49 Smoking	67 Fireworks	87 Dynamite
14 Motorcycles	32 Saccharin	50 Tractors	70 Football	88 Skyscrapers
15 Motor Vehicles	33 Sodium Nitrite	51 Chem. Fertilizers	71 Hunting	89 Bridges
16 Railroads	34 Food Preservatives	52 Herbicides	72 Jogging	90 Dams
17 General Aviation	35 Food Irradiation	53 DDT	73 Mountain Climbing	
18 SST	36 Earth Orbit Satellite	54 Pesticides	74 Mushroom Hunting	

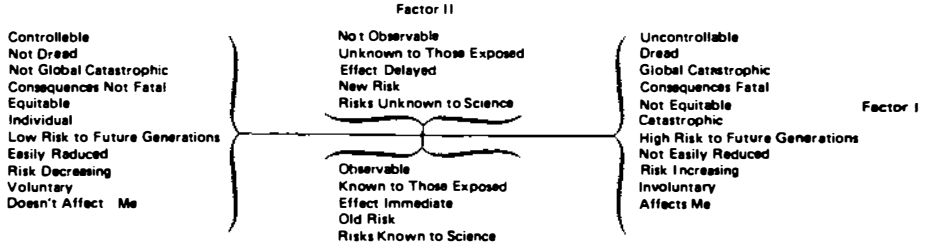

FIGURE 5-1 Factor analysis by Slovic et al. (1980), which illustrates the multiattribute nature of risk perception.
SOURCE: Slovic et al. (1980).

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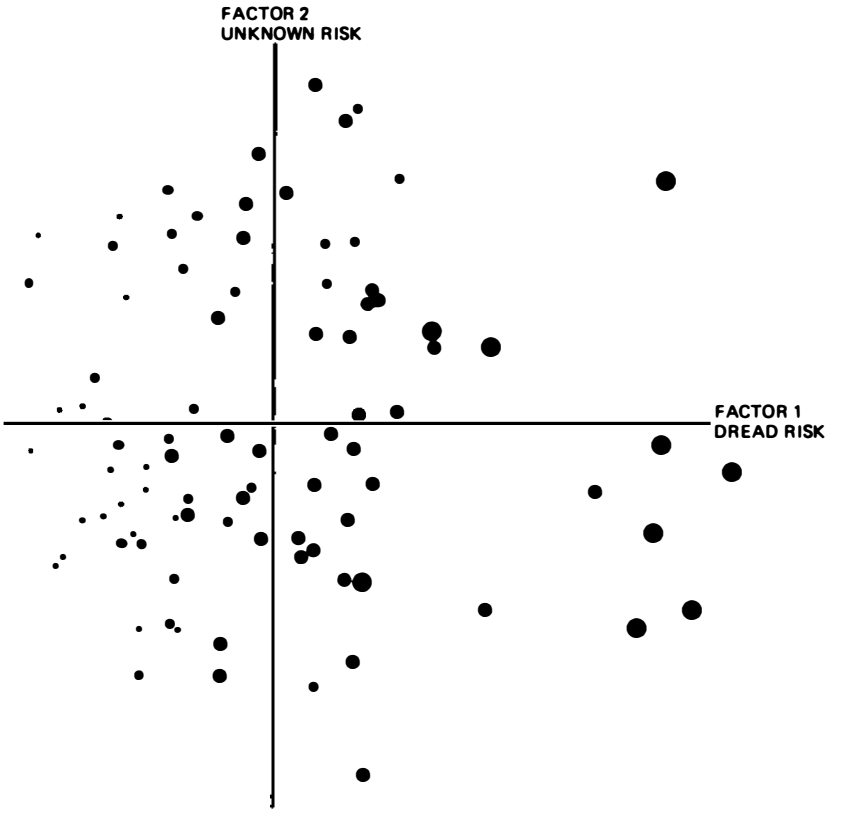

FIGURE 5-2 Correlation of people's attitude toward degree of regulation that they feel is necessary and location in the factor space.
SOURCE: Slovic et al. (1985).

Nuclear Regulatory Commission (U.S. NRC), concluded that they had no regulatory jurisdiction over uranium mill tailings after mill operations were terminated because the tailings material did not contain more than 0.05 percent by weight of uranium plus thorium and thus was not itself licensable material (Hendrie, 1978). The AEC also concluded that it had no jurisdiction over the transfer of tailings by mill licensees to other parties, although it suggested that local authorities might have an interest in the subsequent use of the tailings.

In 1966, the Public Health Service, the Federal Water Pollution Control Administration, and the AEC stated in a Joint

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TABLE 5-2 Breakdown by Regulatory Agency of the Average Cost to Save a Life in Risk Management Programs (Values in table are the numbers of risk management programs)

Regulatory Agency	Average Cost to Save a Life			Total Number Programs
	Less than $170,000	$170,000 to $3 Million	Above $3 Million	
National Highway Traffic Safety Administration	22	7	0	29
Department of Health and Human Services	4	1	0	5
Consumer Product Safety Commission	4	2	0	6
Environmental Protection Agency	4	1	5	10
Occupational Safety and Health Administration	0	0	7	7
Total number of programs	34	11	12	57

SOURCE: After Graham and Vaupel (1981).

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Agreement that mill owners should develop plans for stabilizing and containing mill tailings piles and that those plans should be submitted to the appropriate state agency for approval. The federal agencies, however, would take no more than an advisory role (Agreement, Joint Federal Agency Position Regarding Control of Uranium Mill Tailings, 1966). Although the state of Colorado moved to establish regulatory standards, in the main public access to tailings sites was not limited until 1969, and tailings were used as construction materials in roads, houses, and public buildings (Dennis, 1984).

During the early 1970s increasing public concern about possible risks associated with mill tailings resulted in the passage of the Uranium Mill Tailing Remedial Control Act in 1978. The remedial program for inactive sites, to be funded from federal and state revenues at a 90/10 ratio (100 percent federal on Indian land), was to clean up 24 former processing sites and an estimated 3000 to 5000 "vicinity properties" that were not covered by the legislation established to clean up the misuse of tailings in Grand Junction. A vicinity property is one located near the processing site that has become contaminated with radioactive material originating there. Nine of the former processing sites were designated as high priority, six as medium priority, and nine as low priority (Coffman, 1983). All of these sites are located in the western United States except for one in Canonsburg, Pennsylvania, a facility used for recovery of radium from ore over the period 1911-1922 and recovery of radium and uranium from ores and scrap over the period 1930-1957. Together these sites contain approximately 25 million tons of tailings and occupy a total area of approximately 400 ha (1000 acres) (U.S. Department of Energy, 1984). Remedial actions are to be conducted in accordance with negotiated cooperative agreements with affected states and Indian tribes. The facility must be licensed pursuant to NRC regulations that are in accordance with EPA standards. They also require an environmental impact statement or environmental assessment under the National Environmental Policy Act (Groelsema, 1982). Estimates of costs, originally in excess of $100 million, have increased by about a factor of 10. Under this program the liability of uranium producers for the cleanup of inactive sites is limited.

Thirteen of the licensed operating sites contain substantial quantities of tailings that were generated before 1970 when the

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mills were producing uranium for sale exclusively to the government. The law directed the Department of Energy (DOE) to develop a plan for a cooperative program to assist in the stabilization and management of defense-related uranium mill tailings that have been commingled with those resulting from production of uranium for commercial use.

By the end of 1980, the U.S. NRC had issued the Final Generic Impact Statement on Uranium Milling (U.S. Nuclear Regulatory Commission, 1980) and regulations for active sites, including interim standards developed under National Environmental Policy Act (NEPA) requirements (Smith et al., 1985). Industry officials protested that the regulations were too strict and that the U.S. NRC had overstepped its authority in issuing regulations before the EPA standards were promulgated. The Court of Appeals backed the U.S. NRC; Congress ordered EPA to issue standards and prohibited implementation of the mill tailings regulations by the U.S. NRC until EPA issued the standards (Groelsema, 1982). The U.S. NRC issued its regulations (10 CFR Part 40) in October 1985 to conform with EPA's.

CURRENT SITUATION

After Congress assigned EPA the responsibility of establishing general standards to limit radiation exposure from mill tailings, the agency issued regulations for the cleanup of both inactive and active sites in 1983. In late 1984, the U.S. NRC, which has the authority to establish specific regulations that a licensee must meet, undertook steps to conform to the EPA requirements by publishing revised regulations for public comment. Since mid-1982, the DOE has taken the position that the cost of implementing the EPA/U.S. NRC standards is not commensurate with the health benefits to be obtained and that remedial action should be limited to the prevention of migration and erosion of the tailings and their misuse. Nonetheless, the DOE has undertaken a remedial action program to stabilize and clean up various sites in accordance with EPA regulations and estimates (J. Themalis, DOE/Albuquerque Operations Office, personal communication, 1986) that the corrective action program for the inactive sites will be approximately 25 percent completed by October 1, 1986, with two sites fully stabilized. The cost of the corrective program has escalated from the 1978 expectation of $0.5 billion dollars to recent estimates

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as high as $4 billion. Because New Mexico, Colorado, Texas, and Washington are Agreement States, standards issued by such states are required to conform to those issued by the U.S. NRC. Except for groundwater contamination, the U.S. NRC promulgated regulations (10 CFR 40) in October 1985 to conform with the EPA Standards; Agreement States will have three years to conform their regulations with those of the U.S. NRC.

CHOICE OF RISK-MANAGEMENT STRATEGY

Before undertaking risk-management activities related to an environmental hazard such as uranium mill tailings, it is important to select a general risk-management strategy that is philosophically acceptable to society, compatible with the science of the problem, and administratively, politically, legally, and economically feasible. While scientific and technical matters are important in such a choice, the choice is not fundamentally a scientific or technical question. Rather it involves a choice of values and a variety of political considerations.

In performing analysis in support of risk management decisions, it is important to choose and explicitly identify the general risk-management strategy that will be employed, since otherwise scientifically and socially consistent choices may not result. A wide variety of alternative decision criteria are possible (Goldstein, 1981; Lave, 1981; Schulze and Kneese, 1981; Stone, 1974). Choice of a specific criterion is a value decision, not a scientific question. For example, in some circumstances society may deem a benefit-cost formulation the appropriate decision standard. In others the decision may be rights based and have little or nothing to do with economic issues. A few of the many decision criteria that might be applied to the case of uranium mill tailings are summarized in Table 5-3.

In its analyses of the airborne radiological hazards posed by uranium mill tailings (U.S. Environmental Protection Agency, 1983a, 1983c), the EPA has not explicitly identified the decision criterion that it has applied. The standards that were adopted appear not to have been made strictly on overall social benefit-cost considerations; the discussions of Chapters 3 and 4 suggest that there may be more cost-effective ways to reduce radon exposures received by members of the general public—or even by persons

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TABLE 5-3 Examples of Some of the Alternative Decision Criteria That Might be Applied in Approaching a Risk-Management Decision on Uranium Mill Tailings

UTILITY-BASED CRITERIA

Deterministic Benefit-Cost: Estimate the benefits and costs of the alternatives in economic terms, then choose the one with the highest net benefit (i.e., maximum benefit minus cost).

Probabilistic Benefit-Cost: Same as deterministic benefit-cost but use expected values of uncertain net benefit.

Cost Effectiveness: If total costs and benefits cannot be estimated, but regulation is deemed necessary, then a specific alternative may be chosen on the basis of minimising the cost to achieve a given objective. Implicit in this approach is some assumption that the investments needed to get to the stated objective are worthwhile, but a formal cost-benefit criterion is not applied.

Bounded Cost: Sometimes also termed the "regulatory budget approach"; this strategy sets a maximum budget that society can affort to devote to risk-management activity and then hopes that resources will be spent in such a way as to maximise the amount of risk reduction achieved within the budgetary constraint. To the extent that this hope is realised, this criterion is not logically different from the cost-effectiveness criterion.

Minimize Chance of Worst Possible Outcome . . . Maximise Chance of Best Possible Outcome: While these and similar criteria are utility based, they may not be stated in terms of utility maximisation. Political and behavioral considerations frequently dictate the use of such criteria. Of course, this could be viewed as equivalent to maximising a multiattribute utility, but operationally this is rarely done.

RIGHTS-BASED CRITERIA[a]

Zero Risk: Independent of what it costs, and how big the risks are, eliminate, or do not allow, the risk.

Bounded or Constrained Risk: Independent of the costs, constrain the level of risk so that it does not exceed a specific level, or more generally, so that it meets a set of specified criteria.

Approval/Compensation: Allow risks to be imposed only on people who have voluntarily given consent, perhaps after compensation.

TECHNOLOGY-BASED CRITERIA

Best Available Technology: Do the best job of reducing the risk that is possible with current technological capability. In practice, this criterion is almost never applied as a pure strategy. Rather some "feasibility/affordability" criterion is added, making it a mixed criterion or perhaps even a pure cost-effectiveness criterion.

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TABLE 5-3 (continued)

HYBRID CRITERIA
Often hybrids of utility and rights-based criteria are applied. For example, a bounded-risk criterion may be used to set an upper limit that, for ethical reasons, cannot be exceeded, and a benefit-cost criterion may be applied below that level. An example often employed in the context of radiological risk is ALARA (As Low as Reasonably Achievable), which combines a technology-based criteria with a fuzzily stated utility-based criteria.

[a]Strictly speaking all these criteria could be described by a utility function and thus could arguably be considered utility based. Nevertheless, in popular usage we believe the word "rights" better communicates the essence of these criteria.

who live close to piles—than through the level of pile emission control that the standards require.

This does not, however, mean that the radon control that has been required is necessarily inappropriate. Appropriateness can only be evaluated against a specific decision criterion. Suppose, for example, that a rights-based criterion was employed rather than some benefit-risk tradeoff criterion. Under a rights-based criterion one might adopt the view that *any* anthropogenic radon emission from tailings piles is unacceptable and should be controlled, independent of the relative size of its contribution to the indoor and outdoor background levels and independent of the costs of control.

However, without an explicit statement of the risk management decision criterion that is being applied, it is difficult to evaluate the EPA's supporting analysis for scientific adequacy and consistency. For example, if EPA were applying a utility-based decision criterion, such as a benefit-cost criterion, the ratios of the level of radon emitted by piles to both the level of, and magnitude of fluctuations in, ambient outdoor and indoor radon concentrations become critically important data. These data are important in this case because, once the contribution from the pile is less than the natural variation in the background level, it is highly unlikely that further attention to pile emissions can be justified under a utility-based criterion such as benefit-cost. On the other hand, under a rights-based criterion that simply views all pile emissions as unacceptable—independent of their incremental contribution

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to population radon exposures—these ratios become effectively irrelevant.

"One-Time Solution" and "Adaptive Look-Ahead"

In undertaking risk analysis and risk management, most people start out in a fairly linear manner. That is, models of the processes that give rise to exposure and effects are created, consequences of these models are examined, and then a seemingly appropriate risk-management strategy is chosen. Such a choice is often based—although the fact may not be realized or acknowledged—on a fairly firm idea of the class of strategy to be adopted, and the risk analysis is used to fine tune and/or justify the strategy.

Implicit in what is required to implement many risk management strategies is the assumption that enough is either known today, or can be guessed, to produce a solution that will be appropriate for all time in the future—that is, there is a tendency to try to produce single definitive solutions (Clark, 1980). The panel refers to strategies that require such prior knowledge for successful implementation as "one-time solution" strategies.

Uranium mill tailings are a source of chronic low-level airborne radiological risk that will be around for many tens of thousands of years and, in some cases, are also a source of nonradiological and radiological risk of local groundwater contamination that may last as long or longer. The conditions around present mill tailing pile locations may change profoundly within even the next few hundred years. Human settlement and activity patterns may shift. In response to either anthropogenic or natural causes, climatic patterns may change. Large changes in world prices, in the relative economic advantages of different countries, in resource availability, and in the price and capabilities of technology may combine to convert piles from undesired "waste" into attractive sources of valued materials.

A probabilistic assessment and decision analysis could be constructed with events of uncertain magnitude occurring at uncertain times along a time line. Subjective probability distributions that describe the timing and consequences of these events could be elicited from so-called experts. In fact, however, by the very nature of these variables, no one can be particularly expert about their likely values. This strategy would also face the problem of how to handle comparisons of consequences that occur at different

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times—that is, the problem of future discounting. This problem is more complicated than the question of how to value a cancer death today versus a cancer death later in this generation or in some future generation. Since medical and other technological capabilities evolve with time, given exposures in the future may lead to markedly different consequences than they do today.

From the discussion in the preceding chapters, it is clear that enough is known today about the science of the problem of uranium mill tailings and its associated uncertainties to know that, if such a one-time solution assessment model were built, virtually any answer could be obtained from it—depending on the particular set of plausible assumptions and subjective distributions put into it. Under such circumstances, quantitative modeling has little or nothing to contribute, and, in the panel's view, modelers have an obligation to say this and to suggest that alternative strategies be explored.

Potentially attractive alternatives to one-time solution strategies are those that the panel calls "adaptive look-ahead" strategies. These strategies start by acknowledging that, given the current ambiguities about both the science and the alternative possible future states of the world, it is not possible today to select an approach to risk management that will be optimal for all future times. Instead, an attempt can be made to select an approach that will be reasonable over the near term and that, in addition, allows us and our descendants the option of future flexibility—that is, of moving to alternative approaches that may prove more desirable as the future unfolds and as scientific understanding and technical capabilities improve. The term "look-ahead" is used to distinguish such strategies from the sort of unthinking incrementalism that has characterized so many of our nation's earlier attempts to deal with environmental problems. The historian Joel Tarr (1985) has characterized this traditional approach, which has tended to sweep today's problems under a rug only to have them re-emerge as even more serious problems tomorrow, as "the search for the ultimate sink."

Under an adaptive look-ahead strategy one does not ignore the future. Rather one explicitly acknowledges that the future is not very certain. One tries to do the best job possible of estimating the likelihood of alternative possible future developments and then to select good risk-management strategies for today that are also

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designed to preserve options and reasonable flexibility for future decision makers.

