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ANNUAL OPERATING REPORT

OF

SAN ONOFRE NUCLEAR GENERATING STATION UNIT 1

FOR 1979

ENVIRONMENTAL MONITORING AND RADIATION EXPOSURE REPORTS

SOUTHERN CALIFORNIA EDISON COMPANY
SAN DIEGO GAS AND ELECTRIC COMPANY

DOCKET NO. 50-206

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ENVIRONMENTAL RADIOLOGICAL MONITORING PROGRAM SUMMARY
SAN ONOFRE NUCLEAR GENERATING STATION - UNIT 1

Docket No. 50-206
San Diego County, California

Reporting Period JANUARY - MARCH, 1979

Medium or Pathway Sampled (Unit of Measurement)	Type and Total Number of Analyses Performed	Lower Limit of Detection (LLD)	All Indicator Locations Mean(f) Range	Location with Highest Annual Mean		Control Locations Mean(f) Range	Number of Nonroutine Reported Measurements
				Name, Distance and Direction	Mean(f) Range		
Air Filters (pCi/m^3)	β, 64	0.003	0.029 (64/64) (0.007-0.078)	Units 2/3 Switchyard	0.036 (13/13) (0.021-0.078) 0.6m/110°MAG	Huntington Beach 0.028 (12/12) (.007 - .066)	0
	^{131}I, 64	0.04	< LLD	--	--	Huntington Beach < LLD	0
Air Filters Quarterly composite (pCi/m^3)	Gross α, 5	0.0001	0.0017 (5/5) 0.0010-0.0025)	Visitor Center 0.1 mi. 15°MAG	0.0025 (1/1)	Huntington Beach 0.0017 (1/1)	0
	Sr-90, 5	0.001	< LLD	--	--	< LLD	0
	^{7}Be, 5	0.008	0.09 (5/5) (0.07-0.10)	Huntington Beach 30 mi. 300°MAG Visitor Center 0.1 mi. 15°MAG	0.10 (1/1) (1/1)	Huntington Beach 0.10 (1/1)	0
	^{137}Cs, 5	0.0004	0.0019 (5/5) (0.0015-0.0025)	Visitor Center 0.1 mi. 15°MAG	0.0025 (1/1)	Huntington Beach 0.0019 (1/1)	0
Direct Radiation (mR/Qtr.)	Accumulated Dose 14	10 mR	38.1 (14/14) (32.8-44.3)	Camp Las Pulgas 8.8 mi. 105°MAG	44.3 (1/1)	Huntington Beach 44.1 (1/1)	0

ENVIRONMENTAL RADIOLOGICAL MONITORING PROGRAM SUMMARY
SAN ONOFRE NUCLEAR GENERATING STATION - UNIT 1

Docket No. 50-206
San Diego County, California

Reporting Period April - June, 1979

Medium or Pathway Sampled (Unit of Measurement)	Type and Total Number of Analyses Performed	Lower Limit of Detection (LLD)	All Indicator Locations Mean(f) Range	Location with Highest Annual Mean		Control Locations Mean(f) Range	Number of Nonroutine Reported Measurements
				Name, Distance and Direction	Mean(f) Range		
Air Filters (pCi/m^3)	Gross Beta,61	0.003	0.034 (61/61) (0.015-0.11)	2/3 Switchyard 0.6 mi. 110°MAG	0.038 (26/26) (0.021-0.090)	0.030 Huntington Beach (0.022 -0.040)10/10	0
	^{131}I, 61	0.04	< LLD (0/61)	---	---	Huntington Beach < LLD (0/10)	0
Air Filters Qtrly. Comp. (pCi/m^3)	Gross Alpha,5	0.0001	0.0004 (4/5) (0 - 0.0011)	Visitor Center 0.1 mi. 15°MAG	0.0015 (2/2) (0.0005-0.0025)	Huntington Beach 0.0003 (1/1)	0
	^{90}Sr, 5	0.001	<LLD (0/5)	---	---	Huntington Beach < LLD (0/1)	0
	^{7}Be, 5	0.01	0.07 (4/5) 0 - 0.11	Visitor Center 0.1 mi. 15°MAG	0.095 (2/2) (0.09 - 0.10)	Huntington Beach 0.07 (1/1)	0
	^{137}Cs, 5	0.0009	0.0012 (3/5) (0-0.0026)	Visitor Center 0.1 mi. 15°MAG	0.0026 (2/2) (0.0025-0.0026)	Huntington Beach < LLD (0/1)	0
Direct Radiation (mR/Qtr)	Accumulated Dose, 15	10 mR	29.0 (15/15) (23.6 - 36.1)	Camp Las Pulgas 8.8 mi. 105°MAG	40.2 (2/2) (36.1 - 44.3)	Huntington Beach 33.9 (1/1)	0

ENVIRONMENTAL RADIOLOGICAL MONITORING PROGRAM SUMMARY
SAN ONOFRE NUCLEAR GENERATING STATION - UNIT 1

Docket No. 50-206
San Diego County, California

Reporting Period JULY-SEPTEMBER, 1979

Medium or Pathway Sampled (Unit of Measurement)	Type and Total Number of Analyses Performed	Lower Limit of Detection (LLD)	All Indicator Locations Mean(f) Range	Location with Highest Annual Mean		Control Locations Mean(f) Range	Number of Nonroutine Reported Measurements
				Name, Distance and Direction	Mean(f) Range		
Direct Radiation (mR/Qtr.)	Accumulated Dose, 16	10 mR	35.5 (16/16) (29.6 - 46.6)	SSW Site Bndry, 0.1 mi. 130°MAG	46.6 (1/1)	Huntington Beach 37.3 (1/1)	0
Drinking Water Filtrate (pCi/ℓ)	Gross Alpha,9	5.0	0.7 (2/9) (0 - 4)	San Clemente 3.5 320° MAG	0.4 (1/9) (0 - 4.0)	Huntington Beach (0 - 2) (1/3)	0
	Gross Beta, 9	0.5	10.4 (9/9) (4.4 - 20)	Tri-Cities 8.7 320°MAG	11.78 (8/8) (9 - 17)	Huntington Beach (6.- 8)6.1 (3/3)	0
Drinking Water Filtrate Qtrly. Composite (pCi/ℓ)	Gross Alpha, 3	5.0	< LLD (0/3)	--	--	Huntington Beach < LLD (0/3)	0
	Gross Beta, 3	0.5	10 (3/3) (8 - 12)	Tri-Cities 8.7 320°MAG	14.0 (3/3) (12-17	Huntington Beach 6.8 (4.4-8) (3/3)	0
Drinking Water Solids (pCi/ℓ)	Gross Alpha, 9	5.0	< LLD (0/9)	San Clemente 3.5 320°MAG	0.07 (2/9) (0 - 0.4)	Huntington Beach < LLD (0/3)	0
	Gross Beta, 9	0.5	1.0 (9/9) (0.4 - 2.6)	Tri-Cities 8.7 320°MAG	1.61 (8/8) (0.6 - 2.6)	Huntington Beach 1.0 (0.9-1.2) (3/3)	0

ENVIRONMENTAL RADIOLOGICAL MONITORING PROGRAM SUMMARY

SAN ONOFRE NUCLEAR GENERATING STATION - UNIT 1

Docket No. 50-206
San Diego County, California

Reporting Period JANUARY-SEPTEMBER, 1979

Medium	# of Sampling Locations	Total No. of Samples	# of Locations Above Local Background	SAMPLING LOCATION WITH HIGHEST MEAN* Name	Location	Low	Mean	High	Units
AIR SAMPLING									
Weekly	5 (1)	β 190	0	Units 2/3 Switchyard	0.6 mi 110°MAG	0.021	0.036	0.090	pCi/m³
	5 (1)	I-131 190	0	(All samples were below detection limits)					pCi/m³
Quarterly Composite	5 (1)	α 15	0	Visitor Center	0.1 mi. 15°MAG	0.0005	0.0013	0.0025	pCi/m³
	5 (1)	Sr-90 15	0	(All samples were below detection limits)					pCi/m³
	5 (1)	Be-7 15	0	Visitor Center	0.1 mi 15°MAG	0.09	0.10	0.10	pCi/m³
	5 (1)	Cs-137 15	0	Visitor	0.1 mi 15°MAG	0	0.0017	0.0026	pCi/m³
DIRECT RADIATION									
Quarterly	16 (2)	46	0	SSW Site Boundary	0.1 mi 130°MAG	46.6	46.6	46.6	Accumulative Dose mR/Qtr.

U.S. NUCLEAR REGULATORY COMMISSION

Revision 1
April 1977

REGULATORY GUIDE

OFFICE OF STANDARDS DEVELOPMENT

REGULATORY GUIDE 1.113

ESTIMATING AQUATIC DISPERSION OF EFFLUENTS FROM ACCIDENTAL AND ROUTINE REACTOR RELEASES FOR THE PURPOSE OF IMPLEMENTING APPENDIX I

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A. INTRODUCTION

Section 20.106, "Radioactivity in Effluents to Unrestricted Areas," of the Commission's regulations in 10 CFR Part 20, "Standards for Protection Against Radiation," establishes limits on concentrations of radioactive material in effluents to unrestricted areas. Paragraph 20.1(c) of 10 CFR Part 20 states that, in addition to complying with the limits set forth in that part, licensees should make every reasonable effort to maintain releases of radioactive materials in effluents to unrestricted areas as far below the limits specified as is reasonably achievable (ALARA).

Section 50.34a, "Design Objectives for Equipment to Control Releases of Radioactive Material in Effluents - Nuclear Power Reactors," of 10 CFR Part 50, "Licensing of Production and Utilization Facilities," sets forth design objectives for equipment to control releases of radioactive effluents from light-water-cooled nuclear power reactors. Section 50.36a, "Technical Specifications on Effluents from Nuclear Reactors," of 10 CFR Part 50 further provides that, in order to keep power reactor effluent releases as low as is reasonably achievable, each operating license will include technical specifications on effluent discharge limits, operating procedures for installation, use, and maintenance of effluent control equipment, and requirements for reporting measured releases of radionuclides to the environment.

Appendix I, "Numerical Guides for Design Objectives and Limiting Conditions for Operation to Meet the Criterion 'As Low As Is Reasonably Achievable' for Radioactive Material in Light-Water-Cooled Nuclear Power Reactor Effluents," to 10 CFR Part 50 provides numerical guidance for radioactive effluent design objectives and technical specification requirements for limiting conditions of operation for light-water-cooled nuclear power plants.

To implement Appendix I, the staff has developed a series of guides that present methods acceptable to the staff for calculating preoperational estimates of effluent releases, dispersion of the effluent in the atmosphere and different water bodies, and the associated radiation doses* to man. This guide describes basic features of calculational models and suggests methods of determining values of model parameters for the estimation of aquatic dispersion of both routine and accidental releases of liquid effluents. The methods described herein are general approaches that the NRC staff has adopted for the analysis of routine and accidental releases into various types of surface water bodies. Models for the ground-water pathway are not covered in this guide. Those few cases where the ground-water pathway makes a significant contribution to the dose estimates will be analyzed on a case-by-case basis. Standards for analysis of releases to ground water are currently being developed by the American Nuclear Society and will be published by the American National Standards Institute.

B. DISCUSSION

Radioactive material in aqueous effluents may be released from nuclear power stations, either routinely or accidentally, into a variety of receiving surface water bodies, including nontidal rivers, lakes, reservoirs, cooling ponds, estuaries, and open coastal waters. This material is dispersed by turbulent mixing and by streamflow in rivers, by tidal or nontidal coastal currents in estuaries and coastal waters, and by internal circulation or flow-through in lakes, reservoirs, and cooling ponds. Parameters influencing the dispersion patterns and concentration reduction near a site include the direction and speed of flow of currents, both natural and plant-induced, in the receiving water; the intensity of turbulent mixing; the size, geometry, and bottom topography of the water body; the location of effluent discharge in relation to the receiving water surface and shoreline; the amount of recirculation of previously discharged effluent; the characteristics of suspended and bottom sediments; the sediment sorption properties; and radioactive decay.

This guide describes calculational models acceptable to the NRC staff for estimating aquatic dispersion of routine or accidental releases of radioactive material from a nuclear power station to a surface water body. The models discussed include both simplified models having straightforward analytical solutions and more complex models requiring numerical solution. In general,

*In this guide, the term "dose," when applied to individuals, is used instead of the more precise term, "dose equivalent," as defined by the International Commission on Radiological Units and Measurements (ICRUM).

the modeling techniques discussed represent adaptations of work currently available in the literature. Because of increasing environmental concerns during the past decade, considerable effort has been expended in advancing the state of the art of water quality simulation and thermal plume modeling. The models discussed herein draw heavily from this body of information.

Although specific models are considered, they are intended to represent specific classes of models. Furthermore, discussions of particular techniques for determining model parameters are intended to provide guidance and to stress the desirability of determining these parameters from physical principles or measurements. Applicants may, however, use modeling techniques other than those considered herein. In particular, physical hydraulic models that may have been constructed for hydrothermal studies or other purposes may often be used as reliable predictive tools for radionuclide dispersion. Tracer release studies conducted in situ can provide accurate predictions without need for a model.

The degree of realism inherent in each model described in this guide depends on the ability of that model to account for the physical processes involved and the validity of model coefficients and assumed future flow fields. As a general rule, more complex models are capable of yielding more realistic results. However, a realistic model requires realistic input data, and little is gained by using highly sophisticated calculational models when the input parameters are ill-defined. The simplest models are closed-form analytical solutions of the governing transport equations. Such solutions are possible only for simplified cases. It is seldom possible to obtain analytical solutions for time-dependent flow fields or for complex receiving-water geometry. Consequently, any analytical solution should be carefully assessed by the applicant to ascertain the conditions under which the model might be a valid predictive tool. Simplified models do not necessarily produce conservative results. If such models are used, it is the applicant's responsibility to demonstrate their degree of conservatism. The staff's position on such demonstration is presented in Section C of this guide.

In identifying liquid pathways to man, applicants should identify the location of water users, the types of uses, and the usage estimated out to a distance of 50 miles from the site. Because of high usage rates along many streams and estuaries, the effects of water usage on the spatial and temporal distribution of flows should be estimated. In addition, water usage upstream of a nuclear plant can alter flows at or downstream of the plant. This guide presents an acceptable methodology for evaluating water usage and the consequences thereof on streams and estuaries receiving routine or accidental releases of radionuclides from nuclear power plants.

The ability of suspended and bottom sediments to absorb and adsorb radioactive nuclides from solution, thereby concentrating otherwise dilute species of ions, may create a significant pathway to man. Sorption by sediment is also an important mechanism for reducing the area of influence of plant releases. Unfortunately, the state of the art in evaluating sediment-related effects is less advanced than in other engineering disciplines. Consequently, the transport models discussed in this guide do not explicitly include sediment uptake mechanisms. Until reliable generalized sediment uptake and transport models become available, the NRC staff will rely on existing field studies and the staff's and consultants' experience to determine the level of conservatism or realism of the applicant's estimates. If the applicant elects to take credit for removal of certain ions from the surface waters by the process of sediment uptake, verification using site-related field data will be necessary.

C. REGULATORY POSITION

1. The transport and water use models described in Appendix A of this guide are acceptable to the NRC staff for use in calculating the potential annual average radiation doses to the public that may result from radioactive material in liquid effluents routinely released to surface water bodies. No general models for transport in ground water are included, as such analyses are considered to be site-specific. The models in Appendix A are also acceptable to the staff for analyzing the dispersion and dilution of accidental spills of radioactive material in liquids to surface water bodies. Standards for ground-water analysis, currently being prepared by the American Nuclear Society for publication by the American National Standards Institute, will be evaluated for acceptability by the NRC staff when completed.

2. Although specific models are cited in Appendix A of this guide, the citations are intended to provide guidance in the selection of model types rather than to specify models. Applicants may use models other than those described in Appendix A, but should justify fully the analytical techniques, assumptions, and level of conservatism of the model ultimately chosen.

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*Lines indicate substantive change from previous issuance.

3. The choice of a specific model, values of input parameters, and assumed future flow fields is the responsibility of the applicant. The NRC staff recognizes that the applicant may choose initially to use simplified models employing demonstrably conservative assumptions. The most conservative assumption would be no dilution of the radwaste discharge and a travel time of zero. In some cases, this extreme approach may be acceptable for calculation of dose to man. Should the results of this initial analysis support a conclusion of compliance with Appendix I of 10 CFR Part 50, no further effort is indicated. However, if compliance is not demonstrated by the simplified analysis, more refined and more realistic analyses of liquid transport may be undertaken. The NRC staff will also consider such analyses acceptable provided the applicant establishes the realism of the model, coefficients, parameters, and flow field.

D. IMPLEMENTATION

The purpose of this section is to provide information to license applicants and licensees regarding the NRC staff's plans for implementing this regulatory guide.

This guide reflects current Nuclear Regulatory Commission practice. Therefore, except in those cases in which the license applicant or licensee proposes an acceptable alternative method, the method described herein for complying with specified portions of the Commission's regulations is being and will continue to be used in the evaluation of submittals for operating license or construction permit applications until this guide is revised as a result of suggestions from the public or additional staff review.

APPENDIX A

LIQUID EFFLUENT TRANSPORT AND WATER USE MODELS

This appendix describes transport and water use models for calculating the radiation doses to the public that may result from radioactive material in liquid effluents released to surface water bodies. Symbols used in this description are defined in the "List of Symbols" following the appendix.

1. INITIAL DILUTION

Initial dilution of liquid radioactive effluents (e.g., dilution upon discharge to the receiving water body) is often accomplished by using relatively high-velocity surface or submerged jets or multiport diffusers. Mathematical modeling of such discharges requires solution of the conservation equations applicable to buoyant jets. These equations are solved routinely as a part of the near-field analysis for thermal discharges. Initial dilution rates for waterborne radionuclides should be obtained directly as an integral part of the thermal analysis.

Applicable near-field models are in common usage throughout the industry and are not discussed in detail herein. Jirka et al. (Ref. 1) and Dunn et al. (Ref. 2) discuss in detail the theory and ranges of applicability of near-field models. These references should be consulted for guidance in determining the modeling approach to be used for a specific problem. General remarks on surface and submerged jet discharges are presented below.

For surface discharges, acceptable initial dilution analyses may be obtained from the models of Stolzenbach and Harleman (Ref. 3), Stolzenbach et al. (Ref. 4), Prych (Ref. 5), Shirazi and Davis (Ref. 6), and Pritchard (Refs. 7 and 8). Dilution estimates for surface discharges require a careful assessment of the adverse effects of shoreline and bottom interference. Methods for estimating the magnitude of these boundary effects under given receiving water conditions are discussed in detail in Reference 1.

Estimates of dilution from submerged discharges require careful analysis of the flow conditions in the immediate vicinity of the discharge. The two possible flow conditions, stable or unstable, depend on the discharge and receiving water characteristics. Under stable conditions the discharge, upon reaching the free surface, spreads laterally in the form of a stable density current. As a result, there is little re-entrainment of previously discharged water. Such stable discharges can be characterized as "deep-water." Unstable or "shallow-water" discharges are characterized by counterflow which causes re-entrainment of previously mixed effluent into the discharge jet. Application of deep-water jet models to shallow-water discharges can result in serious overestimation of initial dilution. References 1, 9, and 10 discuss in detail the behavior of stable and unstable discharges and stability criteria for various types of submerged discharges. In practice, the results of a stability analysis will determine whether a "deep-water" or "shallow-water" model should be used for a given discharge-receiving water system.

For deep-water (stable) conditions the commonly used submerged jet models of Koh and Fan (Ref. 11) and Hirst (Ref. 12) and similar models are applicable provided the thickness of the buoyant surface layer is taken into account.

For shallow-water (unstable) conditions the above models (and other similar deep-water models) are not applicable and their use will result in predicting excessively high dilution. The models of Lee et al. (Ref. 10) and Jirka and Harleman (Ref. 9), with appropriate stability analyses, are directly applicable to either deep- or shallow-water discharges.

2. NONTIDAL RIVERS

a. Model Formulations

(1) Steady-State Stream Tube Model

Application of the models herein is restricted to those portions of the river removed from influences of the discharge. Initial dilution near the point of discharge is usually controlled by turbulent mixing induced by momentum effects of the discharge jet. Techniques for the determination of initial dilution were discussed in Section 1 of this appendix.

For nontidal rivers the flow is assumed to be uniform and approximately steady. Under these conditions, the diffusive transport in the flow direction may be neglected compared with the advective transport (Ref. 13). It has been shown that far-field transport of dissolved constituents in rivers can be satisfactorily treated by a two-dimensional model in which vertical variations of velocity and concentration are averaged out (Refs. 14, 15, and 16). Such a model, however, retains transverse variations of river bottom topography and velocity.

Consider a section of a steady natural stream as shown in Figure 1. The origin of the coordinate system is placed on the near shore. The x-axis is taken positive in the downstream direction, the z-axis is directed vertically downward from the water surface, and the y-axis is directed across the stream. The steady-state mass balance equation for a vertically mixed radionuclide concentration may be written (Ref. 14) as follows:

$$ud\frac{\partial C}{\partial x} = \frac{\partial}{\partial y}\left(K_y d\frac{\partial C}{\partial y}\right) - (\lambda d)C \qquad (1)$$

where

- C is the radionuclide concentration (activity/volume);
- d is the stream depth;
- K_y is the lateral turbulent diffusion coefficient (vertically averaged, two dimensional);
- u is the stream velocity; and
- λ is the decay coefficient and is = (ln 2)/half life.

Since, for a real stream, u and d will be functions of the transverse coordinate y, Equation (1) will generally not have a closed-form analytical solution. A more tractable form of the equation is obtained through introduction of a new independent variable q, defined by

$$q = \int_0^y (ud)dy \qquad (2)$$

The quantity q is the cumulative discharge measured from the near shore. Hence, as $y \to B$, $q \to Q$ where B is the river width and Q is the total river flow.

Substitution of Equation (2) into Equation (1) yields the following transport equation:

$$\frac{\partial C}{\partial x} = \frac{\partial}{\partial q}\left[(K_y ud^2)\frac{\partial C}{\partial q}\right] - \frac{\lambda}{u}C \qquad (3)$$

In the decay term, the velocity u may be replaced, to a good approximation, by the sectional mean value $\bar{u}$. If this is done, the decay term may be removed through the transformation

$$C(x,q) = \chi(x,q)e^{-\frac{\lambda x}{\bar{u}}} \qquad (4)$$

The result is the following transport equation for the nondecaying concentration χ:

$$\frac{\partial \chi}{\partial x} = \frac{\partial}{\partial q}\left[(K_y ud^2)\frac{\partial \chi}{\partial q}\right] \qquad (5)$$

The quantity $K_y ud^2$ is known as the "diffusion factor." Yotsukura and Cobb (Ref. 14) have shown that the variable diffusion factor may be replaced by a constant factor $\overline{K_y ud^2}$, where

$$\overline{K_y ud^2} = \frac{1}{Q}\int_0^Q K_y ud^2 dq$$

is the discharge-weighted mean value. Equation (5) may now be written

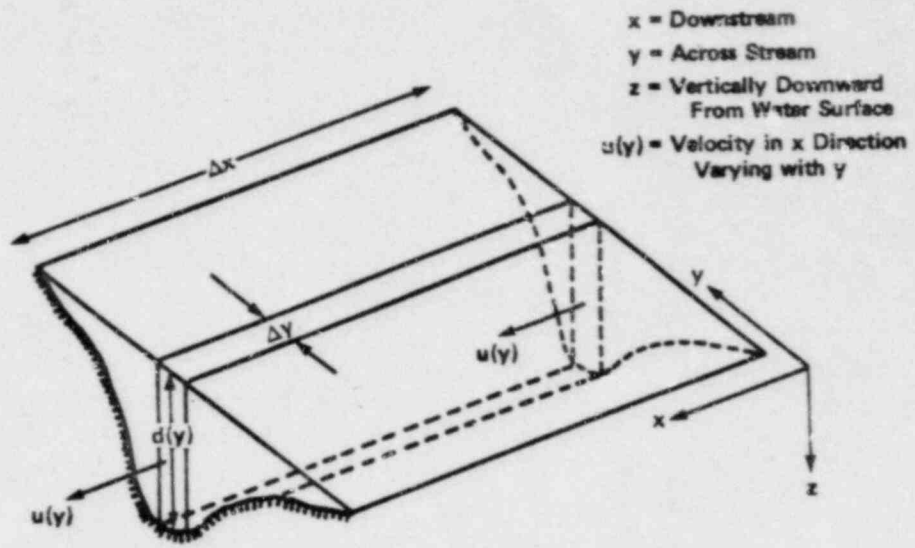


FIGURE 1. MODEL OF AN INFINITESIMAL STREAM TUBE IN A NATURAL STREAM
(Redrawn from Yotsukura and Cobb, Ref. 14.)

$$\frac{\partial \chi}{\partial x} = D \frac{\partial^2 \chi}{\partial q^2} \qquad (6)$$

where

$D = \overline{K_y u d^2}$ = constant diffusion factor

Equation (6) is a standard diffusion equation which has a closed-form analytical solution.

Assume a steady vertical line source discharge emitting a constant W Ci/sec is located at $x = 0$, $y = y_s$. Since there is a one-to-one correspondence between the transverse distance y and the cumulative discharge q, the vertical line source discharge may be located at $x = 0$, $q = q_s$. A closed-form solution to Equation (6) that satisfies the condition that there be no flux of material across the bounding surfaces is given by

$$\chi = \frac{W}{Q}\left[1 + 2 \sum_{n=1}^{\infty} e^{-\frac{n^2\pi^2 Dx}{Q^2}} \cos\frac{n\pi q_s}{Q} \cos\frac{n\pi q}{Q}\right] \qquad (7)$$

This expression, although of different form, is equivalent to Equation (14) of Reference 14.

If the liquid effluent is injected as an area source perpendicular to the river flow, the solution may be obtained by integration of Equation (7) over the source dimensions. If the source is located in the river between distances y_{s1} and y_{s2} (cumulative discharges q_{s1} and q_{s2}), the area source solution may be obtained from Equation (7) by integration with respect to q_s between the limits $q_{s1} < q_s \leq q_{s2}$.

$$\chi_{Area(\perp)} = \frac{W}{Q}\left[1 + 2 \sum_{n=1}^{\infty} e^{-\frac{n^2\pi^2 Dx}{Q^2}} \frac{\sin n\alpha}{n\alpha} \cos\frac{n\pi}{2}\left(\frac{q_{s2} + q_{s1}}{Q}\right) \cos\frac{n\pi q}{Q}\right] \qquad (8)$$

where

$$\alpha = \frac{\pi}{2}\left(\frac{q_{s2} - q_{s1}}{Q}\right)$$

Note that the more familiar solutions for the concentration, as a function of x and y in a uniform, straight, rectangular channel of constant velocity U, can be obtained immediately from Equations (7) and (8) through the transformation of the terms within the brackets:

$$\begin{Bmatrix} D/Q^2 \\ q \\ Q \end{Bmatrix} \longrightarrow \begin{Bmatrix} K_y/UB^2 \\ y \\ B \end{Bmatrix}$$

The more general forms given by Equations (7) and (8), however, are preferable, since they are applicable to irregularly shaped channels. Yotsukura and Sayre (Ref. 16) have recently generalized Equations (5) and (6) so that these can be applied to any nonuniform channel with a minor modification to the diffusion factor.

(2) Transient Release Model

In many cases, routine releases of radioactive effluents are batched and infrequent, rather than continuous. In such cases, it may be important to calculate concentrations as a function of both time and space. The concentration in a straight, rectangular channel corresponding to the instantaneous release of a finite quantity of material from a vertical line source at $x = 0$ and $y = y_s$ is:

$$C = \frac{M}{(4\pi K_x t)^{1/2} A} \exp\left\{-\frac{[x - ut]^2}{4K_x t} - \lambda t\right\}\left[1 + 2 \sum_{n=1}^{\infty} \exp\left(-\frac{n^2\pi^2 K_y t}{B^2}\right) \cos\frac{n\pi y_s}{B} \cos\frac{n\pi y}{B}\right] \qquad (9)$$

where

A is the cross-sectional area;

B is the channel width;

K_x is the longitudinal turbulent transport coefficient (vertically averaged, two dimensional);

M is the amount of activity released (in curies);

t is the time after the release;

and the other terms are as previously defined.

Note that this solution accounts for turbulent diffusion in the direction of flow, which may be important for short-duration releases.

