Ecological Effects of Transuranics in the Terrestrial Environment

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This chapter explores the ecological effects of transuranium radionuclides in terrestrial environments. No direct studies that relate the level of transuranic contamination to specific changes in structure or function of ecological systems have been carried out. The only alternative approach presently available is to infer such relationships from observations of biota in contaminated environments and models. Advantages and shortcomings of these observations as well as those of the direct experimental approach are discussed. Searches for ecological effects of plutonium contamination in terrestrial ecosystems adjacent to the Rocky Flats plant (Colorado) and at the Nevada Test Site have not positively demonstrated plutonium-induced perturbations. These studies were carried out in areas containing of the order of 10 to 1000 $\mu Ci\ {}^{239}Pu/m^2$ in the upper 3 cm of soil. Simple calculations suggest that ${}^{239}Pu$ applications on the order of 1 Ci/m^2 may be required to cause significant mortality in plant populations. Models and calculations indicate that over 1 mCi ${}^{239}Pu/m^2$ would likely be required to produce subacute mortality in mammals. Additional research applicable to ecological effects is suggested.

To grasp the ecological implications of transuranium elements in the environment, we must understand their chemical, physical, and biological behavior through time. We must also understand the effects on biological systems of these elements when they are dispersed into the environment. Knowledge of the biological effects is particularly lacking. This may seem surprising in view of the large research efforts that have been devoted to the biological effects of plutonium and other transuranics. The lack of quantitative understanding in the area of ecological effects is not so surprising, however, when the complexities of the problem are considered. Such complexities include the environmental behavior of transuranics, which is dependent on the physical and chemical form of the nuclides as well as on the nature of the ecosystem. Of major importance is the dose to certain tissues, but dose distribution is especially complex for relatively insoluble alpha emitters. A high-level application of transuranic may have little radiation effect if energy is not deposited in critical cells.

Although we know a great deal about the effects of plutonium on experimental mammals (Bair and Thompson, 1974), we know very little about its effects on the other classes of animals that have important functions in natural systems and even less about its effects on plants. Also, very little is known about the general biological effects of the other transuranics. The effects of X- and gamma radiation on major plant and animal phyla have been studied in depth, but the extrapolation of X- and gamma radiation information to insoluble alpha emitters is seriously complicated by dosimetry and the relative biological effectiveness (RBE) of alpha particles for most actinides. Recent reviews on plutonium and other actinides in the environment have very little to say about ecological effects; rather, they dwell primarily on distribution and behavior (Romney and Davis, 1972; Martell, 1975; Hakonson, 1975; Hanson, 1975).

There are three basic approaches to the study of ecological effects of transuranics: (1) direct experiments in which radionuclides are applied at various levels to study systems, (2) observations of populations that occupy contaminated areas, and (3) modeling and extrapolation from applicable research data. Each approach has inherent advantages and shortcomings. The direct experimental approach might enjoy a relatively high degree of credibility and accuracy, but it has not been used with transuranics for reasons of safety and lack of public acceptance. Examination of contaminated areas is quite feasible and has been done at such places as the Nevada Test Site, Enewetak, and Rocky Flats (in Colorado). This approach is less than ideal, however, because of the usual lack of good experimental control and the common presence of more than one potentially toxic substance, which lead to uncertainties in data interpretation. The third approach can be used when needed with existing data, but accuracy may be poor because of the complexity and uncertainty associated with parameter values.

Ecosystems can probably tolerate higher levels of radioactivity from most, if not all, of the transuranics than from the more biologically mobile fission products, such as ^{90}Sr and ^{137}Cs. Low solubility, lack of essential nutrient analogues, and the virtual lack of penetrating radiations for most transuranics form the basis for this opinion. However, critical experiments to make this comparison have not been done; there is some concern that biological incorporation of long-lived transuranics in the environment may slowly increase with time, and it is known that very low levels may be carcinogenic.