In its risk-management decisions on uranium mill tailings, EPA has not made a clear choice of the kind of strategy that it will adopt. The Agency appears to have wanted to pursue a one-time solution strategy but to have recognized that the science of the problem makes this infeasible. Accordingly, EPA called for an approach that will be good for at least a thousand years. The time scale of a thousand years appears to have been chosen without serious scientific justification. Further, the discussion in Chapter 4 suggests that assured pile integrity, without periodic human attention, over a period of a thousand years is a scientifically unrealistic goal.

Generic and Site-Specific Solutions

In addition to deciding explicitly how to deal with possibly changing circumstances over time, it is also necessary in selecting an approach to risk management to decide how to deal with possible variations over space. One strategy is to demonstrate that such variations are of second-order importance to the decisions being made and approach the problem with generic models or solutions. This strategy has several advantages. It is analytically simple. Perhaps more important, it is politically and administratively attractive because it reduces the number of opportunities for parties opposed to the implementation of the risk-management strategy to intervene in decisions in ways that dilute the stringency of the control. For both of these reasons, this approach is frequently adopted by EPA and other U.S. regulatory organizations. However, in order to use this strategy and still legitimately claim that the regulations are based in the science of the problem, one must be able to demonstrate that the simplifying assumptions that are made in using a generic approach introduce errors in the outcomes that are relatively unimportant in the context of the regulatory conclusions that are being drawn.

In the case of the uranium mill tailings regulations, EPA has explicitly adopted a generic approach. The Agency synthesizes many of the inputs to its generic model from aggregations of data taken from individual sites. The resulting generic model is used to make regulatory decisions that the Agency considers appropriate for the generic site, and these are then directly applied to all real

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sites. However, the supporting analysis (U.S. Environmental Protection Agency, 1983a, 1983c) does not adequately demonstrate that the scientific factors that were considered important in the decision for the generic model do not differ at real sites in ways that are important to the decisions being made. The panel analysis in Chapters 2 and 4 makes it clear that there are enormous variations in important physical variables between sites. In these circumstances it is essential to be able to demonstrate the appropriateness of a generic model to the circumstances that apply at specific sites *if* regulation is to be based on the science of the problem.

Performance Versus Design Standards

Once a decision has been made to use government standards as the approach to managing the risks of an environmental hazard like uranium mill tailings, a choice must be made between adopting a performance standard, a design standard, or some hybrid. While the choice is fundamentally philosophical, several different kinds of consideration often enter. These include questions of technical and administrative feasibility, cost, performance verification, and enforceability.

In the uranium mill tailings case, EPA has stated the standard in terms of a performance criterion—but the implementation by the U.S. NRC has basically evolved as a design standard. The advantage of this approach is that it is administratively simple and minimizes the opportunities for dilution of the standard. In principle, once corrective action has been taken no subsequent monitoring is required. The disadvantages are that there is no direct verification of the degree of effectiveness of the controls put in place at any given pile; the performance index (radon flux at the surface of the pile) is not so readily or unambiguously measured as some other values such as airborne concentration and not so clearly related to exposure; the cost effectiveness of the control in terms of radon exposure reduction achieved per dollar spent in control is not readily apparent.

One possible alternative would be a performance standard based on measurements of airborne radon concentration. This approach has the advantages that compliance is readily verified through empirical observations and the amount of control effort expended can be adjusted to fit the local circumstances. It also has

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the advantage that, as circumstances change over longer periods of time, modifications in the control strategy may automatically be required to remain in compliance. The approach has the disadvantages that it requires greater local decision making and thus may offer opportunities to dilute the level of control, it requires a significant amount of (straightforward) data collection and analysis, it may require periodic maintenance as the pile and its cover evolve over time, and it may be difficult to enforce in the future.

Internal Consistency and Public Trust

Risk management is most easily and effectively undertaken in an atmosphere in which the general public has a high degree of confidence and trust in the social institutions charged with the development and implementation of management programs. Thus, for example, the recent National Science Foundation workshop (Covello et al., 1986) on problems in public risk communication spent a considerable portion of its time addressing the issue of public trust. One ingredient that is essential for the long-term maintenance of public trust is internal consistency in the risk-management positions and standards that are adopted.

Unfortunately, a number of inconsistencies appear to have developed in the approach that EPA has adopted toward uranium mill tailings. This appears to have resulted from the fact that, as discussed above, no clear decision about risk-management strategy and decision criteria was made. Different criteria have been implicitly adopted at different stages in the EPA analysis. The results are thus not entirely internally or externally consistent. Two examples illustrate the problem. The first involves the contradictions that result from adopting a "one-time" thousand-year solution. The problem will be around for tens of thousands of years, but the available science indicates that solutions without regular surveillance and occasional active intervention may not yield reliable results over periods as short as decades. The second involves the contradictions between the radon emission levels that result from regulations designed to control mill tailings piles and those radon levels recently found acceptable by the EPA for some sources under Section 112 of the Clean Air Act (U.S. Environmental Protection Agency 1983d, 1984). To the extent that the EPA does not adopt a standard approach to all decisions that it makes involving public exposures to radioactive materials (as for

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example the International Commission on Radiation Protection has historically attempted to do), or explain explicitly the rationale for adopting different approaches in different contexts, then similar inconsistencies can be expected to develop elsewhere.

Role of Analysis in Environmental Risk Management

The nominal objective of risk analysis and other quantitative policy analysis is usually to provide insight into, and improved understanding of, a problem. However, the actual motivations are frequently quite different (Morgan, 1984). This is particularly true when analysis is performed by environmental regulatory organizations faced with the task of developing and promulgating scientifically based risk-management regulations on a problem that is complex and involves considerable uncertainty.

Most legislation authorizing risk-management activities in the United States requires that standards be based on a clear scientific rational. Since the science in such cases is often incomplete, and considerable uncertainty is involved, problems frequently result.

Whatever the decision criteria, there often comes a time when the science is unable to provide guidance on how to proceed. At some point the duly constituted social decision maker must make a judgment call and must say "I think prudence dictates that we do the following."

A number of forces now operating in our society make it difficult for decision makers to be explicit about such judgment calls. In such circumstances, either knowingly or unknowingly, detailed technical models are sometimes developed and used to justify as scientific what is in fact a judgment call.

There is considerable scientific uncertainty associated with the uranium mill tailings problem. However, the discussion in Chapter 3 has made it clear that uncertainty about the health consequences of chronic radon exposures at ambient levels are certainly known to within a factor of 10 and probably to within a factor of 2. The discussions in Chapters 3 and 4 indicate that, while exposures received by persons living in the vicinity of many specific piles are now poorly known and commonly cannot be modeled to within roughly a factor of 10, a simple and inexpensive measurement program could provide reasonable estimates of local population exposures.

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Discussions in previous sections of this report have shown that the analysis advanced by the EPA in support of its uranium mill tailings regulations (U.S. Environmental Protection Agency, 1983a, 1983c) makes a number of fairly arbitrary assumptions without fully exploring their implications and without examining the consequences of making at least equally plausible alternative assumptions or using alternative model formulations that might also be justified. Whatever the actual motivations, there is the outward appearance in this case of analysis having been undertaken primarily "to get an answer"—that is, as a substitute for making an explicit judgment call—rather than as a vehicle to provide general insight on the uranium mill tailings problem.

The EPA appears to believe that a variety of uncertainties associated with the uranium mill tailing problem preclude the selection of a risk-management strategy based solely on scientific considerations. In other words, any decision on how to manage the risks of uranium mill tailings piles will require a judgment call that, while the science is considered, will have to go beyond the science in order to choose a socially prudent course of action. On the basis of its review of the available science, the panel shares this view. It recognizes that the EPA is the socially authorized decision maker that should make this judgment. However, the panel believes that the scientific assessment and the social judgment should be separated and made explicit—rather than be combined. Such a separation has several advantages:

1. Quantitative risk analysis does not have to produce "the answer" and thus can be freely used to explore systematically the implications of the uncertainties, of alternative model formulations, and of alternative risk-management strategies. Such an exploration may suggest important aspects of the problem that might otherwise receive too little attention, or may even suggest risk-management opportunities that had not previously been apparent.

2. The risk-management alternatives among which a choice must be made are more likely to be explicitly enumerated, and the value judgments inherent in the choice are more likely to be identified. This systematic consideration may in turn lead to a socially better decision, since masking these choices and judgments in the guise of scientific assessment may preclude their receiving the attention that they deserve.

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3. Disagreements about matters of value are more likely to be explicitly identified and less likely to be masked in arguments about fact. When scientific-looking models are used to mask judgment calls, arguments about values masquerading as arguments about science are almost inevitable.

CONCLUSIONS AND RECOMMENDATIONS

On the basis of the preceding discussion, the panel reaches the following conclusions and makes the following recommendations:

1. Viewed in the perspective of the wide variety of risks that face U.S. society, simple order-of-magnitude arguments and comparisons suggest that the health risks posed by exposure to radon from uranium mill tailings piles are trivial for the average U.S. citizen and range from small to modest for most persons who live close to uncontrolled piles. However, if persons were to live right at the edge of a few uncontrolled piles that involve particularly unfavorable exposure conditions, their risk could be significant.*

2. While the potential risks from uranium mill tailing piles involve both airborne radiological hazards and radiological and nonradiological hazards from groundwater contamination, the former have received more attention in most quantitative risk assessments that have been done in the United States.

3. Risk assessments of uranium mill tailings that have been undertaken in the United States have not made adequate use of modern techniques for the characterization and treatment of the substantial scientific uncertainties that are involved.

4. The controls now being implemented for inactive piles will be more than sufficient to manage the risks that they may pose for the next few decades. However, further risk analysis is needed to understand how best to manage the risks of piles not now classified as inactive and to understand the requirement for ongoing observation and corrective intervention on piles now being controlled.

* For this case, the incremental lung-cancer risk per 70-year lifetime is of the order of 0.06, or on an annual basis roughly 8×10^{-4}. For purposes of comparison, this is roughly two-thirds of the individual lung-cancer risk of smoking, or three times the per capita risk of motor vehicle accidents.

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Recommendation: Several model quantitative risk analyses should be undertaken that:

- Explore the problem of how best to control piles not included in the current control program,
- Explore questions related to corrective intervention for piles included in the current control program.

These analyses should be pile specific. They should place attention on groundwater as well as airborne radiological hazards. They should make full use of modern techniques for the characterization and analysis of scientific uncertainty. They should evaluate the costs, risks, and benefits of alternative control strategies. At least one of the studies should address the risks involved in moving an uncontrolled pile.

5. A number of alternative, and philosophically very different, decision criteria for risk-management decision making are possible. The choice of the decision criteria to be used in a risk-management decision is a value based on choice not a scientific question. In making risk-management decisions, and in performing analysis in support of such decisions, it is wise to be explicit about the decision criteria that will be used.

Recommendation: In future regulatory reviews of uranium mill tailing risk management, the U.S. Environmental Protection Agency and the U.S. Nuclear Regulatory Commission should be explicit about the decision criteria that they choose to apply.

Recommendation: If regulation is to be based on the science of the problem, generic models must be validated for actual conditions.

6. Given the conclusions reached in Chapters 2 and 4, it is clear that a tailings risk-management strategy based on "adaptive look-ahead" (that is, a continuing modest program of observation and corrective intervention when it is required) is more consistent with the available scientific understanding of uranium mill tailings piles than is a strategy that seeks a "one-time" solution.

Recommendation: In future regulatory reviews of uranium mill tailing risk-management, the U.S. Environmental Protection Agency and the U.S. Nuclear Regulatory

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Commission should address the need for ongoing observation and corrective intervention that the science of the problem indicates will be necessary.

7. Public trust in risk-management organizations is an essential ingredient for their long-term success. One of the prerequisites of public trust in the long run is consistency in regulations and in the analysis and other arguments used to support them. In a variety of ways, the standards developed by EPA for risk-management of uranium mill tailings, and the analyses and other arguments that have been developed to support these standards, appear to display scientific and philosophical internal inconsistency and inconsistency with other EPA regulatory decisions.

Recommendation: In its future regulatory reviews of uranium mill tailing risk management, the U.S. Environmental Protection Agency should strive to achieve greater internal consistency in its approach to this problem and greater consistency between the way in which it deals with risks from tailings and the way in which it has dealt with other similar risks.

8. Most legislation authorizing risk-management in the United States requires that decisions be based on a clear scientific rationale. However, incomplete understanding and scientific uncertainty often require the decision maker to make "judgment calls." For a variety of reasons there may be pressures to mask such judgment calls by burying them inside a large quantitative risk analysis. There are good scientific and philosophical reasons why this practice should be avoided and why the implications of alternative possible assumptions and approaches should be systematically explored.

Recommendation: In future regulatory reviews of uranium mill tailing risk management, the U.S. Environmental Protection Agency and the U.S. Nuclear Regulatory Commission should work to separate and identify the "judgment calls" explicitly, to identify their implications clearly, and to justify the choice compared with other possible alternatives.

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Appendix A
Visits

Date	*Attendance*	*Installation or Event*
21 May	Panel	Homestake Mining Mill, Grants, New Mexico
21 May	Panel	Anaconda Tailings Pile, Grants, New Mexico
22 May	Panel	Kerr McGee Quivera Mine, Grants Area
22 May	Panel	Kerr McGee Tailings Pile, Grants Area
22 May	Panel	Philips Tailings Pile, Grants Area
Late May 84	G. Morgan	Abandoned Tailings Pile, Mexican Hat, Utah
Late May 84	G. Morgan	Energy Fuels Nuclear Facility, Inc. Blanding, Utah*
Late May 84	G. Morgan	Atlas Minerals Site, Moab, Utah*
Late May 84	G. Morgan	Abandoned Tailings Pile, Grand Junction, Colorado
Late May 84	G. Morgan	Abandoned Tailings Pile, Gunnison, Colorado
4 Sept 84	H. James	Dawn Mining Company, Ford, Washington
20 Feb 85	S. Wiltshire	Atlas Minerals Site, Moab, Utah*

*Inspection performed at site boundary.

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Appendix B
Presentations to Panel

Person	*Organization*	*Date*	*Subject*
D. G. LeClaire	DOE	8 Mar. 84	DOE Defense Waste Program
A. F. Kluk	DOE	8 Mar. 84	DOE Commingled Mill Tailings Study
C. Welty, Jr.	DOE	8 Mar. 84	Uranium Mill Tailings Standards
G. Burley	EPA	8 Mar. 84	EPA Rule-making Process
D. Sollenberger	U.S. NRC	8 Mar. 84	NRC Licensing Process
W. Mills	U.S. NRC	8 Mar. 84	Health Concerns
L. Swent	American Mining Congress (AMC)	8 Mar. 84	Miner Health Effects
E. Still	AMC/Kerr McGee	8 Mar. 84	Public Risks Associated with UMT
R. Beverly	AMC/Umetco	21 May 84	Welcome
G. Swanquist	Homestake	21 May 84	Uranium Milling Process
L. Boggs	AMC	21 May 84	Reaction to EPA, U.S. NRC, DOE Requirements
E. Still	Kerr-McGee	21 May 84	Radon Emissions
L. Cook	Chevron	21 May 84	Particulate Emissions
E. Kennedy	Homestake	21 May 84	Effects on Water Resources
J. Velasques	United Nuclear	21 May 84	Effects on Surface Water
R. Beverly	AMC/Umetco	21 May 84	Tailings Pile Stabilization
J. Cleveland	Kerr-McGee/N.M. Radiation Technical Advisory Council	21 May 84	Related New Mexico Public Health/ Safety Regulations
T. Vogt	Mobil	21 May 84	In-situ Uranium Leaching
P. Jackson	PNL	19 July 84	Radon Measurements

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M. Momeni	San Diego State Univ.	19 July 84	Radon Measurements
J. Hartley	PNL	19 July 84	DOE/UMTRAP Program
M. Foley	PNL	19 July 84	Longer Term Stabilisation
G. Sehmel	PNL	19 July 84	Windblown Particle Study
R. Nelson	PNL	19 July 84	Leachate Seepage
D. Brookins	Univ. of New Mexico	20 Sept. 84	Mineralogy and Chemistry of of Sandstone Type Uranium Ores
F. Cross	PNL	20 Sept. 84	Radon Effects in Animals
W. Kellogg	NCAR	20 Sept. 84	Long-Term Climatic Effects
S. Wells	Univ. New Mexico	20 Sept. 84	Long-Term Geomorphic Effects
V. Rogers	Rogers & Assoc. Eng. Corp	21 Sept. 84	Processing/ Containment Alternatives
D. Dreesen	(formerly ANL)	21 Sept. 84	Thermal Stabilisation of Tailings
T. Tamura	ORNL	21 Sept. 84	In-situ Grouting of Tails
I. Nirdosh	Lakehead Univ.	21 Sept. 84	Processing Alternatives
R. Guimond	EPA	3 Dec. 84	Withdrawal of Proposed Standards (40 CFR 61)
C. Welty, Jr. J. Thiessen	DOE	3 Dec. 84	DOE Views on Withdrawn Standard
D. Chambers	SENES Consultants	4 Dec. 84	UMT Probabilistic Assessments
P. Longmire	Environmental Improvement Division State of New Mexico	4 Dec. 84	Effects of Using UMT as Backfill
L. Stieff	Stieff Research and Development Co.	25 Feb. 85	Radionuclide Distribution Measurement Techniques
E. Landa	USGS	22 Apr. 85	Radionuclide Mobilisation Mechanisms
E. LeTourneau	Dept. of Health and Welfare, Canada	22 Apr. 85	Canadian Radon Surveys
A. Kluk	DOE	22 Apr. 85	DOE Litigation
D. Nikoden	DOE	10 Sep. 85	DOE Forecasting of Uranium Production

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T. Chung	DOE	10 Sep. 85	DOE Forecasting of Uranium Production
M. Mlay	EPA	10 Sep. 85	EPA Groundwater Regulation

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Appendix C
Key Radiation Units

	S.I.	Alternative
Activity	becquerel (Bq)	curie (Ci)
Absorbed Dose	gray (Gy)	rad
Dose Equivalent	sievert (Sv)	rem