The case of a more general time-dependent release may be obtained by integrating Equation (9) with respect to time:

$$C = \int_0^t \frac{Wf(\tau)}{(4\pi K_x)^{1/2} A(t-\tau)^{1/2}} \exp\left\{-\frac{[x - u(t-\tau)]^2}{4K_x(t-\tau)} - \lambda(t-\tau)\right\} \left[1 + 2\sum_{n=1}^{\infty} \exp\left(-\frac{n^2\pi^2 K_y (t-\tau)}{B^2}\right) \cos n\pi\frac{y_s}{B} \cos n\pi\frac{y}{B}\right] d\tau \qquad (10)$$

where the release rate is $Wf(\tau)$ curies/sec. In general, Equation (10) must be solved by numerical quadrature.

Near the source, convergence of the Fourier series terms in Equations (9) and (10) may be extremely slow. However, in this region, the effects of the far shore are not usually important, and the series solution may be replaced by a single image source at the near shore (see the transient lake solution, Equations (19) and (20), Section 3.a.(2)(b) of this appendix). In this case, the solutions do not involve infinite series and present no convergence problems.

b. Model Applications

(1) Steady-State Stream Tube Model

Application of the model requires determination of stream channel geometry, the cross-stream distribution of flow, and the diffusion factor at representative river cross-sections downstream of the effluent discharge. In addition, definition of stream discharge is necessary (see Section 6 of this appendix).

The preferred method of determining the flow cross-sectional distribution is by current-meter measurements using standard stream-gaging techniques. Because it is not always practical to obtain velocity measurements at every river cross-section at which concentration distributions are desired, transverse velocity distributions may be estimated from observed stream bottom profiles and the application of steady-state flow equations such as Manning's formula to channels of compound cross-section (Refs. 17 and 18).

Evaluation of the diffusion factor $\overline{K_y u d^2}$ requires a separate determination of the diffusion coefficient K_y. For steady open-channel flow, K_y can be determined from hydrodynamic properties of the channel by using Elder's empirical formula (Ref. 19):

$$K_y = \beta u^* d \qquad (11)$$

where

d is the river depth;

u^* is the shear velocity; and

β is a dimensionless constant.

(The user is not restricted to this formula. A number of alternative approaches have been published.)

For straight natural stream channels β has a value of approximately 0.23 (Refs. 14 and 16). For curved channels, however, secondary flows can lead to increased lateral mixing and the value of β is larger (Refs. 20-22). Fischer (Ref. 20), for example, has shown that the lateral mixing coefficient is increased in bending streams, varying inversely as the square of the radius of curvature. In general, to obtain realistic transport estimates, values of the lateral mixing coefficient should be determined by onsite tracer studies. Although transverse variations of K_y have not been adequately confirmed in field tests, longitudinal variation of K_y in a sharp bend has been reported by Sayre and Yeh (Ref. 22). Equations (7) and (8) may be modified as follows to account for a diffusion factor that varies in the direction of flow:

$$\overline{Dx} \rightarrow \int_0^x D(x)dx$$

If the diffusion factor is known for each river cross-section of concern, the integral can be evaluated by simple numerical integration. If the variation in D(x) is small over the river stretch under consideration, then Equations (7) and (8) may be used directly, with the quantity D being interpreted as the mean value over the river reach.

It is useful to write Equation (7) in dimensionless form.

$$\tilde{x} = 1 + 2 \sum_{n=1}^{\infty} e^{-n^2\pi^2\tilde{x}} \cos n\pi\tilde{q}_s \cos n\pi\tilde{q} \qquad (12)$$

where

$\tilde{q} = q/Q$ is the dimensionless cumulative discharge;

$\tilde{x} = x/\frac{w}{Q}$ is the dimensionless concentration relative to the fully mixed value; and

$\tilde{x} = \frac{Dx}{Q^2}$ is the dimensionless downstream distance.

The utility of the dimensionless form is illustrated in Figure 2, which shows near- and far-shore concentrations resulting from a near-shore point discharge. For a given downstream location and given flow parameters, the dimensionless concentration for either shoreline may be obtained directly from the two curves. The near-shore concentration exhibits the expected $x^{-1/2}$ dependence for two-dimensional mixing until the influence of the far shore is felt. Both curves in Figure 2 approach unity (complete sectional mixing) for large values of $\tilde{x}$. Hence, for a given set of flow parameters, the downstream distance to sectional homogeneity ("mixing distance") can be estimated directly. (Note that the mixing distance for a shoreline discharge is four times the mixing distance for a centerline discharge.)

(2) Transient Release Model

The transient release model is formulated in this guide only for the case of a vertical line source in a straight rectangular channel, since its primary purpose is to furnish information on the time-dependent behavior of non-continuous releases. However, the model can be extended to treat other source configurations in stream tube coordinates as employed in Section 2.a.(1) of this appendix.

Application of the model requires the determination of the longitudinal turbulent diffusion coefficient K_x, in addition to the parameters necessary for the steady-state model in the previous section. The longitudinal dispersion coefficient should be obtained by site-specific tracer experiments. However, crude estimates of K_x may be obtained from the following formula, which is similar to that for the lateral diffusion coefficient (Ref. 19):

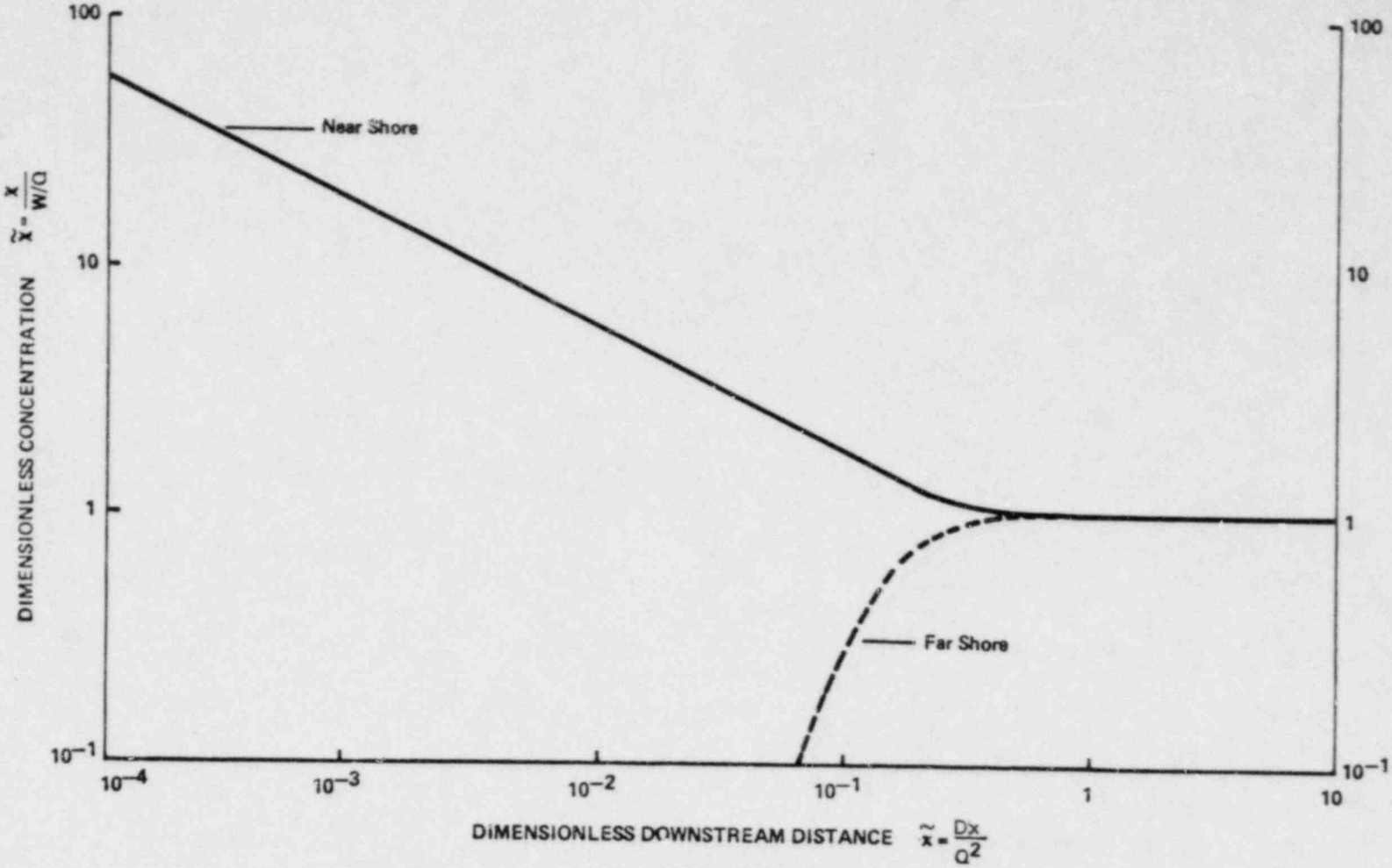


FIGURE 2.

DIMENSIONLESS CONCENTRATION OF NONDECAYING RADIONUCLIDE DISCHARGE AT THE SHORELINE VS DIMENSIONLESS DOWNSTREAM DISTANCE

$$K_x = \beta u^* d, \qquad (18)$$

where

d is the river depth;

u^* is the shear velocity; and

β is a dimensionless constant.

(Again, the user is not restricted to the use of this formula. A number of alternative approaches have been published.)

For straight rectangular stream channels, β has a value of about 5.93. The value of β, however, increases in curved channels and in general must be determined by field studies (Refs. 20-22). The two-dimensional coefficient, K_x, is usually much smaller in magnitude than the one-dimensional coefficient, E, as described later.

The usefulness of the transient model, even for simplified rectangular geometry, is that it allows analysis of the dispersion of material released in a realistic fashion. In the case of short-duration batch releases, spreading in the direction of flow may be an important mechanism for effluent dispersion, which is not included in the steady-state continuous release model.

3. OPEN COASTS

a. Great Lakes

(1) Discussion

Field studies in the Great Lakes have shown that coastal currents are predominantly parallel to the shore and have typical speeds of 10 to 20 cm/sec (0.2 to 0.4 knots). These currents usually persist in one direction for several days. Then, in direct response to wind shifts, they quickly reverse and persist in the opposite direction for several days. The stagnation time at reversal seldom exceeds a few hours (Refs. 23 and 24).

The studies further suggest that each reversal of the coastal current is accompanied by a large-scale mass exchange with offshore waters that effectively removes pollutants from the shore zone. Possible physical mechanisms responsible for this behavior are discussed by Csanady (Refs. 25 and 26). Observations near pollutant discharges have shown a well-defined pollutant plume hugging the shoreline for several days, then relatively quick dispersal offshore, followed by redevelopment of the plume in the opposite direction. Throughout this sequence, the buildup of pollutant concentration in the vicinity of the discharge is small (Refs. 23 and 27). A stable coastal current of, say, 10 cm/sec that persists for about three days before reversal causes an upcoast or downcoast excursion of an effluent plume which is of the order of 25 km (about 16 miles).

In view of the above findings, it is possible to construct a quasi-steady-state model valid for distances of about 25 km and time scales on the order of a few days. For those cases in which lateral mixing and mass exchange occur during flow stagnation periods, extension of the model beyond these limits should provide conservative results. It should be emphasized, however, that knowledge of Great Lakes coastal circulation patterns is far from complete. The existing data base is inadequate to conclude that the behavior described above is applicable to the entire Great Lakes system. A general knowledge of near-shore current climatology is needed. It is therefore recommended that modeling efforts be accompanied by time series current measurements at the site. Such measurements should be of sufficient duration to resolve the important time scales of flow variability. Of particular importance are field studies to define the extent and frequency of near-shore fumigation occurring at a given site.

(2) Analytical Models

(a) Steady-State Model

Analytical models of routine releases of liquid effluents along open coasts are usually based on Gaussian-like solutions to the steady-state diffusion equation. The form of each solution may differ in detail, depending on the number of dimensions retained, the location of the bounding surfaces, and the discharge configuration.

A simple diffusion model for a steady point-source discharge into a lake having a known steady longshore current, u, may be formulated by neglecting the longitudinal diffusion and time dependence in the dissolved constituent transport equation as follows:

$$u \frac{\partial C}{\partial x} = \epsilon_y \frac{\partial^2 C}{\partial y^2} + \epsilon_z \frac{\partial^2 C}{\partial z^2} - \lambda C \tag{14}$$

where ϵ_y and ϵ_z are the lateral and vertical turbulent diffusion coefficients, respectively. The remaining symbols are as defined in Section 2.

The decay term may again be removed through the transformation

$$C(x,y,z) = \chi(x,y,z) e^{-\lambda x/u} \tag{15}$$

resulting in the following equation for the nondecaying concentration χ:

$$u \frac{\partial \chi}{\partial x} = \epsilon_y \frac{\partial^2 \chi}{\partial y^2} + \epsilon_z \frac{\partial^2 \chi}{\partial z^2} \tag{16}$$

It is assumed that the discharge is located at the point $(0,y_s,z_s)$, i.e., at the origin of the x-axis and a distance y_s from the shoreline and z_s beneath the water surface. For a large lake of constant depth d and straight shoreline the solution is

$$\chi = \frac{W}{2\pi u \sigma_y \sigma_z} f(\sigma_z,z,z_s,d) f(\sigma_y,y,y_s) \tag{17}$$

where

$$f(\sigma_z,z,z_s,d) = \sum_{m=-\infty}^{\infty} \left\{ \exp\left[-\frac{(2md + z_s - z)^2}{2\sigma_z^2} \right] + \exp\left[-\frac{(2md - z_s - z)^2}{2\sigma_z^2} \right] \right\}$$

$$f(\sigma_y,y,y_s) = \exp\left[-\frac{(y_s-y)^2}{2\sigma_y^2} \right] + \exp\left[-\frac{(y_s+y)^2}{2\sigma_y^2} \right]$$

$$\sigma_y = \sqrt{\frac{2\epsilon_y x}{u}}, \quad \sigma_z = \sqrt{\frac{2\epsilon_z x}{u}}$$

This model may be used for curved shorelines by substituting, for y and y_s, the corresponding distances normal to the shoreline.

In Equation (17), the condition that there be no flux of material through the bounding surfaces is ensured by placement of an image source of strength W (Ci/sec) at $y = -y_s$ and an infinite series of image sources along the z-axis.

Equation (17) is a basic expression that can be modified to yield solutions for a variety of discharge configurations into bounded water bodies. For the sake of simplicity, the present discussion is limited to point discharges. However, if W is interpreted as source strength per unit length, or per unit area, line and plane sources, respectively, may be treated by integration of Equation (17) over the source dimensions.

The predictive capabilities of this model are limited because of the spatial variations in the flow field under actual conditions and because there are large uncertainties in the diffusion coefficients ϵ_y and ϵ_z (or, equivalently, in the standard deviations σ_y and σ_z). Studies in the Great Lakes and other large lakes suggest that "representative" near-shore values of ϵ_y are roughly in the range of 500 to 1000 cm^2/sec (0.5 to 1.1 ft^2/sec) and that ϵ_z is in the range 1 to 30 cm^2/sec (0.001 to 0.030 ft^2/sec) (Refs. 23 and 28). These values are typical only

of the near-shore zone. Furthermore, there is evidence to suggest that the ε_y is reasonably constant for discharge plume widths exceeding about 50 m (~ 165 ft) (Ref. 23). Hence Richardson's "four-thirds power law" should not be used to describe the lateral diffusion coefficient without justification on the basis of site-specific tracer studies.

Figure 3 shows centerline and shoreline values of χ/W calculated from Equation (17) for the case of a point source discharging at the surface 500 m (1640 ft) offshore into a 10 cm/sec (0.3 ft/sec) current. The horizontal and vertical mixing coefficients are 1000 cm^2/sec (1.1 ft^2/sec) and 5 cm^2/sec (0.005 ft^2/sec), respectively. The depth is 10 m. The centerline concentrations decrease inversely with distance from the source, x^{-1}, for about the first 10 km (6 miles), beyond which the concentration decrease is approximately $x^{-1/2}$. The dilution factor, D_F, is given by

$$D_F = W/(xq_p) \tag{18}$$

where q_p is the volumetric discharge rate of the effluent.

The dilution factor, for example, at 10 km (6 miles) downcurrent is approximately 7 for a 52 m^3/sec (1,830 ft^3/sec) discharge.

This result suggests that, for a nondecaying substance, the downstream concentration reduction in lake plumes parallel to the shore is rather small. This is consistent with observations reported for several of the Great Lakes (Ref. 23). It should be kept in mind that the dilution calculated above is for the far field and does not include possible additional dilution arising from initial mixing in the near field.

For a given location, the presence of a plume might be periodic. Therefore, long-term average dilution factors can be estimated from the above model by multiplying the solution by an appropriate flow-field frequency function. As discussed previously, observations suggest that the directional distribution of Great Lakes coastal currents is approximately bimodal. In such a case, long-term dilution factors would be about twice those calculated from Equation (17). It is emphasized, however, that the presence of reversing currents at a given site should be demonstrated by field observations of flow patterns before credit is taken for concentration reduction attributable to intermittent plume behavior.

(b) Transient Source Model

For other than a continuous source, the transient form of the constituent transport equation must be solved. In this case, diffusive transport in the direction of flow may be important, especially for short-duration releases, whereas it is unimportant in the case of continuous releases.

For an instantaneous release of a finite quantity of material from a vertical line source at $x = 0$, $y = y_s$, into a lake of known steady longshore current u, a simple transient model can be formulated:

$$C = \frac{M}{4\pi\sqrt{K_x K_y}\, td} \exp\left\{-\left(\frac{[x - Ut]^2}{4K_x t} + \lambda t\right)\right\}\left\{\exp\left(\frac{-[y - y_s]^2}{4K_y t}\right) + \exp\left(\frac{-[y + y_s]^2}{4K_y t}\right)\right\} \tag{19}$$

where

d is the depth;

M is the amount of activity released (in curies);

t is the time after the release;

and the other terms are as previously defined.

The case of a more general time-dependent release may be obtained by integrating Equation (19) with respect to time:

FIGURE 3.

DISPERSION VALUES AT SURFACE FOR LIQUID EFFLUENTS IN A LARGE LAKE

$$C = \int_0^t \frac{Wf(\tau)}{4\pi\sqrt{K_x K_y}\, d(t-\tau)} \exp\left\{-\left(\frac{(x-u(t-\tau))^2}{4K_x(t-\tau)} + \lambda(t-\tau)\right)\right\}\left\{\exp\left(\frac{-(y-y_s)^2}{4K_y(t-\tau)}\right) + \exp\left(\frac{-(y+y_s)^2}{4K_y(t-\tau)}\right)\right\} d\tau \qquad (20)$$

where the release rate is $Wf(\tau)$ curies/sec. In general, Equation (20) must be solved using numerical quadrature.

Equations (19) and (20) are also useful for releases into rivers in the region near the source, where the effects of the far shore are unimportant.

(3) Numerical Models

Analytical solutions to the diffusion equation are strictly applicable only to cases of steady uniform flow. In coastal regions having complex geometry and time-dependent nonuniform flow, analytical models might not be adequate for predicting realistic concentration values. In such cases multi-dimensional numerical models are more suitable. The use of such models is becoming increasingly common in water-quality simulation.

Typical acceptable numerical models are the two-dimensional, vertically integrated models developed by Leendertse and co-workers (Refs. 29-34), Codell (Ref. 35), Loziuk et al. (Ref. 36), and Erasian (Ref. 37).

These and other numerical models fall into two broad categories, depending on the method in which the advective velocity field is obtained. The Leendertse, Codell, and Loziuk models, for example, compute the velocity field from the following vertically integrated two-dimensional equations of mass and momentum conservation:

$$\left.\begin{aligned} \frac{\partial U}{\partial t} + U\frac{\partial U}{\partial x} + V\frac{\partial U}{\partial y} - fV &= -g\frac{\partial \zeta}{\partial x} + \frac{\tau_x^s}{H} - \frac{gU(U^2+V^2)^{1/2}}{C_h^2 H} \\ \frac{\partial V}{\partial t} + U\frac{\partial V}{\partial x} + V\frac{\partial V}{\partial y} + fU &= -g\frac{\partial \zeta}{\partial y} + \frac{\tau_y^s}{H} - \frac{gV(U^2+V^2)^{1/2}}{C_h^2 H} \\ \frac{\partial \zeta}{\partial t} + \frac{\partial}{\partial x}(HU) + \frac{\partial}{\partial y}(HV) &= 0 \end{aligned}\right\} \qquad (21)$$

where

C_h is the Chezy coefficient;

f is the Coriolis parameter;

H is the depth from water surface to bottom;

(U,V) are the vertically averaged x and y component velocities;

ζ is the water surface location above an undisturbed level datum; and

τ_x^s, τ_y^s are the x and y component surface stresses.

The resulting velocity field then becomes the advective mechanism in the following vertically averaged conservation equation for the dissolved constituent concentration C:

$$\frac{\partial}{\partial t}(HC) + \frac{\partial}{\partial x}(HUC) + \frac{\partial}{\partial y}(HVC) = \frac{\partial}{\partial x}\left(HK_x\frac{\partial C}{\partial x}\right) + \frac{\partial}{\partial y}\left(HK_y\frac{\partial C}{\partial y}\right) - H\lambda C \qquad (22)$$

where K_x and K_y are the dispersion coefficients in the indicated directions.

The Eraslan model, on the other hand, requires synthesis of the flow field from current measurements. Use of this technique requires a careful analysis of the flow data to ensure that the resulting velocity field conserves mass. The velocity field is then applied to the integral form of the conservation equation for the dissolved constituent in question (donor cell method) (Ref. 37).

b. Oceans

Modeling techniques for estimating radionuclide transport in ocean coastal waters are similar to those applicable to near-shore waters in the Great Lakes. The primary differences in behavior between the two systems results from the greater temporal and spatial variability in flow occurring in ocean coastal waters. This variability results primarily from two factors. The first and more readily defined factor is the major influence of astronomical tidal currents, which are negligibly small in the Great Lakes. The second factor, whose effects are important but much more difficult to quantify, is the influence of meteorological driving forces. These forces include the direct effects of both meso-scale and synoptic-scale wind systems and the indirect effects of seasonal variations in heating or cooling and coastal river discharges. As a result of these factors, the flow variability in oceanic coastal waters contains components having magnitudes and characteristic time scales greater than those of the near-shore waters of the Great Lakes.

In practice, the choice of transport modeling techniques applicable to a given ocean coastal region depends, to a large extent, on the level of knowledge of local near-shore current climatology. A particular model choice and range of model parameters should be demonstrated to include, to the extent practical, the effects of the important scales of flow variability. For synoptic scale fluctuations in flow patterns, it will often be necessary to perform transport calculations for conditions "typical" of various seasons or wind patterns.

For a given set of conditions, however, the choice of modeling techniques is further determined by the interpretation of the role of tidal currents in the mixing process. The interpretation depends on the averaging period used to define the velocity field. If the averaging time is long compared to the tidal period, tidal currents cannot contribute to the advective transport, since their contributions to the mean flow field have been removed by the averaging. Tidal effects would be contained solely in "tidally averaged" turbulent diffusion coefficients. This result is largely a mathematical artifact that assigns the actual advective effects of tidal currents to large-scale turbulent diffusion. Nevertheless, if detailed descriptions of the field of radionuclide concentrations are not required, it is possible to construct quasi-steady-state transport models that are valid for time scales larger than the tidal period and smaller than those associated with major nontidal fluctuations in flow.

For regular shoreline geometry, or discharges removed from the shoreline, steady-state Gaussian models based on Equation (17) may be used (Refs. 38-41). The results based on these models require careful interpretation, however, because of the large uncertainty in input parameter values, particularly the turbulent diffusion coefficients. Since these coefficients arise from time averaging, their values for any given case will depend on the averaging period used to define the mean velocity field. Furthermore, there is evidence to indicate that in the ocean the rate of spread of a contaminant plume depends upon the plume age. Hence, in general, turbulent diffusion coefficients will be time and space dependent. The methodology for obtaining reasonable estimates for these coefficients is based primarily on the interpretation of the results of tracer studies in the light of modern turbulence theory (Refs. 38-45).

More realistic detailed descriptions of radionuclide transport in ocean coastal waters will require the use of numerical models. The advantage of such models is that they are applicable to fully time-dependent flow fields in receiving waters having complex geometry. In particular, these models have the capability of treating tidal currents as advective rather than diffusive mechanisms ("real-time" models), hence removing a large element of uncertainty in the determination of turbulent diffusion coefficients.

Typical acceptable numerical models (Ref. 29-37) were discussed in Section 3.a.(3). In the "real-time" modeling approach, tidal currents are explicitly included as advective transport mechanisms. Leendertse and co-workers (Refs. 29-34) have shown that in this case, reasonable estimates of longitudinal and lateral turbulent dispersion coefficients may be based on Elder's (Ref. 19) formulas for steady open-channel flow.

The applicability of numerical models and the techniques for establishing horizontal mixing coefficients are discussed further in Sections 4.c and 4.d.

4. ESTUARIES

Transport of contaminants in estuaries differs from that in rivers because of oscillatory tidal advection and the nontidal gravitational circulation induced by salinity differences. An important consequence of these differences is that there is transport of material upstream from the discharge point in estuaries, the maximum upstream penetration being limited to the general region of oceanic salt intrusion.

a. One-Dimensional Models

For purposes of radionuclide transport prediction, reduction of the estuarine problem to a single dimension (longitudinal) produces satisfactory results, except in the lower reaches of the estuary, where circulation is clearly two or three dimensional. The one-dimensional simplification is accomplished by averaging over the estuary cross-section. The resulting constituent transport equation is

$$\frac{1}{A}\frac{\partial}{\partial t}(AC) + \frac{1}{A}\frac{\partial}{\partial x}(A\bar{U}C) = \frac{1}{A}\frac{\partial}{\partial x}\left(AE\frac{\partial C}{\partial x}\right) - \lambda C \qquad (23)$$

where

$A(x,t)$ is the cross-sectional area;

$E(x)$ is the sectionally averaged, one-dimensional longitudinal dispersion coefficient; and

$\bar{U}(x,t)$ is the sectionally averaged longitudinal velocity.

Both simple and elaborate methods of solving Equation (23) exist.

The simplest models depend on the "tidally averaged" approximation, in which the tidal oscillations are not included explicitly, but are considered to be responsible for large-scale longitudinal diffusion. The more elaborate "real-time" models consider the actual tidal flow to be advective, with longitudinal diffusion occurring through motions having time scales considerably shorter than a tidal cycle. Each type of model is discussed below.

(1) Analytical Model (Steady State)

The least elaborate one-dimensional model assumes a constant cross-sectional area A, a constant (tidally and sectionally averaged) longitudinal dispersion coefficient E_L, and a constant fresh water velocity U_f. For this case Equation (23) reduces to

$$E_L\frac{d^2C}{dx^2} - U_f\frac{dC}{dx} - \lambda C = 0 \qquad (24)$$

where C is the time and sectionally averaged concentration. The solution (Ref. 46) to Equation (24) for a source at $x = 0$ and the boundary conditions $C = 0$ at $x = \pm\infty$ is

$$C = \frac{W}{AU_f\sqrt{1 + \frac{4\lambda E_L}{U_f^2}}}\exp\left(\frac{U_f}{2E_L}\left[1 \pm \sqrt{1 + \frac{4\lambda E_L}{U_f^2}}\right]x\right) \qquad (25)$$

The sign within the exponential is negative downstream from the source (x positive) and positive upstream from the source (x negative).

In terms of dimensionless variables, Equation (25) reduces to

$$\Gamma = \Gamma_{max}\exp\left(\frac{N \pm \sqrt{N^2+4}}{2}\xi\right) \qquad (26)$$

where

$$N = \frac{U_f}{\sqrt{\lambda E_L}}$$

$$\Gamma = \frac{A\sqrt{\lambda E_L}}{W} C$$

$$\Gamma_{max} = \left(\frac{\lambda E_L}{U_f^2 + 4\lambda E_L} \right)^{1/2}$$

$$\xi = \sqrt{\frac{\lambda}{E_L}}\, x$$

Figure 4 illustrates this dimensionless equation evaluated for ξ and N.

Several features of Equation (26) are evident from Figure 4. The dimensionless source concentration Γ_{max} depends not only on the source strength and freshwater flow, but also on the diffusivity E_L and the decay constant λ. This dependency is explained by the fact that a steady concentration is maintained by a balance between the discharge source, the net advective-diffusive transport away from the source, and a local sink due to radioactive decay.

The upstream and downstream curves have equal but opposite slopes for N = 0, since there is no nontidal advection where the net freshwater flow is zero. The curves become skewed in the downstream direction for increasing values of U_f because of the nontidal advection downstream.

