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Ecological Research Needs at Hazardous Waste Disposal Sites

S. I. Auerbach

Environmental Sciences Division
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ENVIRONMENTAL SCIENCES DIVISION

ECOLOGICAL RESEARCH NEEDS AT HAZARDOUS WASTE DISPOSAL SITES

Edited by S. I. Auerbach

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CONTENTS

	Page
ABSTRACT	v
INTRODUCTION	1
BIOLOGICAL MONITORING	3
Accumulation and Availability of Toxicants	3
Biochemical Indicators of Toxicant Exposures	4
Selection of Species for Monitoring	6
CHEMICAL-SOIL-BIOLOGICAL INTERACTIONS	7
MICROBIAL ECOLOGY	11
ECOLOGICAL RISK ANALYSIS	15
INTERACTION BETWEEN FOOD WEBS AND TOXIC SUBSTANCES	18
ECOSYSTEM STABILITY AND RISK ANALYSIS	21
REFERENCES	24

ABSTRACT

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The Superfund Amendments and Reauthorization Act of 1986 includes sections calling for new efforts in ecological research specifically related to assessing threats to ecosystems from exposures to hazardous wastes. This report describes some new and developing areas of ecological research applicable to the assessment of hazards associated with chemically contaminated sites, to various cleanup approaches and their potential consequences, and to biological monitoring programs established for surveillance after remedial actions have been taken.

The ecologically relevant impacts of exposure to hazardous wastes in the environment are manifested as a deterioration of community structure and/or function at the ecosystem level or as a loss of a commercially important or ecologically critical population. The application of biological markers (biochemical indicators) offers a promising method for monitoring and predicting the long-term effects of contaminants at the population level. Research is needed to develop a quantitative relationship between the chemical contaminants and changes in the suites of biological markers. A realistic definition of exposure regimes requires quantitative and qualitative information on the composition of leachates generated from the buried waste and an understanding of how their composition is altered during transport through soil to the exposed organisms.

Little is known of the rates and pathways of microbial degradation of contaminants in the subsurface, especially of the complex contaminant mixtures that are characteristic of many hazardous waste disposal sites. There is a need to determine the impact of these wastes on the naturally occurring microbial flora and the potential for *in situ* biodegradation as well.

Research is needed to develop improved exposure models for both aquatic and terrestrial ecosystems that can be used in routine

assessments. Ecological exposure involves an array of complex processes that are not included in existing hazardous waste assessment models. Dealing with the potentially disruptive effects of toxic substances on ecosystems requires a knowledge of the flows of energy and matter and of the related ecological food web. The prediction of possible disruptive effects on a whole food web requires a detailed knowledge of the structure of the web and of the many dynamic feedback effects in the total ecosystem.

INTRODUCTION

S. I. Auerbach

The release of toxic substances from contaminated sites or waste disposal facilities can pose a threat both to human health and to the integrity of the ecosystem. The Superfund Amendments and Reauthorization Act of 1986 includes sections calling for basic research to detect toxic substances in the environment and to assess and quantify the risks to human health and the environment resulting from exposure to such substances. Protection of the environment and its various ecosystems is an explicit component of the legislation (see Sect. 121 of the Act).

Evaluating the potential for exposure to contaminants in the environment is a challenging task because of the diversity and multiplicity of the routes of exposure (air, water, food, soil particles), the large differences in the biological availability of contaminants in these media; and the complexity of molecular, biochemical, and toxicological interactions within exposed organisms, populations, and ecosystems. This complexity limits our capability either to quantify exposure to chemical or physical agents in the environment or to assess the ecological significance of such exposure. Exposure to environmental insults cannot be quantified readily by measuring body burdens of contaminants. Many deleterious chemicals are rapidly metabolized, are excreted, or are toxic at concentrations below analytical detection limits. Furthermore, the relationship between body burden and toxic response is complex and highly variable. Assessing the significance of exposure to complex mixtures of chemicals, the most realistic scenario, is even more problematic. Constructing inventories of chemical contaminants is often prohibitively difficult, time consuming, and expensive. Such inventories may ignore possible synergistic or antagonistic interactions of chemicals, which can invalidate predictions based on the toxicity of individual chemicals.

Understanding and predicting the toxicological effects of chemical contaminants on populations, communities, and ecosystems requires information not only on the types and amounts of chemicals released to the environment, but also on the organisms and biological processes that are exposed to the chemicals. The research programs outlined in this report will provide information critical to the assessment of hazards associated with chemically contaminated sites, to various cleanup approaches and their potential consequences, and to biological monitoring programs set up for surveillance after remedial actions have been taken. In addition, this program of studies will enhance our understanding of the interrelationship between organisms and their environment as well as the overall functioning of ecosystems, both of which are important for improving ecotoxicology as a predictive science.

The ecologically relevant impacts of exposure to chemical contaminants in the environment are manifested as a deterioration of community structure or function at the ecosystem level or as a loss of a commercially important or ecologically critical population. The adverse effect on the individual animal is of concern only insofar as it presages a population-level effect. However, the complexity of ecosystem interactions imposes a long latency period before deterioration of the population or ecosystem is apparent. To adequately protect the environment, such deterioration must be detected and evaluated early in the temporal sequence between exposure and irreparable ecosystem damage. There are a number of ecologic research areas that bear directly or indirectly on this challenge. They cover a broad range of activities including (a) the development and use of promising new biochemical techniques, (b) the development and application of new concepts in ecological risk analysis, (c) ecotoxicology, and (d) applications of microbial ecology in biotechnology.

BIOLOGICAL MONITORING

J. F. McCarthy and B. T. Walton

Biological monitoring is an important area in ecotoxicology that is especially relevant to the assessment of hazards associated with chemicals at waste disposal sites. The biological monitoring approach can be used to detect many chemicals which are present in amounts that are below analytical detection limits in air, water, soil, or sediments but which can be detected in biota due to their enrichment over ambient concentrations (Martinucci et al. 1983). In addition, the biological availability of chemicals present at a site can be determined through analysis of native animals for signs of toxicant exposure (Rowley et al. 1983). Because plants and animals serve as indicators of contaminant exposure over time and space, exposure histories can often be reconstructed from body burdens of contaminants (NAS 1986). In addition, toxicity testing of environmental samples, such as effluent discharges and receiving waters, is sometimes used instead of abiotic residue analyses to meet the requirements of national and state discharge permits (Roop and Hunsaker 1985; Bergman, Kimerle, and Maki 1986).