Direct Experiments

The literature dealing with effects of ionizing radiation on plants and animals is massive. Important reviews and bibliographies include the BEIR report (National Academy of Sciences–National Research Council, 1972), the UNSCEAR report (United Nations, 1972), and the bibliography by Sparrow, Binnington, and Pond (1958). The vast majority of this literature, however, is based on laboratory studies with X- or gamma radiation. A far smaller body of literature exists on radiation effects on natural populations. Whicker and Fraley (1974) reviewed field studies dealing with the effects of ionizing radiation on terrestrial plant communities, and Turner (1975) prepared a similar review for native animal populations. This literature also is restricted primarily to X- and gamma radiation, but it provides a substantial basis for understanding dose–effect relationships.

A major problem in applying this information to transuranics is that of determining the equivalent dose to critical tissues which would result from a given level of contamination. Of the 17 transuranic nuclides listed as being of some importance in the nuclear industry to the year 2000 (Energy Research and Development Administration, 1976), 13 are alpha emitters with generally infrequent emission of weak (mostly <0.07 MeV) photons. The other 4 are beta emitters with accompanying weak photon emissions. Alpha–weak-photon emitters include the particularly important nuclides ^{238}Pu, ^{239}Pu, ^{241}Am, ^{242}Cm, and ^{244}Cm. Alphas from these nuclides have energies of 5 to 6 MeV and ranges in air and biological tissue of roughly 4 cm and 40 μm, respectively (Walsh, 1970), which lend considerable complexity to the problem of dosimetry.

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From field studies in plutonium-contaminated areas, most of the plutonium associated with vegetation appears to be surficial and not incorporated within tissues. Therefore critical tissues (meristem for growth and flower bud for reproduction) may receive a widely variable dose from surface contamination, depending on the location of the material and the thickness of epidermal tissue layers. I am not aware of any studies designed to show the detailed histological distribution of transuranics in and on plant tissues in contaminated-field environments.

The effective dose to animal tissues is equally difficult to determine. The dose from inhalation and ingestion of transuranics is subject to many variables. Absorption, translocation, deposition, and retention are affected by the physical and chemical forms of the nuclide and physiology of the animal (International Commission on Radiological Protection, 1972). The environmental chemistry of plutonium is extremely complex (Wildung et al., 1977), and our overall understanding is inadequate (Dahlman, Bondietti, and Eyman, 1976).

A few studies have been conducted in which simulated fallout particles containing beta and beta–gamma emitters were administered to field plots. The studies by Murphy and McCormick (1973) and Dahlman, Beauchamp, and Tanaka (1973) come closer to the kind needed for transuranics in that the problems of dosimetry are circumvented by simply relating effects to the level of fallout simulant applied. Murphy and McCormick applied ^{90}Y-coated albite particles to experimental granite outcrop plant communities. The effects on the reproductive potential of *Viguiera porteri* treated with 0, 205, and 526 mCi/m^2 were measured. Dahlman, Beauchamp, and Tanaka applied ^{137}Cs fused to silica sand particles to 100-m^2 plots in a fescue meadow. The levels applied (22 mCi/m^2) caused measurable decreases in seed production of *Festuca arundinacea*. A similar study using ^{90}Y-tagged sand grains to produce effects on crop plants was conducted by Schulz (1971). Fallout simulants containing ^{137}Cs were also applied to field plots at Oak Ridge, Tenn., to study the effects on arthropods and small mammals (Auerbach and Dunaway, 1970).

For research findings to be integrated and understood, however, it is highly desirable to estimate the dose to critical tissues from the levels of simulants applied. In the studies cited, beta-particle doses were estimated by thermoluminescent dosimeters and various computations. The Stanford Research Institute developed fallout-particle simulants for the field studies and measurement and computational techniques for beta dosimetry (Lane, 1971; Brown, 1965; Mackin, Brown, and Lane, 1971). Similar technology could probably be applied to alpha emitters for their use in field studies.

