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ASSESSING THE TRANSPORT AND FATE OF BIOENGINEERED MICROORGANISMS
IN THE ENVIRONMENT

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INTRODUCTION

The risk that a bioengineered microorganism deliberately released into the environment will have unanticipated adverse health and environmental effects is a function of: (a) its ability to proliferate in the environment in which it is released, (b) its rate or probability of transport to other environments, (c) its ability to establish and proliferate in the new environment, and (d) its effects on the invaded environment. This paper specifically deals with the problems of environmental transport and fate. Given that the microbe in question can survive and proliferate in the environment in which it is released, how likely is it to disperse away from the release site? How far and how rapidly will the transported microbes move? Can they survive long enough to reach target environments or hosts? To answer these questions, it is necessary to identify the various pathways through which microorganisms can move, to quantify their dispersal rates along these pathways, and to estimate their survival rates in air, water, and soil.

In this paper we review the methods currently available for quantifying the transport and fate of microbes in atmospheric and aqueous media and assess their adequacy for purposes of risk assessment. We review the literature on transport and fate of microorganisms, including studies of: (a) pathways of migration, (b) the survival of microorganisms during transport, and (c) methods that have been used to model microbial transport and fate. In addition, we review the transport and fate models that have been used in environmental risk assessments for radionuclides and toxic chemicals and evaluate their applicability to the problem of assessing environmental risks of bioengineered microorganisms.

It should be emphasized that this paper is strictly concerned with the transport of microorganisms between patches of favorable environment separated by unfavorable patches within which growth and reproduction are not possible (e.g., cornfields separated by forest). Phenomena occurring within favorable patches, such as proliferation and plasmid exchange, are outside the scope of this review. In addition, it should be recognized that the overall risk assessment process for bioengineered organisms will probably resemble the process employed in assessing other forms of risk. Both generic assessments of technologies and site-specific assessments of particular applications may be performed; hence, no single model will be adequate for all assessments. Limits on time, resources, and scientific understanding will, as has been the case for other environmental stresses, make it impossible to develop highly accurate numerical risk assessments based on models that faithfully reproduce every process that influences the fate and transport of microbes. The availability, accuracy, and precision of data are as important in choices of models as is mechanistic realism (Barnthouse et al. 1984).

OVERVIEW OF INFORMATION ON THE TRANSPORT AND FATE OF MICROORGANISMS IN THE ENVIRONMENT

Key factors in the transport of microorganisms in the environment include the: (1) size of the source population, (2) rate of dispersal of organisms along environmental pathways, and (3) survival of the organisms during transport. The size of the source population and the ability of the organisms to survive during transport will differ depending on the type of organisms released. Inadvertent or accidental release of

genetically-engineered microorganisms will probably involve small source populations. Microorganisms engineered to produce pharmaceuticals or industrial chemicals will be grown under controlled conditions and they can be altered or weakened to minimize their chances of survival following accidental release. However, microorganisms designed for use in agriculture or pollution control may well be altered to increase their rate of survival and growth in the environment. They may be released in large numbers, and may be transported in a viable state over long distances.

There are three major pathways for the transport of microorganisms in the environment. These are transmission by air, by water, and by animal vectors. The dispersal of pathogenic microorganisms has been assumed to be dominated by one pathway, usually direct contact (Langmuir 1980), but it has recently become evident that multiple dispersal modes are common (e.g., Langmuir 1980, Brachman 1980). It is likely that multiple dispersal modes are also important in the transport of nonpathogenic microorganisms. Disease transmission by direct contact between infected and uninfected hosts, while the dominant dispersal pathway for many pathogens, is outside the scope of this paper.

Aerial dispersal

Airborne microorganisms can originate from both human and nonhuman sources. Bacteria are injected into the atmosphere by human activities that produce dust (e.g., vehicle operation, manufacturing, and construction) and from sources such as solid waste and sewage treatment plants (Lighthart et al. 1979). Microorganisms in water can become

airborne through wave action or bubble bursting, and microorganisms in soil or vegetation can be transported into the atmosphere through wind or thermal convection.