Research is needed to support the scientific basis on which biological monitoring programs are conducted, especially in the areas of biological accumulation and availability of toxicants, biochemical indicators of toxicant exposure (biomarkers), and species selection for field studies.

Accumulation and Availability of Toxicants

The properties of environmental persistence and biological accumulation were among the first to be identified as being characteristic of substances having the potential to cause environmental problems. Barker (1958) recognized this nearly 30 years ago when he described the potential for robins to receive lethal doses of DDT when feeding on earthworms that had accumulated residues of this compound from contaminated soil and decomposing leaf litter. Of

particular concern in hazard assessment is the potential for human exposures to toxic chemicals buried at waste disposal sites due to the movement of such chemicals through food chains. In addition, an understanding of the fundamentals of bioaccumulation in aquatic and terrestrial ecosystems is critical to species selection in biological monitoring programs.

The relative importance of food as a route of contaminant exposure for wildlife is an unresolved issue. Numerous studies indicate that direct uptake of contaminants from water is probably more significant than uptake from food for aquatic species (Kay 1984; Southworth et al. 1982; Spacie and Hamelink 1984); however, the importance of trophic associations in terrestrial species is less definitive (Walton and Edwards 1986; Beyer 1986). Research is needed to delineate the relative importance of trophic associations in the aquisition of contaminant body burdens for various taxa, especially in terrestrial ecosystems.

Biochemical Indicators of Toxicant Exposures

The application of biochemical indicators (biological markers) in free-living animals offers a promising method of predicting the long-term effects of exposure to environmental contaminants at the population level. There is considerable evidence in the literature to show that animals from chemically contaminated environments exhibit increased levels of P-450 enzymes or have detectable levels of DNA or protein adducts (Varanasi, Stein, and Hom 1981; Martineau et al. 1986). Free-living, or feral, animals experimentally exposed to contaminants respond with predictable changes in several biological markers (Kurelec et al. 1977; Stegeman et al. 1986; and Shugart et al. 1986). Research is needed, however, to develop a quantitative relationship between exposure to chemicals and changes in suites of biological markers, and to demonstrate that these biological markers can be used to predict a physiologically or ecologically important response such as impairment of reproductive function, loss of stamina, diminished ability to compete effectively in the environment, or increased susceptibility to disease

or parasites. Validation of these relationships will require a combination of (a) long-term controlled laboratory experiments to establish cause-effect relationships between exposure, changes in biological markers, and concomitant changes in toxicologic end points; and (b) long-term field studies documenting the detection of biological markers in naturally exposed animals and the consequent changes at the population or community level. Changes in fecundity, growth, or survival observed in laboratory studies can be incorporated in existing ecosystem simulation models to predict population and trophic responses, which can then be compared with observed field data.

Progress in the application of biological markers to natural populations of animals can be closely linked with the development of biological markers for detecting and understanding human exposure. Advances in analytical chemistry and analytical biology can be directly utilized in research on feral vertebrates, such as small mammals, birds and fish, although some markers will have to be modified (e.g., monoclonal antibodies may differ between species).

Since feral animal species share the environment with humans, they may be exposed to environmental contaminants through many of the same routes as are human populations. By proper selection of feral animals with different habitats, specific sources and routes of exposure can be identified and compared: fish for water exposure, burrowing animals for soil-borne contamination, and herbivores for warning of potential contamination of garden produce.

One of the essential research needs outlined in the Superfund legislation is identification of the hazardous waste disposal sites that pose the greatest danger to public health. Analysis of biological markers in feral animals living around waste disposal sites or in adjacent areas occupied by human populations could efficiently identify those sites having high levels of biologically available toxicants (Talmage, Shugart, and Walton 1986). Appropriate species could be collected and tested without the publicity that would attend the sampling of human populations (or of their pets). Body tissues could be analyzed destructively if necessary, thus removing some of the limitations in the application of biological markers in humans.

Studies on biological markers in feral species should be coordinated from their inception with surveys of biological markers in humans and with epidemiological studies of human health effects. With a sufficiently large and diverse data base, it may be possible to use cost-effective surveys of easily sampled feral species to improve epidemiological studies, better define the impacts of toxic waste sites on human health, and document the amelioration of hazards as technological improvements in waste treatment are developed and implemented.

Selection of Species for Monitoring

The success of any biological monitoring program depends in part on the selection of appropriate species for analysis. Although ideal criteria for species selection have been published (NAS 1986), additional data are needed to serve as a basis for choosing organisms appropriate for field monitoring at specific sites for specific kinds of contaminants. Preliminary comparative studies of terrestrial species at Oak Ridge National Laboratory indicated that the degree of contact with soil as well as food preferences may be important determinants of exposure to contaminants in the field (Talmage, Shugart, and Walton 1986). Species comparisons are needed under field conditions for various types of contaminants to determine the most appropriate species for specific biological monitoring programs.

CHEMICAL-SOIL-BIOLOGICAL INTERACTIONS

S. E. Herbes

The potential ecological and toxicological effects of contaminants released from waste disposal areas depend upon the concentrations and chemical forms of the contaminants to which the organisms are exposed. Except for those that come into direct contact with buried waste (e.g., burrowing animals and plants whose roots penetrate into the waste), the organisms will be exposed to contaminant mixtures in waste leachate. A realistic definition of exposure regimes requires quantitative and qualitative information on the composition of leachates generated from the buried waste and an understanding of how this composition is altered during transport through soil to the exposed organisms.

Listed below are some of the major unresolved issues of leachate chemistry and soil-chemical interactions which currently limit our ability to define exposures to contaminants that are present in leachates from hazardous waste disposal sites.

1. Leachates from hazardous wastes are often complex mixtures of dozens of potentially toxic contaminants. Few toxicologic studies on mixtures of more than two potentially hazardous compounds have been undertaken (e.g., Giddings, Herbes, and Gehrs 1985; Bergman, Kimerle, and Maki 1986). While similar compounds may interact in an additive manner, the ecological effects of more disparate mixtures may differ significantly from effects of individual contaminants. In particular, virtually no data are available on the effects of mixtures containing both organic compounds and metals. Studies are thus needed to characterize complex mixtures of contaminants and to determine the toxicologic effects of leachates from hazardous wastes generated under conditions that simulate as realistically as possible conditions of waste disposal sites.