Bq	becquerel	SI unit of radioactivity 1 Bq = 1 disintegration per second (dps)
Ci	curie	Alternate unit of radioactivity 1 Ci = 3.7×10^{10} dps
Gy	gray	SI unit of absorbed dose 1 Gy = 1 joule/kilogram = 100 rad
rad	rad	Alternate unit of absorbed dose 1 rad = 100 ergs/gram
rem	rem	Alternate unit of dose equivalent 1 rem = 1 rad $\times$ Q, where Q = the quality factor, 20 for alpha irradiation
Sv	sievert	SI unit of dose equivalent 1 sievert = 100 rem
WL	working level of radiation exposure	1 WL = 1.3×10^5 MeV of potential alpha energy per liter of air for radon daughter products = 100 pCi of radon per liter of air
WLM	Working Level Month	Exposure at 1 WL for 170 hours 1 WLM = 0.7 rad = 14 rem

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Appendix D
Operation of Conventional Uranium Milling Processes

Acid Leaching

In the acid leaching method, U^{4+} is first oxidized to U^{6+} by an oxidant such as MnO_2 or $NaClO_3$ in conjunction with reactions involving iron as shown below. In practice, sulfuric acid and elemental iron, the oxidant, are agitated with a slurry of uranium-rich solids. Some relevant reactions are

$$U^{4+} + MnO_2 \rightleftharpoons UO_2^{2+} + Mn^{2+}$$

or

$$3H_2O + 3U^{4+} + ClO_3^- \rightleftharpoons Cl^- + 3UO_2^{2+} + 6H^+$$

$$Fe + H_2SO_4 \rightleftharpoons FeSO_4 + H_2$$

$$FeSO_4 \rightleftharpoons Fe^{2+} + SO_4^{2-}$$

The Fe^{2+} reacts with MnO_2:

$$2Fe^{2+} + MnO_2 + 4H^+ \rightleftharpoons 2Fe^{3+} + Mn^{2+} + 2H_2O$$

or with $NaClO_3$:

$$6Fe^{2+} + NaClO_3 + 6H^+ \rightleftharpoons 6Fe^{3+} + NaCl + 3H_2O$$

and the following reactions occur simultaneously:

$$UO_2^{2+} + 2SO_4^{2-} \rightleftharpoons UO_2(SO_4)_2^{2-}$$

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$$UO_2(SO_4)^{2-} + SO_4^{2-} \rightleftharpoons UO_2(SO_4)_3^{4-}$$

The uranium in solution thus consists of a mix of UO_2^{2+}, $UO_2(SO_4)_2^{2-}$, and $UO_2(SO_4)_3^{4-}$ after the leach. Usually, excess sulfate ion is added to convert all UO_2^{2+} to one of the uranyl sulfate oxyanions. The acid-leach method is most commonly used for uranium ores with a low carbonate content, such as many of the ores from the Grants, New Mexico, Mineral Belt. Some of the advantages of the acid-leach method are

- Uranium is readily extracted;
- Much of the sulfuric acid can be reused; and
- Some oxidant (e.g., MnO_2) is already present in the ore, thus facilitating oxidation.

Some of the disadvantages are

- The sulfuric acid can damage milling equipment without proper precautions;
- High-calcite-content ore cannot readily be processed by acid leach; and
- The very low pH of the sulfuric acid leach solution allows radium (as Ra^{2+}) to be put into solution, which is of extreme environmental concern.

Alkaline Leaching

In alkaline leaching, after the initial crushing of the ore, the oxidation of U^{4+} to U^{6+} is carried out in mild steel vessels, under pressure, using oxygen from the air as an oxidant in the presence of sodium carbonate:

$$2UO_2 + O_2 + 6Na_2CO_3 + 2H_2O \rightleftharpoons 2Na_4UO_2(CO_3)_3 + 4NaOH$$

but this reaction is not satisfactory as the NaOH will attack the uranyl carbonate complex; therefore $NaHCO_3$ is added so that

$$2UO_2O_2 + 2Na_2CO_3 + 4NaHCO \rightleftharpoons 2Na_4UO_2(CO_3)_3 + 2H_2O$$

The leachate is filtered to remove solids and other foreign matter, and the uranium-rich solution (the pregnant liquor) and

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sent to a product-recovery vessel. The following reaction then results in the formation of a first-step yellowcake:

$$2Na_4UO_2(CO_3)_3 + NaOH \rightleftharpoons Na_2U_2O_7 + 6Na_2CO_3 + 3H_2O$$

The $Na_2U_2O_7$ (yellowcake) at this step contains about 75 percent U_3O_8 and 5 to 6 percent V_2O_5 and 2 to 3 percent carbonate. The impure yellowcake is roasted at this point to produce a calcined yellowcake, which, when rinsed with water, will be purged of its contaminant vanadium and carbonate. The vanadium is stored and sold as a by-product. The sodium content of the yellowcake is lowered by washing with sulfuric acid-ammonium sulfate, which yields a purified yellowcake with only 0.5 percent sodium. The resultant yellowcake contains about 87 percent U_3O_8. This is washed, dried, and packaged.

The advantages of the alkaline leaching method are

- Oxidation costs are lowered because O_2 is the oxidant;
- The high calcite content of the ore facilitates dissolution;
- Radium loss is low because it is not soluble in the alkaline leach; and
- Resin is not involved.

The disadvantages are

- The crushing costs are greater because the ore must be very finely ground; and
- For low-calcite ores, the alkaline leach does not remove uranium as thoroughly as acid leach.

Agitation leaching processes are carried out in cylindrical vessels with air or mechanical agitation. A pulp is made from the wetted, finely ground ore. This method is used for continuous processing with a high yield in a relatively short retention time, lower chemical consumptions (relative to percolation leaching), and little uranium loss from fine material. This method has higher costs, higher water consumption, low-grade pregnant solutions, and a liquid-solid separation process has to be used.

In percolation leaching, a coarser feed is used, which yields a high-grade pregnant solution. Vats with false bottoms are used in which the pregnant solution is filtered through various materials. This method has lower capital costs, lower amounts of solutions,

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low water consumption, higher-grade pregnant solutions, and delivery of filtered pregnant solutions. Percolation leaching is not suited for fine materials, and it has a longer retention time and a higher reagent consumption.

Complex ores are composites of ores of different types, which necessitates use of both acid- and alkaline-leaching techniques. While the different ores can be separated from one another, this is time consuming and expensive. The alternative taken is to mix high-calcite ore with a larger volume of low-calcite ore, followed by acid leaching.

For solution mining of complex ores, an alkaline leach is first used, followed by an acid leach for a shorter time. The alkaline leach is used first because if removes fewer metals from the ore and gangue than acid leach, thus ensuring a longer operation. When the ore has an overall low-calcite content, the alkaline leach will be followed by a short period of acid leaching to get the alkaline-resistant uranium. Pretesting of the ore is done to determine the amount of acid leach to use so as not to release too much radium to the host aquifer.

Recovery Processes

Uranium is stripped from the pregnant liquor by any of several exchange processes. The most common methods are the resin-in-pulp (RIP) method, the solvent extraction method, and the general ion-exchange process. In each method, advantage is taken of the fact that UO_2^{2+} or other U-oxyanions can be exchanged between sites. If R is an organic fraction, and X the mobile ion, then a general reaction for exchange is

$$4RX + [UO_2(SO_4)_3]^{4-} \rightleftharpoons R_4UO_2(SO_4)_3 + 4X^-$$

Loading refers to the reaction going to the right; *stripping* refers to the reaction going to the left. The pregnant solutions are continuously loaded and stripped in recovery tanks until the uranium is concentrated sufficiently for final recovery.

In the ion-exchange processes, the uranium sulfate complexes have a high affinity for certain anion exchange resins. If R is the resin, the following reaction is typical:

$$4RCl + UO_2(SO_4)_3^{4-} \rightleftharpoons R_4UO_2(SO_4)_3 + 4Cl^-$$

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The resins are commonly made of ammonium anion salts attached to a cross-linked styrene-divinylbenzene copolymers backbone (see discussion in Brookins et al., 1981). While effective, this resin, because of the simplicity of its oxidation, releases nitrogen to the mill tailings, which produce nitrates, which move readily in the surface environment.

In the acid leach, the pH is kept below 1.9 so that bisulfate ion (HSO_4^-) is dominant over sulfate ion (SO_4^{2-}). This is done because HSO_4^- competes more strongly for uranium complexes than will SO_4^{2-}. When uranium is saturated on the resin, it can be removed by acid nitrate or chloride ions. Uranium is then precipitated as an uranate with ammonia, sodium hydroxide, or magnesia.

Sands and slimes must be separated from the pregnant leach solution during standard extraction. The RIP method was developed by the U.S. Atomic Energy Commission in the 1950s to speed up operations. Sands are easily separated from uranium-rich solutions, although slimes are not. The RIP circuit contains parallel banks of vats connected in series. Each vat is filled with baskets of a strong anion exchange resin. The baskets rise and fall in the vats, and the pregnant liquor penetrates the resin, resulting in uranium entrapment. Slime coating, which would prevent uranium entrapment, is prevented by constant agitation of the vats. Both pH and Eh are controlled. Melanterite ($FeSO_4 \cdot 7H_2O$) is used for Eh control to prevent removal of vanadium and other cations, except uranium. This allows uranium to be separated as an oxyion $[UO_2(SO_4)_3]^{4-}$. The pregnant liquor cascades from one vat to the next by gravity feed. In a normal 14-vat circuit, 10 tanks are used for loading (by adsorption) and 4 for stripping. When one tank is loaded, it is then stripped and an eluted tank is used for loading. This continues until, from the last vat, a solution rich in uranyl sulfates, sulfate, and ferric cations emerges.

Precipitation is carried out by two steps to purify and clarify the pregnant eluate. First, ferric ions are removed by

$$Fe_2(SO_4)_3 + H_2SO_4 + 4Ca(OH)_2 + 6H_2O \rightleftharpoons 2Fe(OH)_3 + 4CaSO_4 \cdot 2H_2O$$

and the gypsum produced is sold. The solution left is filtered, and magnesia is added as a source of OH^- ions to neutralize any acid left and to precipitate uranium as yellowcake.

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In solvent extraction, organic extractants are used to recover uranium from acid-leach liquors. A phosphate compound such as alkyl orthophosphoric acid or alkyl pyrophosphoric acid can remove uranium by cation exchange by (where R is the alkyl group):

$$UO_2^{2+} + 2R_2HPO_4 \rightleftharpoons UO_2(R_2PO_4)_2 + 2H^+$$

To reverse this reaction, and to strip the uranium from the organic phase, hydrochloric acid is commonly used, after which the solution is evaporated to recover the HCl. The concentrated uranyl chloride is then diluted and precipitated as ammonium diuranate, which is then converted to U_3O_8. Sodium carbonates can also be used to strip the uranium from the organic phase by

$$UO_2(R_2PO_4)_2 + 3Na_2CO_3 \rightleftharpoons 2NaR_2PO_4 + Na_4UO_2(CO_3)_3$$

After first destroying the sodium uranyl carbonate by sulfuric acid, the uranium can be precipitated by addition of ammonia.

Other Stripping Methods

Uranium extraction from acid-leach liquors can also be achieved by addition of organonitrogen compounds:

$$2(R_3HN)_2(SO_4) + UO_2(SO_4)_3^{4-} \rightleftharpoons (R_3HN)_4UO_2(SO_4)_2 + 2SO_4^{2-}$$

Uranium extraction from the organic phase can be achieved by (1) addition of acidic nitrate or chloride solution, (2) sodium carbonate solution striping, or (3) magnesium oxide stripping.

Recovery of Uranium from Phosphates

All but one of the uranium mills in the United States are located in the western United States. The remaining mill, where uranium is removed from marine evaporites as a by-product of phosphate milling, is located in Florida. Uranium as U^{4+} can substitute easily into the Ca^{2+} site of apatite, $Ca_5(PO_4)_3(F,OH)$, and must be removed prior to use as a phosphate fertilizer and from waste solutions released into the biosphere.

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When uranium-bearing apatite is treated with sulfuric acid for phosphate recovery, the following reaction takes place:

$$Ca_{3-x}(UO_2)_x(PO)_2 + 3H_2SO_4 \rightleftharpoons xUO_2SO_4 + 2H_3PO_4 + (3-x)CaSO_4$$

The uranyl sulfate remains in solution with the phosphoric acid, from which it can be readily removed by standard hydrometallurgical methods. The uranium recovered is not of interest economically, yet thorough uranium recovery must be carried out for environmental protection.

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Appendix E
Report on Geologic Diversity, Geomorphic Processes and Time Scales of Geomorphic Processes Related to Uranium Tailings Disposal Site Integrity

Submitted to:
Laurence Kulp, Chairman
Board on Radioactive Waste Management
National Research Council

by
Stephen G. Wells
Department of Geology
University of New Mexico
Albuquerque, New Mexico

September 5, 1985

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Geologic Diversity of Uranium Tailings Disposal Sites: Applications to Geomorphic Hazards and Tailings Stability

The geologic setting of uranium tailings disposal site is the summation of past geologic processes operating in that region. Thus, the character of the landscape and processes operating on that landscape are inherited from the geologic past. The dependence of geomorphic processes on the bedrock geology of a given site increases, in general, with the longer time spans and larger areas considered (Figure 1). The discharge of a single flood event and the sediment load that is carried during the event is primarily influenced by drainage networks and hillslope morphology. However, the development of drainage networks and hillslope forms is strong controlled by the underlying bedrock as well as climate and time (Figure 1). The geomorphic processes which affect the stability and integrity of uranium tailings piles and the relation of these processes to bedrock geology will change with temporal and spatial scales (Figure 2).

The majority of uranium production in the United States occurs in the Colorado Plateau physiographic province, which covers approximately 340,000 km^2 (Figure 3). This region displays similarities in the ages of bedrock units, types of structural features, tectonic history, and general geomorphic history. However, when considering smaller scale regions, significant differences appear in the geologic setting, which impact short-term geomorphic processes, such as flooding. For example, the local diversity of geology and its influence on erosion rates, infiltration rates, and sediment yields is illustrated for a small section of the Colorado Plateau in New Mexico (Figure 4, Table 1). Areas adjacent to each other in the area shown in Figure 4 are characterized by significantly different processes; one region where infiltration dominates runoff may be stable whereas another area may be dominated by runoff and be unstable. These differences reflect the differing geologic history in very localized areas (Wells and Jercinovic, 1983).

The influence of bedrock geology on tailings stability must be evaluated as an external variable (Figure 2). That is, the stability of a reclaimed tailings pile is affected by two types of variables:

1. internal - which are related to surface and subsurface conditions of the reclaimed area only, and

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2. external - which are related to all conditions outside the reclaimed area.

In general, internal variables are those governed by engineering practices during the design of reclaimed lands. Long-term geomorphic processes such as base-level change and valley aggradation are determined by the geologic setting, and these processes are external driving forces which may impact the integrity of the tailings pile.

The role of geologic diversity on geomorphic processes has been evaluated for two basic types of lithologies: sandstones and shales. Differences in the relative occurrence and spatial distribution of these two rock types may have a dramatic influence on geomorphic processes operating over different time scales. For example, bedrock lithology within a given drainage basin determines the textural parameters of material entering the fluvial system and being stored in the valley floors. Differences in the grain size of valley sediments affect the threshold for gully incision and valley floor instability. Channels developed on valley flows composed of fine-grained sediments have a lower threshold for gully incision than those channels developed in coarser-grained sediments (Figure 5). This is a result of the relative amount of infiltration and runoff on the valley floors which is determined, in large part, by the texture of the sediment (Wells and Gardner, 1985). Long-term geomorphic processes are also strongly dependent on the bedrock geology. Listed below are the major qualitative influences of bedrock geology on long-term geomorphic processes:

1. response of drainages to base-level lowering; shale-rich areas are easily eroded and are able to transmit base-level changes rapidly throughout the watershed. Sandstones retard the transmission of base-level changes throughout a watershed;
2. resistant sandstone units support older landscapes: because sandstone retards base-level changes, older surficial deposits are maintained;
3. bedrock influences the valley configuration in terms of widening and scouring;
4. bedrock influences the type and amount of sediment supplied to fluvial systems and, therefore affects the fluvial system morphology and processes.

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These differences can be observed in the various uranium mining regions of northwestern New Mexico (Figure 6). Watersheds in the northern portions of the Crown Point district typically have less relief throughout the watersheds due to the predominance of shale and its higher erodibility. Watersheds in the Ambrosia Lake district have higher local relief due to the predominance of resistant sandstone and volcanic-capped mesas. Relief of a watershed significantly affects the hydrologic behavior of floods.

In summary, the relative role of geologic diversity of tailings sites to the site stability is governed by the relationship or dependence of geomorphic processes to the bedrock geology. Effective management of uranium tailings disposal sites should include evaluation of the dependence of geomorphic processes and bedrock geology.

Natural Geomorphic Processes Affecting the Stability and Integrity of Uranium Tailings Disposal Sites

Those geomorphic processes and conditions which influence the stability of tailings disposal sites in the western United States have been listed by Schumm and others (1982) and Longmire and others (1981). Of the 29 types of geomorphic hazards which affect the stability of tailings sites, Schumm and others cite that most are related to fluvial processes such as drainage network rejuvenation (downcutting and expansion), valley aggradation, and channel pattern adjustments (meander growth). In that a majority of tailings sites in major uranium processing regions, such as New Mexico, are located within valley floors or immediately adjacent to them (Table 2), clearly the determination of types and rates of fluvial processes is imperative in effective management of tailings disposal site stability (Table 3). Those processes which potentially affect the tailings site stability may be classified according to the source of their driving forces (Wells and Jercinovic, 1983). Those forces that are sourced from areas or conditions outside the immediate site are external, and those sourced primarily on the reclaimed site are internal (Figure 2).

Because types and rates of geomorphic processes vary throughout the western United States and vary with site geology, hydrology, and geomorphic history, field data and field-based studies are required as a calibration of theoretical models and data. The lack of field-based studies with respect to geomorphic hazards poses

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one of the greatest problems to effective management of tailings sites. Described below are some of the major natural processes that potentially affect tailings integrity, their relative rates of operation, problems in assessing their magnitudes and frequencies, and selected methods used in evaluating process effectiveness on landscape stability.