Because of their small size, bacteria and viruses settle out of the atmosphere at slow rates and can be effectively transported over long distances. At a wind speed of 5 m/s, a 1 μm and a 5 μm bacterium would be transported, respectively, 170 and 7 km while settling 1 m. Bacteria are known to be transported in the atmosphere for hundreds to thousands of kilometers (ZoBell 1942, Bovallius et al. 1978, Bovallius et al. 1980). Virus particles could be transported even more effectively. Windborn viruses may have spread foot-and-mouth disease over 100 km of open sea to England from France (Gloster et al. 1981).

Bacteria associated with aerosols or airborne soil particles settle more rapidly. Again assuming a wind speed of 5 m/s, a 10 μm and a 100 μm particle would be transported, respectively, 830 m and 12 m while settling 1 m. However, association with particles may facilitate the long-range transport of bacteria by increasing their rate of survival (Andow 1984). Bacteria are sensitive to drying and large particles reach hygroscopic equilibrium more slowly than do small particles.

Several approaches have been used to model the aerial dispersal of microorganisms. Gaussian-plume dispersion models, modified to include bacterial death rates, have been used to predict dispersal gradients (Lighthart and Frisch 1976, Peterson and Lighthart 1977). Plant disease dispersal gradients have been modeled with turbulent diffusion models (Gregory 1973) and with empirical models (Gregory 1968, 1973).

Waterborne dispersal

Although many enteric diseases are caused by water-dispersed microorganisms (Wolman and Gorman 1931, Diesch 1970), minimal data are available on dispersal gradients in water. The transport of microorganisms in surface water depends primarily on the movement of the water itself. However, the attachment of bacteria to particles greatly complicates the problem of modeling microbial transport in this medium. Microbe-particle association is much more complicated than the partitioning of chemical contaminants, and is not amenable to modeling using simple partitioning coefficients. Bacteria use a variety of mechanisms to attach to surfaces (Costerton 1984). Moreover, the ability to attach varies among bacterial species (Pringle and Fletcher 1983) and even with culture conditions (Fletcher and McElowney 1984). Some bacteria undergoing starvation increase their hydrophobicity and thus increase their binding to solid surfaces (Kjelleberg 1984).

The transport of microorganisms in groundwater is strongly dependent on the interactions of microbes with the soil. Bacteria are strained or filtered by passing through soil (Gerba and Bitton 1984); both bacteria and viruses are adsorbed to soil particles (e.g., Gerba et al. 1981). In unsaturated soil, bacteria may move only a few centimeters (Hagedorn 1984). Much greater transport is possible in saturated soil (Hagedorn 1984, Lance and Gerba 1984), especially in coarse-textured or fractured strata (Romero 1970). Viable bacteria have been reported to move farther than 900 m in groundwater (Gerba 1984).

Animal vectors

Animal vectors are highly effective dispersal mechanisms for many microorganisms. Maguire (1963) found that algae, ciliates, flagellates, and moss spores are readily transported between water bodies by birds, insects, lizards, and racoons. Fish have been found to carry fecal coliform bacteria and streptococci within their intestinal tracts for up to 14 days when those microorganisms are present in the water (Geldreich 1972). It is likely that other organisms capable of surviving in the intestinal tracts of fish are similarly transported. Plant bacterial diseases (e.g., Schroth et al. 1974, Crosse 1957) and arboviruses (Harris and Maramorosch 1980, Maramorosch and Harris 1981) are often spread by insects. Bacterial attachment to copepods (Hug et al. 1984) or algae (Paerl and Gallucci 1985) could also be important in influencing bacterial transport in water.

Factors affecting the survival of microorganisms during transport

The relative importance of the factors influencing survival of microorganisms in the environment depends on the characteristics of both the environment and the organism. Bacteria that are resistant to starvation and abiotic stress and also have the ability to coexist with antagonistic species may be best able to survive introduction to a new environment (Liang et al. 1982).

Temperature is an important determinant of survival. In many studies, the effect of temperature is confounded with seasonal effects, therefore the effect of temperature and season are often difficult to separate. In general, survival is decreased in the summer or at high

temperatures (e.g., Van Donsel et al. 1967, Porges and Mackenthun 1963, Kibbey et al. 1978). However, there are exceptions (Schaad and White 1974). In a study of the persistence of protozoan spores in soil, Germida (1964) theorized that longer persistence at low temperatures was related to decreased activity of the indigenous soil community. Thus, the slower dieoff of fecal coliforms at lower temperatures in sludge applied to soil (Edmonds 1976) may be due to the effect of the sludge on the indigenous community. Increased temperature may also reduce survival by increasing the likelihood of starvation due to increased metabolic demand.