2. The chemical characteristics of leachates from hazardous waste disposal sites, and their subsequent ecological and toxicological effects, are highly dependent upon the character of the waste and its disposal environment. To fully evaluate the potential effects, the

mechanisms governing leachate generation as a function of the waste's physical and chemical characteristics and disposal environment should be investigated. Certain environmental parameters, including the pH, the kind and amount of dissolved salts, and the presence of potential complexing agents, are known to be important in solubilization of toxic metals. The parameters controlling solubilization of organic contaminants and the interactions between the contaminants that promote or decrease solubilization are less well established. The importance of the disposal environment in controlling contaminant solubilization is recognized in the continuing evolution of the U.S. Environmental Protection Agency's extraction procedure for determining leachable contaminants in waste, from its original promulgation in 1982 (USEPA 1982a) through the most recent proposed modification (Fed. Reg. 1986), which uses a more realistic leaching solution. Additional studies are needed to explain more fully the principles controlling leaching of hazardous waste constituents.

3. The ecological and toxicological effects of contaminants may depend upon their physicochemical form (e.g., dissolved, particle-bound, colloidal, or associated with humics or other macromolecules). Substantial information is available concerning the effect of organic and inorganic form in metal toxicity, which has been incorporated into metal toxicity criteria (e.g., NAS-NAE 1972). Far fewer studies are available for organic compounds; and extremely few exist on complex mixtures of contaminants that are likely to be present in leachates from many hazardous waste disposal sites. Research is therefore required to elucidate the effects of contaminant form on toxicity and the potential for bioaccumulation of leachate constituents by both plant and animal communities.

4. The composition of leachate may be altered substantially due to sorptive attenuation to soil surfaces during passage through soil from the point of generation to the point of exposure of organisms. To develop the capability to predict exposure regimes for biological communities, the effects of sorption on leachate composition must be better understood. At present only relatively simple models of contaminant transport have been evaluated in laboratory or field

studies (e.g., Gillham 1982). Few quantitative data are available with which to evaluate contaminant transport through soil under natural conditions. Processes that may be of particular importance are (a) the effects of flow through macropores and the extent to which sorption coefficients (K_d's) determined with bulk material reflect sorptive attenuation in these pores (Thomas and Phillips 1979; Beven and Germann 1982), and (b) the effect of nonequilibrium kinetics (e.g., sorption by water moving through macropores during rainfall) on attenuation of sorption. Necessary studies include laboratory-scale investigations of compositional changes in leachates during passage through simulated aquifer material, development of more realistic models, and testing of predictive capabilities by monitoring the compositional changes in plumes from existing waste disposal areas. Linkage of compositional changes in effluents with biological or ecological effects is a critical component of these studies.

5. Volatile organic contaminants (VOCs) (e.g., chlorinated ethenes and ethanes) may be removed from leachates by volatilization, either directly into the air or into unsaturated soil pores. Volatilization is known to occur under some geohydrologic conditions; detection of contaminant vapors in soil has been used to detect contaminated groundwater (e.g., Glaccum, Noel, and McMillan 1983). Although volatilization is a significant pathway for removal of VOCs from surface waters, little data are available on the effect of this process in modifying contaminant concentrations in hazardous waste leachate. Furthermore, few data are available that address the importance of this process either in removing VOCs from groundwater or in providing a pathway for potential exposure of roots and burrowing animals in the unsaturated zone to volatile leachate constituents. Studies are needed to assess the importance of volatilization in both contaminant transport and ecological effects.

6. Little is known of the rates and pathways of microbial degradation of contaminants in the subsurface, particularly of the complex contaminant mixtures that are characteristic of many hazardous waste disposal sites. Hazardous waste leachates will undoubtedly

produce modifications in subsurface microbial communities, as populations respond to both toxicants and potential growth substrates present in the leachate. Because groundwater movement is generally slow, the extended contact time between organisms and groundwater contaminants creates the potential for substantial alteration of the contaminants. Such alterations may result in detoxification of the contaminants through degradation to innocuous products such as water and carbon dioxide or may result in the production of equally toxic products, such as transformation of trichloroethylene under anaerobic conditions to the carcinogen vinyl chloride (Kleopfer et al. 1985). Studies are needed to assess how microbial communities adapt to contaminant inputs and to determine what factors control the rates and end products of degradation.

MICROBIAL ECOLOGY

A. V. Palumbo and D. C. White

Research in microbial ecology related to hazardous waste disposal must be focused on two areas. First, the impact of the waste on the naturally occurring microbial flora must be evaluated, particularly as it relates to human health, for increased pathogenicity or formation of toxic metabolites. For example, the presence of hazardous waste could result in selection for heavy metal tolerance and thus in antibiotic-resistant organisms. Additionally, the metabolism of one toxic compound can result in the formation of another toxicant (e.g., formation of vinyl chloride from trichloroethylene) (Barrio-Lange et al., 1986; Vogel and McCarty 1985). The second area of research to focus on is the potential for in situ biodegradation of the waste either by the indigenous microbiota or by introduced, possibly genetically manipulated, bacteria.

The microbial communities that are present in contaminated zones need to be characterized to determine the levels of potentially hazardous bacteria or higher microorganisms. The environment of contaminated zones is likely to be quite different from that of uncontaminated areas; thus, there may be a shift in the bacterial community structure. Contaminated surface soils or subsurface sediments may harbor bacteria which would not normally persist in the environment due to nutrient limitation (Sinclair and Alexander 1984), predation by protozoans (Habte and Alexander 1977), or competition from other bacteria (Geldreich 1972). Conditions in a contaminated zone may result in alleviation of these factors due to increased nutrient supply and/or elimination of possible predators or competitors. Uncontaminated aquifers, subsurface sediments, and groundwater usually contain relatively small populations of largely gram-positive bacteria (Smith et al. 1985) which support even smaller populations of protozoans. With contamination, there is often an increase in bacterial numbers and a shift to gram-negative bacteria (Smith et al. 1985; Smith et al. 1986). Some hazardous waste disposal sites are

contaminated with at least some easily degradable compounds, and there are elevated populations of bacteria, possibly including pathogenic bacteria, at these sites. Enteric bacteria in soil are known to survive longer and possibly multiply when sufficient organic matter is present (Gerba 1985). Even in the absence of pathogenic bacteria, the elevated total bacterial populations may support populations of pathogenic amoeba.