Flooding Hazards

Long-term (tens of years) hydrologic data (e.g., gaging station records) are required for an adequate evaluation of short-term flood potentials on any given river system. The majority of tailings sites are located on small tributary systems to much larger rivers. The larger rivers are often gaged with hydrologic stations, but the smaller ones rarely have gages and long-term records, especially ephemeral channels. Therefore most flood designs are based upon the probable maximum flood (PMF) or 100- and 200-year flood events. These designs are statistically based on short-term rainfall and runoff records, which are typically extrapolated from other regions. Schumm and others (1982) point out that "on numerous occasions PMFs have been exceeded, and recurrence intervals for PMFs vary at least four orders of magnitude." This problem is a function of the short and spatially limited hydrologic record on western drainage basins.

Flood-frequency curve analyses serve as a major method in estimating the magnitude and frequency of flood events (Dunne and Leopold, 1978), and an example of this method is given in Figure 7A. A common technique is to extend the frequency curve beyond the period of record to evaluate the probability of larger flood events (Figure 7B). However, this extrapolation is based on the assumption that all flood events are part of the same hydrologic population with large variance and covariance (Costa and Baker, 1981). However, it is clear that many gaging records contain mixed populations with some flood events different from the population of typical annual peaks (Figure 7C). Some hydrologic events may be related to rainfall or snowmelt; whereas, other events may be classified as outliers and reflect catastrophic conditions atypical of mean annual conditions.

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Patton and Baker (1977) illustrate the dilemma related to effectively evaluating large, potentially hazardous floods by extrapolating flood-frequency curves (Figure 8). In 1954, an extratropical disturbance resulting from Hurricane Alice in the Gulf of Mexico moved into the Pecos River watershed and produced a peak discharge nearly eight times any previously recorded maximum discharge. Extrapolating a curve based on 39 years of record and two methods of curve extrapolation would indicate that the 1954 flood had a probability equivalent to a recurrence interval of either 50 years or one million years (curves A and C, respectively, Figure 8). However, Patton and Baker used techniques of paleohydrologic analyses involving a longer-term record from Holocene stratigraphic record and radiocarbon dating to determine that such a flood event had a recurrence interval of several thousand years. Paleohydrologic analyses of Holocene alluvial stratigraphy poses one of the greatest tools to determining realistic long-term flood records, and these methods are summarized by Costa and Baker (1981). However, few, if any, such studies have been applied to effective evaluation of flood potentials in areas of uranium-tailings disposal sites.

Floods derived from small tributary watersheds are estimated from statistically based predictions, and applications of engineering structures based on such predictions can be misleading and result in higher risks. An example of such a problem is given by Jercinovic (1984) for a small watershed near the Church Rock district (Figure 6). A September 1980 storm event that occurred in a watershed <0.5 km^2 was determined to be smaller than a 2-year, 24-hour event. Settling ponds were designed to collect runoff generated from storms of a 10-year, 24-hour magnitude, which was inferred from empirically and statistically based rainfall-runoff equations. During the much smaller 1980 storm event, the settling ponds were nearly filled to capacity by flood waters and nearly overtopped; thus, significantly smaller events than the one designed for by statistically based equations nearly caused failure of an engineered structure.

Eichert (1970) stated that computer programs used to determine water-profile calculations and flood stages pose difficulties because the programs do not include longer-term scour and fill processes in stream channels (see discussion below). Such computer methods have been used in the calculation of flood potential and erosion potential of cultural features in Chaco Culture National

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Historic Park (Lagasse and others, 1983). This study provides a detailed example of a method used in the evaluation of a 100-year storm on a major ephemeral channel directly north of the Church Rock district (Figures 6, 9; Table 4). Although such a study presents state-of-the-art engineering practices to an ephemeral arroyo complex, the method does not account for the significant changes in channel pattern (braided to meandering) and vertical position, which has occurred over the past 80 years (Love, 1983). These changes would result in misleading calculations of flood risks unless the analysis is undertaken with each major change in channel conditions over decades.

Similar problems using methods described above can be applied to unincised valley floors, which are common in the western U.S. engineering structures designed to protect tailings from PMGs, or 200-year flood events may lose their effectiveness over time due to valley-floor aggradation (Wells and Gardner, 1985). For example, a flood-protection berm was designed for a tailings site, which included a 1.2-m free board above the 200-year event. The valley floor aggradation rate in this region is approximately 0.005 m/yr. If this rate continues over the next 200 years, the valley floor could be within 0.2 m of the flood berm crest, rendering the flood-protection device useless.

In summary, the effectiveness of tailings management strategies related to the evaluation of flood risks on main-stem rivers and tributaries must involve a consideration of:

1. dynamic geomorphic processes that change with time resulting in significant changes in positions of river
2. inadequacy of short-term hydrologic records and their extrapolation via flood frequency curves to estimate flood risks;
3. new techniques in paleohydrologic analyses involving Holocene valley stratigraphy and radiocarbon dating; and
4. increased field-based studies in the analyses of floodingand its consequences.

Changes in Channel Pattern and Lateral Position

Schumm and others (1982) list the following geomorphic processes as hazards to uranium-tailings site stability:

1. channel erosion including stream bed lowering, nickpoint migration, channel widening and meander shifting;

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2. channel deposition including raising of a stream bed by raising, deposits that migrate down-channel, and narrowing of channel by deposition; and
3. pattern changes including meander growth, meander cutoffs, and island-bar shifting in a braided channel.

These processes are dynamic on ephemeral channels in the western United States as well as on perennial streams. For example, the Rio Puerco (of the east) drains large areas in which uranium milling and storage occur and displays all the processes listed above (Figure 10). Wells and others (1983) have demonstrated the dynamic nature of large ephemeral arroyos systems, which resulted in significant changes in channel pattern and position during the past 50 years (Figure 10A). Meander migration rates over this time period vary between 2 and 8 m/yr with the greatest periods of channel adjustments occurring during peak discharges exceeding 3,000 m^3/s (Figures 10B and 10C). During the highest flood discharges, the stream bed of the Rio Puerco was 3 to 6 m below its present level; this indicates that significant vertical as well as lateral changes occurred during large flood events. In addition, studies show that the Rio Puerco changed from a braided to a meandering river during the past 50 years as well.

Wells and Gardner (1985) demonstrated that processes of similar magnitude occurred in the smaller tributaries and proximal to present and planned uranium tailings sites. These magnitudes are summarized below:

Channel incision:	6.5-50 cm/yr
Channel bank erosion:	1.5-8.0 m/yr
Channel floor deposition:	4.0-30.0 cm/yr
Channel bank deposition:	1.5-6.0 m/yr
Pattern change-meander growth:	2.0-8.0 m/yr
-cutoff activity:	3/50 yrs
Metamorphosis:	change from braided to meandering

The major impact of these processes on tailings site stability is related to the proximity of tailings sites to these types of active channels. Some disposal sites, such as those in the Ambrosia Lake uranium mining area of New Mexico or those near Uravan, Colorado, are proximal to active stream channels. Especially sites such as the Jackpile mine of New Mexico, where tailings abut

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active ephemeral channels, may suffer the greatest impact from channel adjustments over time. However, what is clearly not understood at the present time is the magnitude of space over which channel meandering might occur. There are limits to the sizes of meanders at which time cutoffs or stabilized forms occur (Leopold and others, 1964).

As demonstrated above, channel adjustments on typical ephemeral streams occur during large flood events; however, significant channel adjustments occur in response to the amount of sediment delivered into a channel. An example of the magnitude of channel adjustments in response to a large sediment influx is reported by Miller (1985) for the Grants Wash area near the Ambrosia uranium mining area. In the 1930's tailings from a perlite mine were introduced into a discontinuous arroyo which drained approximately 12 km^2. This massive sediment influx nearly filled the arroyo and spilled out to form a fan on the valley floor where the arroyo became discontinuous (Figure 11). However, this sediment moved as a pulse through the arroyo during the past 40 years resulting in a flushing of the perlite tailings from the arroyo to further points downstream. The changes in the morphology of the arroyo channel during the passage of this sediment is given in Figure 12. Thus, it is important to note that 1) sediment influxes cause significant changes to channel forms, 2) the sediment influx may move as a pulse in the downstream direction, and 3) the amount of channel adjustments varies considerably over a channel reach. The impact of short-term filling of a channel is that its total cross-sectional area is significantly reduced, which may enhance local flooding as described above.

Gully Incision and Lowering of Local Base Level

Among the natural geomorphic processes most hazardous to the integrity and stability of tailings disposal sites are those related to drainage network rejuvenation (downcutting), channel incision, and nickpoint migration. This is a result of the magnitude of the rates at which these processes operate (Wells and Gardner, 1985). All these processes operate locally between 0.5 and > 10.0 m/year. Extensive channel and network erosion is typical of valley floor systems in the arid and semiarid western United States for the past 100 years. These unstable channel forms pose a continuing natural threat to the near-channel environment (Graf,

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1983). This situation results from the local relief produced by incision of a gully or arroyo which promotes continued headward migration of the drainage network until an equilibrium is achieved (Begin and others, 1981). Rapid erosion of valley floors resulting in unstable conditions have been attributed to three principle causative factors:

1. human-induced changes due to overgrazing and vegetation reduction,
2. climatic change causing variations in runoff and sediment load, and
3. intrinsic threshold conditions due to the natural evolution of a drainage network.

In the vast majority of settings in which gully incision has occurred or is occuring, a critical valley slope exists above which alluvial valley floors are entrenched. Commonly this results in the migration of the gully upstream through headcut migration, rejuvenating tributaries, and increasing the sediment load downstream. As a result, large deposition lobes occur in downstream areas which, in turn, increase the valley floor gradient. Thus, the cycle is repeated. This threshold has been defined in a variety of settings (Figure 13).

Recent studies (Wells and Gardner, 1985) have demonstrated that the rates of migration of headcuts and drainage network incision are strongly dependent upon soil and sediment properties, especially texture-infiltration relations. Increasing silt and clay content in sediments at the surface of valley floors enhances headward extension rates of gullies (Figure 14). Rates of headcut migration can vary significantly between valley floors as well as on a valley floor due to changes in the character of near-surface deposits of valley floors.In order to insure adequate stability of a disposal site and integrity of the tailings pile, detailed evaluations of gully incision potential, headcut migration rates, and general valley floor stability must be assessed. One major technique employed for site evaluation is the use of repetitive photographs (vertical air and terrestrial). For example, a rejuvenated drainage network and headcut migration of a tributary drainage near a proposed tailings site (Figure 15) has been estimated via air photographic analysis to have migrated nearly 30 m in less than 50 years (Gibbons and others, 1980). Detailed ground surveys show that gully headcut migration can be measured over shorter periods of time, such as

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a year (Figure 16). This suggests that headcut migration is not necessarily limited to certain periods of time but may be more continuous, especially where runoff is available through aquifer dewatering. Begin and Schumm (1979) suggest the following steps in order to evaluate the probability of a valley incision:

1. within a uniform drainage basin, measure the gradients of valley floors and drainage areas above them and record whether they are gullied or ungullied;
2. plot the points on log-log paper with valley gradient on the ordinate, and draw a straight line through the lowermost points representing gullied reaches;
3. determine the slope on the line (which = rf) and calculate the "shear-stress indicator (To) for each gullied valley point using: $To = k A^{rf} S$ (where k = constant, A = drainage area, S - slope).
4. choose the lowest value of To and this represents Tth, then for all data points calculate To/Tth and subdivide the data into classes according to these values;
5. determine the percentage of gullied valleys in each class; this percentage represents the probability of a valley incision for a given To/Tth value.

This empirical method can be compared with that technique discussed by Wells and Gardner (1985) which involves the use of Landsat 3 imagery to predict valley floor stability. With Landsat 3 imagery, distinct spectral characteristics of land-surface texture and infiltration properties can be classified. Those areas with greater sand content and higher permeabilities are more stable landscapes.

Less information is available concerning the stabilization of gully/headcut complexes than the prediction of their probability. Field observations of various methods to control these processes indicate that relatively simple inexpensive methods (headcuts filled with large concrete blocks) are not successful. Engineered structures, such as berms, designed to control headward migration are documented as failing (Wells and Gardner, 1985). For example, an earthen dam constructed between 1935 and 1965 was breached by a gully complex in 1977 in the vicinity of the Crownpoint uranium mining area. By 1981 the breach in the channel had been lowered 3 m and nearly 1 m over 0.5 km upstream (Figure 17).

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This process was probably enhanced by recharge from upstream mine-related aquifer dewatering.

Valley floors in arid and semiarid regions represent one of the most potentially unstable geomorphic settings. However, with the uranium tailings area of New Mexico, the vast majority of tailings piles are sited within this environment.

Piping

Piping is defined as the process of water percolating through the subsurface (vadose zone) resulting in the translocation of clastic particles and the development of tunnels and conduits. Over time, more and more particles are carried away resulting in a depression and vertical orifice feeding a complex subsurface drainage network which typically exits along the bank of a nearby arroyo. Piping can lead to the formation of gullies and tributaries which head on valley floors; thus, piping and gully development are closely related in valley floors. Several essential conditions for piping occur:

1. surface permeability exceeding a subsurface layer permeability;
2. existence of an erodable layer;
3. existence of a hydraulic head (gradient); and
4. outlet for the flow.

Materials subject to piping include alluvial and colluvium which contains clay, silt, and fine sand, montmorillonitic and bentonitic shale, and loess (Figure 18).

Individual pipes may range from a few millimeters to several meters in diameter. When piping becomes extensive, the surface collapses and a pipe/gully network develops. Rates of lateral extension of pipe/gully complexes have been measured in specific locations within the Rio Puerco drainage system near the Laguna-Paguate uranium mining region. These rates equal those geomorphic processes which display the highest magnitudes of rates (Wells and Gardner, 1985). Pipes may extend at an areal rate of 65-75 m^2/yr or up to 10 m/yr lateral growth.

Techniques suggested for piping control include:

1. application of gypsum and hydrated lime to reduce soil dispersion (this is highly dependent upon local soil chemistry), and

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2. controlling drainage systems by regulating sheetwash and channelized flow in the area of piping.

These techniques have been applied to agricultural and range lands but not to dynamically evolving pipe complexes in semiarid valley floors. Heede (1971) determined that given sufficient time many pipe complexes reclaim themselves naturally. This is typical of terrain so severely piped that the topography is primarily coalescing collapsed depressions.

Piping may impact the integrity of tailings in one of two major ways: 1) by enhancing valley floor instability and rejuvenation of drainage networks, and 2) by developing on reclaimed slopes covering tailings piles. Very few studies of processes on reclaimed slopes indicate piping as a major process; however, in a study of a post coal-mined reclaimed slope, piping was observed (Jercinovic, 1984). In a small watershed near the Church-Rock uranium mining district, reclaimed slopes displayed the greatest variation in infiltration responses (0.6-180 cm/hr) and the highest infiltration rate (14 cm/hr). Subsidence and compaction features related to piping of reclained material increased the infiltration. Although this resulted in decreased erosion of the hillslopes, piping produced significant increases in through flow to the ephemeral channels. This water contained much higher concentratons of dissolved constituents than typical surface runoff (Jercinovic, 1984). Thus, piping may force more through-flow water into a reclaimed pile and enhance the possibilities of leakage and discharge into natural systems.

Sheetwash and Rilling

Erosion on slopes occurs under unconfined flow and confined flow conditions, sheetwash and rilling, respectively. These processes will primarily impact reclaimed uranium tailings disposal sites by post-reclamation overland flow; these processes should be expected to have little impact on tailings as an external driving force such as flooding. In addition, reclaimed landscapes can be designed to reduce overland flow and reduce the erosion potential by sheetwash and rilling. However, of these two processes, rilling poses the greatest threat to tailings cover. Once rills form they are less likely to heal; rather, they may coalesce to form larger integrated drainage lines with more erosive forces.

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Available information on reclaimed hillslopes in New Mexico in the proximity of the uranium-tailings disposal areas suggests similar rates of erosion by sheetwash and rilling on natural and reclaimed hillslopes (Table 5). These rates of processes are two orders of magnitude less than processes related to the fluvial system (Wells and Gardner, 1985).

Time Scales of Geomorphic Processes

As indicated in Figure 2, those processes which will affect the stability of tailings disposal sites will vary with time and space. Channel erosion and deposition operate over the scale of decades, whereas drainage network and hillslope evolution operate most effectively over time periods > 10^3 years. Figure 19 illustrates the relative time scales of selected geomorphic processes with respect to the effectiveness of their work. Effectiveness of their work is measured either in the rate or magnitude of process or the volume of material removed or added to a landform. Over the long term, small-scale processes coalesce to form larger scale processes such as drainage network evolution (Figure 19). Because of this relationship of scale and processes, rates at which geomorphic processes operate can be subdivided into 3 major classes which do not reflect the time periods they are most effective over, but the ability to realistically determine the rate of process. *Historic* periods are those where smaller scale processes can be most accurately estimated using field instrumentation, repetitive photographs, and historic notes to document changes in the landscape. These processes and their rates are applicable over a time scale of 200 years and less. *Holocene* geologic time period includes those larger scale processes operating between 200 years and < 10,000 years. The rates of processes are primarily determined by stratigraphic studies combined with radiocarbon dating and landscape reconstructions (Wells and Gardner, 1985) and are applicable to longer term landscape stability near tailings piles. *Pleistocene* geologic time period involves those processes operating and most effective over 10^2 to 10^5 years. Table 6 lists those processes and their rates for these three time scales; the most reliable rates are those for the historic time periods where several techniques to obtain a single rate can be used as a check on the rate. The absence of a given rate reflects an inability to reliably quantify the process as well as the relative effectiveness of the process.

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Figure 20 illustrates the magnitude of these processes over the historic time scale. The relative order of the process magnitude can be utilized to develop a hierarchy of geomorphic hazards which pose the greatest threat to uranium tailings pile integrity. The proposed heirarchy based on studies in the uranium mining region of northwestern New Mexico are:

1. drainage network extension, channel headcutting
2. piping/gullying
3. bank erosion/meander growth
4. drainage network rejuvenation, channel incision
5. channel aggradation
6. valley floor deposition and wind deflation
7. vertical downwasting of slopes

It is important to note that this heirarchy will probably vary from one site to another due to variation in local geomorphic history and physical setting. However, the significance of this heirarchy is the quantification of these processes which illustrates the dynamic behavior of fluvial systems valley floors in the western U.S.