Moisture can be important in determining survival of bacteria in soil (Beard 1940, Young and Greenfield 1973) and air. The resistance of different bacterial strains and genera in soil to dessication is highly variable (Chen and Alexander 1973, Van Donsel et al. 1967, Osa-Afiana and Alexander 1979, 1982). Soil type also affects the survival of introduced species (Mallman and Litsky 1951, Lowe and Gray 1973, Parhad and Rao 1974). The attachment of bacteria to particles may provide some protection to airborne bacteria from dessication (Andow 1984) and adsorption of bacteria to soil may increase their persistence (Alexander 1984).

The presence of other microorganisms may reduce the chance of survival of an introduced species. Survival of bacteria introduced in to sterilized environments is often greater than survival in similar non-sterile environments (e.g., Geldreich, 1972, Sinclair and Alexander 1984). The survival of bacteria in groundwater may be better than in other aquatic systems where the total bacterial population is greater.

For example, Shigella added to wells could be recovered up to 22 days later (Geldreich, 1972) but the same organism survived only 30 minutes to 4 days in river water (Geldreich, 1972). The relative absence of eukaryotic organisms in groundwater may also increase the chances for survival of bacteria because protozoans which can reduce bacterial populations (Habte and Alexander 1977) are not present.

Nutrients can affect the survival of introduced species. Many natural environments are nutrient-poor or have indigenous populations which can effectively outcompete introduced species for nutrients. Thus, resistance to starvation is often important to survival and the addition of organic or inorganic nutrients can increase the chances of survival of introduced species (Carlucci and Pramer 1960, Ko and Chow 1977). However, the addition of nutrients can also decrease survival (Carlucci and Pramer 1960, Klein and Cassida 1967).

Physical and chemical factors such as light and pH can also influence bacterial survival. High light levels reduce bacterial survival in soils (Beard 1940, Van Donsel et al. 1967), in aerosols (Dorsey et al. 1970), and in water (Rheinheimer 1980). Survival of bacteria in water may be decreased by very low pH (<4; Beard 1940, Cuthbert et al. 1955, Van Donsel et al. 1967) or very high (>8 or 9; Nabbut and Kurayivva 1972).

A wide range of survival times has been reported for bacteria in the environment. Bitton et al. (1983) estimated first-order die-off rates for bacteria and viruses by fitting data to an exponential decay model. Estimated decay constants for bacteria in groundwater range from 0.03 to 0.32 d^{-1} (Gerba and Bitton 1984). Higher mortality rates (e.g.,

0.768 d^{-1} , Bitton and Mitchell 1974) have been observed in seawater. Mortality of airborne bacteria is believed to be much higher still. Based on literature data for laboratory cultures, Peterson and Lighthart (1977) used bacterial death rates of 0.0001 to 0.01 s^{-1} (8.64 to 864 d^{-1}) to model the fate of airborne bacteria.

Studies of the survival of bacteria in lake water and soil have shown that the data often fit a first-order exponential decay model for only the initial portion of an experiment. After a period of exponential decline, the number of bacteria often stabilizes (Liang et al. 1982, Sinclair and Alexander 1984). Based on such observations, Alexander (1984) has suggested that the survival of introduced bacteria follows several patterns. Some bacterial species show a rapid decline and are eliminated within 3 days, others show a logarithmic decline over 7-10 days, leaving a few remaining cells which persist much longer. Another pattern is a logarithmic decline for 1-14 days with little subsequent change in numbers.

All reported estimates of microbial mortality rates must be interpreted with caution. Much of the literature on the survival of microorganisms was written before it was known that bacteria can be "injured" so that they are not enumerated using standard techniques. It has been demonstrated that bacteria can become injured in drinking water, surface water, and wastewater (Babich and Stotzky 1980, Bissonnette et al. 1975, Braswell et al. 1977, LeChevallier et al. 1983). Bacteria may survive in this state longer than indicated by some studies, and under some circumstances may again be able to grow and multiply. Hence,

experimentally-determined microbial mortality rates may often underestimate the abundance of viable microorganisms in the environment.