Research on novel means to characterize microbial populations, to detect and enumerate individual species or groups of microorganisms, and to define conditions in the environment is also important. Potentially significant organisms may be present at levels which are too low to detect by conventional plate counts or Most Probable Number (MPN) techniques. Novel techniques such as DNA probes (Sayler et al. 1985), "signature" phospholipid biomarker identification (e.g. Nichols et al. 1986), fluorescent antibody techniques (Stanley, Gage, and Schmidt 1979), and RNA sequence analysis are all potentially valuable methods to characterize microbial communities. Biochemical analysis of the bacterial community may also yield valuable information about conditions in contaminated aquifers. For example, the redox environment of the bacteria can be monitored by measuring the microbial respiratory quinone/isoprenologue ratios (Hedrick and White 1986). The nutritional status of microbial communities can be estimated from the ratios of the bacterial endogenous storage polymer poly-beta-hydroxy alkanoate (PBH) to phospholipid ester-linked fatty acids (PFLA) (Findlay et al. 1985; Findlay and White 1987). Further research is necessary on the application of these techniques to environmental problems. For example, the development of DNA probe techniques that do not require microbial growth could complement analysis by "signature" lipid biomarkers.

At sites that are being considered as candidates for <u>in situ</u> biological rectification, the factors limiting microbial growth need to be investigated. Often the presence of easily degradable substances will result in utilization of much of the oxygen available in the aquifer; therefore, oxygen may be a limiting factor, at least to

aerobic degradation. However, once oxygen is provided in a cleanup effort, often as hydrogen peroxide, inorganic nutrients may become limiting. Microcosm experiments can be used to determine if additional oxygen and/or inorganic nutrients are needed, and these may be coupled with biochemical measures of respiratory quinones and nutritional status to relate the microcosm to the subsurface environment.

There are two strategies involved in the use of microorganisms to degrade toxic waste at hazardous waste sites: (a) stimulation of the growth of existing microorganisms at the site, and (b) introduction of degradative microorganisms to the site. The microorganisms may be either naturally selected organisms or genetically engineered organisms. The effects of growth stimulation of existing microorganisms and introduction of new microorganisms should be investigated and understood before the widespread use of these techniques. Some site-specific research may be needed in every case where _in situ_ treatment is considered. After some experience is obtained, the level of effort necessary at each site may be reduced if some of the potential problems prove to be very improbable. Potential problems include the stimulation of the growth of undesirable microorganisms along with the desirable degradative microorganisms.

Changes in the bacterial community during _in situ_ cleanup efforts should be monitored. Stimulation of microbial populations to high levels, up to 1000 times the size of commonly existing populations, could cause a concomitant increase in populations of pathogenic microorganisms of either prokaryotic or eukaryotic forms. Bacteria that make up a very small portion of the original population may, under the changed conditions, be elevated to levels of concern.

There are likely to be complex mixtures of hazardous materials at many of the waste disposal sites, and the biological transformation of these products should be studied before stimulation of existing populations or introduction of new organisms is attempted. Some transformation products may be of equal or greater concern than the parent compounds. For example, anaerobic degradation of trichloroethylene (TCE) can result in the formation of vinyl chloride (Barrio-Lange et al. 1986).

Investigation of the conditions in groundwater aquifers both before and during *in situ* waste degradation attempts must be made in order to predict what transformation products will be formed. If an *in situ* cleanup of a TCE-contaminated site is being considered, aerobic conditions should be maintained in the aquifer.

Other potentially important research areas include investigations of the potential for genetically engineered microorganisms to spread beyond the area in which they are introduced. This research may include investigations of the transport of bacteria in groundwater and site-specific hydrological studies and modeling of bacterial growth and transport.

Development of predictive models of transport and growth of bacteria coupled to hydrologic and chemical models would also be important for assessing the potential effectiveness of *in situ* rectification. Such models could be used to design rectification measures, including siting of injection and withdrawal wells, to minimize undesirable outcomes such as development of anaerobic zones and clogging of aquifers around the wells.

ECOLOGICAL RISK ANALYSIS

G. W. Suter

Ecological risk analysis determines the probability of undesired effects on nonhuman biota and ecosystems from environmental contamination (Barnthouse and Suter 1986). Like human health risk analysis, it is conveniently divisible into four steps: (a) risk identification, (b) exposure analysis, which estimates the concentration or duration characteristics of the exposure to a toxicant that results from a release, (c) effects analysis, which estimates the response to that exposure, and (d) risk management, which combines the results of the prior steps with policy considerations to formulate a course of action. Of these steps, exposure analysis and effects analysis are important areas of basic scientific research.

Exposures resulting from toxic waste disposal are particularly difficult to predict. In aquatic systems, they can result from a combination of slow, relatively steady, continuous processes (e.g., groundwater recharge of streams) and occasional pulse processes (e.g., overland flow or formation of seeps following periods of high precipitation) that are not well simulated by existing aquatic exposure models. The terrestrial situation is even more complicated, since terrestrial communities developing on, or adjacent to, waste disposal sites can be exposed by the movement of chemical components of the waste or by direct exposure to the waste at the site of disposal. Plants may send their roots into waste or contaminated soil, and animals have been known to burrow into, and even consume, waste (McKenzie et al. 1982; NRC 1981). The problem of chemical uptake by roots is particularly difficult because (a) plants develop roots in buried waste only if it is an appropriate rooting medium, (b) there is no accepted method of translating concentrations of chemicals in soil into plant exposures, and (c) the tests for determining the uptake of chemicals by roots are conducted in hydroponic systems that have questionable relevance to conditions in the field.

Research is needed to develop improved exposure models for both aquatic and terrestrial ecosystems that can be used in routine assessments for the prediction of exposure to components of hazardous wastes. Existing models of the transport of waste constituents have emphasized movement into groundwater and within aquifers. As the above discussion indicates, ecological exposure involves an array of complex processes that are not included in existing hazardous waste assessment models. The development of suitable exposure models for ecological risk analysis must be based on field and laboratory research (see "CHEMICAL-SOIL-BIOLOGICAL INTERACTIONS"), on an awareness of the diverse routes of exposure, and on the particular demands of routine assessment modeling. Strategies need to be improved for developing models that balance accuracy (i.e., the models must give good predictions), generality (i.e., the models must be parameterized with data that are routinely available), and utility (i.e., the models must be useful in effects assessment and must comprehensibly convey their results and the associated uncertainty to decision makers and the public).