I am not aware of any studies in which physically and/or chemically characterized transuranics have been experimentally applied to field plots at levels sufficient to cause measurable ecological effects. The safe conduct of such studies would require an area remote from human habitation and stringent health physics practice and cleanup. Such a study would be expensive, possibly hazardous, and difficult to justify. A greenhouse study involving plants growing on soil that has been heavily contaminated with transuranics is being conducted by A. Wallace and E. M. Romney at the University of California at Los Angeles. One of the objectives of this study will be the effects of alpha particles.

Another investigation that bears on the problem of biological effects of transuranics in the environment is under way at Battelle–Pacific Northwest Laboratories under the direction of R. E. Wildung. Early results indicate radiation toxicity from ^{238}Pu and ^{239}Pu to some strains of soil actinomycetes and fungi at levels of 0.7 μCi/g (soil)

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($\sim 2.5 \times 10^4$ μCi/m^2). Such toxicity was expressed as a decline in microbial numbers. Since microbes perform functions in soil that are important to plant growth, indirect effects to higher plants and animals could be elicited through microbial perturbations from plutonium in soil.

It appears that large quantities of a transuranic nuclide would be required in the field to cause obvious ecological effects. Two very crude calculations, one for higher plants and one for animals, illustrate the approximate levels of ^{239}Pu required to produce, for instance, detectable mortality.

Plant Communities

Assumptions: A grassland plant community requires a dose rate of about 40 rad/day to show measurable changes in diversity (Whicker and Fraley, 1974); the effective decay energy for ^{239}Pu is 53 MeV/d, considering an RBE of 10 (International Commission on Radiological Protection, 1960); a concentration ratio (CR = activity per gram of plant $\div$ activity per gram of soil) of 10^{-4} is assumed (Energy Research and Development Administration, 1976); the ^{239}Pu is assumed to be uniformly distributed within plant tissues and uniformly distributed in the upper 3 cm of soil, which has a bulk density of 1.2 g/cm^3. Surficial contamination is neglected.

Calculations:

Required ^{239}Pu concentration in plant tissue

$$= \frac{(40 \text{ rad/day})(6.25 \times 10^7 \text{ MeV/g-rad})}{(53 \text{ MeV/d})(3.2 \times 10^9 \text{ d/day-}\mu\text{Ci})} = 1.5 \times 10^{-2} \ \mu\text{Ci/g}$$

Required ^{239}Pu concentration in soil

$$= \frac{1.5 \times 10^{-2} \ \mu\text{Ci/g}}{10^{-4}} = 150 \ \mu\text{Ci/g}$$

$^{-4}$ μCi/g

Required ^{239}Pu application to soil

$$= (150 \ \mu\text{Ci/g})(1.2 \text{ g/cm}^3)(3 \text{ cm}^3/\text{cm}^2)(10^4 \text{ cm}^2/\text{m}^2) = 5.4 \times 10^6 \ \mu\text{Ci/m}^2$$

This value is within an order of magnitude of the soil plutonium levels that appear to evoke some toxic effects in plants under greenhouse conditions (R. E. Wildung and T. R. Garland, Battelle–Pacific Northwest Laboratories, personal communication).

Animals

Assumptions: Inhalation of suspended soil is considered the critical route of entry; human and experimental animal data and standards are used; the maximum permissible ^{239}Pu human lung burden of 1.6×10^{-5} μCi/g is achieved with a mean air concentration of 10^{-5} μCi/m^3 (International Commission on Radiological Protection, 1960); the critical concentration of ^{239}Pu in the lung required for subacute death is 1×10^{-2} μCi/g (Bair, 1974); and a mean resuspension factor of 10^{-5}/m is assumed.

Calculations:

Required air concentration

$$= \frac{(1 \times 10^{-2}\ \mu Ci/g)(10^{-5}\ \mu Ci/m^3)}{1.6 \times 10^{-5}\ \mu Ci/g} = 6.3 \times 10^{-3}\ \mu Ci/m^3$$

Required ^{239}Pu application to soil

$$= \frac{6.3 \times 10^{-3}\ \mu Ci/m^3}{10^{-5}/m} = 630\ \mu Ci/m^2$$

If these calculations approach reality, it is clear that very large applications of ^{239}Pu would be required to produce measurable ecological changes, especially in plant communities. Nevertheless, such studies, if done, would carry more credibility than crude extrapolations and simplified calculations.