AVAILABLE MODELS FOR QUANTIFYING ENVIRONMENTAL TRANSPORT PESTICIDE AND TOXIC CHEMICAL MODELS

In many respects, the processes governing the transport and fate of microbes resemble the processes governing the transport and fate of radionuclides, pesticides, and toxic chemicals. Like microbes, chemicals are often transported in association with aerosols, soil, and sediment particles. The movements of microbial particles are governed by the same physical forces that govern the transport of chemicals by moving air and water. As will be shown below, the transport of chemicals, like that of microbes, is complicated by adsorption/desorption kinetics and intermedia transfer (e.g., settling and volatilization). The decay of radionuclides and the biological/chemical transformation of pesticides and toxic chemicals are both analogous to the death of microbes.

A great variety of models has been developed for use in assessing health and environmental risks of these substances. Often, the same models, with minimal modification, have been adapted to all three types of substances. They vary in spatial scale from meters to tens of kilometers and in temporal resolution from seconds to years. They have been developed for both generic and site-specific assessments, with vastly differing data requirements. In this section we briefly describe the purposes and characteristics of four types of models that appear relevant to biotechnology risk assessment: atmospheric transport and deposition, runoff, surface water, and groundwater models.

Atmospheric transport and deposition models

Models for predicting the fate of radionuclides and toxic chemicals in the atmosphere can be divided into two classes: near-field and far-field. Near-field models are intended for use within 50 km of the contaminant source. Although there are many computer codes for calculating near-field behavior of atmospheric pollutants, nearly all are variants of a single basic model, the Gaussian plume. In its simplest form, the Gaussian plume predicts the diffusion and dispersion of a conservative, gaseous plume from a continuous point source elevated above the ground, under constant wind speed, and homogeneous atmospheric conditions, over uniformly flat terrain. Many variations on the basic model have been developed to account for plume buoyancy, atmospheric stratification, wet and dry deposition of particles and aerosols, contaminant degradation or decay, ground-level sources, areal sources, and complex terrain.

Much of the model development and validation has been performed in connection with studies of radionuclide releases (Slade 1968, Hoffman et al. 1978). Several of the computer codes developed for radionuclide assessment have subsequently been modified for use with other pollutants. Two examples are ATM (Raridon et al., in press), and AIRDOS-EPA (Moore et al. 1979, Travis et al. 1983). Both of these models compute ground-level atmospheric concentrations and deposition rates as functions of distance and direction from the pollutant source. Both account for decay or degradation, but neither accounts for complex terrain. Based on the validation studies that have been performed, it has been concluded (Hoffman et al. 1978) that Gaussian plume models can

predict annual average atmospheric concentrations at distances of 10 km or less over relatively flat terrain with a factor of two accuracy. Predictions over time scales of a few hours or days are much less accurate.

A variety of non-Gaussian models have been developed for predicting far-field transport (e.g., Heffter and Taylor 1975, Murphy et al. 1984) and short-term near-field transport over complex terrain (Lange 1975, U. S. Environmental Protection Agency 1977). Although they extend the range of spatial and temporal scales that can be modeled, these codes are extremely demanding in terms of meteorological data requirements and are not very flexible.

Runoff

The basic premise underlying the runoff models used in fate assessments of radionuclides and contaminants is that the surface transport of contaminants is driven primarily by rainfall, overland flow, and erosion. Consequently, the models are essentially models of the rate of soil erosion as a function of rainfall, soil characteristics, slope, and vegetative cover. Properties of the contaminants themselves, such as solubility and degradability, are modeled as simple partition coefficients and first-order rate constants.

Mills et al. (1982) described a screening-level approach to modeling contaminant deposition, runoff volume, and sediment erosion based on the Universal Soil Loss Equation (Wischmeier and Smith, 1965). The information required to employ the approach of Mills et al. (1982) includes estimates of 7 empirically-determined factors describing

rainfall, soil erodibility, slope, cover, and sediment delivery. Tables and figures for estimating these factors are provided in the report. The Universal Soil Loss Equation was originally developed to estimate the annual erosion potential of soils. Because events on shorter time scales are important in determining rates of pollutant loading-to-surface water, a modified version developed by Williams (1975) is used to model runoff and erosion during single storm events. The storm runoff model requires estimates of the runoff volume and peak runoff rate; methods for estimating both parameters are presented in the report (Mills et al. 1982).