Analysis of the ecological effects of pollutants has only recently moved beyond the direct comparison of toxicity test results and environmental concentrations to include models for extrapolation between species, life stages, and levels of biological organization (Barnthouse and Suter 1986). The following are major issues related to effects analysis for ecosystems that require research.

1. Waste leachates typically consist of complex and poorly defined mixtures of chemicals. While there is a consensus that the effects of components of mixtures are additive in lethal aqueous exposures, it is not clear whether sublethal effects can also be modeled as additive. There also is no good evidence concerning the response of plants or ecosystem processes to mixtures of pollutants in soil.

2. Since waste leachates are likely to enter ecosystems slowly and continue for long periods, acclimation and adaptation are likely to occur on the individual, population and ecosystem levels. There is currently no accepted way to consider these processes in risk analyses.

3. There is currently no good way to assess risks to plants from soil pollutants. Toxicity data are inconsistent, and the most consistent test data [seed germination, hypocotyl elongation, and seedling growth in hydroponic culture-(USEPA 1982b)] bear an unknown relationship to the responses of plants in soil.

4. Initial testing requirements and the requirements for additional testing in tiered testing schemes are based on cost and tradition rather than the expected contribution of the data to the reduction of uncertainty in the risk analysis. There is a need for research to develop decision analysis and other procedures for tying the testing requirements to the needs of risk analysis.

5. Many kinds of organisms (e.g., amphibians, reptiles, birds, ungulates, aquatic insects, and mollusks) are seldom included in testing schemes for determining the ecological effects of toxicants. Research is needed to determine the extent to which the more commonly tested organisms can serve as predictors of the responses of these seldom tested organisms.

6. There is no consensus about the level of biological organization at which ecological risk analyses should be conducted, although nearly all testing for ecological effects is conducted at the level of the individual whole organism. Research is needed to determine the relative utility of tests and models of responses at the physiological, individual, population, and ecosystem levels in predicting and explaining responses observed in field experiments and in monitoring of polluted ecosystems.

7. There is no ecological equivalent of epidemiology. Although considerable effort is being devoted to monitoring the effects of pollutants in the field, ecological risk models and procedures have been almost entirely devoted to predictive assessment based on laboratory tests rather than to elaboration of the implications of observed responses in the field. As monitoring techniques become more precise, employing more subtle markers of exposure and response to pollutants (see "BIOLOGICAL MONITORING"), it will become increasingly necessary to develop methods to determine the implications and significance of the monitoring results and to extrapolate the effects into the future.

INTERACTION BETWEEN FOOD WEBS AND TOXIC SUBSTANCES

D. L. DeAngelis

Integral to every ecosystem is a network of feeding relationships called a food or trophic web. Photosynthesizing autotrophs form the basis of such webs, supporting a superstructure of herbivores and carnivores. In addition, there are decomposers that break down the biomass of dead organisms, releasing nutrients that are available for use by autotrophs. A food web normally contains many thousands of species and an even larger number of interconnections between species. A toxicant that is taken up by one part of the food web is likely to spread by cycling through most of the remaining parts of the web.

The interactions of food webs with toxic substances should be looked at from two general points of view: (1) The transformation, transfer, and transport processes performed on the toxic substances by the food webs (commonly lumped under the term "fate") and (2) the possible disruptive effects of the toxic substances on the food web.

Consider first the processes of transformation, transfer, and transport. Toxic substances may be transformed in the food web into new forms that are more dangerous to humans: for example, the methylation of metals such as mercury by microorganisms to produce a more toxic form of the metal. The food web also serves to transfer toxicants from one type of organism to another. To continue with the example of methylmercury, in certain circumstances it can accumulate in biota such as fish and enter the human food web. Finally, the accumulation of toxic substances in mobile animals may result in these substances being transported beyond the initial area of contamination.

To be able to predict the influences of the food web, it is necessary to know its structure, especially the primary paths of flow of matter. The mobilities of critical organisms that may disperse toxic substances must also be known.

The second viewpoint, that of dealing with the disruptive effects of toxic substances to ecosystems, requires not only a knowledge of the

flows of energy and matter (i.e., rate structures, see "ECOSYSTEM STABILITY AND RISK ANALYSIS"), but also general understanding of the complex roles played by organisms. For convenience, we can divide disruptive effects into three gradations, from severe to subtle.

The most severe effects of toxic substances are disruptions to the photosynthetic or decompositional components of a food web. A severe impact on autotrophic production will cause the collapse of the whole food web. Impairment of decomposition will slow nutrient recycling and affect the autotrophs, again with possible effects on all biota in the system. The effects of both autotroph and decomposer incapacitation can be largely predicted from a knowledge of the energy and nutrient flow in the ecosystem.

The intermediate gradation of disruptions due to toxic substances encompasses the effects resulting from the loss of keystone predators, keystone mutualists, and other individual species that play pivotal roles in food web structure. Paine (1966) showed that the loss of a keystone predator could cause drastic changes in the composition of prey species in a marine intertidal community, and this idea has been applied to many other systems. Losses of pivotal species such as insect pollinators or the loss of a keystone mutualist (i.e., plants that play a critical role in maintaining populations of pollinators) can reverberate through a food web. Recognition of the possibility of this second type of disruptive effect requires more than just a knowledge of energy and material flows in the ecosystem; a detailed knowledge of the interconnections between various species, including those that do not involve material transfer, is necessary.

The most subtle gradation encompasses the inconspicuous, perhaps sublethal, effects of toxic substances. These sublethal effects may slow down prey, making them more vulnerable to predation, tip the balance in a competitive relationship, or combine in subtle ways with other factors to change species composition in an ecosystem. Predicting such effects can only be made from a detailed knowledge of the life histories of the various species present in the ecosystem and the conditions in the particular geographic locale.

Both the fate and effects of toxic substances have been the object of many studies, but few studies have addressed whole ecosystems. Single-species laboratory tests have been the principal means of predicting biological accumulation and the effects on individual species. To determine the fates of toxic substances in complex food webs, however, larger scale studies are needed. Since field studies are usually expensive and time consuming, laboratory microcosm studies of simplified ecosystems represent a practical compromise for getting approximate information on the fates of toxic substances in food webs. Mathematical modeling offers an inexpensive way to extend predictions to large, complex systems. Many computer models are now available to do this; however, the models are only as good as the information available to assign their parameter values. The prediction of possible disruptive effects on whole food webs is an extremely difficult subject, limited by the lack of knowledge of the many dynamic feedback effects in a food web. Computer models should be combined with uncertainty analysis to determine some of the possible harmful effects of toxicants on a food web, together with the uncertainty of the occurrence of these effects.