(2) Releases of Short Duration

For releases of short duration, the preceding steady-state model does not apply. In the case of a time-dependent source term, the transport equation is given by

$$\frac{\partial C}{\partial t} + U_f \frac{\partial C}{\partial x} = E_L \frac{\partial^2 C}{\partial x^2} - \lambda C \qquad (27)$$

The solution to Equation (27) for a time-dependent release may be obtained from the solution corresponding to the instantaneous release of a finite quantity of effluent uniformly over the flow cross-section (unit impulse function) (Ref. 47).

The unit impulse solution is given by

$$C = \frac{M}{A\sqrt{4\pi E_L t}} \exp\left(- \frac{(x - U_f t)^2}{4E_L t} - \lambda t \right) \qquad (28)$$

where M is the amount of activity introduced (Ci).

For a more general time-dependent release, results may be obtained by time integration of Equation (28). Assume that, instead of the instantaneous introduction of a finite quantity at x = 0 and t = 0, effluent is continuously discharged at the rate $\frac{dM}{dt} = Wf(t)$ Ci/sec. The concentration distribution resulting from a continuous discharge in the time interval $0<\tau<t$ is given by

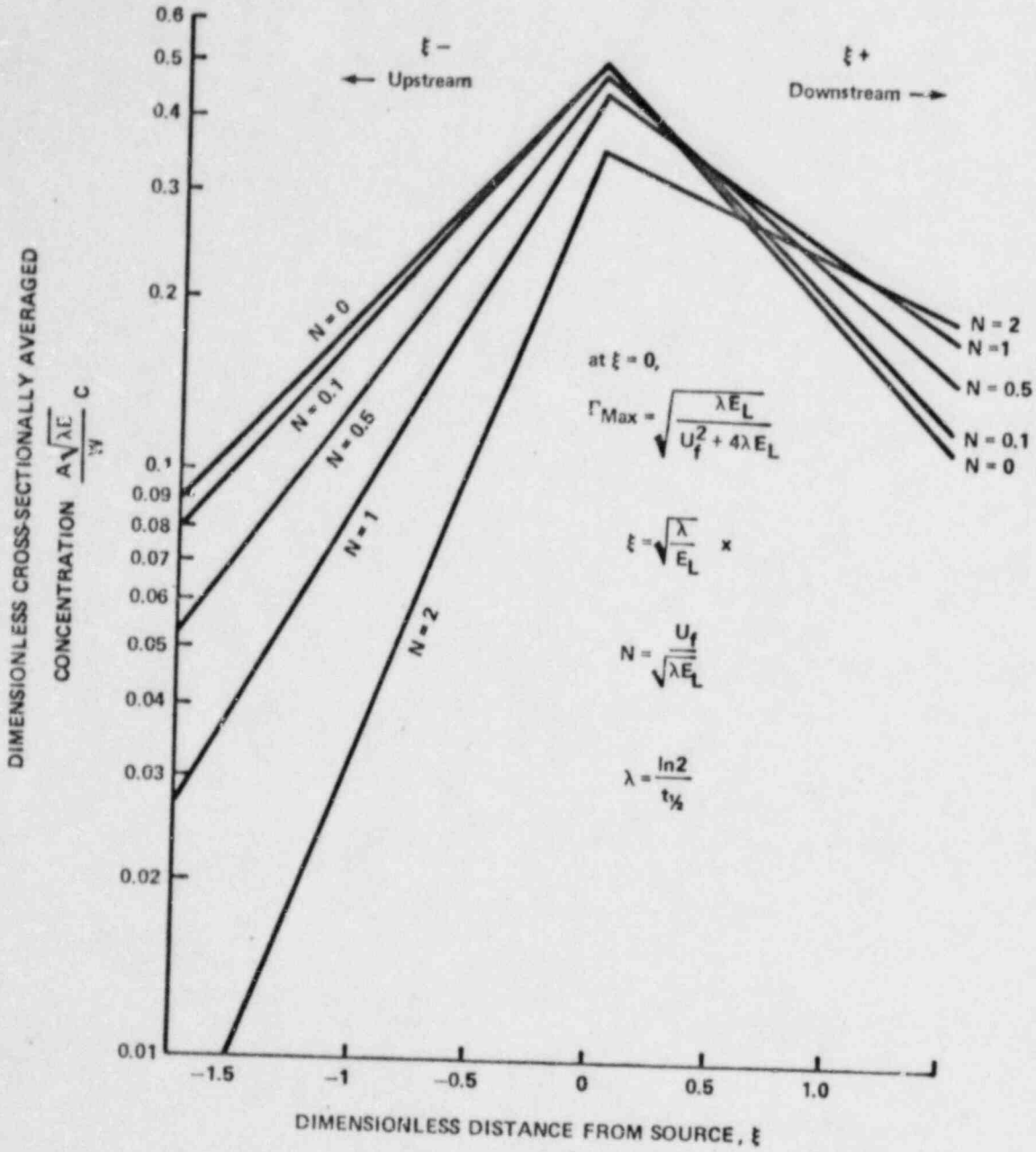


FIGURE 4.
DIMENSIONLESS PLOT OF CONTAMINANT CONCENTRATION VS DISTANCE FROM SOURCE (FROM O'CONNER ET AL., REF. 46)

$$C = \frac{W}{A\sqrt{4\pi E_L}} \int_0^t \frac{f(\tau)}{\sqrt{t-\tau}} \exp\left\{ - \frac{[x - U_f(t-\tau)]^2}{4E_L(t-\tau)} - \lambda(t-\tau) \right\} d\tau \qquad (29)$$

From Equation (29) the concentration distribution corresponding to a square pulse release of amplitude W and duration t_D is

$$C = \frac{W}{A\sqrt{4\pi E_L}} \int_0^{t_D} \frac{1}{\sqrt{t-\tau}} \exp\left\{ - \frac{[x - U_f(t-\tau)]^2}{4E_L(t-\tau)} - \lambda(t-\tau) \right\} d\tau \qquad (30)$$

Equation (30) may be integrated to give the following solution in terms of exponentials and error functions:

$$\left.\begin{aligned} C &= \frac{W}{2A\Omega} \exp\left(\frac{Ux}{2E_L}\right) g(x,t) \text{ for } 0 < t < t_D \\ C &= \frac{W}{2A\Omega} \exp\left(\frac{Ux}{2E_L}\right) [g(x,t) - g(x,t-t_D)] \text{ for } t > t_D \end{aligned}\right\} \qquad (31)$$

where

$$g(x,t) = \left[\operatorname{erf}\left\{\frac{x+\Omega t}{\sqrt{4E_L t}}\right\} \pm 1\right] \exp\left(\frac{\Omega x}{2E_L}\right) - \left[\operatorname{erf}\left\{\frac{x-\Omega t}{\sqrt{4E_L t}}\right\} \pm 1\right] \exp\left(-\frac{\Omega x}{2E_L}\right)$$

$$\Omega = \sqrt{U^2 + 4\lambda E_L}$$

The function $g(x,t - t_D)$ has the same form as $g(x,t)$, with $(t - t_D)$ replacing t. The sign within the brackets is chosen negative downstream of the source (positive x) and positive upstream of the source (negative x).

Equation (31) holds for any pulse duration t_D. In the limit as $(t,t_D) \to \infty$, the solution reduces to the steady-state solution given by Equation (25).

Release rates other than square pulses are most easily computed by solving Equation (29) directly, using numerical quadrature. (See Sections 2.a.(2) and 3.a.(2) of this appendix.)

(3) Tidally Averaged Numerical Models

To simulate constituent transport in many types of estuaries, it is necessary to include detail beyond the capabilities of analytical models. For example, the distribution of sources and sinks (both man-made and natural) may be important.

Additionally, the estuary may have a nonuniform cross-section and tidal mixing properties that vary along its length.

The next level of sophistication above the analytical models are one-dimensional numerical models, which can account for variable cross-sections, inputs, withdrawals, and tidally averaged longitudinal diffusion. These models solve what is essentially the finite difference equivalent of Equation (23) in either the steady-state or transient (but tidally averaged) form. Models similar to the EPA AUTOSS and AUTOQD models fall into this category (Ref. 48).

The estuary is considered to be divided into variable-length segments. Each segment is coupled to the next upstream and downstream segment, as well as to external sources and sinks. Typically, the boundary conditions are chosen so that the concentrations of the first and the last segments are known constants. This is the most realistic assumption for this model, provided the model is extended to the headwaters of the estuary and to the ocean. In practice, the model can easily be extended from the headwaters to the ocean by fine grid spacing in the area near the discharge and by coarse spacing farther away--in the regions of less interest.

A useful application of this model occurs where there are recycle streams such as municipal water withdrawal and return. The recirculation with partial or total removal of certain radionuclides could be important for heavily used tidal and nontidal waterways.

(4) Intratidal Numerical Models

The tidally averaged models are often subject to error because of uncertainty in the longitudinal dispersion coefficient. A more acceptable approach is the intratidal model, in which velocity, water level, and concentration in the estuary are simultaneously solved for, the tidal velocity being retained explicitly as an advective transport mechanism. In such a model, the longitudinal diffusion coefficient is better defined on the basis of physical principles and is less important than in the tidally averaged case. The model solutions are suitable for digital computation and do not require excessive computer resources.

Included are models such as the Dailey-Harleman (Ref. 49) one-dimensional finite element model, the Lee-Harleman (Ref. 50) finite difference model, and the Eraslan (Ref. 51) one-dimensional donor cell model. Basically, these models solve the one-dimensional equations of mass, momentum, and constituent conservation,

$$b\frac{\partial \zeta}{\partial t} + \frac{\partial Q}{\partial x} - \text{Source} = 0 \tag{32}$$

$$\frac{\partial Q}{\partial t} + \bar{U}\frac{\partial Q}{\partial x} + Q\frac{\partial \bar{U}}{\partial x} + g\frac{\partial \zeta}{\partial x}A + \frac{gQ|Q|}{AC_h^2 R_h} = 0 \tag{33}$$

$$\frac{1}{A}\frac{\partial}{\partial t}(AC) + \frac{1}{A}\frac{\partial}{\partial x}(A\bar{U}C) = \frac{1}{A}\frac{\partial}{\partial x}(AE\frac{\partial C}{\partial x}) - \lambda C \tag{34}$$

where

b is the width of the estuary at the water surface;

C_h is the Chezy coefficient;

R_h is the hydraulic radius; and

ζ is the water surface location above an undisturbed level datum.

Concentration boundary conditions can be treated realistically in the intratidal formulation. The upstream boundary is usually the concentration at the head of the tide. The downstream boundary, however, usually differs according to whether the tide is flooding or ebbing. During ebb tide, the downstream boundary is chosen so that all constituents leave by advection. During flood tide, the entering concentration must be specified. This is determined by the physical situation assumed. If the downstream boundary is the ocean, the concentration of constituents in ocean water can be the input. If the downstream boundary is a bay or other water body where a discharged constituent can accumulate, an approximation of this concentration must be made.

An advantage of the intratidal model is its ability to simulate releases coordinated with the tide. If the source of contaminant is close to the mouth of the estuary, it may be advantageous to discharge only during ebb tide to flush the contaminant rapidly out of the estuary. Such operation could not be simulated with a tidally averaged model.

b. Multi-Dimensional Models

In very wide estuaries and embayments, the one-dimensional assumption is not realistic. For such conditions, both transverse and longitudinal velocity components are important, and concentration gradients across the channel approach those along the channel. To simulate this case with one-dimensional models, unreasonably large longitudinal dispersion coefficients must be used.

Two-dimensional vertically averaged numerical models are more suitable for these situations. Typical acceptable models were discussed in Section 3 of this appendix.

In some cases, numerical models which simulate vertical concentration variations rather than horizontal variations may be more appropriate, for instance, in the salinity intrusion region of an estuary, or a highly stratified reservoir.

Although three-dimensional numerical models are currently being developed, their complexity and relatively high cost of observation are rarely warranted for the purpose of computations under Appendix I.

c. Exchange Coefficients

The sectionally averaged, one-dimensional longitudinal dispersion coefficient, E, includes a combination of several individual processes. In the prototype, these processes include molecular exchange and flow- and wind-induced turbulent exchange. In most cases, these processes make a relatively minor contribution to the overall dispersion, which is in part an artifact resulting from the temporal and spatial averaging of the three-dimensional transport equations. In general, the greater the simplification of the model, the larger the exchange coefficient must be to simulate the prototype.

The simplest model is the tidally averaged one-dimensional model. The actual three-dimensional oscillating flow is drastically simplified into a one-dimensional system in which the advective transport is determined by the fresh water flow rate. The diffusive transport includes the effects of deviations from sectional homogeneity and "turbulence" components of time scales on the order of the tidal period or less.

In the intratidal models, the true oscillating or unidirectional flow is simulated and is treated as part of the advective process. The diffusive term includes the effects of deviations from sectional homogeneity of the concentration and velocity fields. However, in this case, tidal effects are no longer included in the turbulence field.

In the case of two-dimensional intratidal models, it is, in general, only necessary to include deviations of velocity and concentration in the vertical direction since, with sufficient resolution, the lateral flow field is simulated correctly.

In all cases, the most acceptable procedure for choosing the diffusion coefficient is to adjust the model to match observed prototype conditions, usually through tracer study results. In general, the more refined the model, the less empirical tuning is necessary because the turbulent transport coefficients are more firmly based on physical principles.

(1) Tidally Averaged Models

If the tidally averaged model is used, the determination of the diffusion coefficient is empirical and must be based on the observed dispersion of a known tracer, with prototype and model constituent concentrations being matched. The tidally averaged longitudinal dispersion coefficient E_L may be determined from Equations (25), (28), or (29) by a trial and error procedure where E_L is changed until the model concentrations match observed values of the tracer.

In the case of tidally averaged numerical models covered under paragraph 4.a(3), it is possible to restructure the finite difference equations to solve for E_L with input of observed concentration of the tracer. The calculated values of E_L may then be used for subsequent concentration computations.

As a rough approximation to the dispersion coefficient, lacking any field data, a formula by Hetling and O'Connell (Ref. 52) based on data in the salinity intrusion region of the Potomac River estuary may be used:

$$E_L = 1680\, V_{max}^{4/3} \qquad (35)$$

where

E_L is the sectionally and tidally averaged, one-dimensional longitudinal dispersion coefficient (in ft^2/sec), and

V_{max} is the maximum tidal velocity (in knots).

It must be cautioned that this equation can only be relied upon for order of magnitude estimates and is not necessarily conservative.

(2) One-Dimensional Real-Time Models

Less "tuning" is necessary for real-time models than for the tidally averaged models. In the well-mixed region of the estuary, Taylor's formula (Ref. 53) for dispersion is acceptable for reasons discussed in Section 4.d of this appendix.

The salinity intrusion region of the estuary is still poorly defined on physical grounds. Since the presence of gravitational circulation casts doubts on the applicability of sectional averaging, it is this region for which tuning is most important.

An approximation for the longitudinal dispersion coefficient that is applicable to the whole length of the estuary is based on the work of Thatcher and Harleman (Ref. 54). This approach is based on a combination of Taylor's dispersion formula applicable to the well-mixed portion of the estuary combined with an empirical correlation for mixing in the salinity intrusion region based on observed salinity distributions. The dispersion coefficient is

$$E(x,t) = K\left|\frac{d(\frac{S}{S_o})}{d(\frac{x}{L})}\right| + 77nU_t R_h^{5/6} \qquad (36)$$

where

L is the length of estuary;

n is Manning's coefficient (local);

R_h is the hydraulic radius (local);

S is the salinity (local);

S_o is the salinity at mouth;

U_t is the RMS velocity (local); and

x is the distance from mouth.

The factor K is given by

$$K = 0.00215 V_{max} L E_D^{-0.25} \qquad (37)$$

and is shown in Figure 5.

The quantity E_D is the so-called "estuary number" and is given by

$$E_D = \frac{P_T F_D^2}{Q_f T} \qquad (38)$$

where

F_D is the densimetric Froude number evaluated at the estuary mouth;

P_T is the tidal prism, in ft^3;

Q_f is the freshwater flow rate; and

T is the tidal period.

The dispersion formula given in Equation (36) may be used with good results as a first approximation in the tuning of a real-time model.

In an oscillatory flow such as a hydroelectric or pump storage reservoir where there is no salinity intrusion region, the Taylor formula alone may be used as a first approximation:

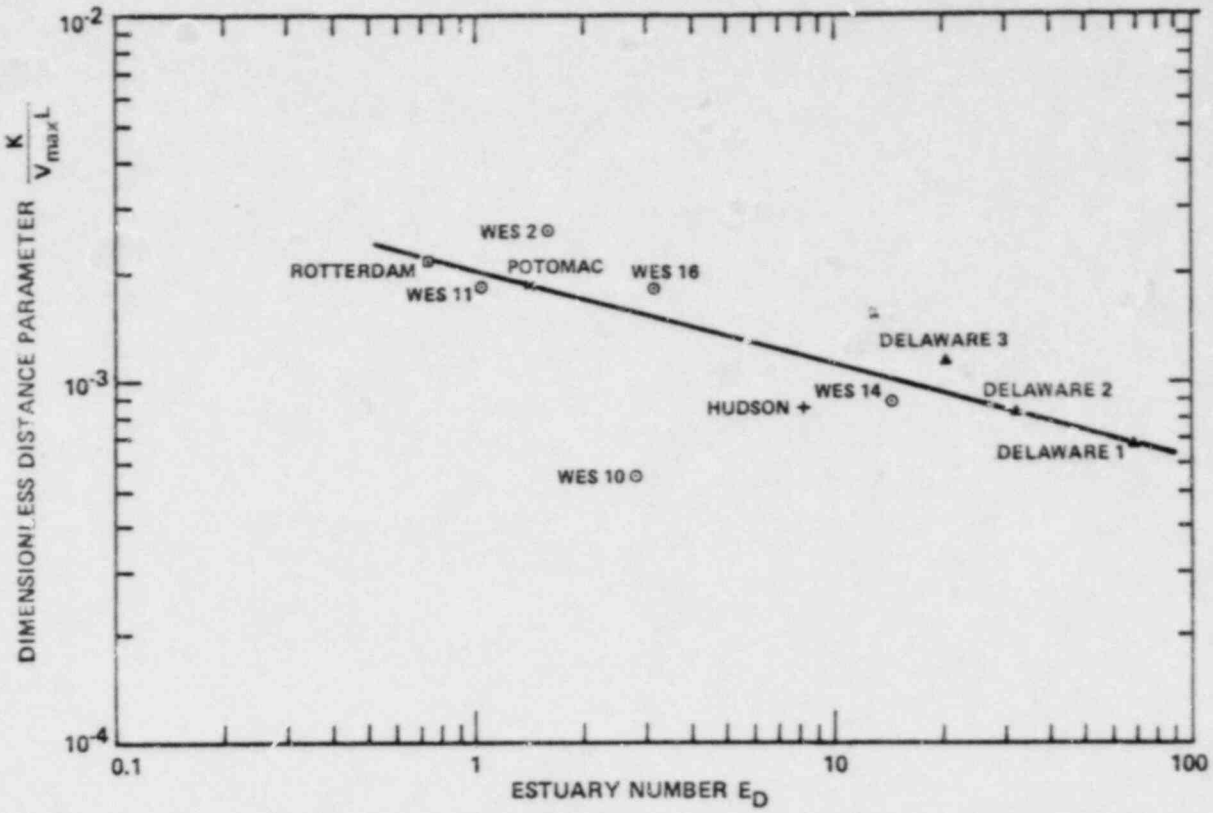


FIGURE 5.
CORRELATION OF DISPERSION PARAMETER TO DEGREE OF STRATIFICATION
(FROM THATCHER, et al., REF. 54)

$$E(x,t) = 77nU_t R_h^{5/6} \qquad (39)$$

These approximations are suitable only for periodic oscillating flows and not necessarily for unidirectional flows (as will be further explained in Section 4.d of this appendix). Tuning of the dispersion coefficient should be performed after the friction coefficients are adjusted to simulate the observed flows, since Equation (39) is a function of Manning's coefficient.

(3) Two-Dimensional Models

The best approximation for the dispersion coefficients in the two-dimensional estuary model appears to be the one adapted for the one-dimensional real-time model discussed above. For example, as a first approximation in the simulation of Jamaica Bay, which can be classified as a well-mixed two-dimensional estuary, Leendertse (Ref. 29) used Elder's formula with an additional factor for the effect of wind-induced turbulence.

As with the one-dimensional models, tuning is necessary to match prototype observations and should be performed after roughness factors in the hydraulic part of the model have been established. In the salinity intrusion regions of the estuary, initial estimates based on the method described in Thatcher and Harleman (Ref. 54) should be adequate.

d. Applicability of Models

Choice of one of the above models involves several factors. Although these models are classified as estuary models, they may be used for other types of water bodies, such as reservoirs with hydroelectric power generation flow reversals and, in most cases, unidirectional rivers. However, the assumptions implicit in one-dimensional models may be invalid in some cases, as explained below.

Fischer (Ref. 55) analyzed the mechanics of dispersion in nontidal rivers and concluded that Elder's formula yielded dispersion coefficients that were low by as much as two orders of magnitude. He concluded that longitudinal dispersion in rivers is primarily due to velocity variations in the lateral direction, while Elder's formula accounts only for velocity variations in the vertical direction. An approximate diffusion coefficient for unidirectional rivers derived by Fischer (Ref. 55) is

$$E = 0.3\,\overline{u'^2}\,\frac{\ell^2}{R_h u^*} \qquad (40)$$

where

ℓ is approximately the cross-sectional distance from the point of maximum velocity to the further bank;

R_h is the hydraulic radius;

u'^2 is the mean squared deviation of the river flow from the sectional mean velocity $\bar{u}$; and

u^* is the shear velocity.

The one-dimensional model is valid only for downstream distances corresponding to travel times greater than the so-called "Taylor period":

$$t \geq 1.8\,\frac{\ell^2}{R_h u^*} \qquad (41)$$

at which point the constituent introduced at t = 0 is sufficiently well mixed in the cross-section for the transport to be considered one-dimensional.

In an oscillating tidal flow in a wide, shallow estuary, however, the tidal period is large compared to the vertical mixing time scale, but small compared to the lateral mixing time scale. In this case, velocity variations in the lateral direction add little to the longitudinal mixing, and Equation (39) is appropriate.

5. IMPOUNDMENTS

There are two basic types of cooling ponds. The first is a closed-loop system in which the thermal effluent is cooled in the pond and recirculated through the power plant condensers. Some water ("blowdown") must be removed from the pond to limit the dissolved solids concentration to an acceptable level. Fresh water ("make-up") must be added to the pond to compensate for evaporation and blowdown.

The second type of cooling pond is a flow-through system in which there is little or no recirculation of effluent through the power plant. The effluent is discharged to the pond which, in turn, discharges to a larger body of receiving water. The pond serves as a holding reservoir, allowing effluent to cool before entering the receiving water.

The source of radionuclides may either be located on the impoundment or upstream on a tributary of the impoundment. For the simplest models, this distinction is irrelevant because concentration is based on flow through the impoundment and does not depend on the placement of the input. In the case of the upstream plant, the source term W is the rate of radionuclide entering with the flow at the boundary of the reservoir.

Figure 6 illustrates a closed-loop cooling pond. Water for cooling is drawn through the intake, circulated through condensers, and returned to the pond via the discharge. There are two important hydrological parameters of this system. The first is the internal recirculation time constant associated with the flushing of the pond by the makeup and blowdown streams. The second is the time constant associated with the decay of radioisotopes.

Figure 7 illustrates the flow-through pond. The hydraulics of this pond are simpler than the closed-loop pond, since no recirculation occurs between intake and discharge. In this case, the only hydraulic time constant is that associated with the travel time from the plant discharge to the receiving water.

a. Simple Analytical Models

Simple models may be used to obtain conservative estimates of the radioisotope concentrations. Four models can be used to describe all cooling ponds: the completely mixed model, the plug-flow model, the partially mixed model, and the stratified model. In each case, the effect of evaporation is neglected.

(1) Completely Mixed Model

Figure 8 shows the first case (the closed loop), in which the pond is represented as a completely mixed tank. All inputs of material makeup are instantaneously mixed throughout the tank, so that the concentration is homogeneous.

By performing a mass balance on the volume of the pond, a solution for concentration is obtained, assuming zero initial concentration and complete mixing:

$$\frac{C}{C_o} = \frac{\frac{q_b}{V_T\lambda}}{\frac{q_b}{V_T\lambda} + 1}\left[1 - \exp\left(-\frac{q_b}{V_T\lambda} - 1\right)\lambda t\right] \qquad (42)$$

where

q_b is the pond blowdown rate and

V_T is the volume of the pond.

The concentration C_o is the steady-state concentration that would exist for a nondecaying substance and is given by

$$C_o = \frac{W}{q_b}$$

where W is the rate of addition of radioactivity (in Ci/sec).