While erosion is the basic driving force which moves substances deposited on the soil surface, substance-specific factors such as solubility, sorption, and degradation also affect rates of removal and delivery to surface water. Mills et al. (1982) present modifications of the basic soil-erosion model designed to account for the overland transport of nitrogen, phosphorus, and pesticides. The pesticide model includes both degradation and soil-water partitioning. Given estimates of: (1) pesticide application, (2) pesticide disappearance (sum of first-order rates for volatilization and degradation), and (3) octanol-water partitioning coefficient, the model calculates the total pesticide loss for a given storm event.

More complex models of runoff and erosion are available (Donigian 1981, Basta and Bower 1982, Crawford 1982). In these models, the Universal Soil Loss Equation and its empirically-derived parameters are replaced by mechanistic descriptions of hydrological processes such as

interception, evapotranspiration, infiltration, percolation, and interflow. The hydrologic cycle is modeled in continuous time, using meteorological data defined on an hourly basis. Some of the models can follow multiple areas or catchments (e.g., several fields containing different crops or a field surrounded by forest). Provisions may be made for simulating the effects of erosion control technologies on sediment and contaminant transport. Some of the most sophisticated models (e.g., Onishi et al. 1982; Patterson et al. 1983; Hydrocomp International, 1979) link runoff models for multiple catchments to surface water models. In the Unified Transport Model (Patterson et al., 1983) the runoff and surface water models are also linked to an atmospheric transport model that estimates contaminant deposition on land and water. It should be noted that, while the modeling of hydrological processes in these models is very detailed, contaminant/sediment partitioning, volatilization, and degradation are often modeled using the same partitioning coefficients and first-order rate constants employed by Mills et al. (1982).

Surface water models

The objective of surface-water modeling is to predict the movement, dilution, partitioning, and degradation of contaminants in surface water. Typical problems for which surface-water models have been developed include estimation of: (1) the position and size of plumes and/or mixing zones for point-source discharges, (2) the downstream movement rate, dispersion, and fate of contaminants following spills or other pulse-discharge events, and (3) average concentrations as a function of distance for continuous point-source or nonpoint-source discharges. As with runoff models, the basic driving forces in

surface-water models are the movement of water and sediment as determined by the law of conservation of mass and momentum. These basic physical principles are coupled with contaminant-specific estimates of sediment-water partitioning, volatilization, and transformation/degradation. Again, as with runoff models, a variety of models with varying complexity, simulation capability, and information requirements have been developed.

The best known surface water model is the Exposure Analysis and Modeling System (EXAMS, Baughmann and Lassiter 1978; Burns et al. 1982). The EXAMS model represents a water body as a series of linked compartments, within which contaminants are sorbed and desorbed from particles, volatilized, and degraded. Each compartment is assumed to be homogeneous, and contaminant inputs are assumed to be constant. The number and types of compartments and the connections between them are defined by the model user. The outputs from EXAMS consist of estimates of steady-state contaminant concentrations in each model compartment. The EXAMS model has been subjected to validation studies both in the laboratory (Lassiter 1982) and in the field (Games 1982).

Although EXAMS employs simple steady-state hydrodynamics, its environmental descriptions and process chemistry are complex. Many removal processes are modeled as second-order, and multiple ionization states of a contaminant can be simultaneously tracked. To accommodate spatial variation in the rates of chemical processes, water bodies are vertically stratified and horizontally segmented according to four compartment types. Because the contaminant and site-specific information available for many assessment problems is insufficient to utilize the

full capabilities of EXAMS, a number of scaled-down models employing the EXAMS concept have been developed. Mills et al. (1982) describe simple analytical models for all types of surface-water modeling problems; for rivers, lakes, and estuaries. Assumptions employed in these models include: the systems are at steady-state (e. g., stream flows are not time-varying) and are either completely-mixed or can be represented as a set of coupled, completely-mixed segments. Various additional simplifying assumptions are used to develop models for different classes of rivers, lakes, and estuaries; readily-evaluated classification criteria are provided for determining which model is appropriate for a given system. Di Toro et al. (1982) described a conceptually similar approach to modeling the fate of contaminants in lakes and rivers, including a detailed treatment of sediment-water partitioning, settling, and resuspension. Travis et al. (1983) described a steady-state river model that computes statistical distributions of contaminant concentrations as functions of stochastically-varying environmental parameters.