ECOSYSTEM STABILITY AND RISK ANALYSIS

S. M. Bartell

An integrated approach involving manipulative experiments, monitoring studies, and ecosystem modeling should be used to evaluate and extend current methods for estimating the ecological risk posed by toxic chemicals in terrestrial and aquatic ecosystems (e.g., O'Neill et al. 1982; O'Neil, Bartell, and Gardner 1983). The state of an ecosystem is intrinsically multidimensional. Thus, its description requires the simultaneous consideration of a wide range of structural and functional characteristics. Therefore, an ecosystem's response to, and recovery from, anthropogenic stress should be measured in reference to a comprehensive set of system descriptors. General methods of risk estimation might usefully derive from a quantitative understanding of the patterns of energy and material flux that define the rate structures* of ecosystems. Ecosystem research should therefore focus on quantification of changes in the underlying rate structures of ecosystems in relation to exposure to toxic chemicals. Implementation of this integrated approach would contribute to development of an increased capability for forecasting the ecological risk posed by toxic chemicals in ecosystems.

Estimators of ecological risk should derive from a basic understanding of the response of natural systems to perturbation. Stability theory provides a conceptual framework for understanding an ecosystem's response to, and recovery from, anthropogenic stress. Ecosystem stability can be divided into two measurable components (Webster, Waide, and Patten 1975): (1) resistance, the ability of the system to resist changes in the face of perturbation, and (2) resilience,

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*The pattern of flux along individual system pathways, when normalized to total system inputs (or outputs) and determined for all known system pathways, defines a rate structure for energy or material flow through an ecological network. System pathways may be defined by physical processes (e.g., advection) or ecological processes (e.g., uptake, predation, excretion) connecting the components of the network. A distinct rate structure can result for each material of interest (e.g., kilocalories of energy, nitrogen, lead).

the rate of ecosystem recovery following perturbation. Resistance and resilience are related to the patterns of energy and material flux through the network of biological and abiotic components that constitute the system (i.e., the rate structures). Resistance can be determined by the presence of large, frequently abiotic, components with slow turnover rates, which can buffer the effects of variations in the environment (Webster, Waide, and Patten 1975). Smaller, more dynamic ecosystem components (e.g., populations of heterotrophs) contribute to the system's resilience (O'Neill 1976). An inverse relationship between resistance and resilience has been proposed (Webster, Waide, and Patten 1975). Resistance and resilience should be predictable from the rate structure of ecosystems.

The stability theory of Webster, Waide, and Patten (1975) has been revised to emphasize the characteristic spatial and temporal scales of different ecosystems (Waide, in press). These differences in scale must be normalized when comparing different ecosystems (e.g., forests and lakes). Waide used total organic mass, total energy dissipation, and ecosystem turnover structure as scaling factors. Ecosystem turnover can be approximated by the turnover rate of the primary producers, calculated as the ratio of net primary production to autotrophic biomass. Further analysis of this scaling method and an examination of alternatives may provide the long-needed ability to make meaningful comparisons across disparate ecosystem types. This ability is required if we are to understand the relative risks posed by toxicants to different ecosystems.

State-space analysis is emerging as a powerful conceptual tool in ecology (e.g., Holling 1973) and ecotoxicology (Johnson, in press; Waide et al. 1980; Schindler et al. 1980) that facilitates quantitative summarization of the multidimensional nature of ecosystem response and recovery following stress. Using this method, the changes in the rate structure of an ecosystem determine a succession of ecosystem states. Each state can be represented by a single vector in a multidimensional phase-space. The dynamic behavior of the ecosystem state vector describes a trajectory through the phase-space. System response to perturbation can be quantified by comparing the magnitude and direction

of displacement of trajectories of systems perturbed by toxic chemicals with those of nondisturbed reference systems. Values of selected ecosystem descriptors can be used to define trajectories in phase-space. The temporal patterns of displacement of these trajectories can be used to measure system resistance and resilience in relation to chemical stress.

Correlations among descriptors of ecosystem structure and function may change in relation to natural driving variables such as light and temperature and to the patterns of disturbance. Changes in correlation structure in response to perturbation may indicate a change in the underlying rate structure of the system that can determine the system's responses to future disturbance. Therefore, an altered correlation structure, whether due to prior perturbation or simply to normal variation in driving variables, implies at least the potential for differences in response to new stresses. Analysis of correlation structures may lead to predictions about the resistance and resilience that could be extrapolated across ecosystem types.

The ratio of production to respiration in unstressed, steady-state ecosystems will on average equal 1.0. Exogenous perturbations can disturb this balance and alter the production:respiration ratio (e.g., Odum 1985). This response may be viewed as a natural consequence of the nonequilibrium thermodynamic nature of ecosystems. These complex, organized systems are sustained by the inflow and dissipation of free energy. These ecosystems exist in states of locally minimized entropy production, and perturbation of these systems would be expected to lead to greater energy dissipation. Changes in total ecosystem production and respiration in relation to toxicant exposure can thus serve as useful measures of energy flow at the ecosystem level, an integration of changes in the underlying rate structure of the ecosystem.

One result of an integrated experimental, monitoring, and modeling effort will be to increase our ability (a) to predict ecosystem responses to anthropogenic perturbations, based on the known effects of such perturbations on specific ecosystem components, and (b) to characterize the properties of terrestrial and aquatic ecosystems that determine their state and their resistance to, or ability to recover from, toxicant stress.

REFERENCES

Barker, R. J. 1958. Notes on some ecological effects of DDT sprayed on elms. J. Wildl. Manage. 22:269-274.

Barnthouse, L. W., and G. W. Suter II (eds.). 1986. User's manual for ecological risk assessment. ORNL-6251. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Barrio-Lange, G., F. Z. Parsons, R. S. Nassar, and P. A. Lorenzo. 1986. Sequential dehalogenation of chlorinated ethenes. Environ. Sci. Technol. 20:96-99.

Bergman, H. L., R. A. Kimerle, and A. W. Maki. 1986. Environmental Hazard Assessment of Effluents. Pergamon Press, New York.