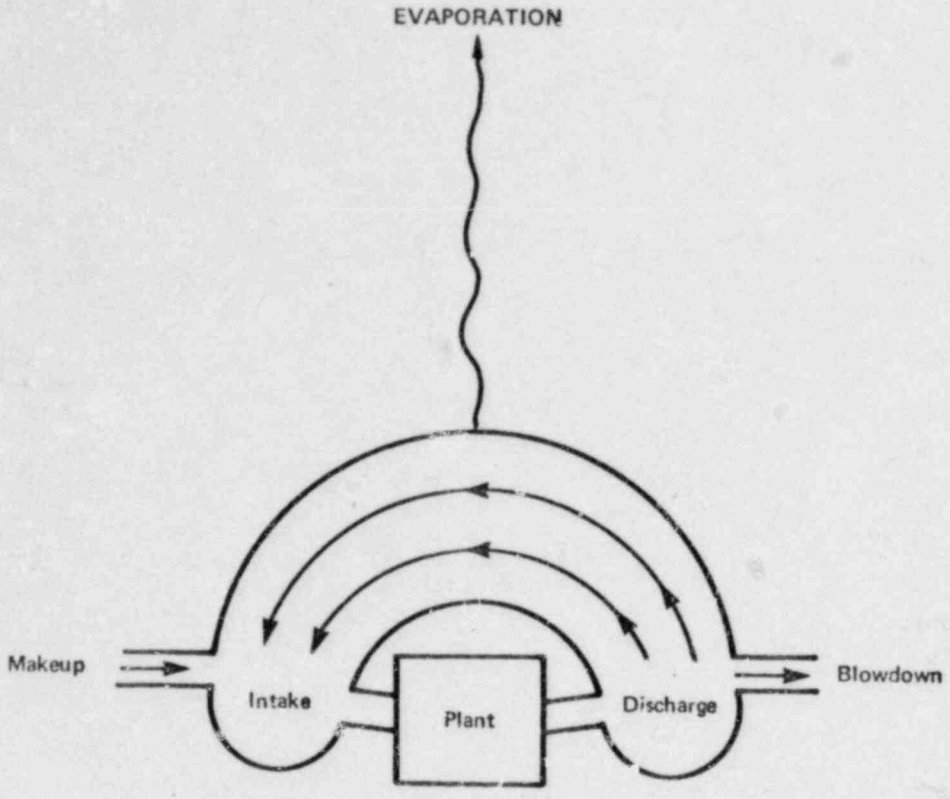


FIGURE 6.
CLOSED-LOOP COOLING POND

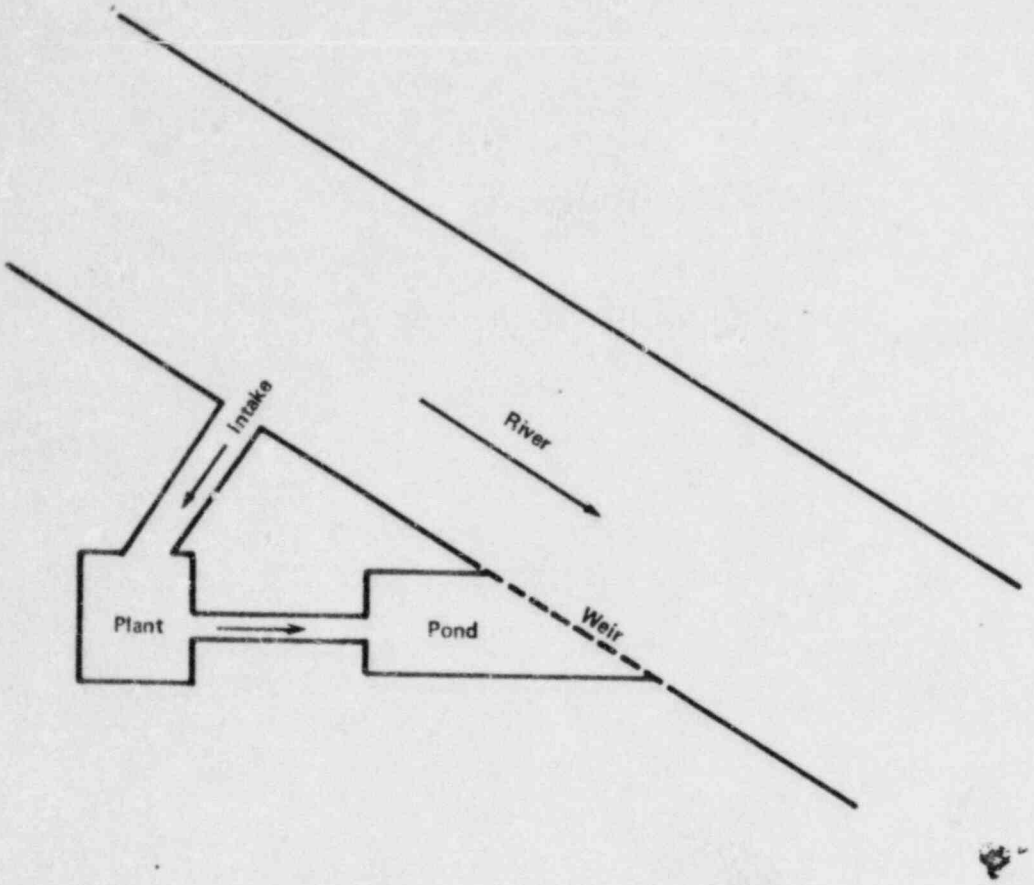


FIGURE 7.
FLOW-THROUGH COOLING POND

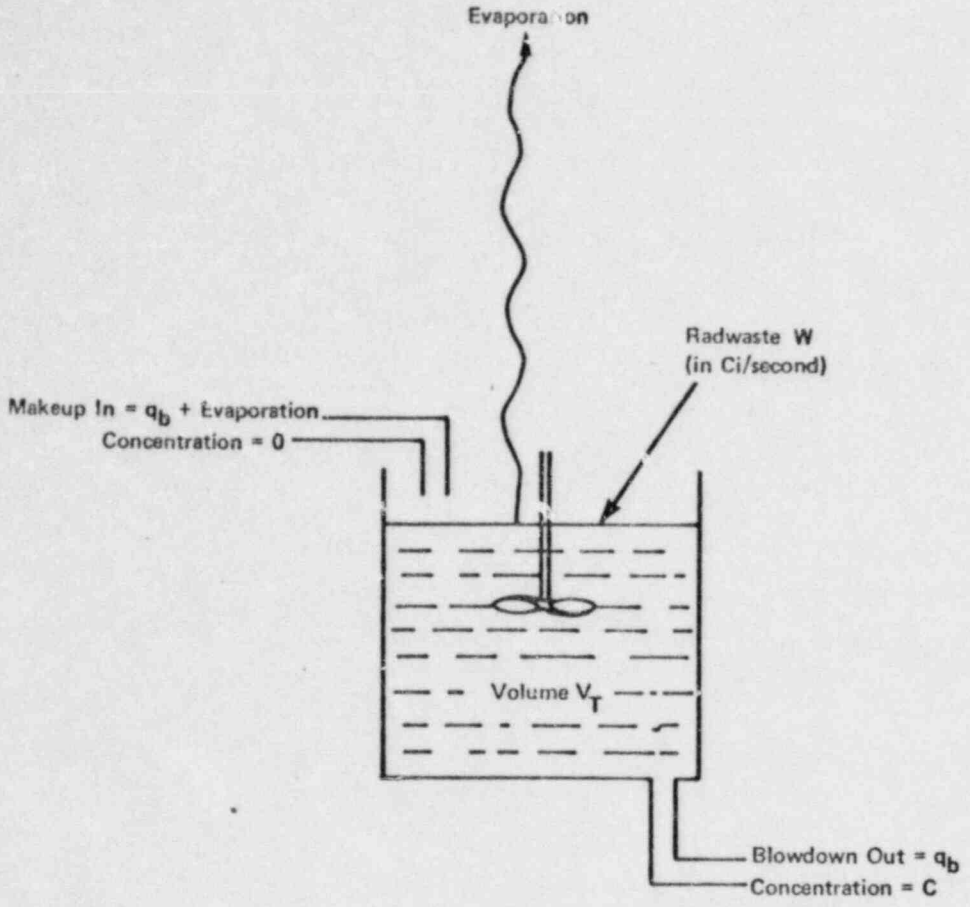


FIGURE 8..
COMPLETELY MIXED MODEL

For the steady-state concentration of a decaying substance, Equation (42) reduces to

$$\frac{C}{C_o} = \frac{\frac{q_b}{V_T \lambda}}{\frac{q_b}{V_T \lambda} + 1} \qquad (43)$$

In terms of the half-life, $t_{\frac{1}{2}}$, of the added radioactivity, Equations (42) and (43) reduce to

$$\frac{C}{C_o} = \frac{\tau}{\tau + \ln 2} \left(1 - \exp\left\{ - [\tau + \ln 2] \frac{t}{t_{\frac{1}{2}}} \right\}\right) \qquad (44)$$

(at steady state) $$\frac{C}{C_o} = \frac{\tau}{\tau + \ln 2} \qquad (45)$$

where

$$\tau = \frac{t_{\frac{1}{2}} q_b}{V_T} \quad \text{and}$$

$$t_{\frac{1}{2}} = (\ln 2)/\lambda$$

The dimensionless variable τ is the radioisotope half-life expressed in multiples of the flushing time (V_T/q_b). Figure 9 illustrates the steady-state concentration defined by Equation (45) as a function of τ. This figure shows that for small half-lives (compared with the flushing time, V_T/q_b), the concentration depends strongly on half-life, but for long half-lives, this dependence diminishes.

The completely mixed tank model (see Figure 8) is adequate for substances with long half-lives, where the internal circulation time is short compared with the half-life. In this case, the concentration in the pond is fairly homogeneous, satisfying the "completely mixed" limitation of this model.

(2) Plug-Flow Model

For the flow-through cooling pond, the plug-flow model illustrated in Figure 10 may be adequate if it is presumed that there is no recirculation. The concentration is expressed as

$$C = C_o \exp(-\lambda V_T/q_b) = C_o \exp(-\ln 2/\tau) \qquad (46)$$

where q_b is now the flow rate through the pond.

This expression is evaluated in Figure 9 and compared with the completely mixed case. Agreement is best for large τ. Although not easily seen in Figure 9, the models deviate for small τ.

(3) Partially Mixed Model

Where a significant part of the flow is due to both blowdown and plant pumping, neither of the above models is adequate. A suitable model that includes both the plant pumping rate q_p and blowdown rate q_b is illustrated in Figure 11. The recirculation factor R is defined as

$$R = \frac{q_b}{q_p}$$

The steady-state concentration is then defined as

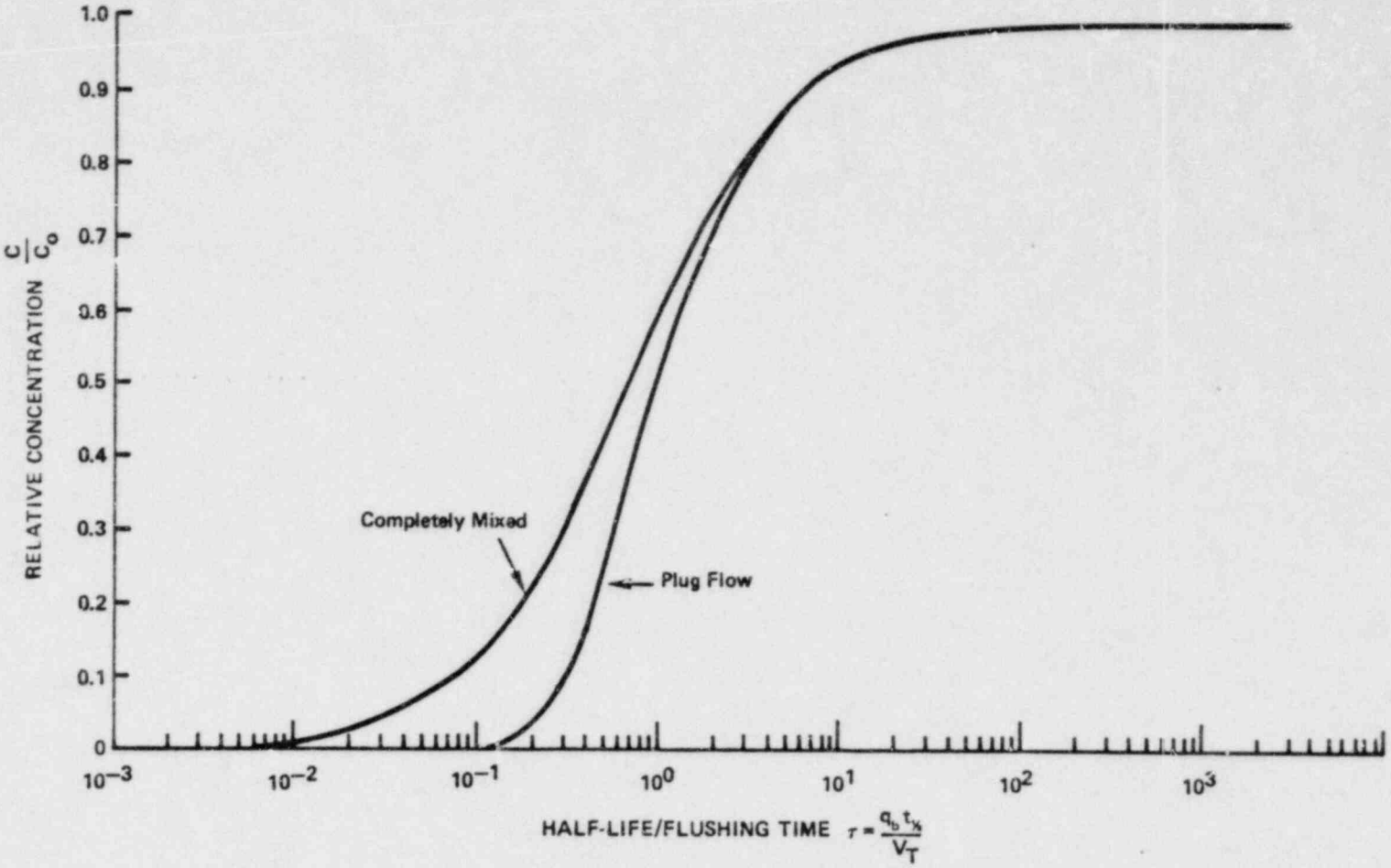


FIGURE 9.
STEADY-STATE CONCENTRATIONS IN COMPLETELY MIXED AND PLUG-FLOW MODELS

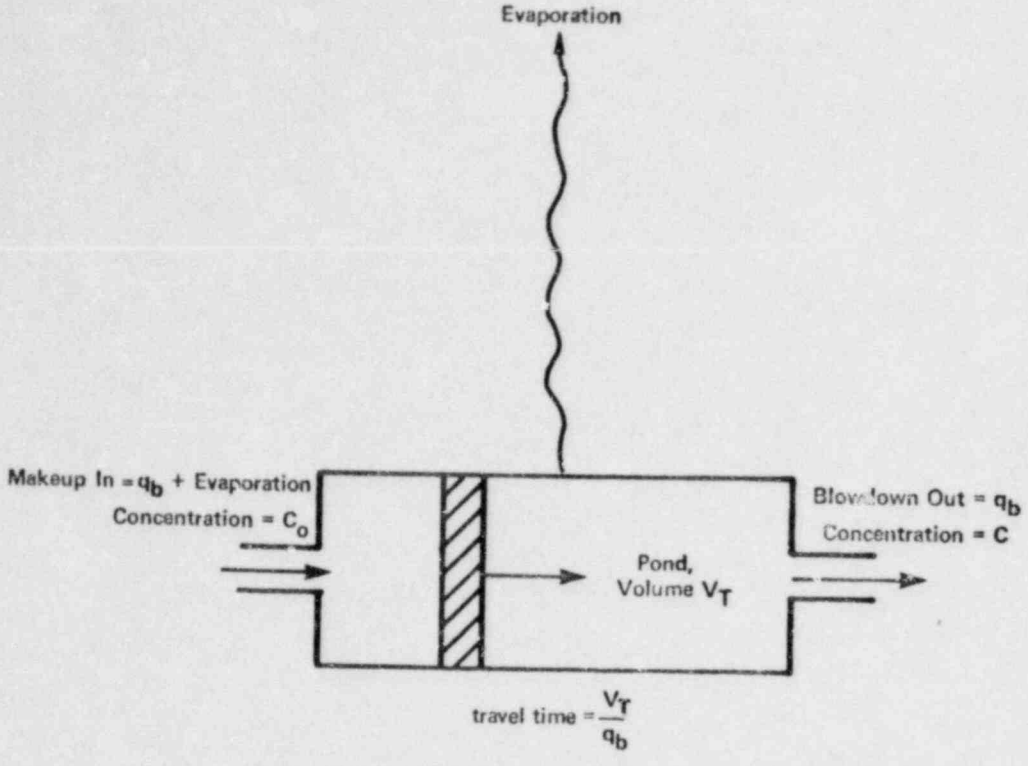


FIGURE 10.
PLUG-FLOW MODEL

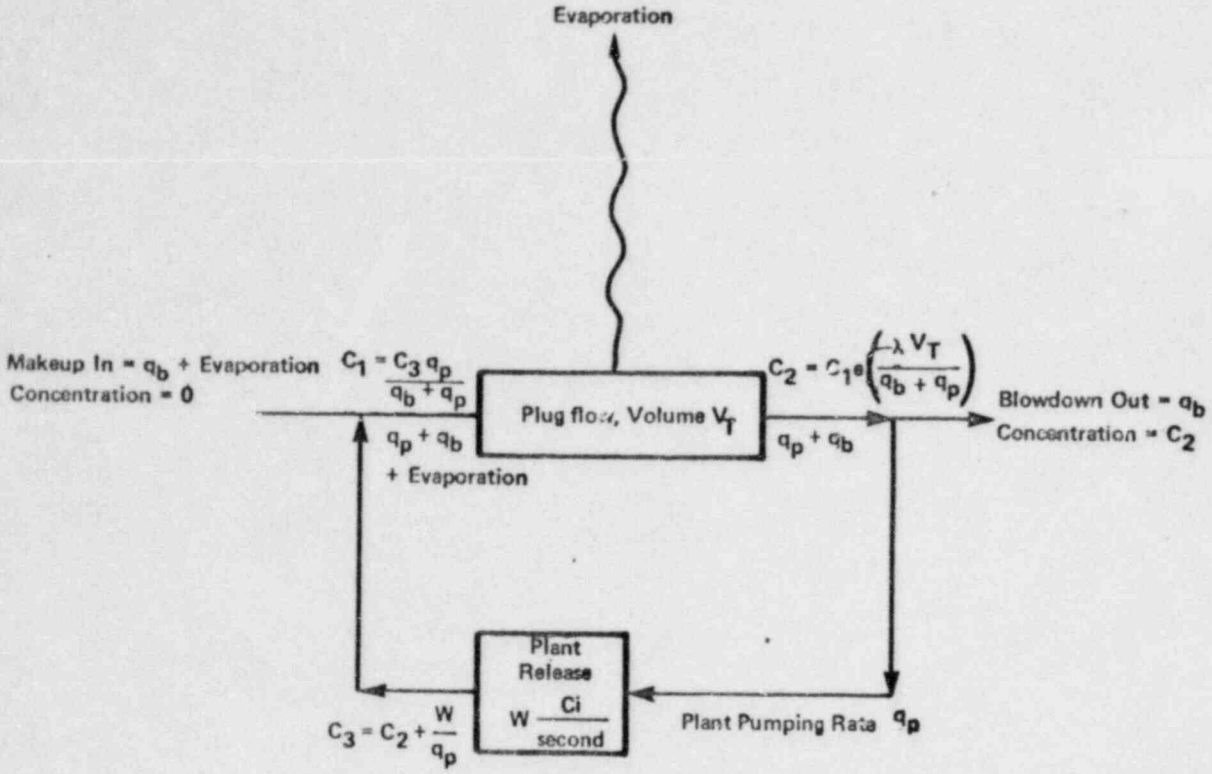


FIGURE 11.
PARTIALLY MIXED MODEL

$$\frac{C}{C_o} = \frac{R}{(R+1)\exp\left[\frac{R}{\tau(R+1)}\ln 2\right] - 1} \qquad (47)$$

Figure 12 shows the relative concentration C/C_o as a function of τ and R. Notice that the asymptotic forms of Equation (47) for R = 0 reduce to the completely mixed case:

$$\frac{C}{C_o} = \frac{\tau}{\tau + \ln 2} \qquad (45)$$

and for R = ∞ to the plug-flow case:

$$\frac{C}{C_o} = \exp\{-\ln 2/\tau\} \qquad (46)$$

(4) Stratified Reservoir Lumped Parameter Model

A simple model based on an approach of Trent (Ref. 56) is useful for a gross approximation of the mixing processes in stratified reservoirs that have seasonal turnover. The lake is assumed to have two layers, each totally mixed (Figure 13). Water can flow into and be withdrawn from either layer, but the layers do not mix during stratified flow periods. The volume of each layer is assumed constant during the period of stratification or during unstratified flow. Turnover is assumed to mix the two layers totally and instantaneously. This assumption is reasonable for systems in which the turnover time is small compared to the residence time.

Input data needed for this model are as follows:

Stratified Period

1. Length of stratified period
2. Volume of epilimnion and hypolimnion (constant over period)
3. Inflow and withdrawal (same), either layer
4. Concentration in inflow
5. Half-life of constituent

Unstratified Period

1. Length of unstratified period
2. Total volume of reservoir (constant over period)
3. Inflow and withdrawal (same)
4. Concentration in inflow
5. Half-life of constituent

The model is initialized with concentration C(o) at time t = 0. The first period, $(0<t<t_1)$, corresponds to stratified conditions. Concentration in the epilimnion during this period is

$$C_E = \frac{a - [a - b\,C(o)]e^{-bt}}{b} \qquad (48)$$

$$a = \frac{C_i q_i}{V_E},\quad b = \frac{q_i}{V_E} + \lambda$$

where

C_i is the input concentration in the upper layer;

q_i is the inflow to the upper layer;

V_E is the volume of epilimnion; and

λ is the decay constant and is = $\ln 2/t_{\frac{1}{2}}$.

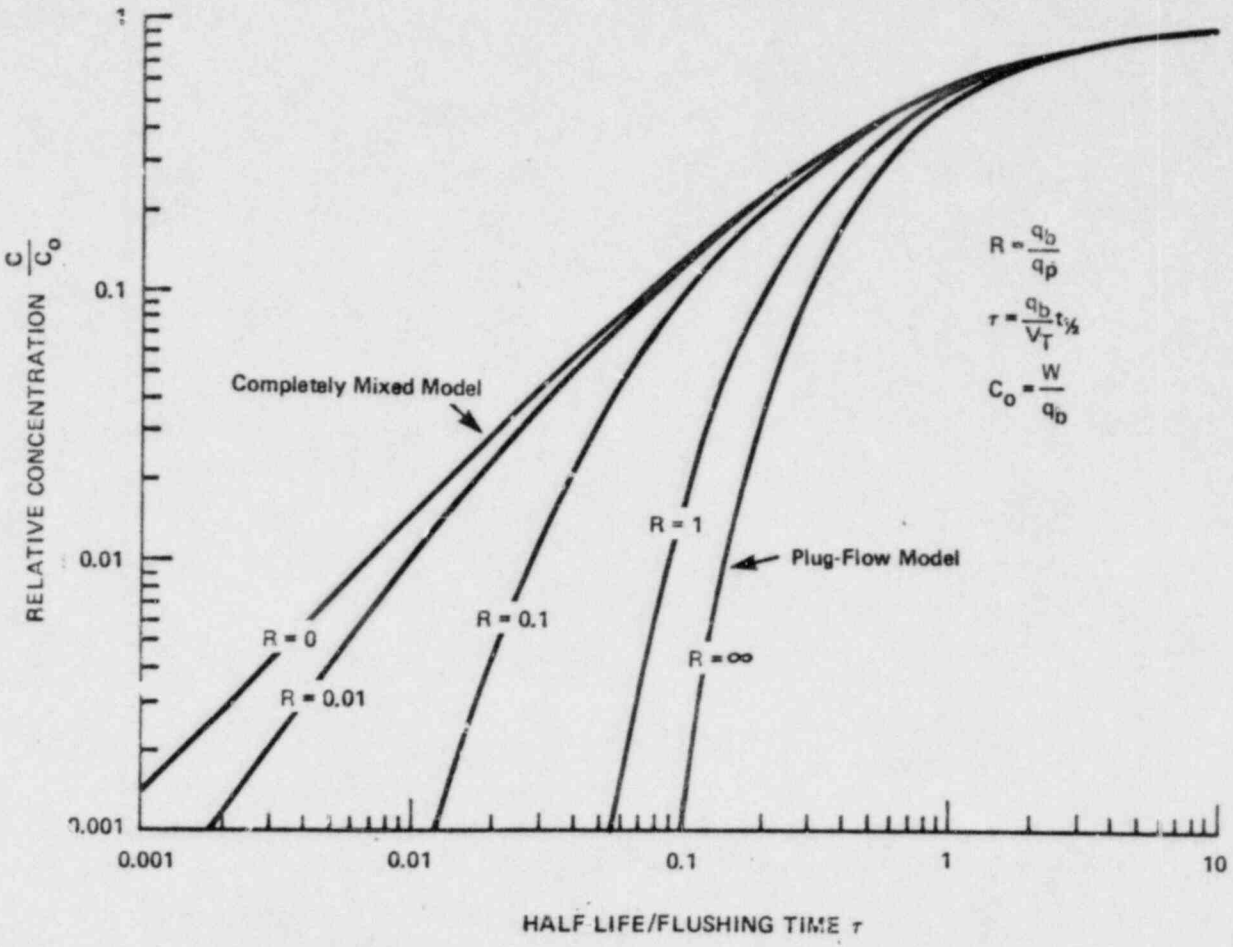
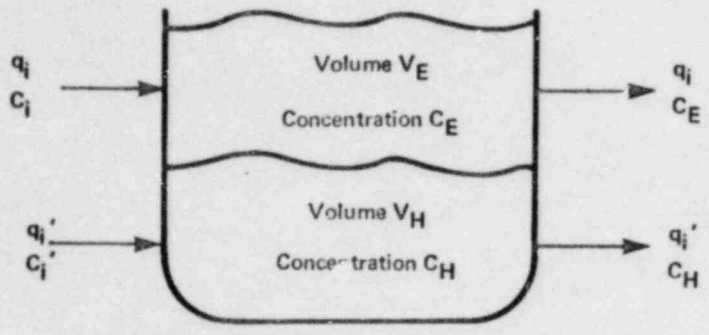


FIGURE 12.
CONCENTRATION IN PARTIALLY MIXED CASE

a. Stratified



b. Unstratified

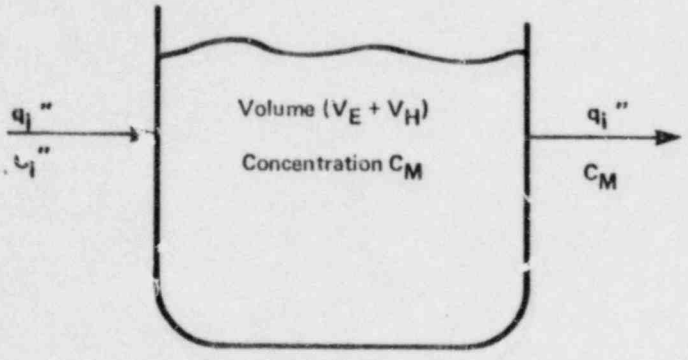


FIGURE 13.

LUMPED PARAMETER STRATIFIED
RESERVOIR MODEL

The concentration in the lower layer during this period is

$$C_H = \frac{a' - [a' - b'C(o)]e^{-b't}}{b'} \qquad (49)$$

$$a' = \frac{C_i' q_i'}{V_H}$$

$$b' = \frac{q_i'}{V_H} + \lambda$$

where

C_i' is the input concentration to the lower layer;

q_i' is the inflow to the lower layer; and

V_H is the volume of hypolimnion.

During turnover, instantaneous mixing of the two layers is assumed as

$$C_T = \frac{C_E V_E + C_H V_H}{V_E + V_H} \qquad (50)$$

During the unstratified flow period, $(t_1<t<t_2)$, the concentration is

$$C_M = \frac{a''-(a''-b''C_T)e^{-b''(t - t_1)}}{b''} \qquad (51)$$

$$a'' = \frac{C_i'' q_i''}{V_T}, \quad b'' = \frac{q_i''}{V_T} + \lambda$$

where

C_i'' is the input concentration for the total pond;

q_i'' is the inflow; and

V_T is the total volume.

Subsequent seasonal stratification cycles repeat with Equation (48), but with the latest fully mixed concentration substituted for C(o).

Simple methods such as the well mixed, partially mixed, and stratified models covered in this and the previous section are most suitable for estimating concentrations in reservoirs, ponds, and lakes that are downstream of the radionuclide discharge and consequently already partially mixed. In such a case, the hydraulics of the pond are less important and simple methods may suffice.

More elaborate models may be required, however, for a direct radionuclide discharge to the reservoir. In this case, the hydraulics of the reservoir may strongly affect the way in which the radionuclide releases disperse.

(5) Buildup of Isotopes Using Simple Models

For large cooling ponds with relatively small blowdown rates, the concentrations of the longer-lived isotopes may build up over a period of several years (exclusive of sediment uptake effects). It may be desirable to know the rate of this buildup, since the flushing time would be a significant fraction of the useful life of the plant.

Since the only concern is long-lived isotopes, the completely mixed model is analyzed for transient conditions.

If the concentration in the pond is initially zero, then

$$\frac{C}{C_o} = \frac{\tau}{\tau + \ln 2} \left(1 - \exp\{-[\tau + \ln 2]t/t_{\frac{1}{2}}\}\right) \qquad (44)$$

Figure 14 shows the buildup to steady-state concentration C_{SS} as a function of dimensionless time θ (time/flushing time) for several values of τ. Notice that Equation (44) becomes fairly insensitive to τ for large τ. This fact is illustrated more clearly by observing the time to reach some arbitrary fraction, say 99% of steady state:

$$1 - \exp\left[-(\tau + \ln 2)\frac{t}{t_{\frac{1}{2}}}\right] = 0.99 \text{ and} \qquad (52)$$

$$\theta_{99} = \left(\frac{tq_b}{V_T}\right)_{99} = -\tau\left(\frac{\ln 0.01}{\tau + \ln 2}\right)$$

Figure 15 illustrates the 99% buildup time as a function of τ, clearly showing how, for very long half-lives compared with pond flushing times, the time depends only on the flushing time of the pond.

(6) Hydraulics of Ponds Using Simpl lodels

The simple models presented here must be used with caution for several reasons. Large portions of a pond may be unused for dilution if the pond is unstratified and irregularly shaped. If the same pond becomes stratified during certain times of the year, however, previously unused sections may become useful because of density flows (Ref. 57) and because of the strong mixing induced by seasonal turnover.

In a flow-through cooling pond without recirculation, stratification may be detrimental because the thermal effluent and the radioisotopes may be confined to the upper layer, thereby reducing the effective volume of the pond. Thus, definition of the effective volume of a pond may be difficult.

It should be possible, however, to pick a conservative volume for a "worst case" calculation. Calculation of the steady-state concentration of isotopes whose half-lives are long should cause little error because the concentrations approach that of a conservative substance

$$C_o = \frac{W}{q_b}$$

regardless of pond hydraulics.

b. Numerical Models

Stratified reservoir models are in most cases numerical evaluations of the one- or two-dimensional equations describing convection and diffusion in stratified flow. The simplest numerical models are the one-dimensional diffusion models, such as the MIT deep reservoir model (Ref. 57). In such models, concentration is assumed to be horizontally uniform. Vertical diffusion and advection are modeled. The flow field is calculated by the equation of continuity and by accounting for inflows and outflows such as dams, tributaries, and outfalls to and from the different layers of the reservoir. Such models are useful where there is strong stratification, especially where the reservoir is used for direct condenser cooling. In such cases, the stratification is reinforced by the additional heat, discharge is usually to the surface, withdrawal is from the hypolimnion, and vertical gradients are more pronounced than horizontal gradients. These models are less accurate for reservoirs that have seasonal turnover and subsequent unstratified periods. Furthermore, these models are incapable of simulating certain important effects, such as horizontal mixing in the vicinity of a power plant discharge.