Many dynamic surface-water models have been developed for use in situations where detailed simulation of contaminant distributions in space and time are desired. The basic principles and uses of dynamic surface-water models are described in engineering textbooks (e.g., Thomann 1972). In these models, time-dependent equations of motion for water molecules, sediment particles, and contaminants are solved numerically. Examples include: (1) plume models for calculating contaminant or temperature distributions in the vicinities of point sources (Koh and Fan 1970) and (2) the various water quality simulation models described by Basta and Bower (1982) and Boutwell and Roberts

(1983). The physical and chemical processes incorporated in these models are the same ones incorporated in steady-state models such as EXAMS. However, dynamic models can simulate the effects of time-varying contaminant inputs and hydrological parameters on contaminant concentrations in time and space. Although a few three-dimensional models have been developed, most dynamic water quality models are one- or two-dimensional. Because the results obtained from dynamic models are highly site-dependent, the models are designed for site-specific applications, where detailed characterization of such parameters as; basin morphometry, background water quality, geochemistry, and hydrology is feasible.

Groundwater transport

Rainfall that does not evaporate or run off infiltrates into the soil. Hence, as a consequence of mass balance, many of the models developed for predicting contaminant runoff (e.g., Mills et al. 1982, Onishi et al. 1982, Patterson et al. 1983) also can predict infiltration of contaminated water into the soil. Except for multimedia models such as the Unified Transport Model (Patterson et al. 1983), these models do not predict the rate of movement of contaminants through the unsaturated soil column to groundwater. For this purpose, specialized models of hydrological processes in the soil column are needed. In addition to climatological characteristics (e.g., rainfall, temperature, and evapotranspiration) such models must account for soil characteristics (e.g., effective porosity, organic content, and permeability) as functions of soil depth. The best known such model is SESOIL (Bonazountas and Wagner 1981). Given data on (1) rates of contaminant

deposition, (2) site meteorology and soil characteristics in one - three layers, and (3) contaminant-specific estimates of partitioning and degradation, SESOIL estimates steady-state rates of contaminant delivery to an aquifer underlying the unsaturated zone. Information on other unsaturated zone models can be obtained from a digitized data base compiled by the International Ground Water Modeling Center (1982).

A different class of models is used to predict the movement of water and dissolved contaminants in aquifers. The rates of movement, dispersion, and diffusion of water and dissolved contaminants in saturated soil or rock are functions of hydraulic gradient, hydraulic conductivity, and effective porosity. Screening-level predictions of the rate and direction of movement of contaminants in groundwater can be obtained from models developed by Yeh (1981), Code11 et al. (1982), Nelson and Schur (1980), Enfield et al (1982), and Donigian et al. (1983). All of these models are based on analytical solutions to hydraulic equations in homogeneous media. Many of these models account for the effects of degradation and retardation due to rock/solution partitioning. They are intended for use in predicting aquifer contaminant concentrations and travel times to order-of-magnitude accuracy.

Real subsurface environments are highly heterogeneous. Models of groundwater movement in heterogeneous media have been developed for use in evaluating the suitability of sites for underground disposal of hazardous wastes. In these models, the subsurface environment is divided into compartments having with varying characteristics. Reviews of such models have recently been compiled by Moiser et al. (1980), Bachmat et al. (1978), and Science Applications, Inc. (1981). Although

spatially-complex subsurface models allow realistic simulation of contaminant migration patterns, they require detailed site-specific hydrogeological surveys and may not be appropriate for assessing risks of bioengineered microorganisms applied to the vegetation or soil surface.

EVALUATION OF AVAILABLE METHODS FOR MODELING MICROBIAL TRANSPORT AND FATE

It is apparent from the above considerations that the capability for modeling the transport and fate of microbes in the environment varies considerably among pathways and organism types. Of the three pathways of primary interest in microbial fate assessment, suitable models appear to be available for two: aerial and waterborne transport. The fundamental force responsible for the transport of both microbes and hazardous chemical substances is the mass movement of air, water, and suspended particles. Many of the models developed for assessing the environmental risks of radionuclides, pesticides, and toxic chemicals are, therefore, either directly applicable to or adaptable for use in assessing environmental risks of bioengineered microorganisms.