Contaminated Environments

The approach of examining ecosystems that have been accidentally contaminated with transuranics is feasible and probably desirable. Because of the lack of direct experimental data and the inherent complexity and uncertainty in computational models, we should look at areas that have been contaminated to ^{239}Pu activity levels that significantly exceed worldwide fallout levels. Several such areas exist or have existed in the past. These include Rocky Flats, Trinity, several areas at the Nevada Test Site, and various sites on Enewetak atoll (in the Pacific). In addition, plutonium releases to the environment have occurred from nuclear facilities at Oak Ridge, Hanford, Mound Laboratory, Los Alamos, Savannah River, Idaho National Engineering Laboratory, and from bomber crashes in local areas in Greenland and Spain.

If sufficiently careful searches for ecological changes in contaminated areas prove to be negative, then it probably can be concluded that the observed levels had no detectable consequence. Such data should be examined in the light of laboratory information for additional assurance. If biological perturbations are discovered in contaminated areas, then it may or may not be possible to assign causal factors. In many contaminated sites, more than one toxic substance may be present, or other factors may be responsible for changes. It may be possible to offset these problems if a proper control area is available, but this should be determined before the initiation of any search for effects.

A comparison of various biological measurements between two ecologically similar study areas of substantially differing ^{239}Pu levels at Rocky Flats was conducted by T. F. Winsor* and C. A. Little† of Colorado State University and G. E. Dagle of Battelle–Pacific Northwest Laboratories (Whicker, 1976; Little, 1976). The ^{239}Pu readings from soil in the principal study areas ranged from 100 to over 20,000 d/min per gram in the upper 3 cm (2 to 400 $\mu Ci/m^2$). In addition, comparative data were obtained from control areas containing only worldwide fallout plutonium of the order of 0.1 d/min per gram

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(0.002 μCi/m^2). Biological measurements were made, including vegetation-community structure and biomass; litter mass; arthropod community structure and biomass; and small mammal species occurrence, population density, biomass, reproduction, and whole carcass and organ masses. In addition, small mammals were examined by X ray for skeletal sarcomas, microscopy for lung tumors, and necropsy for general pathology and parasite occurrence.

Although minor differences in certain biological attributes between study areas were observed, none could be related to plutonium levels. Pathological conditions and parasites were found in some rodents, but occurrence frequencies between control and contaminated areas were similar. No evidence of cancers or other radiogenic disease was found. These observations and measurements, combined with intensive field observations over a period of 5 years, led to the conclusion that plutonium contamination at Rocky Flats has not produced demonstrable ecological changes. Furthermore, the levels of plutonium observed in tissues of plants and animals in contaminated areas were insufficient to produce the doses that would be required to produce obvious biological changes.

Subcellular biological changes, such as chromosome aberrations, cannot be ruled out at Rocky Flats. Even if chromosome aberration frequencies were increased in the more highly contaminated areas, however, population-level changes would likely not persist because of the surrounding reservoir of normal genetic information. The possibility of long-term biological effects cannot be discounted either, although this would appear highly unlikely, nor can we conclude that a similar level of plutonium contamination spread over a much larger area would be without ecological consequence. The latter idea, discussed by Odum (1963), stems from the fact that population effects in a highly localized area can be readily masked by immigration of unaffected individuals and propagules from the surrounding area, emigration of affected individuals, and gene pool mixing between the contaminated and surrounding areas. Such masking might not operate, at least to the same degree, for a large area. The validity of any future studies of animals in small-size contaminated areas might be increased if a barrier were erected to prevent the animals from moving into or out of the study area.