Beven, K., and P. Germann. 1982. Macropores and water flow in soils. Water Resour. Res. 18(5):1311-1325.

Beyer, W. N. 1986. A reexamination of biomagnification of metals in terrestrial food chains. Environ. Toxicol. Chem. 5:863-864.

Fed. Regist. 51 114:21648-21693, (June 13, 1986).

Findlay, R. H., P. C. Pollard, D. J. W. Moriarty, and D. C. White. 1985. Quantitative determination of microbial activity and community nutritional status in estuarine sediments: Evidence for a disturbance artifact. Can. J. Microbiol. 31:493-498.

Findlay, R. H., and D. C. White. 1987. A simplified method for bacterial nutritional status based on the simultaneous determination of phospholipid and endogenous storage lipid poly-beta-hydroxy alkanoate. J. Microbiol. Methods 6:113-120.

Geldreich, E. E. 1972. Water-borne pathogens. pp. 207-241. IN R. M. Mitchell (ed.), Water Pollution Microbiology. John Wiley & Sons, New York.

Gerba, C. P. 1985. Microbial contamination of the subsurface. pp. 53-67. IN C. H. Ward, W. Giger, and P. L. McCarty (eds.), Ground Water Quality. John Wiley & Sons, New York.

Giddings, J. F., S. E. Herbes, and C. W. Gehrs. 1985. Coal liquefaction products. Environ. Sci. Technol. 19(1):14-18.

Gillham, R. W. 1982. Applicability of solute transport models to problems of aquifer rehabilitation. pp. 6-16. IN D. M. Nielsen (ed.), Proceedings of the Second National Symposium on Aquifer Restoration and Groundwater Monitoring. National Water Well Association, Worthington, Ohio.

Glaccum, R., M. Noel, and L. McMillan. 1983. Correlation of geophysical and organic vapor analyzer data over a conductive plume containing volative organics. pp. 421-427. IN D. M. Neilson (ed.), Proceedings of the Third National Symposium on Aquifer Restoration and Groundwater Monitoring. National Water Well Association, Worthington, Ohio.

Habte, M., and M. Alexander. 1977. Further evidence for the regulation of bacterial populations in soil by protozoa. Arch. Microbiol. 113:181-183.

Hedrick, D. B., and D. C. White. 1986. Microbial respiratory quinones in the environment. I. A sensitive liquid chromatography method. J. Microbiol. Methods 5:243-254.

Holling, C. S. 1973. Resilience and stability of ecological systems. Annu. Rev. Ecol. Syst. 4:1-24.

Johnson, A. R. Evaluating ecosystem response to toxicant stress: A state space approach. IN American Society for Testing and Materials Tenth Symposium on Aquatic Toxicology and Hazard Assessment, American Society for Testing and Materials, Philadelphia, Pennsylvania (in press).

Kay, S. H. 1984. Potential for biomagnification of contaminants within marine and freshwater food webs. Technical Report D-84-7, U.S. Army Engineers Waterways Experiment Station, Vicksburg, Mississippi.

Kleopfer, R. D., D. M. Easley, B. B. Haas, Jr., T. G. Deihl, D. E. Jackson, and C. J. Wurrey. 1985. Anaerobic degradation of trichloroethylene in soil. Environ. Sci. Technol. 19(3):277-280.

Kurelec, B., S. Britvic, M. Rijavec, W. E. C. Muller, and R. K. Zanh. 1977. Benzo(a)pyrene monooxygenase induction in marine fish. Molecular response to oil pollution. Marine Biol. 44:211-216.

Martineau, D., A. Lagace, P. Beland, R. Higgins, D. Armstrong, and L. Shugart. 1986. Pathology of stranded beluga whales (Delphinapterus leucas) from the St. Lawrence Estuary, Quebec, Canada. J. Comp. Pathol. (in press).

Martinucci, G. B., P. Crespi, P. Omodeo, G. Osella, and G. Traldi. 1983. Earthworms and TCDD (2,3,7, 8-Tetrachlorodibenzo-p-dioxin) in Seveso. pp. 275-283. IN J. E. Satchell (ed.) Earthworm Ecology. Chapman and Hall, London and New York.

McKenzie, D. H., L. L. Cadwell, C. E. Cushing, Jr., R. Harty, W. E. Kennedy, Jr., M. A. Simmons, J. R. Soldat, and B. Swartzman. 1982. Relevance of biotic pathways to the long-term regulation of nuclear waste disposal: A report on tasks 1 and 2 of phase I. NUREG/CR-2675, vol. 1. Pacific Northwest Laboratory, Richland, Washington.

National Academy of Sciences and National Academy of Engineering (NAS-NAE). 1972. Water quality criteria 1972, EPA-R3-73-033, U.S. Environmental Protection Agency, Washington, D.C.

National Academy of Sciences (NAS). 1986. Ecological Knowledge and Environmental Problem-Solving: Concepts and Case Studies. pp. 81-87. National Academy Press, Washington, D.C.

Nichols, P. D., J. M. Henson, C. P. Antworth, J. Parsons, J. T. Wilson, and D. C White. 1986. Assessment of microbial biomass and community structure in an aerobic halogenated hydrocarbon-degrading unsaturated soil column using phospholipid ester-like fatty acid profiles. Environ. Toxicol. Chem. (in press).

Nuclear Regulatory Commission (NRC). 1981. Draft environmental impact statement on 10 CFR Part 61, "Licensing Requirements for Land Disposal of Radioactive Waste." NUREG-0782. Office of Nuclear Material Safety and Safeguards, Washington, D.C.

O'Neill, R. V. 1976. Ecosystem persistence and heterotrophic regulation. Ecology 57:1244-1253.

O'Neill, R. V., R. H. Gardner, L. W. Barnthouse, G. W. Suter, S. G. Hildebrand, and C. W. Gehrs. 1982. Ecosystem risk analysis: A new methodology. Environ. Toxicol. Chem. 1:167-177.

O'Neill, R. V., S. M. Bartell, and R. H. Gardner. 1983. Patterns of toxicological effects in ecosystems: A modeling study. Environ. Toxicol. Chem. 2:451-461.

Odum, E. P. 1985. Trends expected in stressed ecosystems. BioScience 35:419-422.

Paine, R. T. 1966. Food web complexity and species diversity. Am. Nat. 100:65-75.