For atmospheric pathways, there is a very close correspondence between the processes governing the transport of microorganisms and those governing the transport of chemicals. The transport of airborne microbes, like the transport of particulate contaminants, depends primarily on dispersion, diffusion, and settling as functions of particle size. The same modeling approaches (e.g., Gaussian plume dispersion) have been employed in both cases. For aquatic pathways, the same physical processes control the transport of both contaminants and

microbes. However, the chemical processes responsible for sediment/water partitioning and volatilization of chemical contaminants are distinctly different from the analogous processes affecting microbial transport. Therefore, mechanistic models of contaminant partitioning and degradation in surface water and groundwater are not applicable to microorganisms.

The environmental data required to model the transport of microbes are essentially the same as the data required to model the transport of chemical contaminants. However, the contaminant-specific data employed in these models must be replaced by microbe-specific data. As noted above such processes as the partitioning of microbes between particles and solution and the injection of microbes into the atmosphere from surface water are not amenable to modeling by using the approaches developed for chemicals. Of even greater importance is the need for data on microbial survival. The terms in transport models which correspond to radioactive decay constants or contaminant degradation rates must be replaced by estimates of microbial death rates. It is evident from our review that, although many of the abiotic and biotic factors influencing microbial survival have been identified, few quantitative generalizations are possible.

There appear to be few, if any, existing models applicable to the third major transport pathway for bioengineered microorganisms: animal vectors. Models of pathogen dispersal via obligate animal hosts (e.g., mosquitoes) probably exist in the epidemiological literature, but these may not be applicable to modeling the passive dispersal of microorganisms by foraging or migrating animals.

CONCLUSIONS

Bacteria and viruses are readily transported in the environment by air, water, and animals. Existing models appear to be adequate for predicting the bulk transport of bioengineered microorganisms in air, surface water, and groundwater. Such models are available for a wide range of spatiotemporal scales and assessment requirements. Models developed for plant pathogen assessment and contaminant fate assessment appear adequate for modeling the mass movement of microbes used in agriculture, sewage treatment, and pollution control. In all of these situations the release of a bioengineered microbe is analogous to the application of a pesticide, the dispersal of a naturally-occurring pathogen, or the release of treated water from a sewage treatment plant.

Practical considerations suggest that, when selecting contaminant transport models for potential use in microbial risk assessments, it is desirable to emphasize models that are physics-intensive rather than chemistry-intensive. Model subroutines that estimate contaminant partitioning and degradation rates based on chemical kinetics are useless for quantifying microbial fate and may increase the difficulty of using the model. We recommend that the simpler of the existing models be used for predicting the transport of microorganisms by air and water. Examples of such models include those described by Mills et al. (1982) and Travis et al. (1983). We believe that risk assessments do not need the additional accuracy that is, in principle, achievable with high-resolution models. There seems little reason to require better than order-of-magnitude accuracy in dispersion estimates, given the much larger uncertainties regarding establishment, survival, and animal transport.

Transport of microbes by animals is a uniquely biological phenomenon that has no counterpart in environmental chemistry. Existing models appear inadequate for modeling transport by animals. Although the number of microorganisms carried by animals may be small, transport via this pathway may be highly efficient. Animals are able to transport microorganisms rapidly over long distances. Moreover, there is a high probability that an animal carrying microbes away from a field or pond will travel to another field or pond suitable for colonization.

The key fate-related problems requiring research solutions are biological rather than physical or chemical. These include: (1) the survival of microbes, especially in environments, such as groundwater, where direct observation is difficult, (2) the transport of microbes by animal vectors, and (3) the relationship between the number of viable microbes reaching a hospitable environment and the likelihood of their establishment and proliferation.

For microorganisms capable of surviving a few days while being transported, estimates of bulk transport obtained from models of the type reviewed in this paper may be irrelevant. Given the multiple pathways by which microbes may move, it is inevitable that organisms will be transported from their site of release and deposited on other sites suitable for establishment. If a single viable organism is sufficient to establish a population, then the significant questions involved in microbial risk analysis are neither the abundance of transported organisms, or their dispersal patterns in time and space, but, instead, the probability of establishment of arriving propagules and the expected

time-to-colonization for potentially suitable environments in the vicinity of a release site. In this respect, the migration and establishment of bioengineered microorganisms may be more analogous to carcinogenesis in animals than to the transport of chemicals.

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