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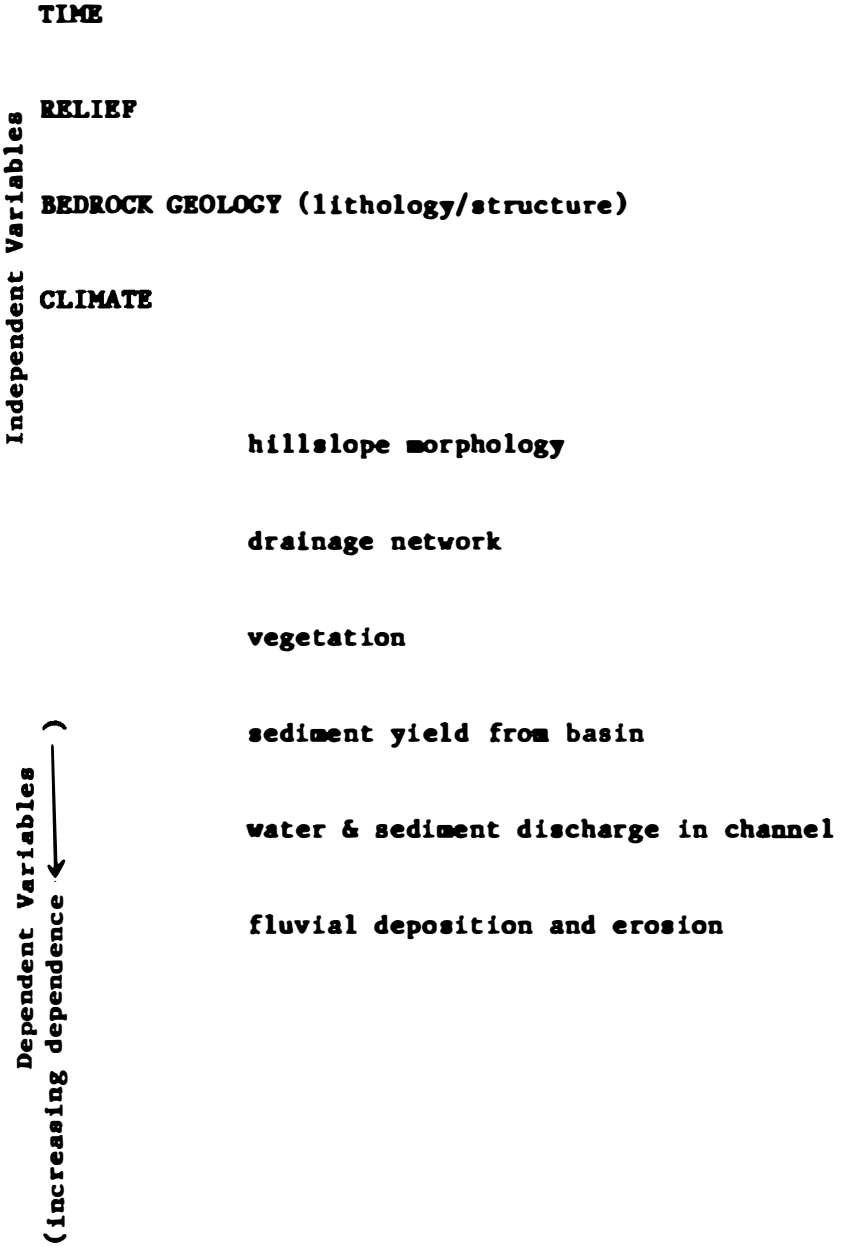

FIGURE 1 Major independent and dependent variables influencing geomorphic processes; hierarchy of dependence given in diagram (increasing from top to bottom).

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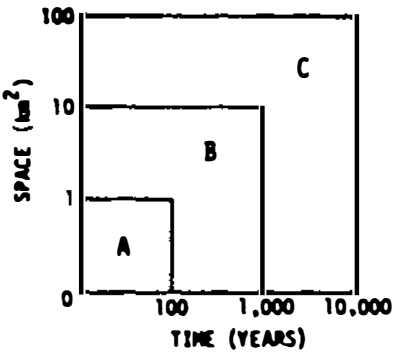

MAGNITUDE CLASSES OF AREA AND TIME		VARIABLES AFFECTING LANDSCAPE STABILITY: INTERNAL	VARIABLES AFFECTING LANDSCAPE STABILITY: EXTERNAL
	A	• HILLSLOPE GEOMETRY, GRADIENT & COMPOSITION • SURFACE ROUGHNESS • HILLSLOPE DRAINAGE DENSITY & FREQUENCY • SURFACE & SUBSURFACE BIOMASS • INFILTRATION CONDITIONS • SEDIMENT YIELD & RUNOFF PER UNIT AREA • VEGETATION DENSITY	• RATE OF ARROYO HEADCUTTING • VEGETATION DENSITY • SEDIMENT YIELD & RUNOFF (TOTAL) • VALLEY-FILL GEOMETRY • BEDROCK HETEROGENEITY • PRECIPITATION INTENSITY, DURATION & FREQUENCY • DRAINAGE DENSITY & FREQUENCY (WATERSHED)
	B	• HILLSLOPE RELIEF & GRADIENT • HILLSLOPE NETWORK • RATES OF WEATHERING & SOIL DEVELOPMENT • DEGRADATION & AGGRADATION RATES	• VEGETATION DENSITY • VALLEY SLOPE • DRAINAGE AREA • CHANNEL PATTERN & GEOMETRY • WATERSHED DRAINAGE NETWORK • BEDROCK GEOMETRY • CLIMATIC PERTURBATIONS
	C	• VEGETATION DENSITY • DEGREE OF SOIL DEVELOPMENT • DEGRADATION & AGGRADATION RATES	• VEGETATION DENSITY • BASE-LEVEL CHANGE • CLIMATIC CHANGE • BEDROCK GEOLOGY

FIGURE 2 Classification scheme for variables in geomorphic systems: internal—driven by forces generated within reclaimed tailings piles—and external—driven by forces outside the limits of the tailings pile. Different time and spatial scales reflect changes in the effectiveness of geomorphic processes (from Wells and Jercinovic, 1983).

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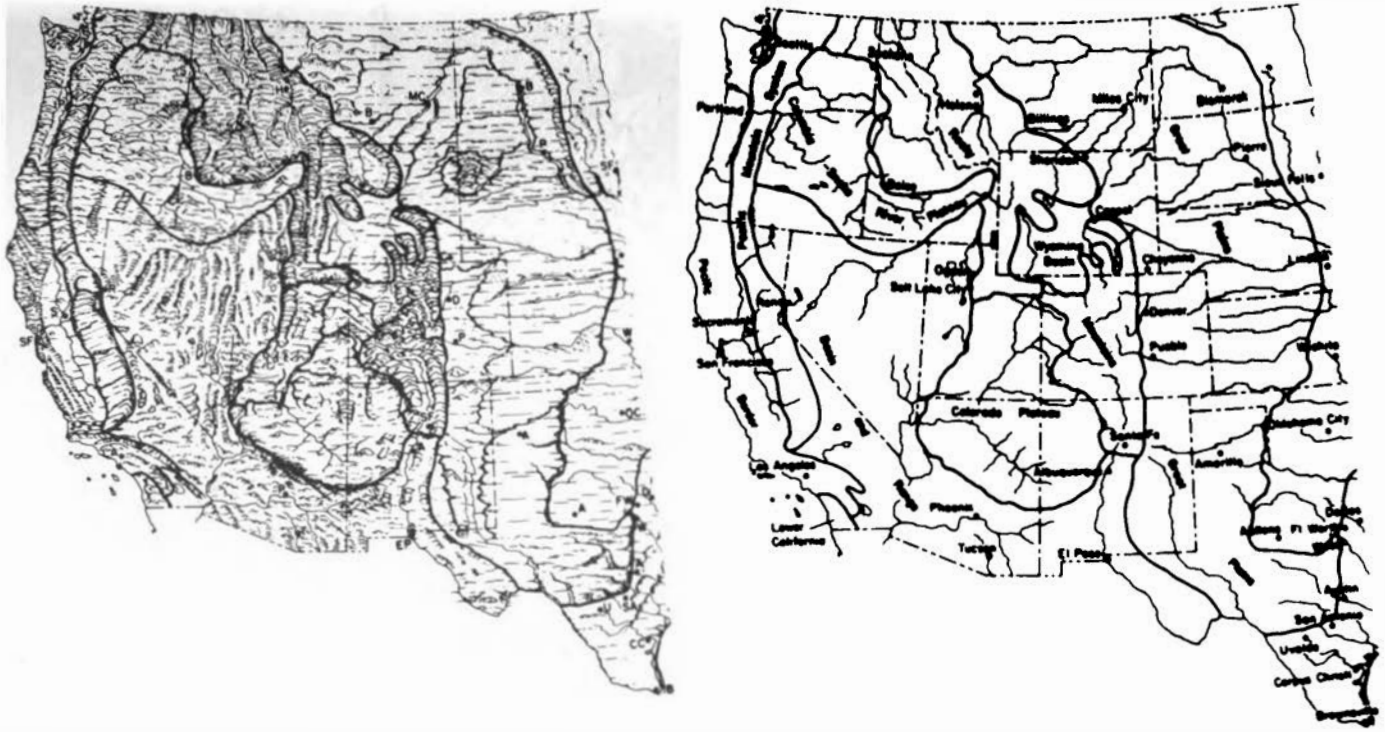

FIGURE 3 **Physiographic province of Colorado Plateau contains the majority of uranium tailings disposal sites; although it has similar regional geologic settings, significant local variations occur.**

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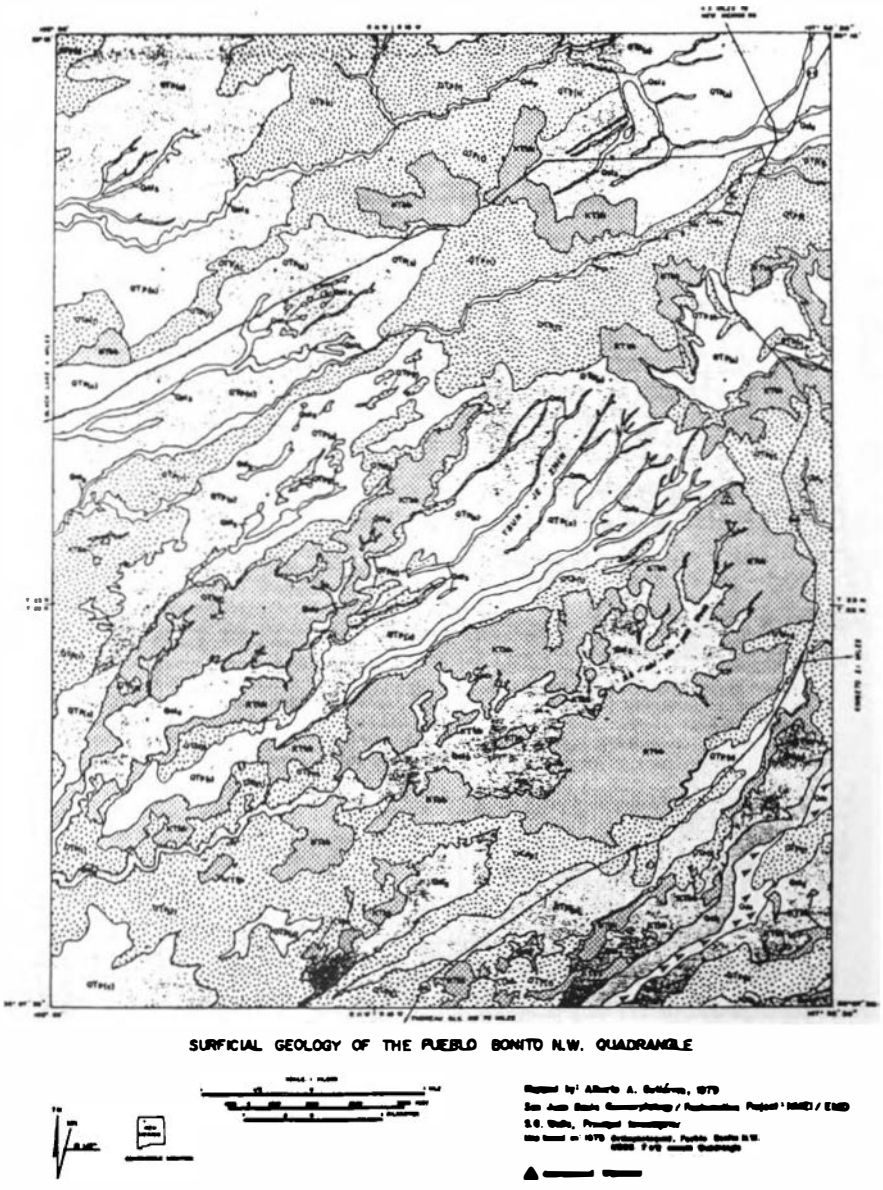

FIGURE 4 Map showing the diverse geologic setting of a small area; compare units shown on map to those described in Table 1 (from Wells and Jercinovic, 1983).

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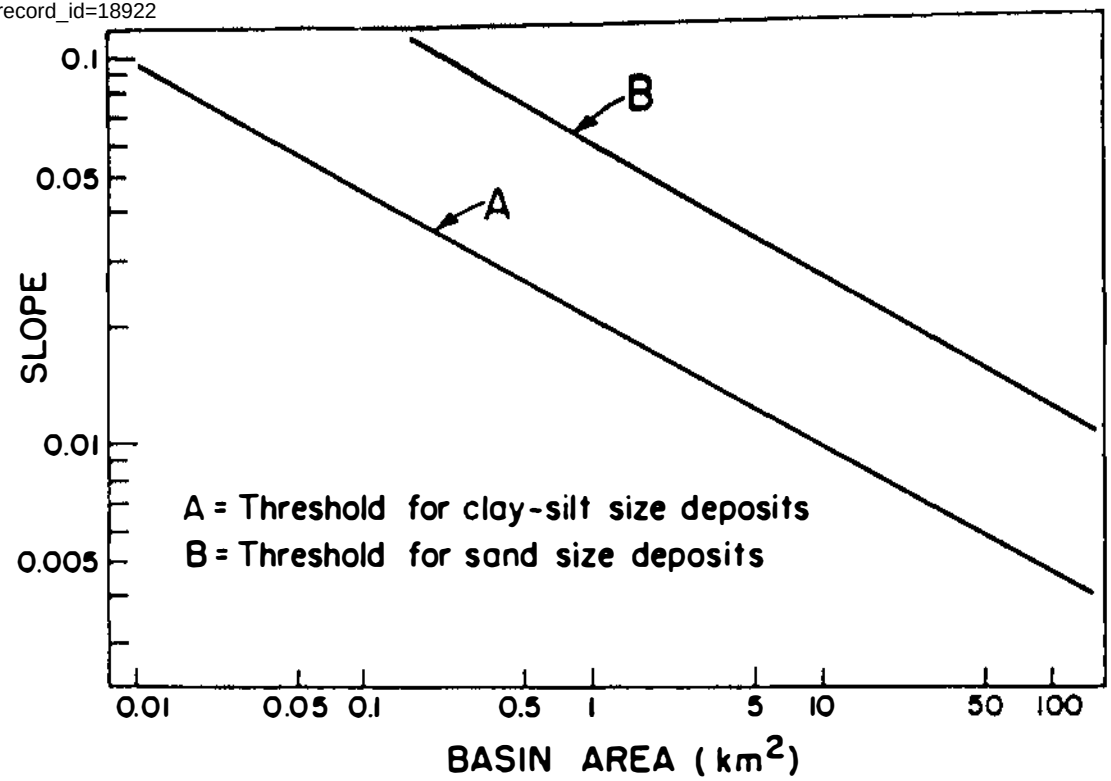

FIGURE 5 Different threshold conditions for gully incision in valley floors composed of fine and coarse grained deposits (from Wells and Gardner, 1985).

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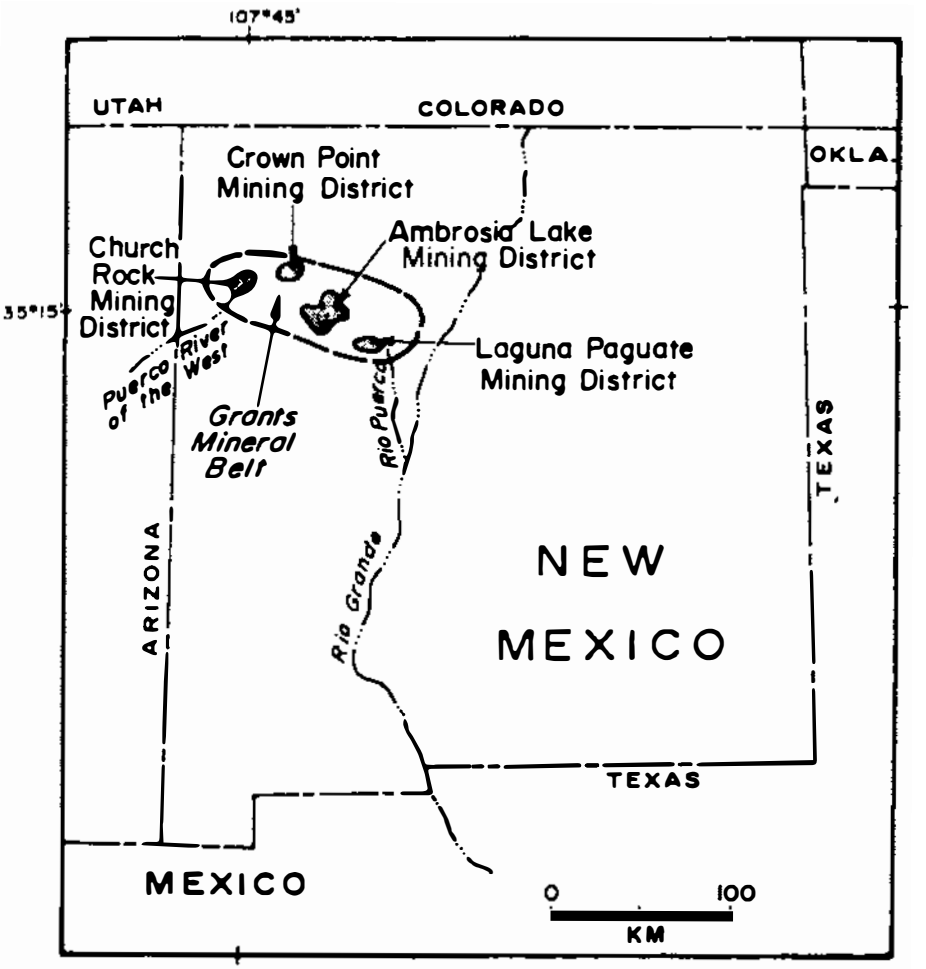

FIGURE 6 Location of major uranium mining and milling districts in northwestern New Mexico.