Roop, R. D., and C. T. Hunsaker. 1985. Biomonitoring for toxics control in NPDES permitting. J. Water Pollut. Control Fed. 57:271-277.

Rowley, M. H., J. J. Christian, D. K. Basu, M. A. Pawlikowski, and B. Paigen. 1983. Use of small mammals (voles) to assess a hazardous waste site at Love Canal, Niagara Falls, New York. Arch. Environ. Contam. Toxicol. 12:283-297.

Sayler, G. S., M. S. Shields, E. T. Tedford, A. Breen, S. W. Hooper, K. M. Sirotkin, and J. W. Davis. 1985. Application of DNA:DNA colony hybridization to the detection of catabolic genotypes in environmental samples. Appl. Environ. Microbiol. 46:1146-1151.

Schindler, J. E., J. B. Waide, M. C. Waldron, J. J. Hains, S. P. Schreiner, M. L. Freedman, S. L. Benz, D. R. Pettigrew, L. A. Schissel, and P. J. Clark. 1980. A microcosm approach to the study of biogeochemical systems. 1. Theoretical rationale. pp. 192-203. IN J. P. Giesy, Jr. (ed.), Microcosms in Ecological Research. DOE Symposium Series 52, CONF-781101. U.S. Department of Energy, Washington, D.C.

Shugart, L., J. McCarthy, B. Jimenez, and J. Daniels. 1986. Analysis of adduct formation in the bluegill sunfish (Leopomis macrochirus) between benzo(a)pyrene and DNA of the liver and hemoglobin of the erythrocyte. Aquat. Toxicol. (in press).

Sinclair, J. M., and M. Alexander. 1984. Role of resistance to starvation in bacterial survival in sewage and lake water. Appl. Environ. Microbiol. 48:410-415.

Smith, G. A., J. S. Nickels, J. D. Davis, R. H. Findlay, P. S. Vashio, J. T. Wilson, and D. C. White. 1985. Indices identifying subsurface microbial communities that are adapted to organic pollution. pp. 210-213. IN N. N. Durham and A. E. Redelfs (eds.), Second International Conference on Ground Water Quality Research. Oklahoma State University Printing Services, Stillwater, Oklahoma.

Smith, G. A., J. S. Nickels, B. D. Kerger, J. D. Davis, S. P. Collins, J. T. Wilson, J. F. McNabb, and D. C. White. 1986. Qualitative characterization of microbial community structure in subsurface material: A unique prokaryotic consortium responsive to organic contamination. Can. J. Microbiol. 32:104-111.

Southworth, G. R., B. R. Parkhurst, S. E. Herbes, and S. C. Tsai. 1982. The risk of chemicals to aquatic environment. pp. 88-153. IN R. A. Conway (ed.), Environmental Risk Analysis for Chemicals. Van Nostrand Reinhold Company, New York.

Spacie, A., and J. L. Hamelink. 1982. Alternative models for describing the bioconcentration of organics in fish. Environ. Toxicol. Chem. 1:309-320.

Stanley, P. M., M. A. Gage, and E. L. Schmidt. 1979. Enumeration of specific populations by immunofluorescence. pp. 46-55. IN J. W. Costerton and R. R. Colwell (eds.), Native Aquatic Bacteria: Enumeration, Activity and Ecology. American Society for Testing and Materials, Philadelphia, Pennsylvania.

Stegeman, J. J., P. J. Kloepper-Sam, and J. W. Farrington. 1986. Monooxygenase induction and chlorobiphenyls in the deep sea fish Coryphaenoides armatus. Science 231:1287-1289.

Talmage, S., L. Shugart, and B. Walton. 1986. Use of small mammals as biomonitors at polycyclic aromatic hydrocarbon contaminated field sites. Abstract 229, pp. 121 IN proceedings of the 7th Annual Meeting of the Society of Environmental Toxicology and Chemistry, SETC, Rockville, Maryland.

Thomas, G. W., and R. E. Phillips. 1979. Consequences of water movement in macropores. J. Environ. Qual. 8(2):149-152.

U.S. Environmental Protection Agency (USEPA). 1982a. Test Methods for Evaluating Solid Waste. SW-846. U.S. Environmental Protection Agency, Washington, D.C.

U.S. Environmental Protection Agency (USEPA). 1982b. Environmental effects test guidelines. EPA 560/6-82-002. U.S. Environmental Protection Agency, Washington, D.C.

Varanasi, U., J. E. Stein, and T. Hom. (1981). Covalent binding of benzo(a)pyrene to DNA in fish liver. Biochem. Biophys. Res. Commun. 103:780-787.

Vogel, T. M., and P. McCarty. 1985. Biotransformation of tetrachloroethylene to trichloroethylene, dichloroethylene, vinyl chloride and carbon dioxide under methanogenic conditions. Appl. Environ. Microbiol. 49:1080-1083.

Waide, J. B. Forest ecosystem stability: Revision of the resistance-resilience model in relation to observable macroscopic properties of ecosystems. IN W. T. Swank and D. A. Crossley, Jr. (eds.), Symposium on Forest Hydrology and Ecology at Coweeta. Springer-Verlag, New York (in press).

Waide, J. B., J. E. Schindler, M. C. Waldron, J. J. Hains, S. P. Schreiner, M. L. Freedman, S. L. Benz, D. R. Pettigrew, L. A. Schissel, and P. J. Clark. 1980. A microcosm approach to the study of biogeochemical systems. 2. Responses of aquatic laboratory microcosms to physical, chemical, and biological perturbations. pp. 204-233. IN J. P. Giesy, Jr. (ed.), Microcosms in Ecological Research. DOE Symposium Series 52, CONF-781101. U.S. Department of Energy, Washington, D.C.

Walton, B. T., and N. T. Edwards. 1986. Accumulation of organic waste constituents in terrestrial biota. pp. 73-86. IN R. C. Loehr and J. F. Malina, Jr. (eds.), Land Treatment: A Hazardous Waste Management Alternative. University of Texas, Austin, Texas.

Webster, J. R., J. B. Waide, and B. C. Patten. 1975. Nutrient recycling and the stability of ecosystems. pp. 1-27. IN F. G. Howell, J. B. Gentry, and M. H. Smith, (eds), Mineral Cycling in Southeastern Ecosystems. ERDA Symposium Series, CONF-740513. U.S. Energy Research and Development Administration (changed to U.S. Department of Energy), Washington, D.C.

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