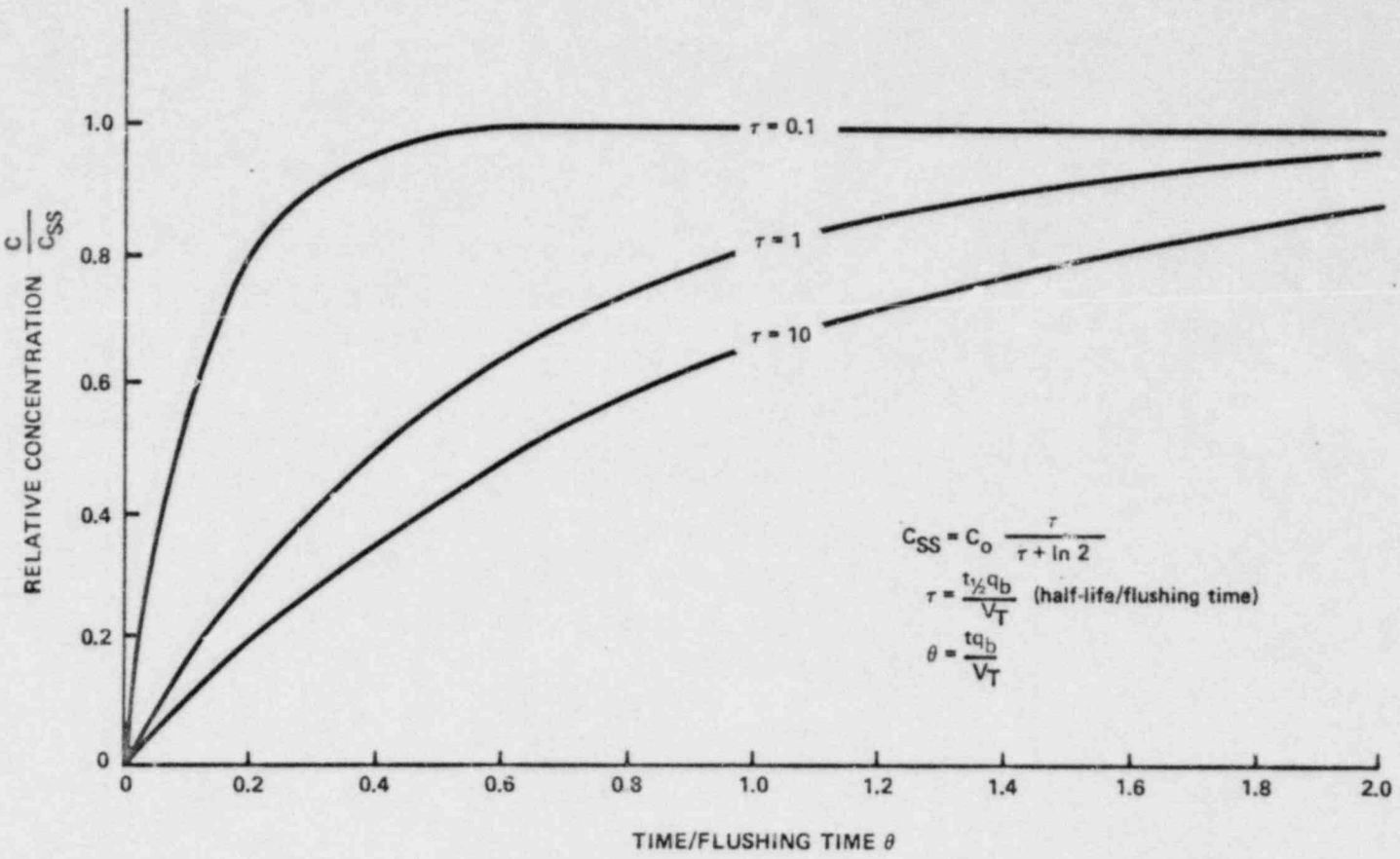


FIGURE 14.
BUILDUP OF RADIOISOTOPES IN COMPLETELY MIXED MODEL (NORMALIZED)

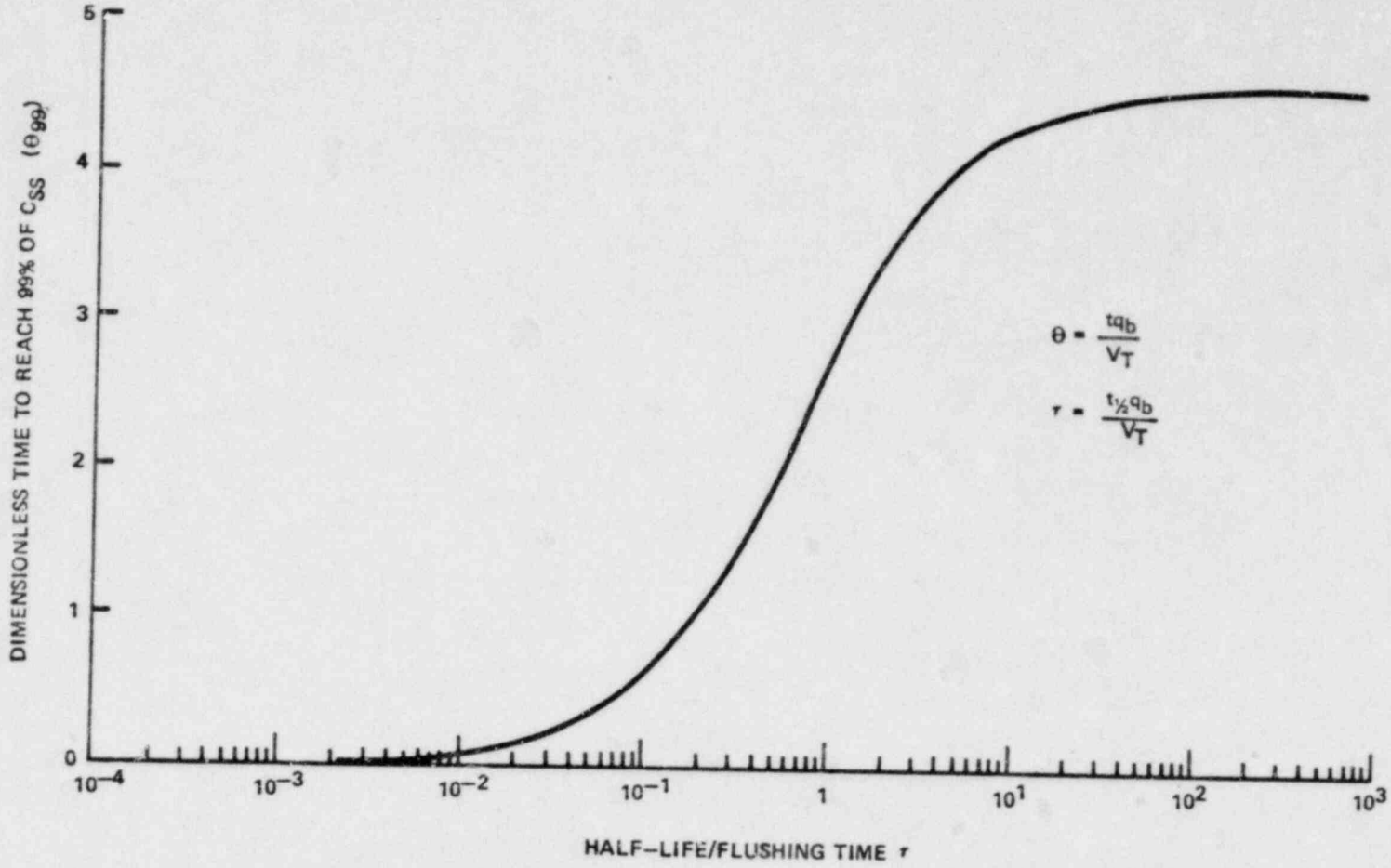


FIGURE 15.
DIMENSIONLESS TIME TO REACH 99% OF STEADY-STATE CONCENTRATION IN COMPLETELY MIXED CASE

Where both vertical and horizontal effects are important, more complicated models are warranted. For example, the EPA Reservoir Model (Ref. 58) allows for the horizontal segmentation of the reservoir. Each reservoir segment is solved in the vertical direction. Inputs from upstream and downstream segments are coupled empirically to account for density flows.

Another useful model is the reservoir model of Ryan and Harleman (Ref. 59). In this model, the one-dimensional horizontal solution in the surface layer is coupled with a one-dimensional solution in the vertical direction. Effects of discharge mixing, surface cooling, and lateral gradients are calculated in the surface layer solution. Vertically stratified flows and selective withdrawals are handled by the vertical solution. The model is most useful for cooling reservoirs where stratification is strong.

6. WATER USE

For identifying liquid pathways to man for realistic evaluations of the doses from normal releases from nuclear power plants, it is necessary to locate water users, identify ty of uses, and estimate usage to a distance of 50 miles from a site. Because of high usage rates along many streams and estuaries, it is also necessary to identify the effects of water usage on the spatial and temporal distribution of flows. In addition, water usage upstream of a nuclear power plant can alter flows at or downstream of the plant. The following is an acceptable methodology to evaluate water usage and the consequences thereof on streamflows. This methodology provides a realistic evaluation of the liquid transport of normal releases of radionuclides from nuclear power plants along streams and estuaries.

a. Users

A schematic diagram of the river basin that locates the following features in relation to the plant site should be prepared: (1) surface water uses* upstream and downstream of the plant site, (2) major tributaries and their junctions, (3) streamflow gaging stations (including their period of record), and (4) major reservoirs and diversions upstream and downstream of the plant site. Approximate contributing drainage areas and types of water use for all points identified should be shown on the diagram or tabulated separately. An example schematic diagram is shown in Figure 16.

From the diagram, key diversion and streamflow stations should be selected to provide the basis for establishing reasonable spatial and temporal distributions of runoff patterns, upstream and downstream of nuclear power plant sites. Historical streamflows at major mainstem and tributary locations should be adjusted for historical diversions and reservoir effects to produce "natural" flows (e.g., flows independent of reservoir and diversion effects). Missing streamflow data (i.e., gaps in records) at critical stations may subsequently be synthesized by direct correlation with nearby streamflow stations or by statistical correlation with many stations using models such as "HEC-4, River Flow Simulation" (Ref. 60). In general, a period of at least 20 to 30 years of record, including the historical drought of record in the region, should be used.

Monthly durations have been found (Refs. 61 and 62) to describe adequately the fluctuations in streamflow without introducing significant errors in long-term estimates of reservoir yields. Similarly, estimates of average annual radionuclide concentrations along a river, based on monthly streamflow averages, produced acceptable values when compared with longer flow periods (see Figure 17). Flow periods for multidirectional estuaries, lakes, and ocean sites were discussed in Sections 3 and 4 of this appendix.

b. Usage

The effects on streamflows at and downstream of a site caused by reservoirs or diversions should be identified. Reservoirs may cause significant changes in the distribution of seasonal runoff. Operating constraints on reservoir storage, outflow, and diversions should be established on a monthly basis for existing conditions of basin development and water demand and for anticipated future conditions throughout the lifetime of the plant. Where proposed reservoirs may significantly affect flow conditions, their effects should be considered by simulating their operation using models such as "HEC-3, Reservoir Analysis" (Ref. 63).

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*Use types include drinking water, irrigation, process water (consumed by such users as breweries and soft drink manufacturers), recreation areas, and fisheries. Ground water users with wells whose zones of influence extend to streams should also be included.

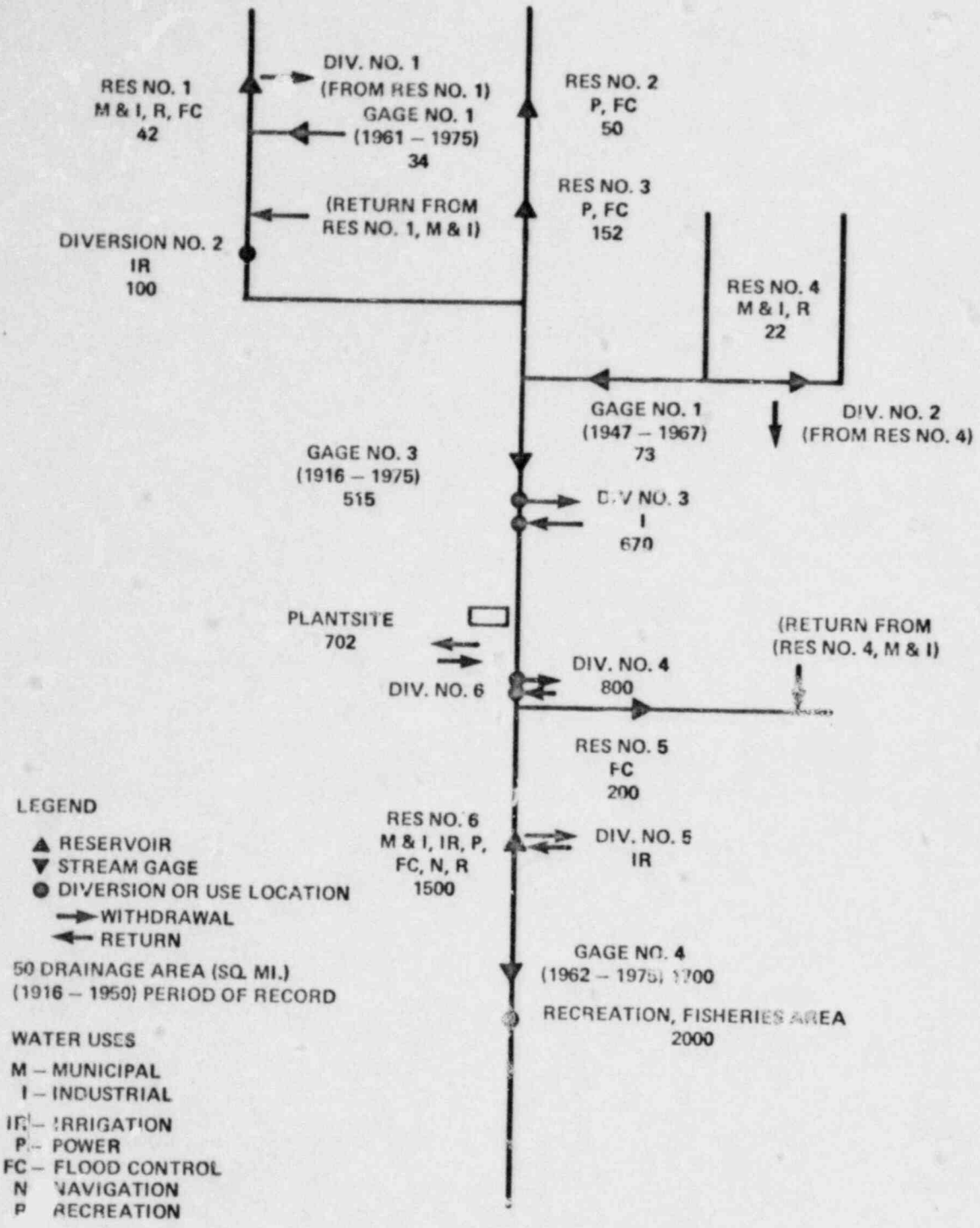

FIGURE 16. TYPICAL SCHEMATIC DIAGRAM OF A RIVER BASIN

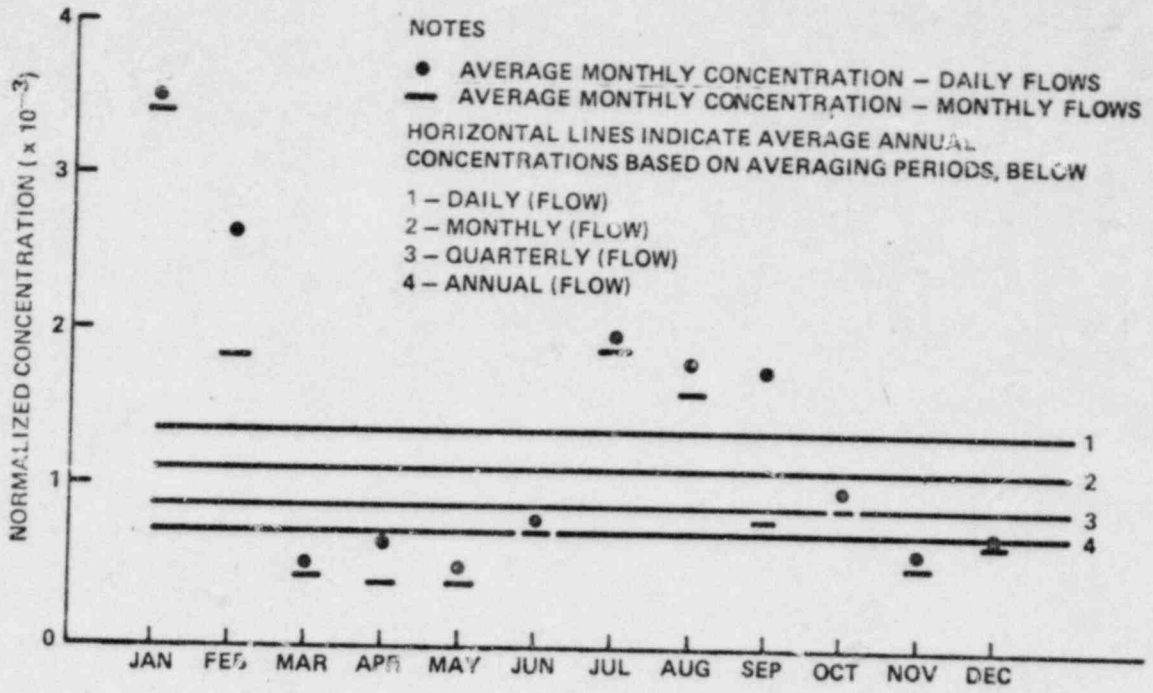

FIGURE 17.

EFFECT OF FLOW AVERAGING PERIOD ON NORMALIZED CONCENTRATION

(DATA – KANKAKEE RIVER 1961)

Many alternative schemes for developing water resources may have been proposed in a particular region, and it is difficult to conclude which, if any, are likely to be constructed. The evaluation described herein should consider any Federal, state or locally authorized projects, any projects adopted by local river basin commissions; or any other projects whose chances of being constructed are considered likely.

The locations of fisheries and recreation areas that will be exposed to normal releases of radionuclides within 50 radial miles of a plant should be identified. Present and projected future contact by humans and fish should be estimated.

The use of river system simulation models such as HEC-3 to adjust historic streamflows and to synthesize existing and potential future water use can provide acceptable estimates of the spatial and temporal distributions of streamflows at the locations for which estimates of average radionuclide concentrations are required. Two basic assumptions are required. First, it is assumed that the period of recorded historical streamflow selected for simulation is representative of conditions to be expected in the future. Secondly, it is assumed that it is necessary to adjust historical streamflows for the effects of reservoirs and diversions. If both of these assumptions are not supported by historical basin water usage, no adjustments are necessary. Furthermore, if projected water use cannot be expected to alter streamflows in a way such as to require basin simulation, no flow adjustments are necessary.

Water use should be estimated, on a monthly basis, for present and projected future conditions. Local, state, or regional agencies often maintain records of such use, and such information should be summarized and appropriately annotated. These same agencies have in many cases made projections of future usage; these projections should be summarized and annotated.

Where existing records or locally projected usage information is not available or is considered unreliable, conservative estimates may be made from population projections and forecasted per capita usage estimates of agencies such as the U.S. Geological Survey or the Water Resources Council. Where population or water use forecasts are at variance with other forecasts submitted by the applicant, the bases for the differences should be provided.

Although conservative estimates are sometimes required to ensure that the consequences of accidents are acceptable, realistic estimates will be adequate for the normal effluent release evaluations required by 10 CFR Part 50, Appendix I.

c. Existing Studies

Many studies of the effects of water resource development schemes have been completed in great detail. For example, many large river basins have been studied by the Corps of Engineers and others. These existing studies, with only minor modifications for plant water use, may be used directly.

7. SEDIMENT UPTAKE AND TRANSPORT MODELS

The ability of suspended and bottom sediments to absorb and adsorb radioactive nuclides from solution is recognized as contributing to important pathways to man through the sediment's ability to concentrate otherwise dilute species of ions. The pathways are by direct contact with the populace and by transfer to the aquatic food chain. Sorption by sediments is also an important mechanism for reducing the area of influence of plant releases.

The state of the art in sediment-related effects is not as advanced as in other engineering disciplines. For this reason, and until reliable models become available, the staff will rely on existing field studies and the staff's and consultants' experience to determine the conservatism or reasonableness of the applicant's analyses and results.

If the applicant elects to take credit for the removal of certain ions from the surface waters by sediment uptake, extensive verification of the techniques used will be necessary because of the lack of existing verified generalized models. The applicant's models will be evaluated based on their adherence to physical principles and their proven reliability in simulating prototype data. Models will be compared, in terms of physical principles, to those listed in the references, in accordance with the classification of receiving water. However, the staff does not accept these models a priori. Because most existing sediment uptake and transport modeling techniques are crude, the applicant should demonstrate that the model is conservative or realistically simulates the prototype. The model verification will be accepted based on the quality of comparison with measurements for water bodies having characteristics

similar to those at the site. Actual measurements of sorption characteristics for pertinent radionuclides should be presented for areas of the water body within the influence of the plant. These measurements should reflect seasonal variations of sources and sinks (spatial and grain size) and the physical and chemical properties of the receiving water.

Estimates of sediment movement should be supported by actual field measurements (by the applicant or others). Events and processes affecting sediment movement and deposition (e.g., floods, storms, wave activity, and estuarine stratification) should be considered. Changes in the character of the receiving water that would influence sediment transport (e.g., dams, jetties, groins, and shoreline changes) should be considered.

LIST OF SYMBOLS

Symbol	Description
A	Cross-sectional area of river or estuary
b	Width of estuary at water surface
B	Width of river
C	Concentration of constituent in water
C_E	Concentration in epilimnion of stratified impoundment
C_h	Chezy resistance coefficient
C_H	Concentration in hypolimnion of stratified impoundment
C_i	Input concentration in the upper layer of a stratified impoundment
C_i'	Input concentration to the lower layer of a stratified impoundment
C_i''	Input concentration for the total impoundment
C_o	Steady-state concentration of a nondecaying substance
C_T	Concentration in total impoundment at time of turnover
C_{SS}	Steady-state concentration of a decaying substance
d	Depth of river
D	Diffusion factor
D_F	Dilution factor
E	Sectionally averaged, one-dimensional longitudinal dispersion coefficient
E_L	Estuarine, sectionally and tidally averaged, one-dimensional longitudinal dispersion coefficient
E_D	Estuary number (dimensionless coefficient)
f	Coriolis parameter
F_D	Densimetric Froude number at downstream boundary of estuary
g	Acceleration of gravity
H	Depth in estuary or open coast measured from water surface to bottom
K	Empirical coefficient used in description of mixing in salinity intrusion region of an estuary
K_x, K_y	Vertically averaged, two-dimensional turbulent diffusion coefficients in x and y directions, respectively
ℓ	Distance from point of maximum velocity to further bank in river flow
L	Length of estuary

Symbol	Description
M	Quantity of radioactivity released in a pulse discharge
n	Manning's coefficient
N	Dimensionless estuarine freshwater velocity
P_T	Tidal prism
q	Cumulative river discharge measured from near shore
Q	Total river discharge
$\tilde{q}$	Dimensionless cumulative river discharge
q_b	Flow rate of blowdown through the pond
Q_f	Freshwater volumetric discharge rate in estuary
q_i	Inflow to the upper layer of a reservoir
q_i'	Inflow to the lower layer of a reservoir
q_i''	Inflow to the total reservoir
q_p	Plant pumping rate
q_s	Cumulative river discharge rate at position of point source
$\tilde{q}_s$	Dimensionless cumulative river discharge rate at position of point source
q_{s1}, q_{s2}	Cumulative river discharges at beginning and end of line source, respectively
R	Recirculation factor for impoundments
R_h	Hydraulic radius
S	Salinity in estuary
S_o	Salinity at downstream boundary of estuary
t	Time
t_d	Duration of pulse discharge
T	Tidal period
$t_{\frac{1}{2}}$	Half-life of radionuclide
u	Velocity in x direction
$\bar{u}$	Sectional mean velocity in x direction
u'	Deviation from sectional mean velocity $\bar{u}$
U	Vertically averaged velocity in x direction
$\bar{U}$	Sectionally averaged longitudinal velocity in river or estuary
U_f	Sectionally averaged fresh water velocity in estuary
U_t	RMS velocity in an oscillating flow
u^*	Shear velocity

Symbol	Description
v	Velocity in y direction
V	Vertically averaged velocity in y direction
V_E	Volume of epilimnion in stratified impoundment
V_H	Volume of hypolimnion in stratified impoundment
V_{max}	Maximum local tidal velocity
V_T	Total volume of impoundment
W	Rate of radioactivity addition, in Ci/sec
x	Longitudinal coordinate
$\tilde{x}$	Dimensionless longitudinal coordinate
y	Lateral coordinate
y_s	Lateral position of a point source
y_{s1}, y_{s2}	Lateral positions of beginning and end of line source in cartesian coordinates
z	Vertical coordinate
z_s	Vertical position of a point source
α	Dimensionless coefficient
β	Dimensionless coefficient
Γ	Dimensionless cross-sectionally averaged concentration
Γ_{max}	Γ at the source position
ϵ_y	Lateral turbulent diffusion coefficient
ϵ_z	Vertical turbulent diffusion coefficient
ζ	Water surface elevation above undisturbed datum
θ	Dimensionless time (ratio of time to flushing time)
λ	Radioactive decay coefficient ($\ln 2/t_{\frac{1}{2}}$)
ξ	Dimensionless longitudinal distance from source in a one-dimensional estuary
σ_y, σ_z	Standard deviations of radionuclide concentrations in y- and z-directions, respectively
τ	Ratio of the half-life to the impoundment flushing time
τ_x^s, τ_y^s	x and y components of surface wind stress
X	Nondecaying concentration
$\tilde{X}$	Dimensionless nondecaying concentration

CITED REFERENCES

1. G. H. Jirka, G. Abraham, and D. R. F. Harleman, Massachusetts Institute of Technology, "An Assessment of Techniques for Hydrothermal Prediction," U.S. Nuclear Regulatory Commission, NUREG-0044, 1976. (Copies may be obtained from the National Technical Information Service, Springfield, Va.)

2. W. E. Cunn, A. J. Policastro, and R. A. Paddock, "Surface Thermal Plumes: Evaluation of Mathematical Models for the Near and Complete Field," Parts One and Two, Argonne National Laboratory, Energy and Environmental Systems Division, ANL/WR-75-3, 1975.

3. K. D. Stolzenbach and D. R. F. Harleman, "An Analytical and Experimental Investigation of Surface Discharges of Heated Water," Ralph M. Parsons Laboratory for Water Resources and Hydrodynamics, Massachusetts Institute of Technology, Rep. No. 135, 1971.

4. K. D. Stolzenbach, E. E. Adams, and D. R. F. Harleman, "A User's Manual for Three-Dimensional Heated Surface Discharge Conditions," Ralph M. Parsons Laboratory for Water Resources and Hydrodynamics, Massachusetts Institute of Technology, Rep. No. 156, 1972.

5. E. A. Prych, "A Warm Water Effluent Analyzed as a Buoyant Surface Jet," Sveriges Meteorologiska och Hydrologiska Institut, Serie Hydrologi, nr 21, Stockholm, Sweden, 1972.

6. M. A. Shirazi and L. R. Davis, "Workbook of Thermal Plume Prediction Vol. 2 - Surface Discharge," U.S. Environmental Protection Agency, National Environmental Research Center, Environmental Protection Technology Series, Rep. No. EPA-R2-72-005b, Corvallis, Ore., 1974.

7. D. W. Pritchard, "Design and Siting Criteria for Once-Through Cooling Systems," presented at American Institute of Chemical Engineers, 68th Annual Meeting, Houston, Tex., 1971.

8. D. W. Pritchard, "Fate of and Effect of Excess Heat Discharged into Lake Michigan with Specific Application to the Condenser Cooling Water Discharge from the Zion Nuclear Power Station," Testimony at the AEC Licensing Hearings for the Zion Nuclear Power Station Operating Permit, Chicago, Ill., 1973.