Extensive searches for ecological changes in contaminated areas have been carried out at the Nevada Test Site (Wallace and Romney, 1972; Allred, Beck, and Jorgensen, 1965; and Rhoads and Platt, 1971). In the majority of these studies, however, the contamination consisted principally of mixed fission products, and, except for the work reported by Rhoads and Platt, the more dramatic ecological effects were generally attributed to nonradiological perturbations. The best opportunities for searching for ecological effects from plutonium alone exist in a number of areas on or adjacent to the Nevada Test Site which have been used for "safety shot" tests. These tests involved detonation by conventional explosives of plutonium in various containment configurations. Some 300 acres containing on the order of 10 μCi Pu/m^2 exist, and a few acres have levels exceeding 6000 μCi Pu/m^2 (Wallace and Romney, 1975; Martin and Bloom, 1976). Studies of small mammals and grazing cattle in these areas have failed to discover any evidence of radiogenic pathology (Romney and Davis, 1972; Smith, Barth, and Patzer, 1976). Varney and Rhoads (1977) have examined shrubs in areas assumed to have been contaminated primarily with plutonium. Although their data implied that such shrubs had increased frequencies of chromosomal aberrations in comparison to controls, the evidence was not conclusive.

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Although, as mentioned, other sites in the world have been contaminated with plutonium, I am not aware of any specific searches for ecological effects at these sites. Competent ecologists have conducted studies on plutonium distribution and behavior within many of these sites, however, and any readily apparent ecological changes would likely have been reported. I am also not aware of any sites at which other transuranics have been released at levels greater than existing plutonium levels.

Another approach to the study of contaminated environments is to examine areas containing above-normal amounts of the naturally occurring radionuclides. Many areas contain substantial quantities of natural uranium and thorium. These primordial radionuclides and many of their progeny are alpha emitters. Possibly some inferences to the transuranics could be made from studies in areas of high natural alpha activity. For instance, the rodents on Morro do Ferro in Minas Gerais, Brazil, which receive an estimated lung dose of 10^3 to 10^4 rem/yr, might provide a good study opportunity (Drew and Eisenbud, 1966). Major problems with such an approach include differences in radiation schemes and chemical properties between the naturally occurring and transuranium radionuclides. We know something about the relative toxicities of ^{239}Pu and ^{226}Ra (Thompson, 1976) but very little about the relative ecological and physiological behavior and toxicities of transuranics with other naturally occurring alpha emitters. Pochin (1976) points out some other difficulties inherent in trying to quantify biological effects of environmental radioactivity at low levels.

On the basis of data summarized by the United Nations (1972), I calculate that the upper 3 cm of soil in the United States averages roughly 0.3 μCi of natural alpha activity per square meter. A similar calculation applied to atypically high natural radiation background areas yielded alpha activities of 7 $\mu Ci/m^2$ in the upper 3 cm near Central City, Colo. (Mericle and Mericle, 1965), 50 $\mu Ci/m^2$ in local areas in Brazil (Eisenbud et al., 1964), and 200 $\mu Ci/m^2$ in the USSR (Maslov, Maslova, and Verkhovskaya, 1967).

A number of genetic and ecological studies have been done in some of these and similar areas. Rats occupying a monazite sand area in Kerala, India, had no discernible phenotypic effects (Gruneberg et al., 1966). There is, however, suggestion of radiation-induced genetic damage leading to mental retardation of humans who occupy the same region in India (Kochupillai et al., 1976). Furthermore, Gopal-Ayengar et al. (1977) report genetic alterations in plants indigenous to the monazite belt in Kerala. Cullen (1968) reported preliminary findings of a human cytogenetic study in Guarapari, Brazil, in which an apparently increased incidence of somatic chromosome aberrations in comparison to a control area was found. A high incidence of multiple-break aberrations was noted which was thought to be compatible with the presence of internal alpha emitters. These findings were apparently corroborated more recently by Barcinski et al. (1975).