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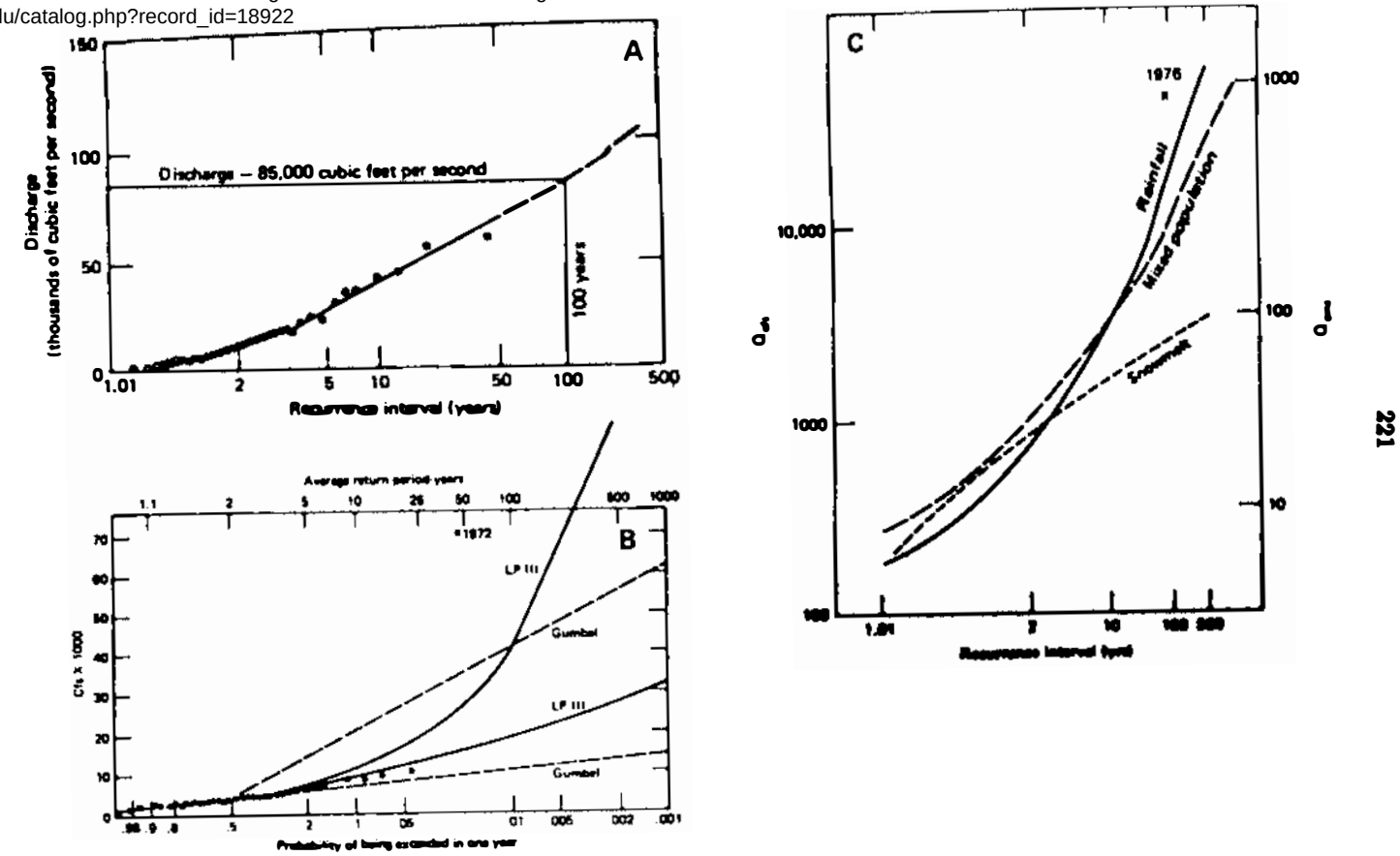

FIGURE 7 (A) Example of flood frequency curve analysis. (B) Methods of extrapolating curves beyond limit of flood record. (C) Different populations of floods due to variations in hydrologic controls. (From Costa and Baker, 1981.)

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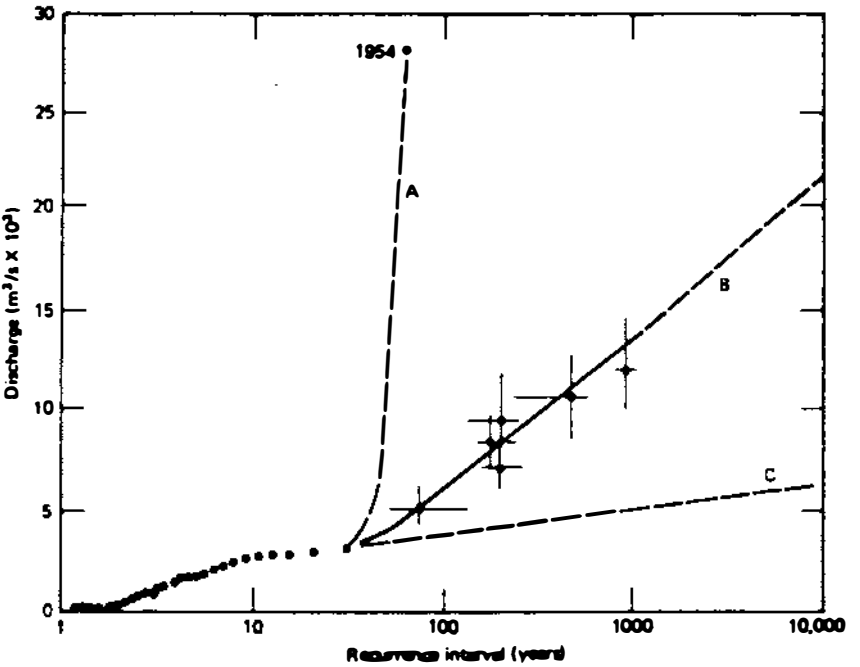

FIGURE 8 Example of problems related to determining the recurrence interval of large storm events (1954 storm in Pecos River watershed). Curves A and C provide misleading calculations; curve B is determined from detailed alluvial stratigraphic work and represents more realistic determinations (from Patton and Baker, 1977).

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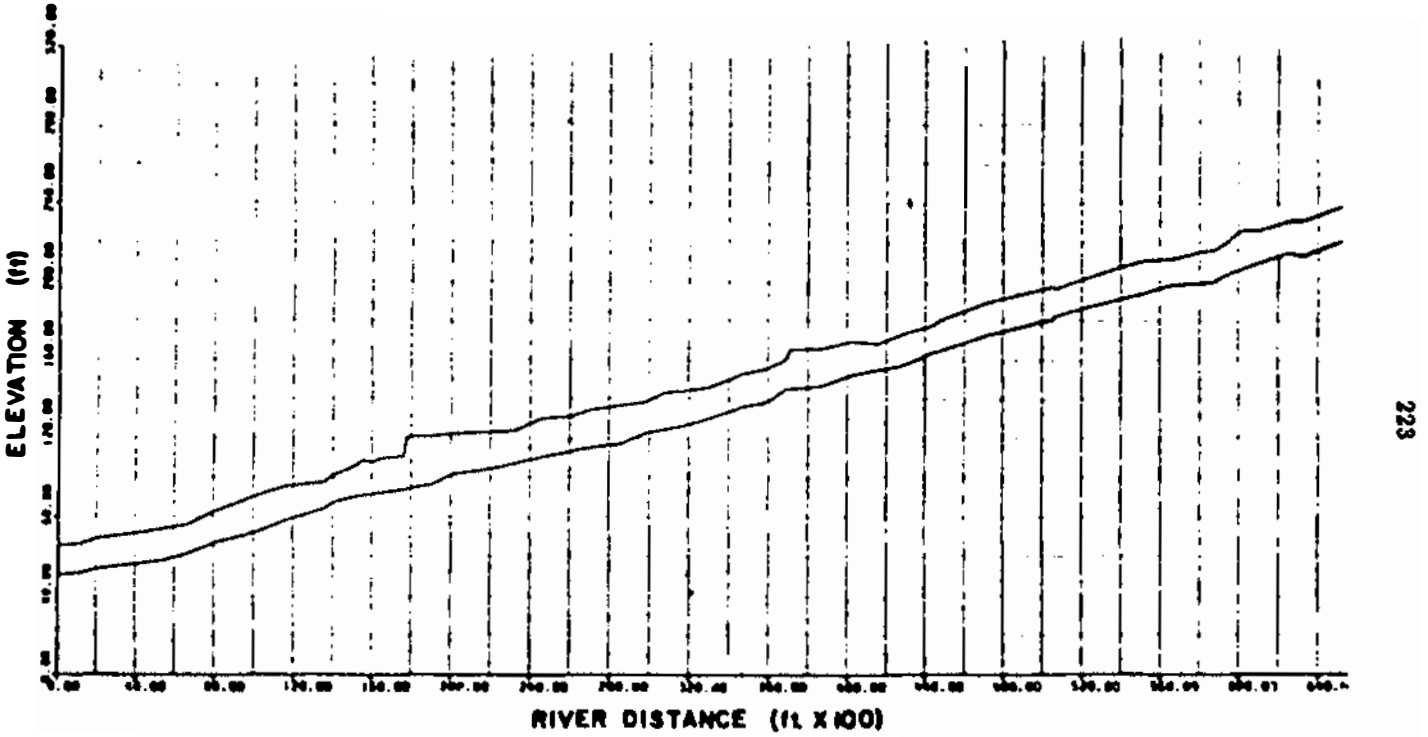

FIGURE 9 Computer-generated profiles of water surface and thalweg of Chaco River (arroyo) during 100-year storm event (Lagasse, 1983).

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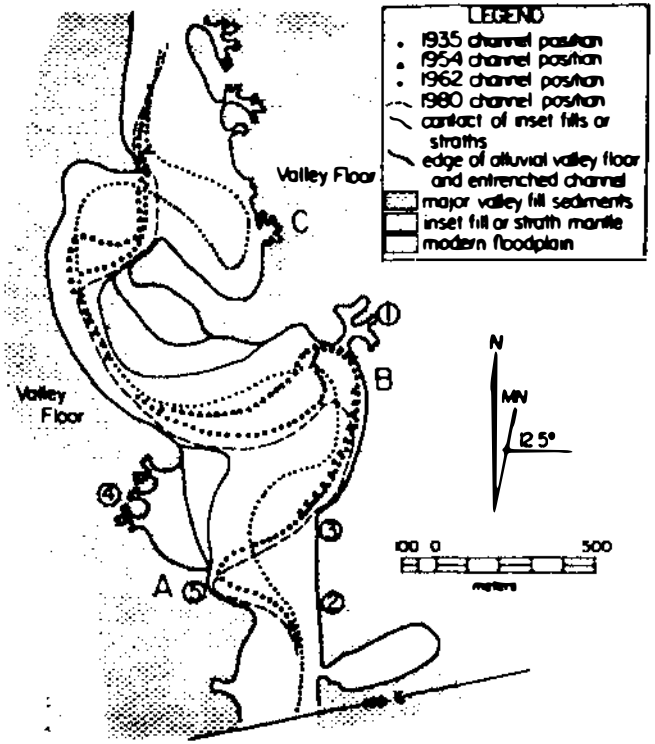

FIGURE 10(A) Changes in planform of Rio Puerco during past 50 years (from Wells and others, 1983).

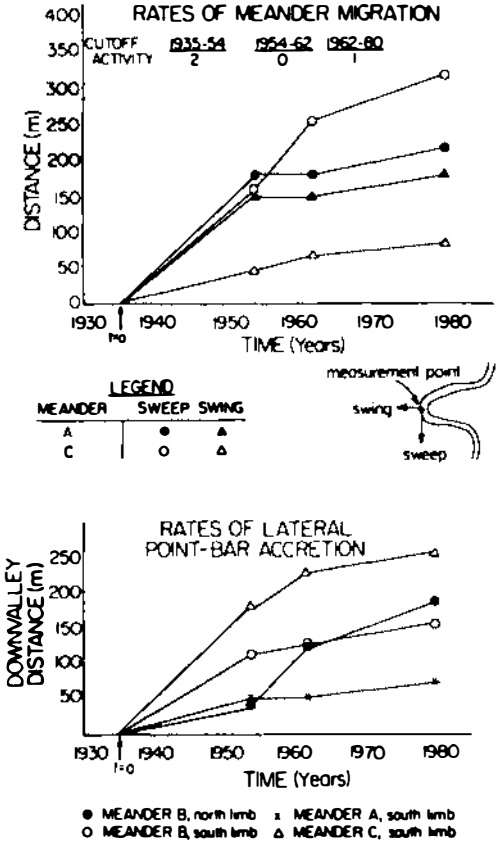

FIGURE 10(B) Rates of change of Rio Puerco channel in past 50 years (from Wells and others, 1983).

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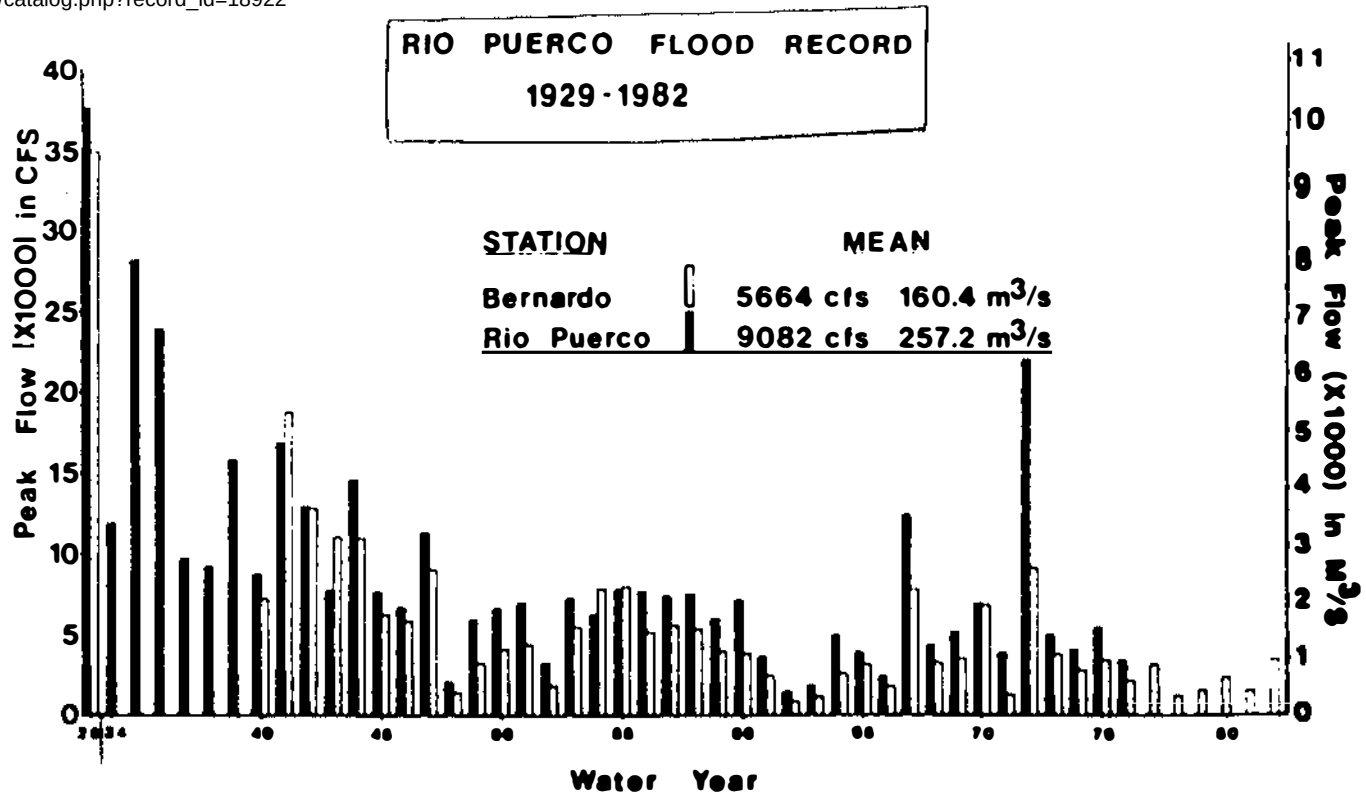

FIGURE 10(C) Flood discharge record for Rio Puerco since 1929 (from Heath, 1983).