9. G. H. Jirka and D. R. F. Harleman, "The Mechanics of Submerged Multiport Diffusers for Buoyant Discharges in Shallow Water," Ralph M. Parsons Laboratory for Water Resources and Hydrodynamics, Massachusetts Institute of Technology, Rep. No. 169, 1973.

10. J. H. W. Lee, G. H. Jirka, and D. R. F. Harleman, "Stability and Mixing of a Vertical Round Buoyant Jet in Shallow Water," Ralph M. Parsons Laboratory for Water Resources and Hydrodynamics, Massachusetts Institute of Technology, Rep. No. 195, 1974.

11. R. C. Y. Koh and L. N. Fan, "Mathematical Models for the Prediction of Temperature Distributions Resulting from the Discharge of Heated Water in Large Bodies of Water," Water Pollution Control Research Series, Rep. No. 16130DW010/70, 1970.

12. E. Hirst, "Analysis of Round, Turbulent, Buoyant Jet Discharged to Flowing Stratified Ambients," Oak Ridge National Laboratory, Rep. No. 4685, 1971.

13. W. W. Sayre and F. M. Chang, "A Laboratory Investigation of Open-Channel Dispersion Processes for Dissolved, Suspended, and Floating Dispersants," U.S. Geological Survey, Professional Paper 433-E, 1968.

14. N. Yotsukura and E. D. Cobb, "Transverse Diffusion of Solutes in Natural Streams," U.S. Geological Survey, Professional Paper 582-C, 1972.

15. N. Yotsukura, "A Two-Dimensional Temperature Model for a Thermally Loaded River with Steady Discharge," Proceedings, 11th Annual Environmental and Water Resources Engineering Conference, Vanderbilt University, Nashville, Tenn., 1972.

16. N. Yotsukura and W. W. Sayre, "Transverse Mixing in Natural Channels," Water Resources Research, Vol. 12, No. 4, August 1976.

17. F. M. Henderson, Open Channel Flow, New York: Macmillan Company, 1959.

18. V. T. Chow, Open Channel Hydraulics, New York: McGraw-Hill Book Company, 1959.

19. J. W. Elder, "The Dispersion of Marked Fluid in Turbulent Shear Flow," Journal of Fluid Mechanics, No. 5, 1959, pp. 544-60.

20. H. B. Fischer, "The Effects of Bends on Dispersion in Streams," Water Resources Research 5(2), 1969, pp. 496-506.

21. N. Yotsukura, H. B. Fischer, and W. W. Sayre, "Measurement of Mixing Characteristics of the Missouri River Between Sioux City, Iowa, and Plattsmouth, Nebraska," U.S. Geological Survey, Water-Supply Paper 1899-G, 1970.

22. W. W. Sayre and T. P. Yeh, "Transverse Mixing Characteristics of the Missouri River Downstream from the Cooper Nuclear Station," University of Iowa, Iowa Institute of Hydraulic Research, Rep. No 145, Iowa City, Iowa, 1973.

23. G. T. Csanady, "Dispersal of Effluents in the Great Lakes," Water Research, vol. 4, 1970, pp. 79-114.

24. Stone and Webster, "1970 Lake Temperature and Current Studies," Report to Power Authority, State of New York, AEC Docket No. 50-333, 1971.

25. G. T. Csanady, "Response of Large Stratified Lakes to Wind," Journal of Physical Oceanography, vol. 2, no. 1, 1972, pp. 3-13.

26. G. T. Csanady, "The Coastal Boundary Layer in Lake Ontario: Part II. The Summer-Fall Regime," Journal of Physical Oceanography, vol. 2, no. 2, 1972, pp. 168-176.

27. C. R. Murthy and G. T. Csanady, "Experimental Studies of Relative Diffusion in Lake Huron," Journal of Physical Oceanography, vol. 1, 1971, pp. 77-24.

28. T. R. Sundaram and R. G. Rehm, "The Seasonal Thermal Structure of Deep Temperate Lakes," Tellus, vol. 25, 1973, pp. 157-168.

29. J. J. Leendertse, "A Water-Quality Simulation Model for Well-Mixed Estuaries and Coastal Seas: Volume I, Principles of Computation," The Rand Corporation, No. RM-6230-RC, 1970.

30. J. J. Leendertse and E. C. Gritton, "A Water-Quality Simulation Model for Well-Mixed Estuaries and Coastal Seas: Volume II, Computation Procedures," The Rand Corporation, No. R-708-NYC, 1971.

31. J. J. Leendertse and E. C. Gritton, "A Water-Quality Simulation Model for Well-Mixed Estuaries and Coastal Seas: Volume III, Jamaica Bay Simulation," The Rand Corporation, No. R-709-NYC, 1971.

32. J. J. Leendertse, "A Water-Quality Simulation Model for Well-Mixed Estuaries and Coastal Seas: Volume IV, Jamaica Bay Tidal Flows," The Rand Corporation, No. R-1009-NYC, 1972.

33. E. C. Gritton, "A Water-Quality Simulation Model for Well-Mixed Estuaries and Coastal Seas: Volume V, Jamaica Bay Rainstorms," The Rand Corporation, No. R-1010-NYC, 1972.

34. J. J. Leendertse and S. K. Liu, "A Water-Quality Simulation Model for Well-Mixed Estuaries and Coastal Seas: Volume VI, Simulation, Observation, and State Estimation," The Rand Corporation, No. R-1586-NYC, 197[illegible].

35. R. B. Codell, "Digital Simulation of Thermal Effluent Dispersion in Rivers, Lakes, and Estuaries," Ph.D. Dissertation, Lehigh University, 1973.

36. L. A. Loziuk, J. C. Anderson, and T. Belytschko, "Hydrothermal Analysis by Finite Element Method," Proceedings of the American Society of Civil Engineers, Journal of the Hydraulics Division, vol. 98 (HY11), 1972.

37. A. H. Eraslan, "Two-Dimensional, Discrete Element, Far Field Model for Thermal Impact Analysis of Power Plant Discharges in Coastal and Offshore Regions," Oak Ridge National Laboratory, Rep. No. 4940, 1975.

38. N. H. Brooks, "Diffusion of Sewage Effluent in an Ocean Current," Proceedings, 1st International Conference on Waste Disposal in the Marine Environment, Pergamon Press, 1960.

39. J. E. Foxworthy, "Eddy Diffusivity and the Four-Thirds Law in Near-Shore (Coastal Waters)," The Allan Hancock Foundation, University of Southern California, Report 68-1, 1969.

40. J. E. Foxworthy, "Eddy Diffusion and Bacterial Reduction in Waste Fields in the Ocean," The Allan Hancock Foundation, University of Southern California, Report 69-1, 1969.

41. R. E. Stewart, H. D. Putnam, R. H. Jones, and T. N. Lee, "Diffusion of Sewage Effluent from Ocean Outfall," Proceedings of the American Society of Civil Engineers, Journal of the Sanitary Engineering Division, vol. 97, SA4, 1971.

42. H. H. Carter and A. Okubo, "A Study of the Physical Processes of Movement and Dispersion in the Cape Kennedy Area," Chesapeake Bay Institute, The Johns Hopkins University, Report No. NYO-2973-1, 1965.

43. G. Kullenberg, "Vertical Diffusion in Shallow Waters," Tellus, vol. 23, 1971, pp. 129-135.

44. G. Kullenberg, "Results of Diffusion Experiments in the Upper Reaches of the Sea," Institute of Physical Oceanography, University of Copenhagen, Rep. No. 12, 1971.

45. A. Okubo, "Oceanic Mixing," Chesapeake Bay Institute, Johns Hopkins University, Tech. Rep. 62, Docket No. NYO-3109-49, 1970.

46. D. J. O'Conner and J. P. Lawler, "Mathematical Analysis of Estuarine Pollution," 55th National Meeting, American Institute of Chemical Engineers, Reprint No. 31a, Houston, Tex., 1965.

47. TRACOR, Inc., "Estuarine Modeling, An Assessment," Environmental Protection Agency, Project No. 16070DZV, Contract 14-12-551, 1971.

48. R. L. Crim and N. L. Lovelace, "Auto-Qual Modeling System," U.S. Environmental Protection Agency, EPA-440/9-73-003, Washington, D.C., March 1973.

49. J. E. Dailey and D. R. F. Harleman, "Numerical Model for the Prediction of Transient Water Quality in Estuary Networks," Ralph M. Parsons Laboratory for Water Resources and Hydrodynamics, Massachusetts Institute of Technology, Tech. Rep. 158, 1972.

50. C. H. Lee and D. R. F. Harleman, "One-Dimensional Real Time Model for Estuarine Water Quality Prediction," Water Quality Office, Environmental Protection Agency, Report 16070DGW, Washington, D.C., 1971.

51. A. H. Eraslan, unpublished users manual, "One Dimensional Model," Oak Ridge National Laboratories, 1975.

52. L. J. Hetling and R. L. O'Connell, "Estimating Diffusion Characteristics of Tidal Waters," Department of the Interior, Federal Water Pollution Control Administration, Middle Atlantic Region, CB-SRBP Technical Paper No. 4, 1967.

53. G. I. Taylor, "The Dispersion of Matter in Turbulent Flow in a Pipe," Proc. Royal Society London (A), vol. 223, May 1954.

54. M. L. Thatcher and D. R. F. Harleman, "A Mathematical Model for the Prediction of Unsteady Salinity Intrusion in Estuaries," R. M. Parsons Laboratory, Massachusetts Institute of Technology, Technical Report No. 144, February 1972.

55. H. B. Fischer, "The Mechanics of Dispersion in Natural Streams," Proceedings of the American Society of Civil Engineers, Journal of the Hydraulics Division, No. 93 (HY6), 1967, pp. 187-216.

56. D. S. Trent, "Mathematical Modeling of Transport Processes in Aquatic Systems," Battelle-Pacific Northwest Laboratories, BNWL-SA-5379, Richland, Wash., April 1975.

57. P. J. Ryan and D. R. F. Harleman, "Prediction of the Annual Cycle of Temperature Changes in a Stratified Lake or Reservoir - Mathematical Model and User's Manual," R. M. Parsons Laboratory, Massachusetts Institute of Technology, Report No. 137, 1971.

58. Water Resources Engineers, Inc., "Mathematical Models for the Prediction of Thermal Energy Changes in Impoundments," Walnut Creek, Calif., prepared for the Water Quality Office, Environmental Protection Agency, December 1969.

59. P. J. Ryan and D. R. F. Harleman, "An Analytical and Experimental Study of Transient Cooling Pond Behavior," R. M. Parsons Laboratory, Massachusetts Institute of Technology, Report No. 161, January 1973.

60. U.S. Army Corps of Engineers, "HEC-4, River Flow Simulation," Hydrologic Engineering Center, Davis, Calif., 1971.

61. L. G. Hulman, "Regional Water Supply Systems Analysis for a Megalopolis," paper presented at the August 1971 ASCE Hydraulics Division Specialty Conference, Iowa City, Iowa, 1971.

62. L. G. Hulman and D. D. Erickson, "Delaware River Basin Modeling," presented at the 17th Annual ASCE Specialty Conference, Logan, Utah, August 1969.

63. U. S. Army Corps of Engineers, "HEC-3, Reservoir Analysis," Hydrologic Engineering Center, Davis, Calif., 1971.

ADDITIONAL REFERENCES

SEDIMENT UPTAKE MODELS

Selected Mathematical Models and Theoretical Development

1. Oak Ridge National Laboratories, ORNL-TM-4751, "A System Analysis Model for Calculating Radionuclide Transport Between Receiving Waters and Bottom Sediments," R. S. Booth, Oak Ridge, Tenn., April 1975.

2. E. K. Duursma and C. J. Bosch, "Theoretical, Experimental, and Field Studies Concerning Diffusion of Radioisotopes in Sediments and Suspended Particles of the Sea," Netherlands Journal of Sea Research, vol. 4, no. 4, 1970, pp. 395-469.

3. D. E. Fields, "Modification of the Unified Transport Model to Allow for Sediment Transport and Chemical Exchange," Proceedings of the Second Annual NSF-RANN Trace Contaminants Conference, Asilomar, Calif., August 1974.

4. J. F. Fletcher and W. L. Dotson, "HERMES - A Digital Computer Code for Estimating Regional Radiological Effects from the Nuclear Power Industry," Hanford Engineering Development Lab., Hanford, Wash., December 1971.

5. E. F. Gloyna, A. A. Yousef, and T. J. Padden, "Radionuclide Transport and Responses to Organic Stress in a Research Flume," Advances in Water Pollution Research, vol. 1, no. 1, Pergamon Press, Great Britain, 1971.

6. Geological Survey Professional Paper 433-F, "Role of Certain Stream Sediment Components in Radioion Sorption," E. A. Jenne and J. S. Wahlberg, U.S. Government Printing Office, 1968.

7. Oak Ridge National Laboratories, ORNL-NSF-EATC-7, "A User's Manual for the FORTRAN IV Version of the Wisconsin Hydrologic Transport Model," M. R. Patterson et al., Oak Ridge, Tenn., October 1974.

8. R. J. Raridon et al., "Computer Model for Chemical Exchange in the Stream System," Proceedings of the First Annual NSF Trace Contaminant Conference, Conf-730802, 1973.

9. Geological Survey Professional Paper 433-A, "Uptake and Transport of Radionuclides by Stream Sediments," W. W. Sayre, H. P. Guy, and A. R. Chamberlain, U. S. Government Printing Office, 1963.

10. Center for Research in Water Resources, Technical Report 19, "Radioactivity Transport in Water, Summary Report," Y. A. Yousef, A. Kudo, and E. F. Gloyna, University of Texas, Austin, Texas, 1970.

Selected Field Studies, Surveillance Programs, and Data

1. Oak Ridge National Laboratories, ORNL-3721, TID4500, "Concentrations, Total Stream Loads, and Mass Transport in the Clinch and Tennessee Rivers," M. A. Churchill et al., (and others in this series), Oak Ridge National Laboratories, August 1965.

2. C. Collison and N. F. Shimp, "Trace Elements in Bottom Sediments from Upper Peoria Lake, Middle Illinois River," Environmental Geology Notes, Illinois State Geological Survey, No. 56, September 1972.

3. S. Davies, F. Cosolito, and M. Eisenbud, "Radioactivity in the Hudson River," Proceedings First Symposium on Hudson River Ecology, Hudson River Valley Commission of New York, 1966, p. 289.

4. National Academy of Sciences, "Marine Sediments and Radioactivity," Chapter Six of "Radioactivity in the Marine Environment," E. K. Duursma and M. G. Gross, National Academy of Sciences, Washington, D.C., 1971, pp. 147-160.

5. U. S. Environmental Protection Agency, RD No. 71-1, "Radiological Surveillance Studies at a Pressurized Water Nuclear Power Reactor," B. Kahn et al., National Environmental Research Center, Cincinnati, Ohio, August 1971.

6. A. Lerman, "Time to Chemical Steady-States in Lakes and Oceans," Nonequilibrium Systems in Natural Water Chemistry: Symposium of the American Chemical Society, Houston, Texas, February 24-25, 1970, Advances in Chemistry Series 106, American Chemical Society, Washington, D.C., 1971.

7. A. Lerman and G. J. Brunskill, "Migration of Major Constituents from Lake Sediments into Lake Water and Its Bearing on Lake Water Composition," Limnology and Oceanography, vol. 16, no. 6, November 1971, pp. 880-890.

8. A. Lerman and R. R. Weiler, "Diffusion and Accumulation of Chloride and Sodium in Lake Ontario Sediments," Earth and Planetary Science Letter, vol. 10, 1970, pp. 150-156.

9. T. F. Lomenick and T. Tamura, "Naturally Occurring Fixation of Cesium-137 on Sediments of Lacustrine Origin," Soil Science Society of America Proceedings, vol. 29, no. 4, July-August 1965, pp. 383-387.

10. F. T. Manheim, "The Diffusion of Ions in Unconsolidated Sediments," Earth and Planetary Science Letter, vol. 9, 1970, pp. 307-309.

11. D. K. Phelps, "Partitioning of the Stable Elements Fe, Zn, Se, and Sm Within a Benthic Community, Anasco Bay, Puerto Rico," Proceedings of the Symposium on Radioecological Concentration Processes, April 22-29, 1966, Stockholm, ed. by B. Aberg and F. P. Hungate, Pergamon Press, pp. 721-733.

12. G. K. Riel, "Radioactive Cesium in Estuaries," Radiological Health Data and Reports, December 1970, pp. 659-665.

13. N. I. Sax et al, "Radioecological Surveillance of the Waterways Around a Nuclear Fuels Reprocessing Plant," Radiological Health Data and Reports, vol. 10, no. 7, July 1969, pp. 289-296.

14. U. S. Environmental Protection Agency, Radiological Health Data and Reports, (published monthly 1959-74), Washington, D.C.

15. U. S. Atomic Energy Reports, BNWL-1341, and 1341-Appended. "Evaluation of Radiological Conditions in the Vicinity of Hanford for 1968," C. B. Wilson (ed.), Pacific Northwest Labs, Richland, Wash. (and others in this series back to 1961), 1970.

16. G. K. Riel, "Distribution of Fallout of Cesium 137 in the Chesapeake Bay," Second International Symposium on Natural Radiation Environment, Houston, Texas, August 1972.

Selected Sediment Transport Methodology

Sedimentation Committee, Hydraulics Division

1. American Society of Civil Engineers, "Sediment Discharge Formulas," Journal of the Hydraulics Division, ASCE, vol. 97, no. HY4, April 1971.

2. U. S. Army Coastal Engineering Research Center, "Shore Protection Manual," 1973.

3. U. S. Geological Survey, "Quality of Surface Waters of the United States," U.S. Government Printing Office, all years.

Selected Symposia

1. Proceedings, Symposium on Disposal of Radioactive Wastes into Seas, Oceans, and Surface Waters, International Atomic Energy Agency, Vienna, Austria, 1966.

2. Proceedings, Second National Symposium on Radio Ecology, Ann Arbor, Mich., 1967.

3. Proceedings, Symposium on Environmental Surveillance in the Vicinity of Nuclear Facilities, W. C. Reinig (ed), Charles Thomas Corp., Springfield, Ill., 1968.

4. Proceedings, Fifth Annual Health Physics Society Midyear Topical Symposium, "Health Physics Aspects of Nuclear Facility Siting," Idaho Falls, Idaho, 1970.

5. Proceedings, Third National Symposium, Radioecology, Oak Ridge, Tennessee, May 10-12, 1971.

6. Proceedings, International Symposium, Radioecology Applied to the Protection of Man and the Environment, Rome, Sept. 1971, EUR 4800-d-f-i-e, Luxembourg, May 1972.

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REGULATORY GUIDE 1.111

METHODS FOR ESTIMATING ATMOSPHERIC TRANSPORT AND DISPERSION OF GASEOUS EFFLUENTS IN ROUTINE RELEASES FROM LIGHT-WATER-COOLED REACTORS

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A. INTRODUCTION

Section 20.106, "Radioactivity in Effluents to Unrestricted Areas," of 10 CFR Part 20, "Standards for Protection Against Radiation," establishes limits on concentrations of radioactive material in effluents to unrestricted areas. Paragraph 20.1(c) of 10 CFR Part 20 states that licensees should, in addition to complying with the limits set forth in that part, make every reasonable effort to maintain radiation exposures, and releases of radioactive materials in effluents to unrestricted areas, as far below the limits specified in that part as is reasonably achievable.

Section 50.34a, "Design Objectives for Equipment to Control Releases of Radioactive Material in Effluents - Nuclear Power Reactors," of 10 CFR Part 50, "Licensing of Production and Utilization Facilities," sets forth design objectives for equipment to control releases of radioactive material in effluents from nuclear power reactors. Section 50.36a, "Technical Specifications on Effluents from Nuclear Power Reactors," of 10 CFR Part 50 further provides that, in order to keep power reactor effluent releases as low as is reasonably achievable, each license authorizing operation of such a facility will include technical specifications that require establishment of operating procedures for effluent control, installation and maintenance of effluent control equipment, and reporting of actual releases.

Appendix I, "Numerical Guides for Design Objectives and Limiting Conditions for Operation to Meet the Criterion 'As Low As Is Reasonably Achievable' for Radioactive Material in Light-Water-Cooled Nuclear Power Reactor Effluents," to 10 CFR Part 50 provides numerical guidance for those design objectives and limiting conditions for operation for light-water-cooled nuclear power plants. To implement Appendix I, the NRC staff has developed a series of guides providing acceptable methods for the calculation of effluent releases, dispersion of the effluent in the atmosphere and water bodies, and associated radiation doses to man. This guide describes basic features of calculational models and assumptions for the estimation of atmospheric transport and dispersion of gaseous effluents in routine releases from land-based light-water-cooled reactors.

The procedures and models provided in this guide will be subject to continuing review by the staff with the aim of providing greater flexibility to the applicant in meeting the requirements of Appendix I. As a result of such review, it is expected that alternative acceptable methods for calculation will be made available to applicants and that calculational procedures found to be unnecessary will be eliminated.

This guide supersedes portions of Regulatory Guide 1.42, Revision 1, "Interim Licensing Policy on As Low As Practicable for Gaseous Radioiodine Releases from Light-Water-Cooled Nuclear Power Reactors," which has been withdrawn (see 41 FR 11891, 3/22/76). *

B. DISCUSSION

The transport and dilution of radioactive materials in the form of aerosols, vapors, or gases released into the atmosphere from a nuclear power station are a function of the state of the atmosphere along the plume path, the topography of the region, and the characteristics of the effluents themselves. For a routine airborne release, the concentration of radioactive material in the surrounding region depends on the amount of effluent released; the height of the release; the momentum and buoyancy of the emitted plume; the windspeed, atmospheric stability, and airflow patterns of the site; and various effluent removal mechanisms. Geographic features such as hills, valleys, and large bodies of water greatly influence dispersion and airflow patterns. Surface roughness, including vegetative cover, affects the degree of turbulent mixing. Sites with similar topographical and climatological features can have similar dispersion and airflow patterns, but detailed dispersion patterns are usually unique for each site.

Most gaseous effluents are released from nuclear power plants through tall stacks or vents near the tops of buildings. Certain plant designs can result in other release pathways. For example, auxiliary equipment and major components such as turbines may be housed outside buildings; releases from these components could occur near ground level.

1. Diffusion Models

Atmospheric diffusion modeling has developed along two basic approaches: gradient-transport theory and statistical theory. Gradient-transport theory holds that diffusion at a fixed point

*Lines indicate substantive changes from previous issue.

the atmosphere is proportional to the local concentration gradient; this theory attempts to determine momentum or material fluxes at fixed points. The statistical (e.g., Gaussian) approach attempts to determine the histories of individual particles and the statistical properties necessary to represent diffusion. Input data for models based on either approach include windspeed, atmospheric stability, and airflow patterns in the region of interest. Several basic models have been developed using these approaches. These models vary according to their treatment of the spatial changes of input data and the consideration of either a variable trajectory model or a constant mean wind direction model.

a. Variable Trajectory Models

Variable trajectory models allow conditions to vary spatially and temporally over the region of interest; thus, they require regional data. The number of sampling locations needed to approximate the regional airflow depends on the meteorological and topographical characteristics of that region.

The particle-in-cell model is a variable trajectory model based on the gradient-transport approach. In this model, "particles" representing the effluent mass are released in groups over the time period of interest. The particles move at the effective transport velocity of the windflow field into which the effluent is released. The effective velocity is determined by the mean and turbulent windflows within the field. The number of particles located at any given time in each cell (volume) of a fixed coordinate grid determines the effluent concentration. Concentration averages are determined from the total number of particles that pass through a cell during the time of interest.

The plume element models, another class of variable trajectory models, are based on the statistical approach to diffusion. These models approximate a continuous release by dividing a plume into a sufficient number of plume elements to represent a continuous plume These elements are released at specified intervals and are tracked over the region of interest. The advective transport of these elements and the diffusion of the elements about their individual centers cause the dispersion of the plume effluent. Concentration averages are calculated by determining the contribution each element makes to the grid of points over which it passes.

b. Constant Mean Wind Direction Models

Constant mean wind direction models assume that a constant mean wind transports and diffuses effluents, within the entire region of interest, in the direction of airflow at the release point. A commonly used version of this model is the Gaussian straight-line trajectory model. In this model, the windspeed and atmospheric stability at the release point are assumed to determine the atmospheric dispersion characteristics in the direction of the mean wind at all distances.

These basic models can be modified to account for various modes of effluent release and for effluent removal mechanisms.

2. Release Mode

At ground-level locations beyond several miles from the plant, the annual average concentrations of effluents are essentially independent of the release mode; however, for ground-level concentrations within a few miles, the release mode is very important.

For a typical nuclear power plant, gaseous effluents released from tall stacks generally produce peak ground-level air concentrations near or beyond the site boundary; near-ground-level releases usually produce concentrations that monotonically decrease from the release point to all locations downwind. Under certain conditions, the effluent plume may become entrained in the aerodynamic wake of the building and mix rapidly down to ground level; under other conditions, the full effect of the elevation of the release may be realized.

Methods have been developed to estimate the effective release height for calculations of effluent concentrations at all downwind locations. The important parameters in these methods include the initial release height, the location of the release point in relation to obstructions, the size and shape of the release point, the initial vertical velocity of the effluent, the heat content of the effluent, ambient windspeed and temperature, and atmospheric stability.

For those effluents that are entrained into the aerodynamic wake of a building, mixing of the effluent into the wake is usually assumed. This mixing zone can constitute a plume with an initial cross section of one-half or more of the cross-sectional area of the building.

3. Removal Mechanisms

As the effluent travels from its release point, several mechanisms can work to reduce its concentration beyond that achieved by diffusion alone. Such removal mechanisms include radioactive decay and dry and wet deposition.

Radioactive decay is dependent on the half-life and the travel time of the radioactive effluent. All effluents can undergo dry deposition by sorption onto the ground surface; however, the dry deposition rate for noble gases, tritium, carbon-14, and nonelemental radioiodines is so slow that depletion is negligible within 50 miles of the release point. Elemental radioiodines and other particulates are much more readily deposited. The transfer of elemental radioiodines and particulates to a surface can be quantified as a transfer velocity (where concentration x transfer velocity = deposition rate). There is evidence that the transfer velocity is directly proportional to windspeed and, as a consequence, the rate of deposition is independent of windspeed since concentration in air is inversely proportional to windspeed.

Dry deposition is a continuous process while wet deposition only occurs during periods of precipitation. However, the dry removal process is not as efficient as the wet removal process. At most sites, precipitation occurs during a small percentage of the hours in a year so that, despite the greater efficiency of the wet removal process, dose calculations for long-term averages considering only dry deposition should not be significantly changed by the consideration of wet deposition. However, wet deposition can be a significant factor in dose calculations for releases from stacks at sites where a well-defined rainy season corresponds to the local grazing season.

Deposition of radionuclides over large bodies of water is not considered in this guide. Such deposition will be analyzed on a case-by-case basis.

C. REGULATORY POSITION

This section identifies types of atmospheric transport and diffusion models, source configuration and removal mechanism modifications, and input data that are acceptable to the NRC staff for use in providing assessments of potential annual radiation doses to the public resulting from routine releases of radioactive materials in gaseous effluents.

The listing of the atmospheric transport and diffusion models below is presented in order of decreasing model complexity and should not be construed as indicating the preference of any one type of model over another. The preferred model is that which best simulates atmospheric transport and diffusion in the region of interest from source to the receptor location, considering the meteorological characteristics of the region, the topography, the characteristics of the effluent source and the effluent as well as the receptor, the availability and representativeness of input data, the distance from source to receptor, and the ease of application.

Models proposed by the applicant and accepted by the NRC staff will be used by the staff in determining environmental technical specifications.

1. Atmospheric Transport and Diffusion Models

The following types of atmospheric transport and diffusion models can be modified for elevated sources and for effective area sources created when effluent is trapped in the building wake cavity in accordance with the source configuration considerations presented in regulatory position 2. Plume rise due to momentum or buoyancy effects can also be incorporated into the calculations. Radiological decay and dry and wet deposition, consistent with the guidelines presented in regulatory position 3, should also be considered.

a. Particle-in-Cell (PIC) Model

The basic equation for each "particle" group in this variable trajectory model, modified from Sklarew (Ref. 1), is:

$$\delta(\overline{\chi})/\delta t + \nabla \cdot V(\overline{\chi}) = 0 \qquad (1)$$

where

t is the travel time;

V is the velocity vector for effective mean wind transport, which includes the mean flow component, $\overline{V}$, and the turbulent flow component, V', such that $V = \overline{V}+V'$; and

$(\bar{x})$ is the average atmospheric concentration produced by a group of particles.