Osburn (1961) observed an increased incidence of morphological anomalies and flower abortion in *Penstemon virens* growing on the more radioactive sites near Central City, Colo. However, the chemical toxicity of thorium and possibly other factors cannot be ruled out as causal. In the USSR, Maslov, Maslova, and Verkhovskaya (1967) reported various deleterious effects on reproduction, parasite infestation, and the general condition of small mammals in areas of high natural radiation. Although radiation was implied as the cause of such effects, it was not the only variable between experimental and control populations. I am not convinced from these studies that naturally occurring alpha emitters, even in the unusually high natural background regions of the world, cause demonstrable ecological consequences. Potential genetic changes in local areas would likely be disadvantageous and selected out of populations (National Academy of Sciences–National Research Council, 1972; Muller, 1950).

Application of Existing Data

Existing data can be used to predict the magnitude of human or ecological hazard from a given level of transuranic contamination. As mentioned, however, computational models that reasonably simulate actual environmental and physiological processes require many parameter values which in themselves vary with circumstances and site. Existing knowledge of appropriate parameter values for plutonium behavior in extensively studied areas, such as the Nevada Test Site, the White Oak Creek drainage at Oak Ridge, and Rocky Flats, appears sufficient to develop models with reasonable credibility. Data that could be applied to most other terrestrial environments, however, are essentially lacking. This is especially true for transuranics other than plutonium.

Complexities involved in computational models have been discussed in considerable detail by Healy (1974), Anspaugh et al. (1975), and Martin and Bloom (1976). Healy (1974) undertook the difficult task of calculating the levels of plutonium in soil which might be considered standard or guideline levels for humans residing on and deriving sustenance from such soils. The standard levels calculated could conceivably result in the attainment of maximum permissible doses for members of the public. The computations were general in application and used available experimental data and conservative assumptions. The conceptual model considered surface soil to be the major reservoir and source of plutonium and considered processes by which the material might reach the critical organs of man. These processes included resuspension, atmospheric dispersion, cloud depletion, deposition, inhalation, ingestion of soil and contaminated foods, skin absorption, and metabolic behavior following intake. The calculations suggested that 4×10^{-4} μCi ^{239}Pu/g or 25 μCi ^{239}Pu/m^2 in the top 3 cm of soil was probably a conservative standard.

Using a similar approach but with site-specific data from the Nevada Test Site, Martin and Bloom (1976) calculated that 3 nCi ^{239}Pu/g (soil) (170 μCi/m^2) could result in the nonoccupational maximum permissible dose to the lung (1.5 rem/yr) of a standard man living over and obtaining food from the soil in question. This model was presented in a lucid and practical way, and the relative degree of confidence that can be placed on each parameter used in the model was made clear. The basic approach relates intake rates for ingestion and inhalation to surface soil concentrations; human metabolic and dose calculations are based on International Commission on Radiological Protection (ICRP) models and recommended parameter values.

In the ecological context, it seems important to consider the concept of the "critical organism." Although our primary concern is focused on man, the general welfare of the human population cannot be separated from environmental quality. Legal, moral, and scientific justification exists for ensuring the protection of species other than man from environmental contaminants. Indigenous species of plants and animals, by virtue of proximity and life habitats, will receive substantially higher radiation dose rates than man at many sites likely to receive transuranic contamination. On the other hand, many wild species, because of shorter normal life-spans, may not live long enough to develop serious pathology from chronic low-level exposures. In addition, for wildlife, society is generally concerned about performance of the population, whereas for humans, we are concerned about the more limiting case of individuals (Auerbach, 1971).

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Historically, assessments of radionuclides in the environment have considered man to be the critical organism. The assumption has often been made that, if adequate protection for man is assured, we need not worry about ecological effects. Auerbach (1971) has addressed this question with the conclusion, "Present knowledge based on these and similar studies of the ecological effects of low-level chronic doses, such as could result from routine reactor releases under current standards, guidelines, and operational experience, indicates that any possible biological effects would be undetectable." Although this philosophy generally appears defensible, especially for reactor effluents as stated, I hope that we do not blindly adopt it for all situations. For example, nuclear-waste disposal could present unanticipated ecological problems in the future, possibly without causing hazardous doses to humans.