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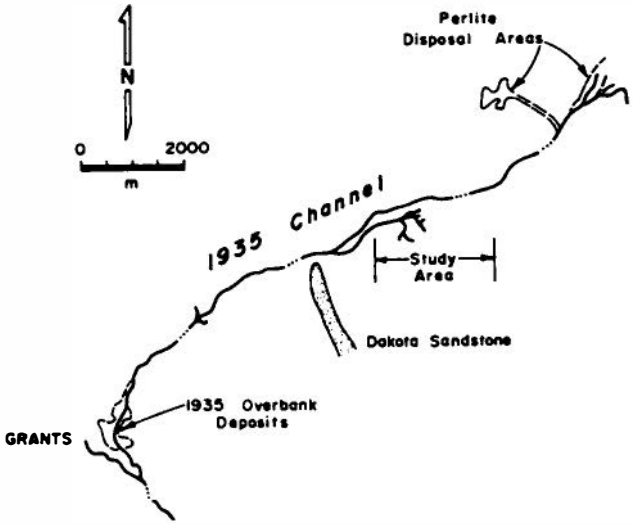

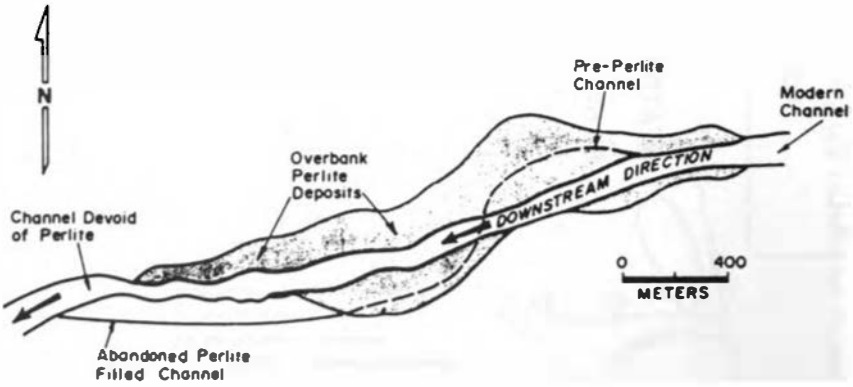

FIGURE 11 Location of massive sediment load moved as a pulse of sediment through an ephemeral arroyo; sediment load was introduced via a perlite tailings failure. Perlite filled channel in places and spread laterally over valley floor (from Miller, 1985).

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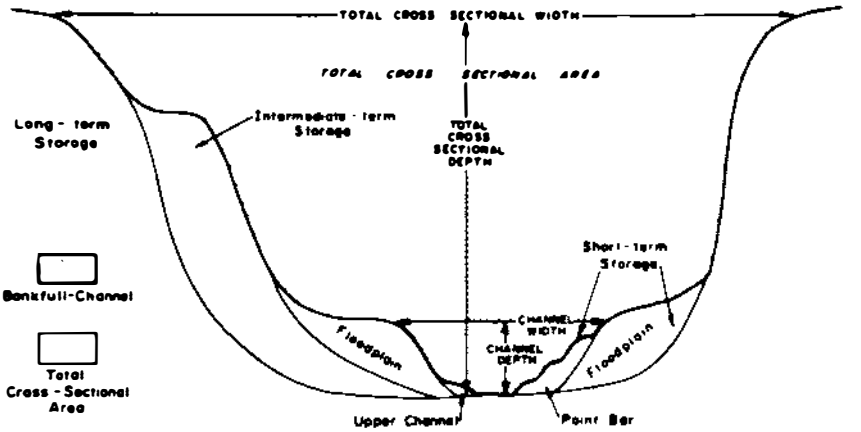

FIGURE 12(A) Channel morphology of an arroyo showing typical positions of flood water and sediment and total cross-sectional which may be reduced during massive sediment loads (Miller, 1985)

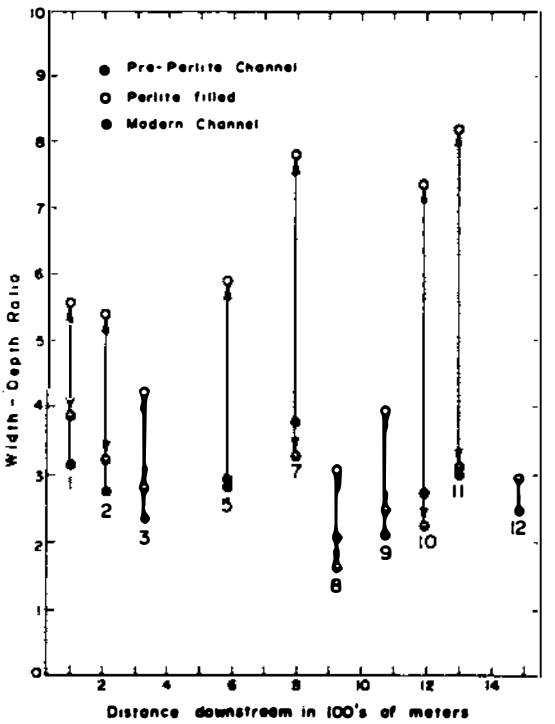

FIGURE 12(B) Changes in the width-depth ratio of an arroyo during massive sediement loads (from Miller, 1985).

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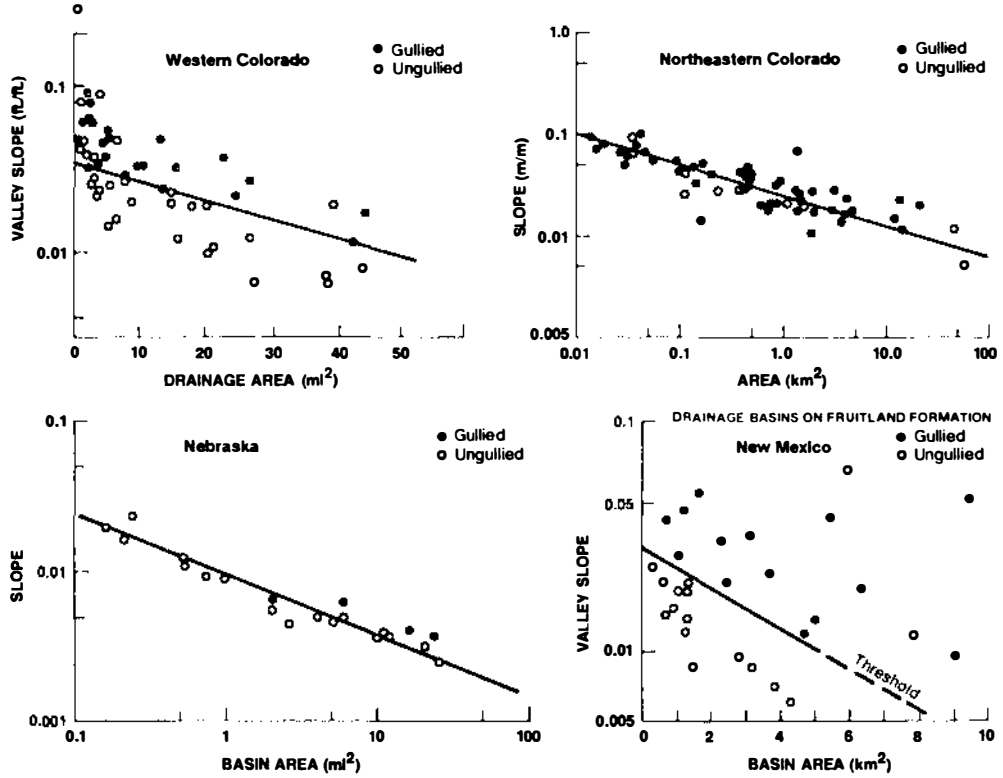

FIGURE 13 Examples of threshold conditions for gully incision in several different geologic settings (from Wells and Gardner, 1985).

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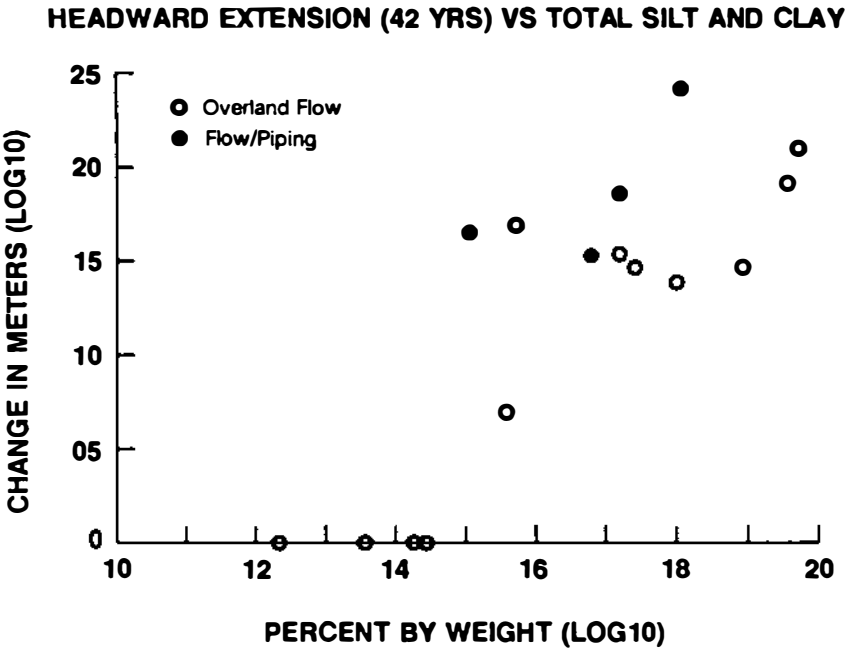

FIGURE 14 Relation between silt/clay content in valley deposits and rate of headward cutting by arroyos (from Wells and Gardner, 1985).

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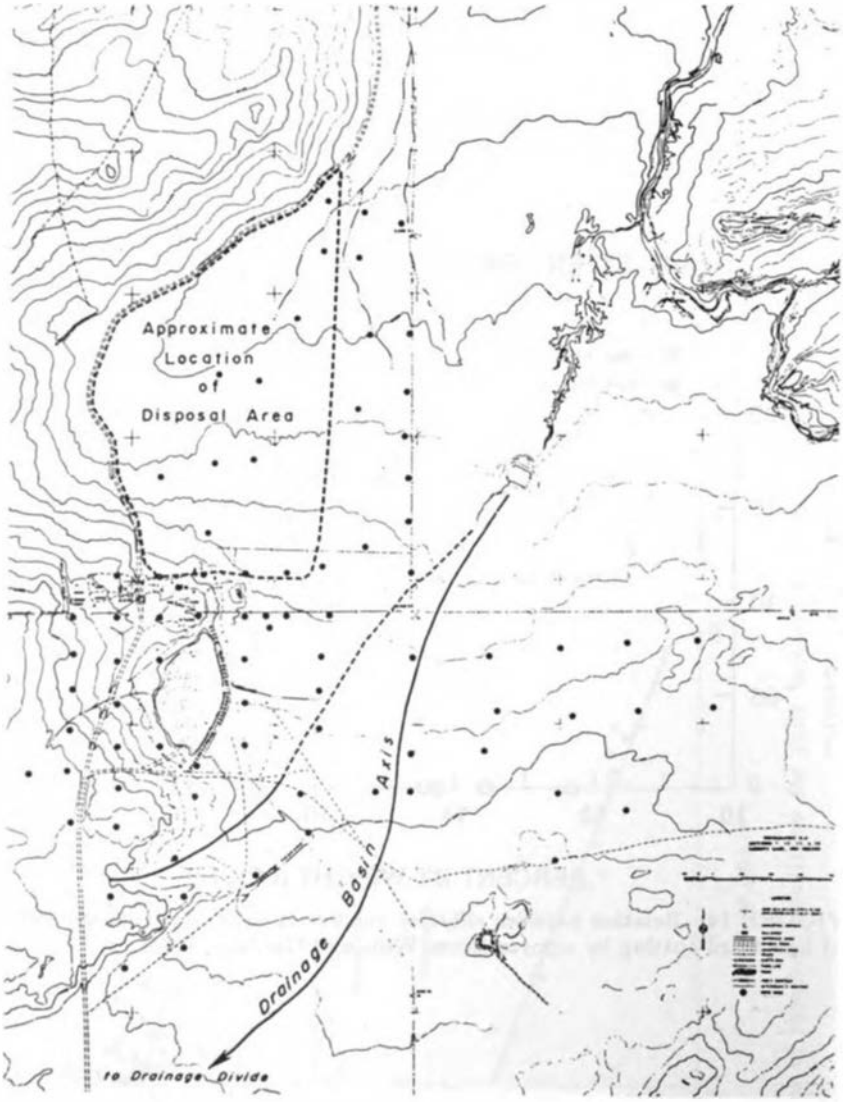

FIGURE 15 Proximity of active headcutting arroyo to planned uranium tailings disposal site in New Mexico; rate of extension is 300 m per 50 years.

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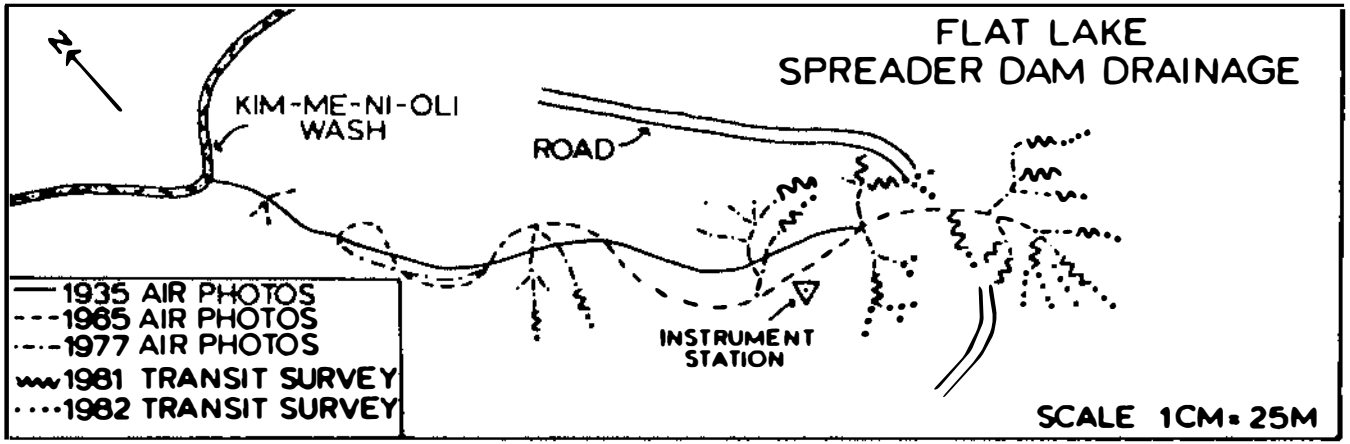

FIGURE 16 **Small-scale changes of arroyo headcutting in an area of active recharge from water derived from uranium-mine related aquifer dewatering (from Wells and Gardner, 1985).**

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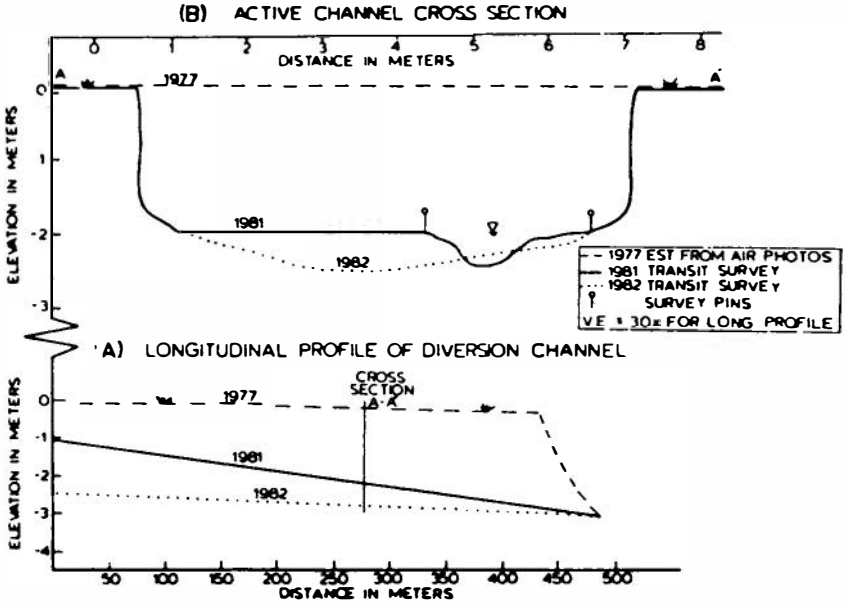

FIGURE 17 Erosional patterns during arroyo headcutting through an engineered berm to arrest arroyo headcutting (from Wells and Gardner, 1985).

FIGURE 18 Large-scale pipes and a gully system developed in Cretaceous shales of northwestern New Mexico.

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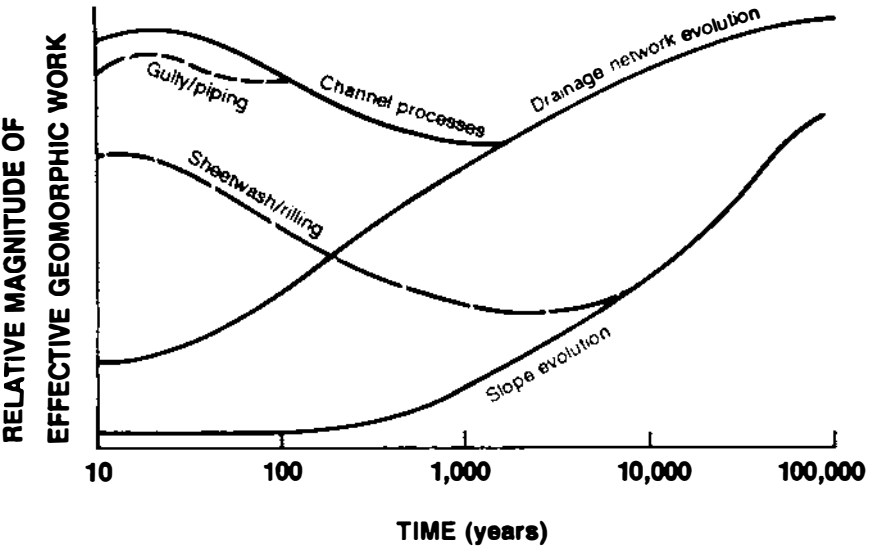

FIGURE 19 Conceptual model illustrating the relative effectiveness of geomorphic processes over different time scales.