Concentration averages for long time intervals are obtained by summing all "particles" passing through each grid cell during the period of interest.

The PIC model uses spatial and temporal variations of wind direction, windspeed, atmospheric stability, and topography as input parameters to define airflow and atmospheric diffusion rates. The representativeness of the input data determines the accuracy of estimates (i.e., fewer data acquisition locations tend to increase the uncertainty of the estimates); therefore, detailed discussion of the applicability and accuracy of the model and input data used should be provided.

b. Plume Element Models

In these types of models, the transport and dispersion of an effluent plume are determined by using a horizontal wind field that can vary in time and space. The diffusion of individual plume elements, according to Gifford (Ref. 2), can be determined from the general Gaussian diffusion model. Commonly used plume segment elements are vertical "disk" segments and three-dimensional "puffs." In using the "puff" version, if it is assumed that the plume spread within a puff along the direction of flow is equal to the spread in the lateral direction, the "disk segment" and "puff" versions of this model would be expected to yield similar results.

An equation for a "puff" version of a fluctuating plume model, as presented by Start and Wendel (Ref. 3), is:

$$\chi/Q = 2[(2\pi)^{3/2}\sigma_H^2\sigma_z]^{-1}\exp[-1/2(r^2/\sigma_H^2 + h_e^2/\sigma_z^2)] \qquad (2)$$

where

$$r^2 = (x - \bar{u}t)^2 + y^2 \text{ and}$$

$$\sigma_H = \sigma_y = \sigma_x$$

and where

h_e is the effective release height;

Q is the effluent emission over the time interval;

t is the travel time;

$\bar{u}$ is the mean windspeed at the height of the effective release point;

x is the distance from center of puff along the direction of flow;

y is the distance from center of puff in the crossflow direction;

σ_x is the plume spread along the direction of flow;

σ_y is the lateral plume spread;

σ_z is the vertical plume spread; and

χ is the atmospheric concentration of effluent in a puff at ground level and at distance x from the puff center.

Concentration averages for long time intervals should be calculated by summing the concentrations of individual elements for the grid of points over which they pass.

The number of elements and the plume spread parameters (σ_x, σ_y, and σ_z) should be selected such that the resulting concentration estimate is representative of the concentration from a continuous point source release. Elements should be followed in the computational scheme until they are beyond the region of interest or until their peak concentration falls below a specified value.

The plume segment model uses spatial and temporal variations of wind direction, windspeed, and atmospheric stability as input parameters to define the transport and diffusion rate of each element. The effectiveness of the meteorological input data in defining atmospheric transport and diffusion conditions is dependent on the representativeness of these data and the complexity of the topography in the site region; therefore, a detailed discussion of the applicability and accuracy of the model and input data used should be provided.

c. Constant Mean Wind Direction Models

The equation for this model, as presented by Sagendorf (Ref. 4), is:

$$(\overline{\chi/Q'})_D = 2.032 \sum_{ij} n_{ij}[NX\bar{u}_i\Sigma_{zj}(X)]^{-1} \exp[-h_e^2/2\sigma_{zj}^2(X)] \qquad (3)$$

where

h_e is the effective release height (see regulatory position 2);

n_{ij} is the length of time (hours of valid data) weather conditions are observed to be at a given wind direction, windspeed class, i, and atmospheric stability class, j;

N is the total hours of valid data;

$\bar{u}_i$ is the midpoint of windspeed class, i, at a height representative of release;

X is the distance downwind of the source;

$\sigma_{zj}(X)$ is the vertical plume spread without volumetric correction at distance, X, for stability class, j (see Figure 1);

$\Sigma_{zj}(X)$ is the vertical plume spread with a volumetric correction (see regulatory position 2.c) for a release within the building wake cavity, at a distance, X, for stability class, j; otherwise $\Sigma_{zj}(X) = \sigma_{zj}(X)$;

$(\overline{\chi/Q'})_D$ is the average effluent concentration, χ, normalized by source strength, Q', at distance, X, in a given downwind direction, D; and

2.032 is $(2/\pi)^{1/2}$ divided by the width in radians of a 22.5° sector.

Effects of spatial and temporal variations in airflow in the region of the site are not described by the constant mean wind direction model. Unlike the variable trajectory models, the constant mean wind direction model can only use meteorological data from a single station to represent diffusion conditions within the region of interest. For Appendix I considerations, the region of interest can extend to a distance of 50 miles from the site. Therefore, if the constant mean wind direction model is to be used, airflow characteristics in the vicinity of any site should be examined to determine the spatial and temporal variations of atmospheric transport and diffusion conditions and the applicability of single station meteorological data to represent:

(1) Conditions between the site and the nearest receptors (generally within 5 miles) and

(2) Conditions out to a distance of 50 miles from the site.

Examples of spatial and temporal variations of airflow to consider for three basic categories of topography are:

(1) At inland sites in open terrain, including gently rolling hills, with airflow dominated almost entirely by large-scale weather patterns, recirculation of airflow and directional biases during periods of prolonged atmospheric stagnation;

(2) At sites in pronounced river valleys, with airflow patterns largely dominated by terrain, restrictions to lateral and vertical spread of the effluent plume, and the diurnal distributions of downvalley and upvalley circulation, with particular attention to the period of flow reversal; and

(3) At sites along and near coasts of large bodies of water, with significant land-water boundary layer effects on airflow, sea (or lake) land breeze circulation (including

distance of penetration, vertical development, temporal variations of wind direction, and conditions during periods of flow reversal), variation of the mixing layer height with time and distance from the shore (e.g., fumigation and plume trapping), and the effects of shoreline bluffs and dunes.

Therefore, adjustments to Equation (3) may be necessary to prevent misrepresentation of actual atmospheric transport and diffusion characteristics that could result in substantial underestimates of actual exposure to an individual or population. Adjustments to Equation (3) should be based on data (e.g., comparison to other sites in the region) or studies that characterize airflow patterns in the region of the site out to a distance of 50 miles.

For all sites, a detailed discussion of the applicability and accuracy of the model and input data should be provided. Use of Equation (3) will be acceptable only if a well-documented and substantiated discussion of the effects of spatial and temporal variations in airflow in the region of the site out to a distance of 50 miles is provided.

2. Source Configuration Considerations

The actual height above ground of the gaseous effluent plume should be considered in making estimates of average effluent concentrations downwind from the release points. An acceptable method to determine the effective plume height is described below. In addition, for effluent plumes traversing irregular terrain under stable or neutral atmospheric conditions, the model described by Egan (Ref. 5) may be used. On the other hand, the model described by Burt (Ref. 6) may be used when stable atmospheric conditions exist.

Source configuration evaluations may consider the effluent release point(s) and adjacent or nearby solid structure(s) in conjunction with the individual direction sector (as described in regulatory position 4) in which the downwind receptor of interest is located.

a. Elevated Releases

For effluents exhausted from release points that are higher than twice the height of adjacent solid structures, the effective release height (h_e) is determined (Ref. 4) from:

$$h_e = h_s + h_{pr} - h_t - c \qquad (4)$$

where

c is the correction for low relative exit velocity (see below);

h_e is the effective release height;

h_{pr} is the rise of the plume above the release point, according to Sagendorf (Ref. 4), whose treatment is based on Briggs (Ref. 7);

h_s is the physical height of the release point (the elevation of the stack base should be assumed to be zero); and

h_t is the maximum terrain height (above the stack base) between the release point and the point for which the calculation is made (h_t must be greater than or equal to zero).

Note that the effective release height is a function of the distance between the release point and the location where the concentration is being calculated.

When the vertical exit velocity is less than 1.5 times the horizontal windspeed, a correction for downwash is subtracted from Equation (4), according to Gifford (Ref. 8):

$$c = 3(1.5 - W_o/\bar{u})d \qquad (5)$$

where

c is the downwash correction;

d is the inside diameter of the stack or other release point;

$\bar{u}$ is the mean windspeed at the height of release; and

W_o is the vertical exit velocity of the plume.

b. Releases Other Than Elevated

For effluents released from points less than the height of adjacent solid structures, a ground-level release should be assumed ($h_e = 0$).

For effluents released from vents or other points at the level of or above adjacent solid structures, but lower than elevated release points, the effluent plume should be considered as an elevated release whenever the vertical exit velocity of the plume, W_o, is at least five times the horizontal windspeed, $\bar{u}$, at the height of release; i.e., as modified from Johnson et al. (Ref. 9):

$$W_o/\bar{u} \geq 5.0 \tag{6}$$

In this case, the release should be evaluated as described in regulatory position 2.a.

If $W_o/\bar{u}$ is less than 1.0 or unknown, a ground-level release should be assumed ($h_e = 0$)

For cases where the ratio of plume exit velocity to horizontal windspeed is between one and five, a mixed release mode should be assumed, in which the plume is considered as an elevated release during a part of the time and as a ground-level release ($h_e = 0$) during the remainder of the time. An entrainment coefficient, E_t, modified from Reference 9, is determined for those cases in which $W_o/\bar{u}$ is between one and five:

$$E_t = 2.58 - 1.58(W_o/\bar{u}) \text{ for } 1 < W_o/\bar{u} \leq 1.5 \tag{7}$$

and

$$E_t = 0.3 - 0.06(W_o/\bar{u}) \text{ for } 1.5 < W_o/\bar{u} \leq 5.0 \tag{8}$$

The release should be considered to occur as an elevated release $100(1 - E_t)$ percent of the time and as a ground release $100E_t$ percent of the time. Each of these cases should then be evaluated separately and the concentration calculated according to the fraction of time each type of release occurs. Windspeeds representative of conditions at the actual release heights should be used for the times when the release is considered to be elevated. Windspeeds measured at the 10-meter level should be used for those times when the effluent plume is considered to be a ground release. If Equation (3) is used, the adjustment described in regulatory position 2.c may be made for the ground release portion of the calculation.

c. Building Wake Correction

For ground-level releases only ($h_e = 0$), an adjustment may be made in Equation (3) that takes into consideration initial mixing of the effluent plume within the building wake. This adjustment, according to Yanskey et al. (Ref. 10), should be in the form of:

$$\Sigma_{zj}(X) = (\sigma_{zj}^2(X) + 0.5D_z^2/\pi)^{1/2} \leq \sqrt{3}\sigma_{zj}(X) \tag{9}$$

where

D_z is the maximum adjacent building height either up- or downwind from the release point;

X is the distance from the release point to the receptor, measured from the lee edge of the complex of adjacent buildings;

$\sigma_{zj}(X)$ is the vertical standard deviation of the materials in the plume at distance, X, for atmospheric stability class, j; and

$\Sigma_{zj}(X)$ is the vertical standard deviation of plume material as above, with the correction for additional dispersion within the building wake cavity, restricted by the condition that

$\Sigma_{zj}(X) = \sqrt{3}\sigma_{zj}(X)$

when

$$(\sigma_{zj}^2(X) + 0.5D_z^2/\pi)^{1/2} > \sqrt{3}\sigma_{zj}(X).$$

3. Removal Mechanism Considerations

Radioactive decay and dry and wet deposition should be considered in radiological impact evaluations. Acceptable methods of considering these removal mechanisms are described below.

a. Radioactive Decay

For conservative estimates of radioactive decay, an overall half-life of 2.26 days is acceptable for short-lived noble gases and of 8 days for all iodines released to the atmosphere. Alternatively, the actual half-life of each radionuclide may be used. The decay time used should be the calculated time of travel between the source and receptor based on the airflow model used.

b. Dry Deposition

Dry deposition of elemental radioiodines and other particulates and attendant plume depletion should be considered for all releases.

Acceptable plume depletion correction factors and relative deposition rates are presented in Figures 2 through 9. These figures are based on measurements of deposition velocity as a function of windspeed as presented in Reference 11 and on a diffusion-deposition model as presented in Reference 12.

Figures 2 through 5 illustrate an acceptable method for considering plume depletion effects for all distances from the source and atmospheric stability classes for ground and elevated release modes. After a given concentration is calculated by using the models in regulatory position 1, the concentration should be corrected by multiplying by the fraction remaining in the plume, as determined from these figures.

Figures 6 through 9 show acceptable values of relative deposition rate ($meters^{-1}$) as a function of distance from the source and atmospheric stability for ground and elevated release modes. The relative deposition rate is the deposition rate per unit downwind distance (Ci/sec per meter) divided by the source strength (Ci/sec).

To obtain the relative deposition per unit area ($meters^{-2}$) at a given point in a given sector, the relative deposition rate must be (1) multiplied by the fraction of the release transported into the sector, determined according to the distribution of wind direction and (2) divided by an appropriate crosswind distance (meters), as discussed below.

Figures 6 through 9 are based on the assumption that the effluent concentration in a given sector is uniform across the sector at a given distance. Therefore, for the straight-line trajectory model, or for any model that assumes uniform concentration across the sector at a given distance, the relative deposition rate should be divided by the arc length of the sector at the point being considered. In addition, for the straight-line trajectory model, the relative deposition rate should be multiplied by the appropriate correction factor discussed in regulatory position 1.c.

For models where concentration at a given distance is not uniform across the sector, the relative deposition at a given point should be calculated as above, but then multiplied by the ratio of the maximum effluent concentration in the sector at the distance being considered to the average concentration across the sector at the same distance.

c. Wet Deposition

For long-term averages, dose calculations considering dry deposition only are not usually changed significantly by the consideration of wet deposition. However, the effects of wet deposition and attendant plume depletion should be considered for plants with predominantly elevated releases and at sites that have a well-defined rainy season corresponding to the grazing season. Consideration of wet deposition effects should include examination of total precipitation, number of hours of precipitation, rainfall rate distributions, and the precipitation wind rose. If the precipitation data indicate that wet deposition may be significant, washout rates and attendant plume depletion should be calculated in accordance with the relationships identified by Engelmann (Ref. 13).

d. Deposition Over Water

For dispersion over small bodies of water, deposition may be assumed to occur at the same rate as over land. For calculations involving radionuclide transport over large bodies of water, deposition should be considered on a case-by-case basis.

4. Meteorological Data for Models

Sufficient meteorological information should be obtained to characterize transport processes (i.e., airflow trajectory, diffusion conditions, deposition characteristics) out to a distance of 50 miles (approximately 80,000 meters) from the plant. The primary source of meteorological information should be the onsite meteorological program (see Regulatory Guide 1.23, Ref. 14). Other sources should include nearby National Weather Service (NWS) stations, other well-maintained meteorological facilities (e.g., other nuclear facilities, universities, or private meteorological programs), and satellite facilities.

Adequate characterization of transport processes within 50 miles of the plant may include examination of meteorological data from stations further than 50 miles when this information can provide additional clarification of the mesoscale transport processes. To augment the assessment of atmospheric transport to distances of 50 miles from the plant, the following regional meteorological data, based on periods of record specified in Regulatory Guide 4.2 (Ref. 15), from as many relevant stations as practicable should be used:

a. Windspeed

b. Wind direction

c. Atmospheric stability

d. Mixing height

e. Precipitation

For input to variable trajectory atmospheric transport models, measured hourly values of windspeed should be used. Calms* should be assigned a windspeed of one-half of the appropriate starting speed, as described in the footnote, for instruments conforming to the recommendations or intent of Regulatory Guide 1.23 (Ref. 14). Otherwise, a windspeed of 0.1 meter/second should be assigned to calms. Hourly wind directions should be classed into at least the 16 compass point sectors (i.e., 22.5-degree sectors, centered on true north, north-northeast, etc.) according to measured values averaged over the time interval.

For input to the constant mean wind direction model, windspeed data should be presented as (1) hourly measured values or (2) windspeed classes divided in accordance with the Beaufort wind scale or other suitable class division (e.g., a greater number of light windspeed classes should be used for sites with high frequencies of light winds). Wind directions should be divided into the 16 compass directions (22.5-degree sectors, centered on true north, north-northeast, etc.). If joint frequency distributions of wind direction and speed by atmospheric stability class, rather than hourly values, are used in this model, calms* should be assigned to wind directions in proportion to the directional distribution within an atmospheric stability class of the lowest noncalm windspeed class. If hourly data are used, calms should be assigned to the recorded wind direction averaged over the time interval. The windspeed to be assumed for calms is one-half of the starting speed of the vane or anemometer, whichever is higher, for instruments conforming to the recommendations or intent of Regulatory Guide 1.23. Otherwise, the windspeed to be assumed for calms is 0.1 meter/second.

Atmospheric stability should be determined by vertical temperature difference (ΔT) between the release point and the 10-meter level, or by other well-documented parameters that have been substantiated by diffusion data. Acceptable stability classes are given in Reference 14.

Appropriate time periods for meteorological data utilization should be based on constancy of the source term (rate of release) and potential availability of the receptor (e.g., man or cow). If emissions are continuous, annual data summaries should be used. If releases are intermittent, consideration should be given to frequency and duration of release. If emissions are

*Calms are defined as hourly average windspeeds below the starting speed of the vane or anemometer, whichever is higher.

infrequent and of short duration, atmospheric dispersion models and meteorological data applicable to the time of release should be considered. Use of annual average conditions for consideration of intermittent releases will be acceptable only if it is established that releases will be random in time. Otherwise the method of evaluation of intermittent releases should follow the methodology outlined in Section 2.3.4 of NUREG-75/087 (Ref. 16). This method uses an appropriate χ/Q probability level, as well as the annual average χ/Q, for the direction and point of interest being evaluated to provide the basis for adjustments reflecting more adverse diffusion conditions than indicated by the annual average. These adjustments are applied to the annual average χ/Q and $D'Q$ for the total number of hours associated with intermittent releases per year. Detailed information for this calculation is given by Sagendorf and Goll (Ref. 17). However, if intermittent releases are limited by technical specifications to periods when atmospheric conditions are more favorable than average for the site, annual average data and annual average dispersion models could be used. For calculations of doses through ingestion pathways, particularly through the cow-milk pathway, meteorological data for only the grazing or growing season should be used.

D. IMPLEMENTATION

The purpose of this section is to provide information to license applicants and licensees regarding the NRC staff's plans for implementing this regulatory guide.

This guide reflects current NRC staff practice. Therefore, except in those cases in which the license applicant or licensee proposes an acceptable alternative method, the method described herein for complying with specified portions of the Commission's regulations will continue to be used in the evaluation of submittals for operating license or construction permit applications until this guide is revised as a result of suggestions from the public or additional staff review.

REFERENCES

1. R. C. Sklarew et al., "A Particle-in-Cell Method for Numerical Solution of the Atmospheric Diffusion Equation and Applications to Air Pollution Problems," Final Report 3SR-844, Vol. 1, EPA Contract 68-02-0006, 1971.

2. F. A. Gifford, "Statistical Properties of a Fluctuating Plume Dispersion Model," in Advances in Geophysics, Vol. 6, F. N. Frankiel and P. A. Sheppard, Editors, Academic Press, Inc., New York, pp. 117-138, 1959.

3. G. E. Start and L. L. Wendell, "Regional Effluent Dispersion Calculations Considering Spatial and Temporal Meteorological Variations," NOAA Tech Memo ERL-ARL-44, 1974.

4. J. F. Sagendorf, "A Program for Evaluating Atmospheric Dispersion From a Nuclear Power Station," NOAA Tech Memo ERL-ARL-42, 1974.

5. B. A. Egan, "Turbulent Diffusion in Complex Terrain" in Lectures on Air Pollution and Environmental Impact Analyses - AMS Workshop on Meteorology and Environmental Assessment, Boston 1975, Dwayne Haugen, Workshop Coordinator, American Meteorological Society, Boston, MA, pp. 123-124, 1975.

6. E. W. Burt, "Description of Valley Model-Version C9M3D," U.S. Environmental Protection Agency Dispersion Program, available from the United States Environmental Protection Agency, Office of Air Quality Planning and Standards, Research Triangle Park, NC 27711, pp. 4-6.

7. G. A. Briggs, "Plume Rise," AEC Critical Review Series, TID-25075, 1969.

8. F. A. Gifford, "Atmospheric Transport and Dispersion Over Cities," Nuclear Safety, Vol. 13, pp. 391-402, Sept.-Oct. 1972.

9. W. B. Johnson et al., "Gas Tracer Study of Roof-Vent Effluent Diffusion at Millstone Nuclear Power Station," AIF/NESP-007b, Atomic Industrial Forum, Inc., 1975.

10. G. R. Yanskey et al., "Climatography of National Reactor Testing Station," Idaho Operations Office, USAEC, IDO-12048, 1966.

11. E. H. Markee, Jr., "A Parametric Study of Gaseous Plume Depletion by Ground Surface Adsorption," in Proceedings of USAEC Meteorological Information Meeting, C. A. Mawson, Editor, AECL-2787, pp. 602-613, 1967.

12. C. A. Pelletier and J. D. Zimbrick, "Kinetics of Environmental Radioiodine Transport Through the Milk-Food Chain," in Environmental Surveillance in the Vicinity of Nuclear Facilities, W. C. Reinig, Editor, Charles C. Thomas Publishers, Springfield, Ill., 1970.

13. R. J. Englemann, "The Calculation of Precipitation Scavenging," in Meteorology and Atomic Energy-1968, D. H. Slade, Editor, USAEC TID-24190, pp. 208-221, 1968.

14. Regulatory Guide 1.23 (Safety Guide 23), "Onsite Meteorological Programs," U. S. Nuclear Regulatory Commission, Washington, D.C.

15. Regulatory Guide 4.2, "Preparation of Environmental Reports for Nuclear Power Stations," U.S. Nuclear Regulatory Commission, Washington, D.C.

16. "Standard Review Plan for the Review of Safety Analysis Reports for Nuclear Power Plants," NUREG-75/087, September 1975, Office of Nuclear Reactor Regulation, U.S. Nuclear Regulatory Commission, Washington, D.C.

17. J. Sagendorf and J. Goll, "XOQDOQ-Program for the Meteorological Evaluation of Routine Effluent Releases at Nuclear Power Stations," Draft, U.S. Nuclear Regulatory Commission, Washington, D.C., 1976.

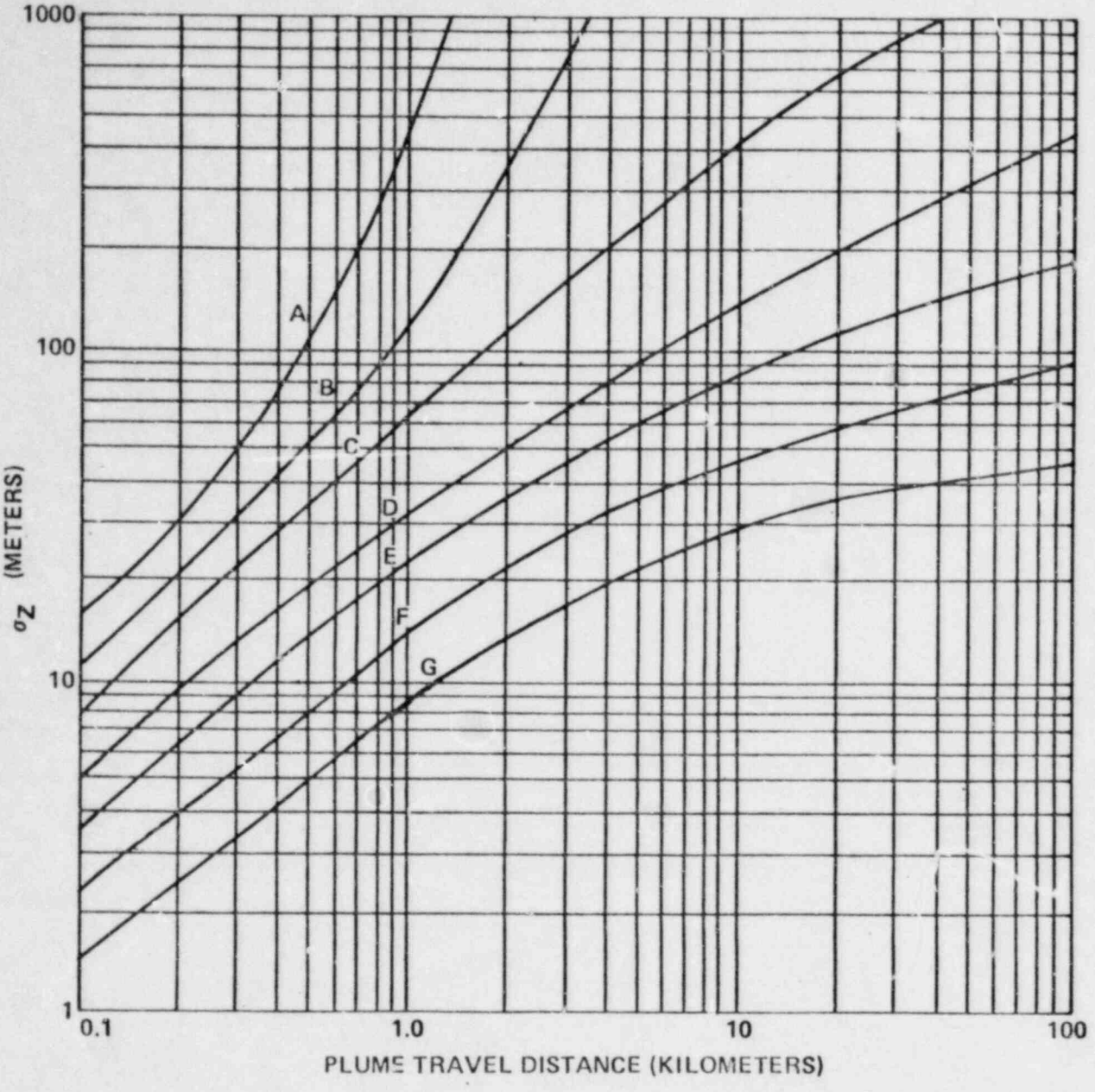

Figure 1. Vertical Standard Deviation of Material in a Plume (Letters denote Pasquill Stability Class)

NOTE: THESE ARE STANDARD RELATIONSHIPS AND MAY HAVE TO BE MODIFIED FOR CERTAIN TYPES OF TERRAIN AND/OR CLIMATIC CONDITIONS (E.G., VALLEY, DESERT, OVER WATER).