Present Status and Directions for Future Work

To add clarity, before discussing research needs and possible directions for future work, I will recapitulate what I think is the status of our knowledge on biological responses to alpha emitters in the environment. Apparently, transuranics have not been experimentally applied to study plots in the field. On the basis of limited observations of terrestrial environments accidentally or inadvertently contaminated with plutonium in the range of 10 to 1000 $\mu Ci/m^2$, no clear-cut ecological effects attributable to plutonium have been found. A few investigations have shown biological differences between areas containing natural alpha radioactivity in the range of 5 to 200 $\mu Ci/m^2$ in the top 3 cm of soil and nearby control areas. It is not clear, however, that the differences are caused by variations in radiation dose. Simulation models and available data imply that humans should be able to occupy and derive sustenance from land areas containing of the order of 20 to 200 μCi $^{239}Pu/m^2$ in the top 3 cm of soil without exceeding the nonoccupational maximum permissible dose to critical organs as recognized by the ICRP. Simplified calculations suggest that ^{239}Pu applications of roughly 1 Ci/m^2 may be required in grassland areas to cause significant mortality in plant populations. I am not aware of computational models relating ecological effects to the level of application of transuranics other than plutonium.

The general lack of confidence in the accuracy of our predictive capability at present appears to justify substantial research efforts in this area. The shortcomings of the three general approaches have been discussed; yet I see no other approaches to the problem. Therefore it seems that enhanced efforts in each area are called for with continual integration of findings from each.

For direct measurements of the relationship between levels of transuranic application and ecological effects, such applications would need to be made under controlled experimental designs. The use of shorter lived transuranics and engineered barriers to prevent unwanted dispersal of the radioactive material would reduce the risks from such an experiment. If such experiments ever become feasible, remote, controlled areas, such as the Hanford Reservation, the Idaho National Engineering Laboratory, and the Nevada Test Site, might be considered. In addition, the application of effect-inducing quantities of transuranics to terrestrial microcosms might be considered. Although direct-application experiments seem needed from a scientific viewpoint, I do not necessarily advocate them.

Areas presently contaminated with substantial quantities of transuranics should be investigated for suitability for long-term study. Areas in which higher levels of transuranics occur without a previous history of contamination with other materials, such as fission products, and for which good control areas exist would seem particularly

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valuable for study. Such areas have existed in the past (e.g., Rocky Flats), but, as a result of public concern, cleanup operations were judged more expedient than biological studies. Cleanup decisions are deserving of greater scientific input because in some cases the operation itself may expose the public to greater risk than leaving the protected material in place.

Areas that contain notably high levels of naturally occurring alpha emitters seem deserving of further study, particularly if it can be shown how results might be integrated with current knowledge of transuranic behavior and effects. Potentially valuable study areas exist in Brazil, Colorado, Wyoming, and the USSR.

In terms of theoretical efforts, it seems clear that more generally applicable models are needed. This will require more data from a greater diversity of environments, however, and a much better understanding of basic transport mechanisms. For example, we need to know how climate, vegetation, soil, and other ecosystem attributes affect model parameters that describe such processes as erosion, resuspension, assimilation, and retention. The substantial quantities of data on the environmental behavior of plutonium in the Nevada desert or in Colorado grasslands have only limited applicability to ecosystems in regions of higher precipitation. Resuspension seems to be a particularly critical process affecting the hazard of deposited transuranics, especially in arid regions. As a final point, our knowledge of the effects of pure alpha emitters on plants is far less than our knowledge on animals and is grossly inadequate. Since plants provide stability and the food base of ecosystems, this deficiency should be corrected.

From a scientific viewpoint, it is clear that additional and redirected research can be justified for transuranium elements in the environment. Social tolerance of environmental contamination with radioactive materials, however, appears to be far lower than biological tolerance. In other words, the level of contamination that appears in many cases to prompt cleanup efforts is considerably lower than that which might be expected to elicit obvious biological change. This argument might be used against continued funding for environmental transuranic research. If this is to be the case, scientists in the field may need to provide stronger justification for their work in the future.

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