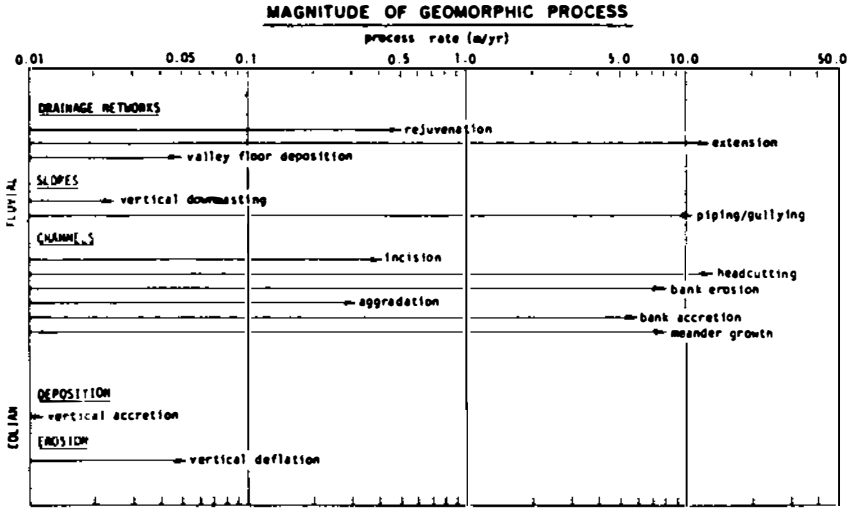

FIGURE 20 Magnitudes of geomorphic processes operating over historic time scale in northwestern New Mexico (from Wells and Gardner, 1985).

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TABLE 1 Local Variations in Geologic Setting and Geomorphic Processes in a Small Region of Northwestern New Mexico (from Wells and Jercinovic, 1983)

Surficial-Process Units	Landforms	Type of Surficial Material	Age of Surficial Materials[1]	Range in Slope Gradient (percent)	Bare (unvegetated) Soil[2] (percent)
QTp(s)	pediment and terrace remnants	fluvial clay, silt, sand, and gravel, usually indurated	Pliocene (?) to Pleistocene	0-1	45-64
	stabilized dunes and sand sheets	eolian sands	late Pleistocene to Holocene	0-1	41-58
QTp(t)	dissected QTp(s)	includes some material reworked from QTp(s)	same as QTp(s)	12-28	85
Qal,	broad vegetated channels	fluvial clay, silt, and sand	middle to late Holocene	1-4	41-58
KTbb	badlands	thin clayey to silty weathered mantle	late Holocene to modern	10-40	94
Qal,	low terraces, alluvial fans, and stabilized dunes	fluvial clay, silt and sand; eolian sand	late Holocene	1-5	35-58
Qal,	active channels	fluvial sand and gravel	present day	0.3-1	90-100
Qds	active dunes	eolian sand	late Holocene to present day	1-10	65-100

[1]Data taken from Schultz (1980) and Wells and Gutierrez (1981).
[2]Data taken from U.S. Department of the Interior, Bureau of Land Management (1976), Figure 33.

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TABLE 1 (continued)

Surficial-Process Units	Landscape Type	Dominant Geomorphic Process(es)	Erosion (vertical downcutting) Type	Rate (cm/yr)	Infiltration[1, 4] Rate (cm/hr)	Detachability[1, 5] Rate (kg/m²/hr)	Estimated Sediment Yield[3, 7] (cm³/m²/yr)	Measured Sediment Yield[1, 8] (cm³/m²/yr)
QTp(s)	stable upland surface	erosional stability, eolian deposition	sheetwash	(undetermined)	>20	100-1,000	0-24	(undetermined)
QTp(t)	transition between upland surface and badland	slope erosion	rilling and sheetwash	0.3-2.0	1-20	10-1,000	810-1,500	2,800-20,000
Qal_1[2]	upland surface	fluvial erosion and deposition	gully incision and sheetwash	–	–	–	–	–
KTbb	badland	slope erosion	creep and sheetwash	0.1-2.0	0-1	1-10	570-1,500	950-26,000
Qal_2	valley floor	slope erosion and fluvial deposition (0.1-0.5 cm/yr)	sheetwash and gully incision	–	1-10	100-1,000	120-310	3,600-18,000
Qal_1[2]	valley floor	fluvial transport	–	–	–	–	430-570	–
Qds	valley floor and upland surface (not differentiated on upland surfaces)	eolian deposition	–	–	20	100-1,000	up to 190 (fluvial)	–

[1]Data taken from U.S. Department of the Interior, Bureau of Land Management (1976)
[2]Data not available on process rates and yields
[3]Determined by the unmodified Pacific Southwest Inter-Agency Committee (PSIAC) method; data taken from U.S. Department of the Interior, Bureau of Land Management (1976)
[4]Taken from Wells (1978) and Gutierrez (1980).
[5]Rate of water that is absorbed into ground using a single-ring constant infiltrometer
[6]Susceptibility of a soil to erosion by running water determined under controlled laboratory conditions
[7]Empirical (PSIAC) method of determining volume of sediment yielded from a land surface under assumed precipitation conditions
[8]Determined in the field during precipitation events

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TABLE 2 Geomorphic Setting of Major Uranium Mills and Tailings Disposal Sites in Northwestern New Mexico (from Wells and Gardner, 1985)

Uranium Mill and Tailings Site	Geomorphic Setting
1. United Nuclear-Churchrock	valley floor and side-slope
2. Phillips-Nose Rock*	valley floor and side-slope
3. Phillips-Ambrosia Lake	valley floor
4. Kerr McGee-Ambrosia Lake	valley floor and side slope
5. Anaconda-San Mateo	valley floor and basalt flow
6. United Nuclear Homestake -San Mateo	valley floor
7. Sohio-Paguate	valley floor
8. Gulf-San Mateo*	pediment and valley floor
9. Bokum-Marquez*	valley floor and terraces

* = planned

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TABLE 3 Major Geomorphic Hazards to Stable Uranium Tailings Disposal Sites (from Schumm and others, 1982)

Zone 1 Landforms

1. Drainage Networks
   a) Erosion
      1. rejuvenation - incision of all channels and headward lengthening into undissected areas of the drainage basin.
      2. extension - headward growth of the drainage network without rejuvenation.
   b) Deposition
      1. valley filling - major deposition causing significant elevation of valley floor.
   c) Pattern change
      1. capture - diversion of runoff and sediment from one channel to another with major channel adjustments (see 3a,b,d below).
2. Slopes
   a) Erosion
      1. denudation - normal slope erosion and retreat
      2. dissection - incision of slope by rills, gullies, or by drainage network extension (1a2 above)
      3. mass failure - all forms of mass movement, slump, landslide, etc.

Zone 2 Landforms

3. Channels
   a) Erosion
      1. degradation - general lowering of stream bed by incision
      2. nickpoint formation and migration - development of break in longitudinal profile that migrates upstream
      3. bank erosion - channel widening or local erosion as related to meander shift.
   b) Deposition
      1. aggradation - general raising of stream bed by deposition
      2. back and downfilling - deposition in channel that migrates upstream or downstream
      3. berming - narrowing of channel by deposition of fine sediments on banks
   c) Pattern change
      1. meander growth and shift - increase of meander amplitude and downstream migration
      2. island and bar formation and shift - bars form by local deposition and become islands when colonized by vegetation; islands as a result of in-channel erosion and deposition can shift position
      3. cutoffs - neck and chute cutoffs of meanders locally steepen gradient
      4. avulsion - major shift of channel position
   d) Metamorphosis - a complete change of channel morphology as expressed by pattern change
      1. straight to meandering - development of bends and probable increase of width; gradient decreases
      2. straight to braided - development of bars and islands, increase of width and reduction of depth, may reflect aggradation
      3. braided to meandering - narrowing, deepening, and development of bends, decrease of gradient
      4. braided to straight - narrowing and deepening
      5. meandering to straight - increase of gradient
      6. meandering to braided - widening, shallowing, increase of gradient, may indicate aggradation.

Zone 3 Landforms

4. Piedmont and Alluvial Plain (alluvial fan, delta, bajada, pediment)
   a) Erosion
      1. dissection - development of channels and incision of surface
   b) Deposition
      1. aggradation - general filling of channels and deposition on surface of fan or delta
      2. progradation - lengthening of delta or fan with upstream deposition
   c) Pattern change
      1. rejuvenation and extension - incision of existing channels and headward growth
      2. development of drainage pattern - growth of drainage network on undissected surface
      3. avulsion - channel shift and bifurcation on fan or delta
      4. capture - diversion or piracy of water and sediment leading to rejuvenation of captor stream.

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TABLE 4 Computer-based Hydrologic Data for 100-Year Event in Chaco River, An Ephemeral Arroyo (from Lagasse and others, 1983)

	Discharge Q cfs	Depth d feet	Velocity V fps	Topwidth Tw feet
Wijiji Area	13300	16.6	5.4	631
Bradley Site	13300	15.6	19.1	148
Narrows Region	13300	16.2	10.1	323
Visitor Center Bridge	23800	15.6	19.5	204
Hungo Pavi Region	22470	19.6	14.0	281
P. Bonito Bridge	19860	18.6	22.5	83
P. del Arroyo Ruin	19860	15.4	17.3	178
Kin Kletso Ruin	19860	13.5	20.0	289

TABLE 5 Rates of Geomorphic Processes in Natural Human-Affected Landscapes (from Wells and Gardner, 1985)

FLUVIAL NETWORKS

Process Type	Hydrologic Condition	
	Pre-mine, Ephemeral	De-watering, Perennial
Headcut Erosion	1.2 m/yr	10.5-95.0 m/yr
Channel Bank Erosion	5.1 m/yr	1.5 m/yr

HILLSLOPES

Hillslope Type	Erosion Rate
Reclaimed, convex shale & sandstone aggregate	0.2 to 0.9 cm/yr (mean = 0.5 cm/yr)
Undisturbed, convex shale	0.3 cm/yr
Undisturbed, concave sand and sandy shale	0.3-2.5 cm/yr

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TABLE 6 Rates of Geomorphic Processes for Three Time Scales; Rates Based on Studies in Northwestern New Mexico (from Wells and Gardner, 1985)

Geomorphic Hazard	Material Property*	Rates of Geomorphic Processes		
		Historic (<200 yr)	Holocene (>200,<10,000 yr)	Pleistocene (>10,000 yr)
I. Fluvial Activity				
A. Drainage networks				
1. Erosion				
a. rejuvenation	bedrock	---	0.3-0.5 cm/yr	0.01-0.04 cm/yr
b. extension	alluvium	up to 50.0 cm/yr	up tp 0.2 cm/yr	---
2. Deposition (valley floor)	alluvium	0.5 cm/yr	0.25-0.30 cm/yr	---
3. Pattern Change		development of drainage networks in valley floors	increased drainage density	reduction in watershed size
B. Slopes				
1. Erosion				
a. vertical downwasting			---	---
	sandstone/sand alluvium	0.3-2.5 cm/yr	---	---
	mudstone/shale	0.1-0.5 cm/yr	---	---
	bedrock	---	0.3-0.5 cm/yr	---
b. gullying/piping (lateral extension)	alluvium	up to 10.5 m/yr	---	---
B. Channels				
1. Erosion				
a. incision	alluvium	6.5-50.0 cm/yr	0.25-0.45 cm/yr	---
b. headcutting	alluvium	1.0-12.5 m/yr	---	---
c. bank erosion	alluvium	1.5-8.0 m/yr	---	---
2. Deposition				

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TABLE 6 (continued)

a. channel floor	---	4.0-30.0 cm/yr	0.025-0.400 cm/yr	---
b. bank accretion	---	1.5-6.0 m/yr	---	---
3. Pattern Change				
a. meander growth	alluvium	2.0-8.0 m/yr	---	---
b. cutoff activity	alluvium	3 per 50 yr	---	---
4. Metamorphosis				
a. complete pattern change		braided to meandering	---	---
b. channel width-depth ratio		decrease	decrease	---
II. Eolian Activity				
A. Deposition (vertical accretion)		0.02-0.1 cm/yr	0.01-0.02 cm/yr	---
B. Erosion				
1. Saltation		0.075 $g/cm^2/yr$	0.0326 $g/cm^2/yr$ ($231.0\ cm^2/m^2/yr$)	---
2. Suspension		0.004 $g/cm^2/yr$	---	---

* = type of material process is operating within

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Appendix F
Demos Model Listing

Variable Impact_twp
Title: Total radon health impact for person living near pile.
Units: cancer risk/lifetime
Description: Per capita lifetime risk of lung cancer for a model subject living next to a tailings pile.
Definition: Impact_inwp + Impact_outwp

Variable Impact_tnp
Title: Total radon health impact for person not living near pile.
Units: cancer risk/lifetime
Description: Per capita lifetime risk of lung cancer for a model subject not living near a tailings pile.
Definition: Impact_innp + Impact_outnp

Variable Impact_innp
Title: Health impact of indoor radon without pile.
Units: cancer risk/lifetime
Description: Per capita lifetime risk of lung cancer from exposure to indoor radon sources in homes not located near piles.
Definition: FT_in*C_innp*F_in*R_pcl

Variable Impact_inwp
Title: Health impact of indoor radon with pile.
Units: cancer risk/lifetime
Description: Per capita lifetime risk of lung cancer from exposure to indoor radon sources in homes located near piles.
Definition: FT_in_home*(C_innp+C_pile)*F_in*R_pcl+FT_in_nhome*C_innp*F_ in*R_pcl

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Variable Impact_outwp
Title: Health impact of outdoor radon (w/pile).
Units: cancer risk/lifetime
Description: Per capita lifetime risk of lung cancer from exposure to outdoor radon sources including a tailings pile.
Definition: (Ft_out_home*(C_back*F_out+C_pile*F_pile)+Ft_out_nhome*(C_back*F_out))*R_pcl

Variable Impact_outnp
Title: Health impact of outdoor radon (no pile).
Units: cancer risk/lifetime
Description: Per capita lifetime risk of lung cancer from exposure to outdoor radon sources (no pile).
Definition: (Ft_out_home+Ft_out_nhome)*C_back*F_out*R_pcl

Variable Impact_pile
Title: Health impact of radon from pile.
Units: cancer risk/lifetime
Description: Cancer risk due to radon from pile via exposure outdoors.
Definition: (Ft_out_home*C_pile*F_pile+Ft_in_home*C_pile*F_in)*R_pcl

Variable Ft_in
Title: Fraction of time spent indoors.
Units: dimensionless fraction
Description: Fraction of total time spent indoors by model subject.
Definition: 1.0-Ft_out_nhome-Ft_out_home

Variable Ft_in_home
Title: Fraction of time spent indoors at home.
Units: dimensionless fraction
Description: Fraction of total time spent indoors at home by model subject.
Definition: (1-InH)*Ft_in

Variable Ft_in_nhome
Title: Fraction of time spent indoors away from home.
Units: dimensionless fraction
Description: Fraction of total time spent indoors away from home by model subject.
Definition: InH*Ft_in

Variable InH
Title: Fraction of indoor time not at home.

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Units: dimensionless fraction
Description: Fraction of indoor time spent away from home.
Definition: .35

Variable Ft_out_home
Title: Fraction of time outdoors at home.
Units: dimensionless fraction
Description: Fraction of total time spent outdoors at home by model subject
Definition: .022*sens_fact

Variable Sens_fact
Title: Sensitivity factor.
Units: dimensionless factor
Description: Sensitivity factor to examine sensitivity of results to subjects spending more than average amounts of time outdoors at home.
Definition: [1, 5, 10, 20]

Variable Ft_out_nhome
Title: Fraction of time spent outdoors not at home.
Units: dimensionless fraction
Description: Fraction of total time spent outdoors but away from home by model subject.
Definition: .013

Variable C_back
Title: Background concentration.
Units: pCi/l
Description: Background outdoor radon concentration.
Definition: lognormal(.5, 2)

Variable C_pile
Title: Pile concentration.
Units: pCi/l
Description: Concentration of radon from the pile.
Definition: lognormal(1, 2)

Variable C_innp
Title: Indoor radon concentration without pile.
Units: pCi/l
Description: Indoor radon concentration.
Definition: lognormal(0.9, 2.8)

Variable F_in

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Title: Indoor equilibrium factor.
Units: dimensionless fraction
Description: Fractional equilibrium of radon with its decay daughters in indoor air.
Definition: uniform(.3, .6)

Variable F_out
Title: Outdoor equilibrium factor.
Units: dimensionless fraction
Description: Fractional equilibrium of radon with its decay daughters in regular outdoor air.
Definition: uniform(.5, .8)

Variable F_pile
Title: Equilibrium factor for fresh pile radon.
Units: dimensionless fraction
Description: Fractional equilibrium of radon with its decay daughters for radon that has just come off the pile.
Definition: uniform(.005, .1)

Variable R_pcl
Title: Exposure risk factor.
Units: Cancers/pCi/l
Description: Lifetime per capita risk of cancer per pCi/l of radon at equilibrium.
Definition: R_wlm*H/D_conv

Variable R_wlm
Title: Risk factor for wlm exposure.
Units: Cancers/wlm
Description: Lifetime per capita risk of cancer per wlm of radon at equilibrium.
Definition: lognormal(0.0002, 2.5)

Variable H
Title: Hours per life.
Units: hrs
Description: Number of hours in an expected lifetime.
Definition: 365.25*24*70

Variable D_conv
Title: Dose conversion factor.
Units: pCih/l/wlm
Description: Number of pCi/l hours in a working level month at equilibrium.

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Definition: 170*100

Variable Ratio_wp
Title: Pile risk/total risk with pile.
Units: dimensionless
Description: Ratio of risk from exposure to radon from just the pile to risk from exposure to all radon including pile.
Definition: Impact_pile/Impact_twp

Variable Ratio_np
Title: Pile risk/total risk with no pile.
Units: dimensionless
Description: Ratio of risk from exposure to just the radon from the pile to risk from exposure to all radon excluding pile.
Definition: Impact_pile/Impact_tnp

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Appendix G
Acronyms and Other Abbreviations

AEC	Atomic Energy Commission
AMC	American Mining Congress
DOE	Department of Energy
DOT	Department of Transportation
EDTA	Ethylenediaminetetraacetic acid
EID	Environmental Improvement Division, State of New Mexico
EIS	Environmental Impact Statement
EPA	Environmental Protection Agency
ERDA	Energy Research and Development Administration
FRC	Federal Radiation Council
FUSRAP	Formerly Utilized Sites Remedial Action Program
ICRP	International Commission on Radiological Protection
LET	Linear Energy Transfer
MSHA	Mine Safety and Health Administration
NCRP	National Council on Radiation Protection and Measurements
NEPA	National Environmental Policy Act
PHS	Public Health Service
UMTRA	Uranium Mill Tailings Remedial Action Program
UMTRCA	Uranium Mill Tailing Radiation Control Act
U.S. NRC	U.S. Nuclear Regulatory Commission

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