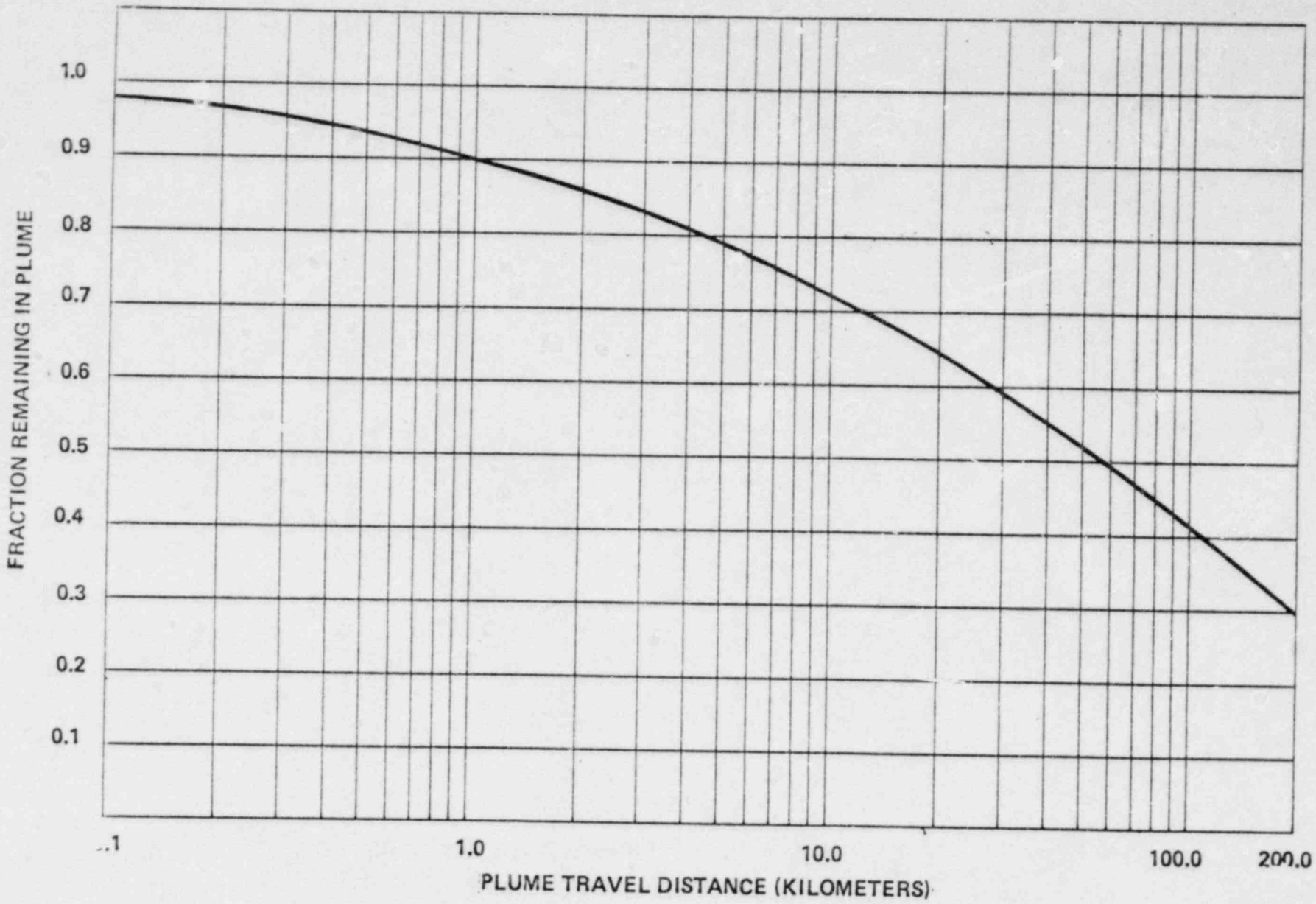

Figure 2. Plume Depletion Effect for Ground-Level Releases (All Atmospheric Stability Classes)

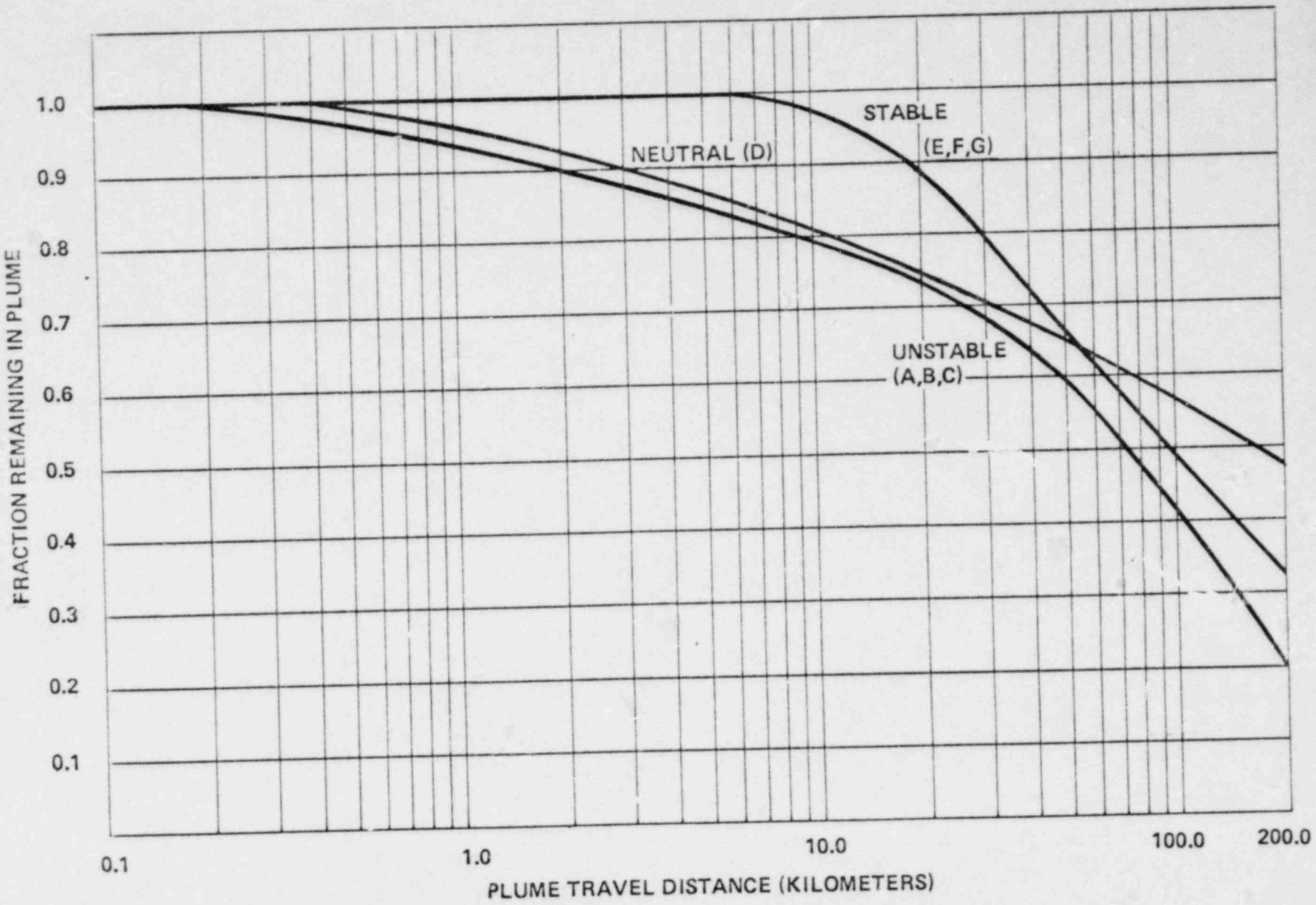

Figure 3. Plume Depletion Effect for 30-m Releases (Letters denote Pasquill Stability Class)

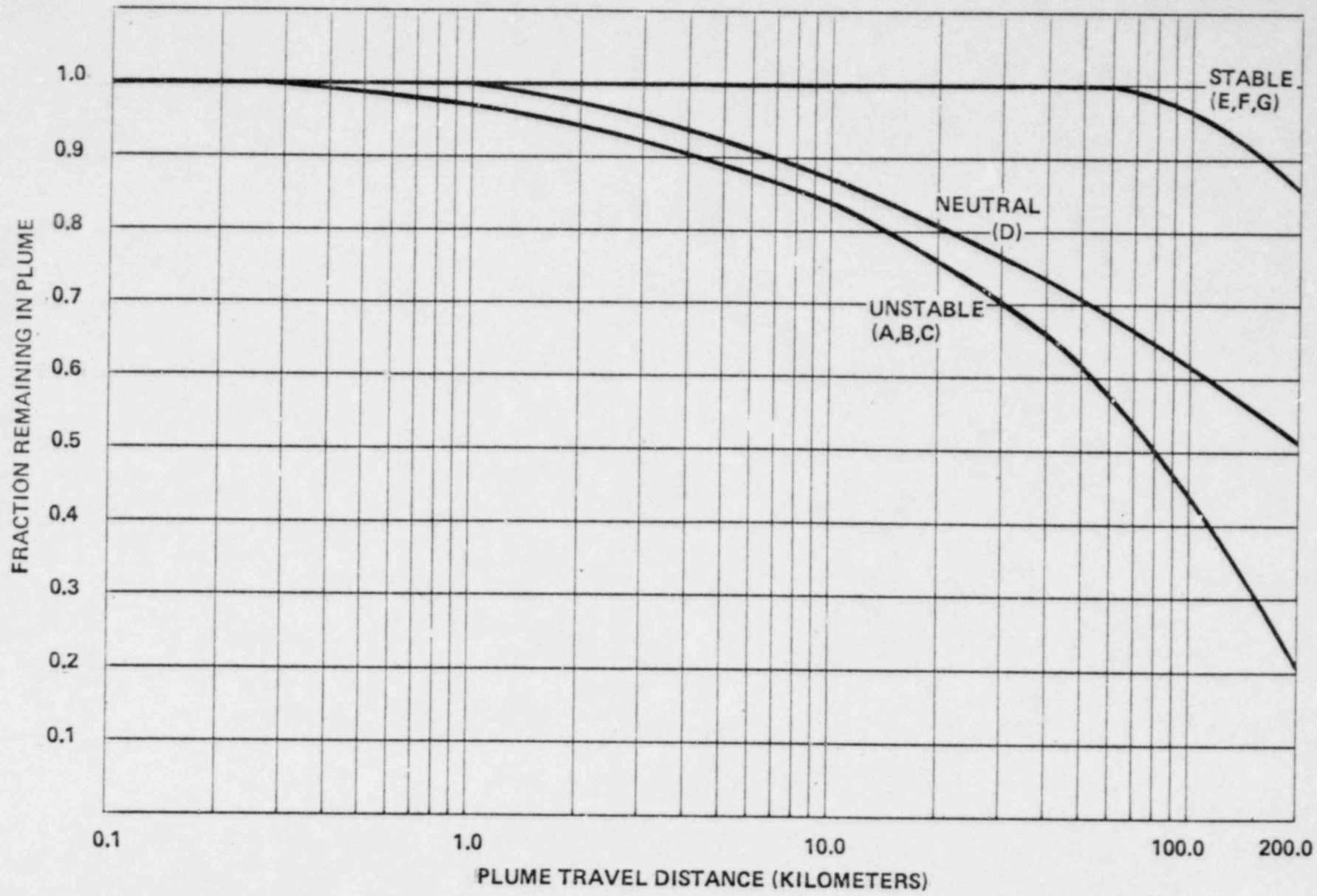

Figure 4. Plume Depletion Effect for 60-m Releases (Letters denote Pasquill Stability Class)

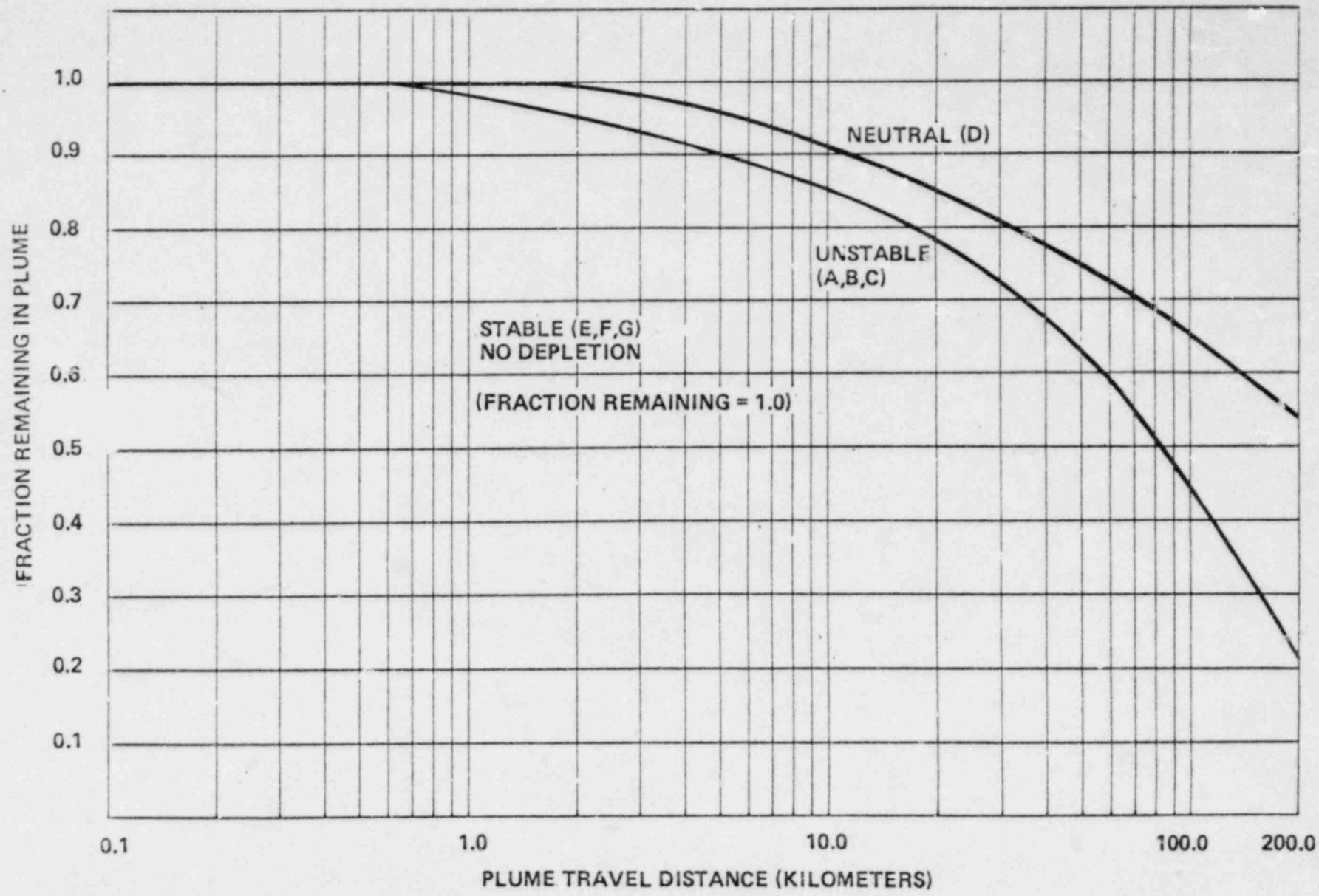

Figure 5. Plume Depletion Effect for 100-m Releases (Letters denote Pasquill Stability Class)

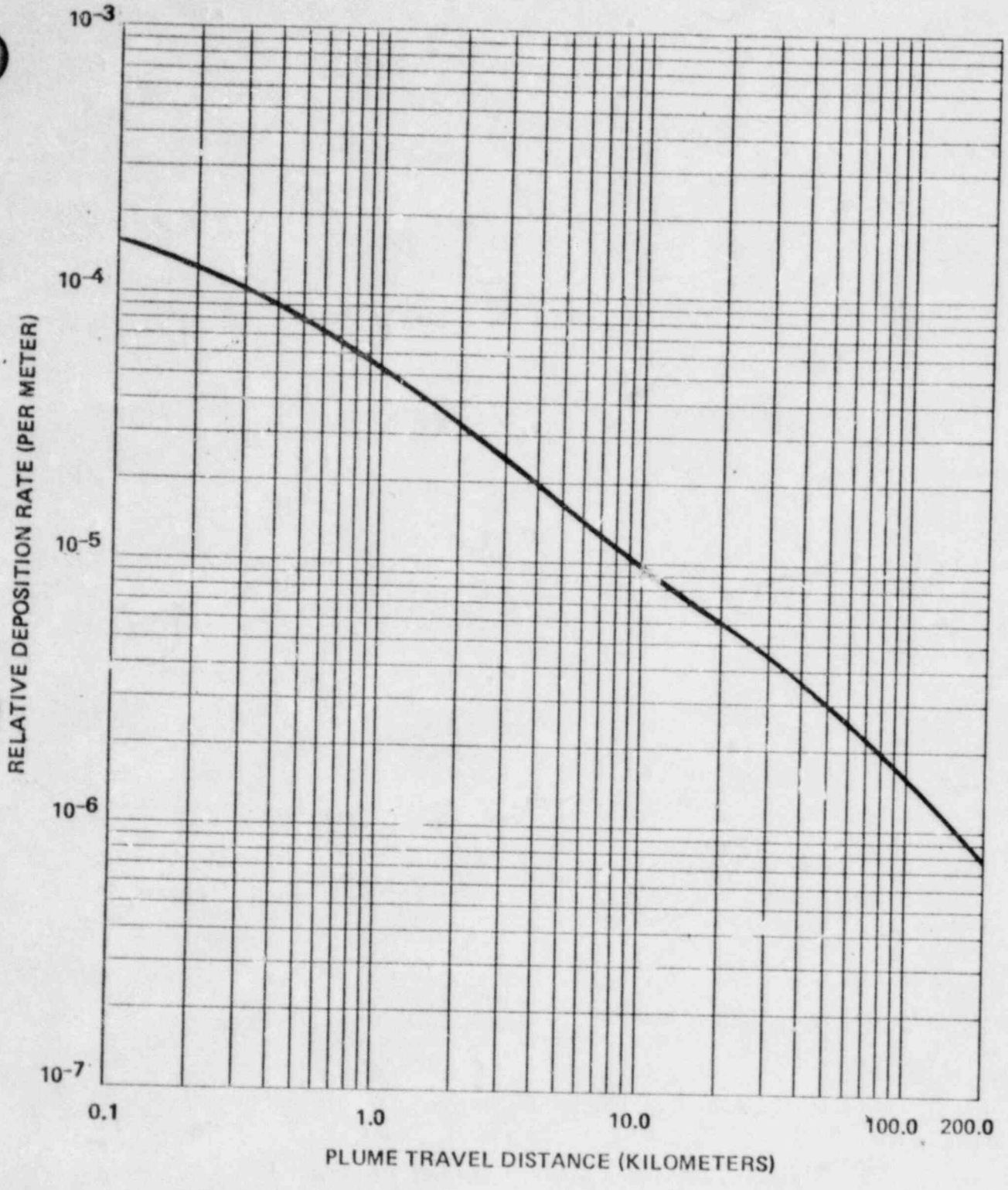

Figure 6. Relative Deposition for Ground-Level Releases (All Atmospheric Stability Classes)

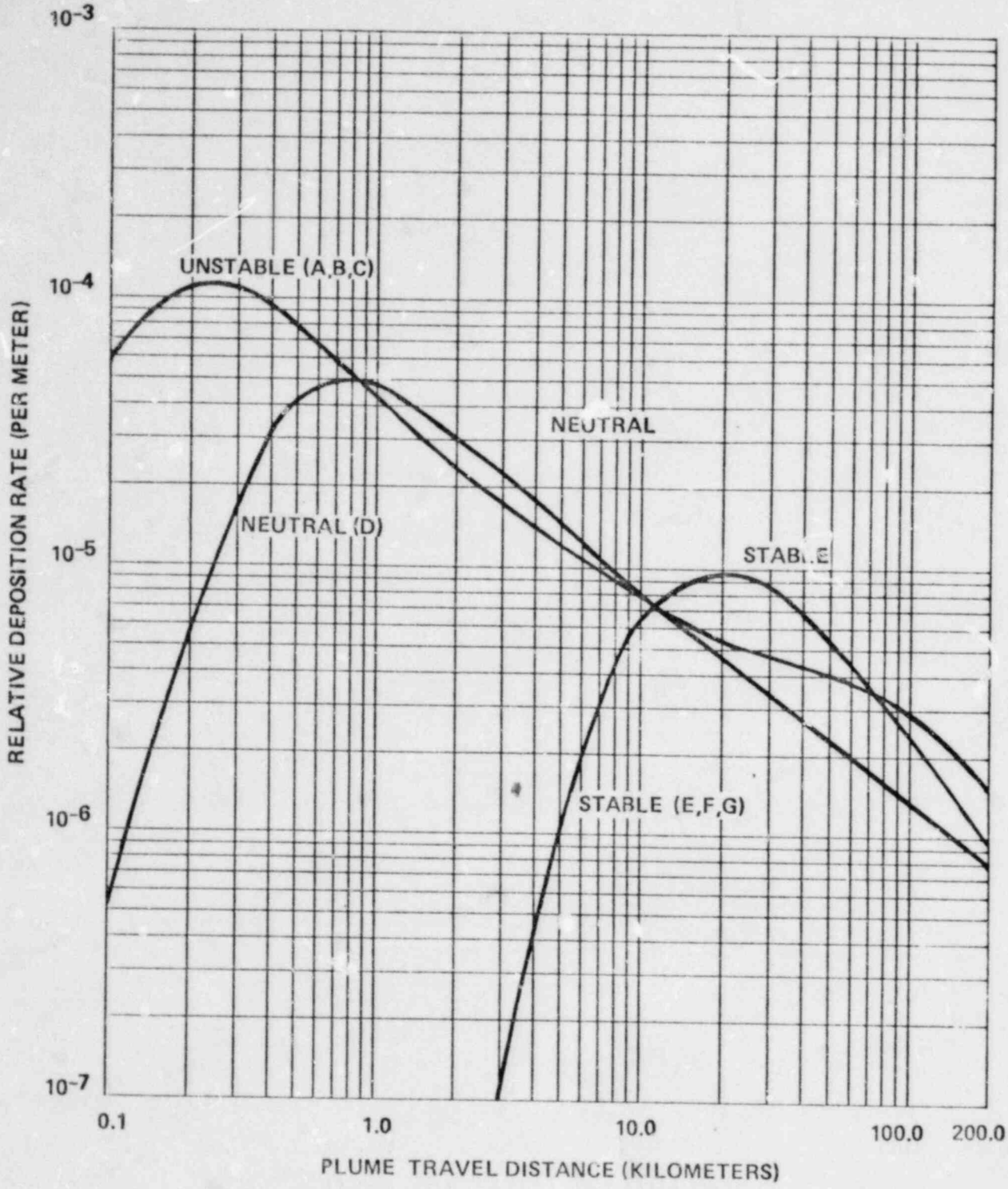

Figure 7. Relative Deposition for 30-m Releases (Letters denote Pasquill Stability Class)

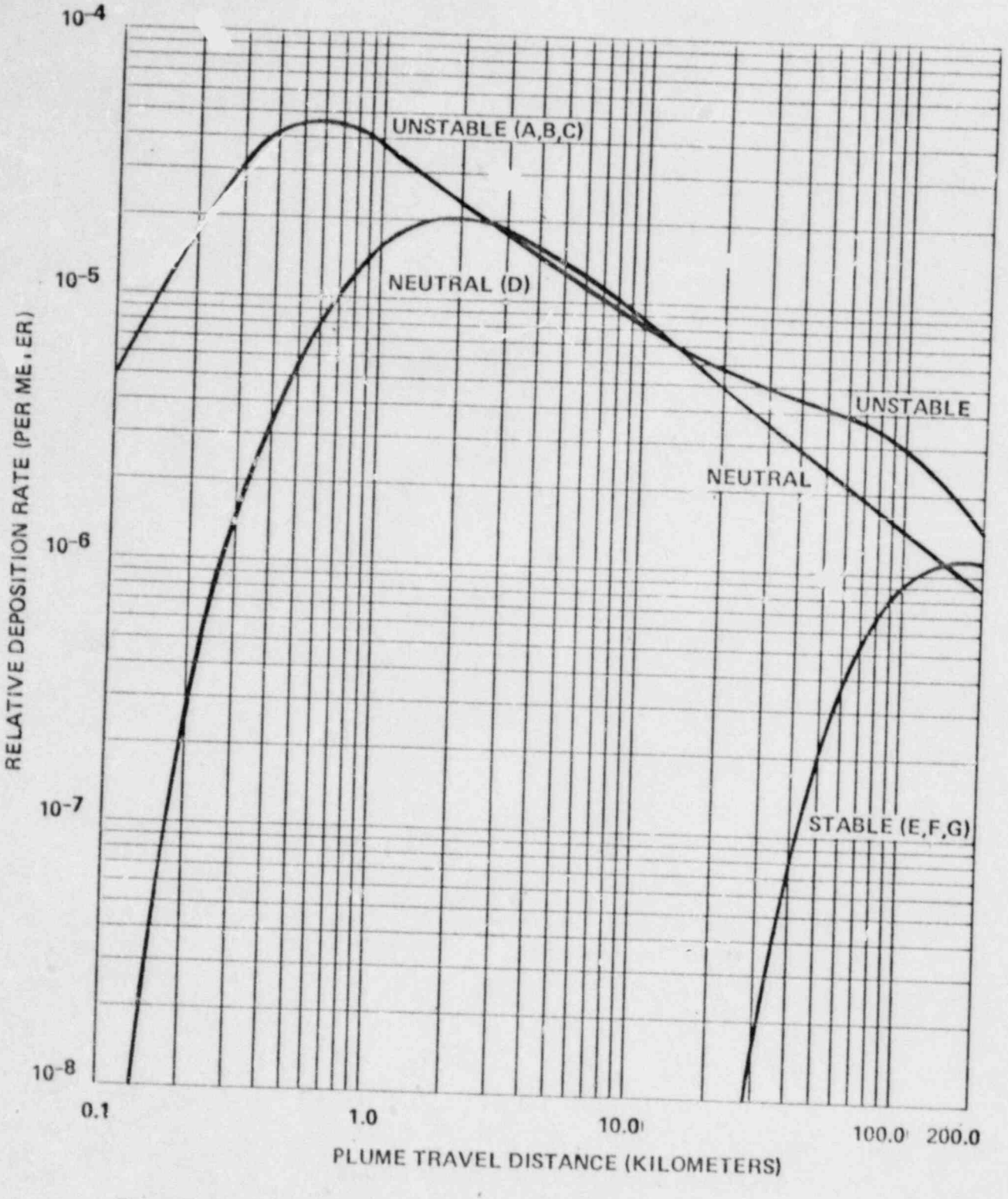

Figure 8. Relative Deposition for 60-m Releases (Letters denote Pasquill Stability Class)

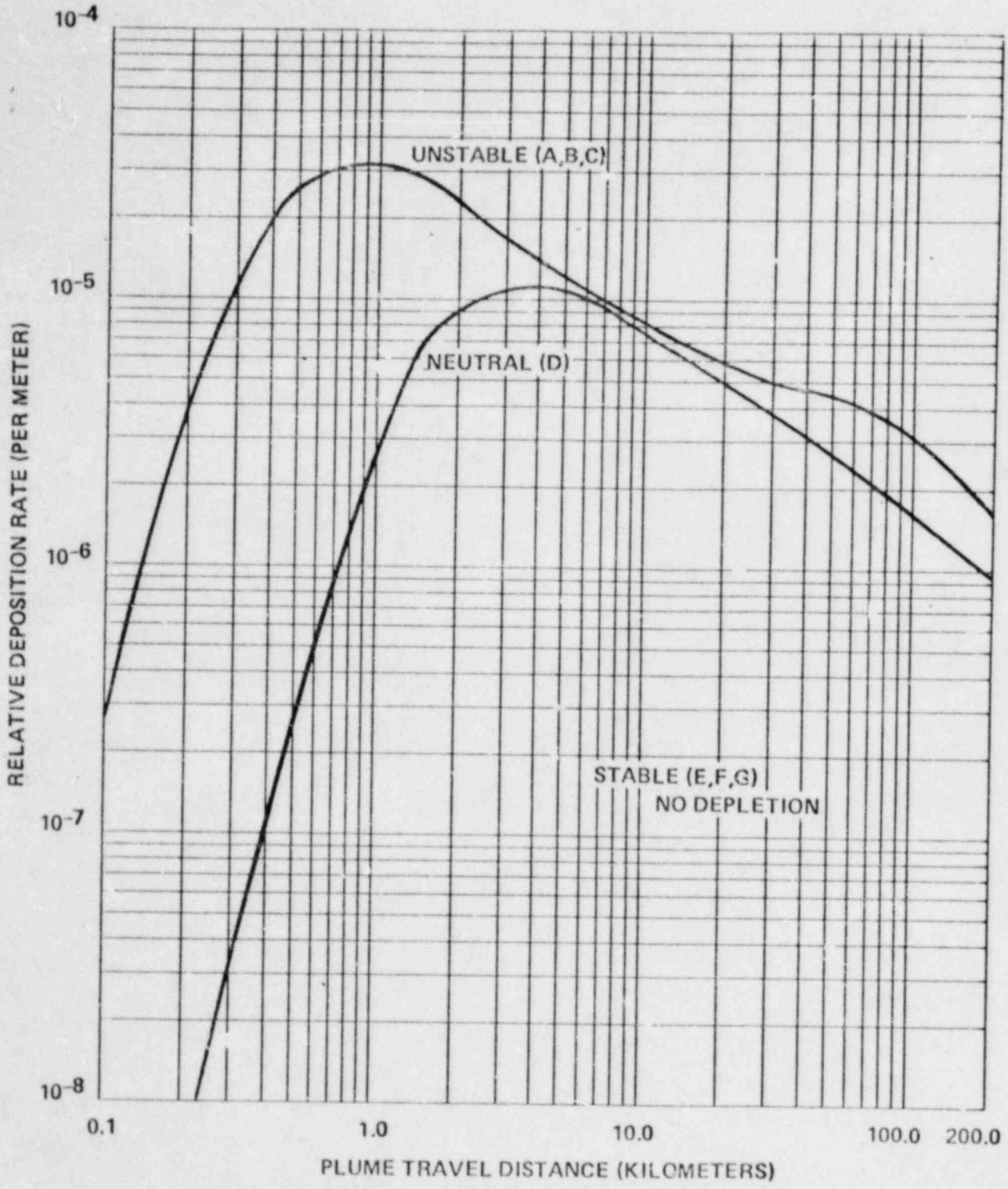

Figure 9. Relative Deposition for 100-m Releases (Letters denote Pasquill Stability Class)