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ABSTRACT

A theoretical evaluation has been made of the consequences of an environmental release of an airborne cloud resulting from a sodium fire in an LMFBR. The evaluation takes into consideration only the hazards of the radioisotopes of the sodium, itself. Problems associated with accompanying fission products were not included. On the basis of conservative assumptions, the data show that the most significant exposure would be to a child through the forage-cow-milk food chain pathway. The whole body dose for an instantaneous direct environmental release of 1 gram of sodium (containing 0.31 Ci of ^{24}Na and 1.8×10^{-4} Ci of ^{22}Na) would be about 0.2 rem; the corresponding dose resulting from a release of 1000 grams of sodium within the reactor containment vessel, followed by a subsequent containment vessel leakage rate of 0.1% per day, would be about 13 rems. Restrictions on the use of milk and milk products would considerably reduce these values. Other possible countermeasures which might be taken to cope with the environmental hazards are also discussed. Supplementing the report are appendices containing data on the properties and environmental behavior of sodium as well as information on computer models for evaluating the environmental transport of sodium and its compounds.

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ENVIRONMENTAL BEHAVIOR OF RADIOACTIVE SODIUM

(Transport, Uptake & Population Dose)

I. Introduction

The Liquid Metal Fast Breeder Reactor (LMFBR) is considered to be one of the types which will play a major role in the future development of large scale electric power stations in the United States. Although a number of investigators have studied the in-plant consequences of the leakage or spillage of sodium from the cooling system and subsequent fires, there is little information in the published literature on the consequences of a major release of sodium into the environment. In particular, there appears to be little information on the environmental behavior and transport of sodium, the major pathways by which it might gain access to man, and the resulting exposures which might be anticipated. It is the purpose of this paper to direct specific attention to the environmental consequences of a major sodium release and to provide estimates of its impact upon the environment and man.

It is assumed that the nature of the release will be in the form of an airborne cloud resulting from a sodium fire. In assessing its environmental consequences, the transport and behavior of the sodium is dealt with starting at the point where the cloud reaches the ground surface. Details of the atmospheric diffusion of the cloud or of the subsequent deposition of the sodium aerosol are not considered. Also not evaluated is the potential environmental impact of the fission

products or other radionuclides which may accompany the released sodium even though in many, if not most, cases it is quite probable that these radionuclides will initially be of greater importance than the radioisotopes of sodium.

Much information for the present study came from environmental Flowshare studies which dealt to some extent with the behavior of ^{22}Na and ^{24}Na , produced as activation products in nuclear detonations. Additional information is based on data from published accounts of the transport and behavior within the environment of stable sodium as well as other radionuclides whose behavior was considered to be representative of sodium. Much of this reference information, plus details of the models used in calculating the resulting population doses from a release of a known magnitude, are given in Appendices to this report.

II. The Reference Accident

For purposes of this study, it is assumed that an accident occurs and that sodium is released to the environment, either instantaneously, or over an extended period through leakage from the containment vessel. For convenience, the instantaneous release is referred to later in this report as "direct release"; the extended release, since it involves leakage through the containment vessel, is referred to as "indirect release". While it is recognized that the occurrence of the necessary sequence of events leading to such an accident is highly improbable, the assumption is made that they do occur. Factors tending to prevent such accidents are many. These include safeguards built into the reactor system (Beusman, 1959; Int. Conf., 1967; IAEA Symposium, 1967; Farmer, 1970), including high integrity components designed to contain the sodium, strong barriers provided to keep sodium and water within the system separated, and inert gas blankets provided to limit combustion even if sodium spillage does occur. In addition, it has been pointed out that any aerosol formed within the containment vessel would agglomerate naturally and plate out on exposed surfaces. Reist, 1969, has estimated that these mechanisms would reduce sodium concentrations in the vessel by a factor of 10^4 within the first day, and by a factor of 10^8 within the first week, following the termination of a sodium fire. Another factor tending to restrict the release is that the residual airborne sodium would be vented through an absolute filter system (First, 1972).

To provide a basis for the calculations in this report, a

"reference" accident is assumed. In selecting the basic data, values were taken which would yield calculations which would cover the worst possible conditions. For example, Yevick and Amorosi (1966) have reported a range of values for the specific activity for ^{22}Na and ^{24}Na in fast reactors. Values taken for the reference accident were the highest reported: 10^{-4} Ci/cm³/Mw(t) for ^{24}Na in the Fermi reactor; 5.8×10^{-8} Ci/cm³/Mw(t) for ^{22}Na in the Dounreay reactor. Other assumptions for the reference accident were based on Gaillanger (1971); Yevick and Amorosi (1966); Reist (1969); Fies (1971); and Hale and Shire (1971).

Details of the conditions for the assumed releases are given in Table II-1.

Table II-1

RELEASE CONDITIONS

Reactor Power: 3,000 Mw(t)

Source of Contamination: Sodium Fire

<u>Release from Containment</u>	<u>Duration of Release</u>	<u>Amount of Sodium Released</u>	<u>Activity Released</u>	<u>Release Height</u>	<u>Atmospheric Conditions</u>
Instantaneous	< 1 min.	1 gram	0.31 Ci, ^{24}Na 1.8×10^{-4} Ci, ^{22}Na	Ground Level or 100 meters	Stable (F)
Continuous	2 hours, 150 hours infinite time	1000 grams	310 Ci, ^{24}Na 0.18 Ci, ^{22}Na	Ground Level or 50 meters	Stable (F) for 2 hour release; Average* conditions for 150 hour or infinite time leakage

Assumptions: The sodium aerosol remains airborne and homogeneously dispersed within the containment during the entire duration of the leakage; the mass median diameter of the aerosol is about 1μ .

*Average refers to 30% stable and 70% unstable conditions.

III. Hazards Evaluation of an LMFBR Sodium Release

The radiation dose following a sodium release is attributable to both the sodium retained in the containment vessel and that which reaches the environment. Calculation of the radiation dose from the containment is straightforward. However, in order to evaluate the environmental hazards from a radionuclide release, several additional parameters must be determined. These parameters are:

- a. initial concentrations in air and water;
- b. dry deposition and washout; and
- c. intake by inhalation and ingestion.

A. Computations

1. External Radiation Dose from the Containment Building

For purposes of this assessment, the containment building is considered to be a cylindrical, shielded volume source within which the sodium aerosol is homogeneously dispersed following the incident. The following dimensions were taken for the building (Hale and Shire, 1971).

Volume, $5.5 \times 10^4 \text{ m}^3$

Radius, $2.5 \times 10^3 \text{ cm}$

Thickness of Steel Shell, 2.5 cm

Thickness of Construction Material (concrete), 6 cm

Etherington (1958) provides an approach through which the dose may be calculated. The dose rate D (r/sec) at a

point P which is z cm from the nearest point of the cylinder (see Figure III-1) is given by:

$$D = \frac{B k p a^2 \exp(-\mu_b z)}{4(z + t_c)} \left[F(\theta_1, \mu' t') + F(\theta_2, \mu' t') \right] \quad (3-1)$$

where B is a build-up factor in the shield (Moteff, 1954)

k is a conversion factor:

$$1.84 \times 10^6 \frac{r/hr}{\text{MeV/sec cm}^2}$$

p is the source strength in $\text{MeV cm}^{-3} \text{ sec}$

$\mu' t'$ is an effective attenuation and equals $\mu_c t_c + \mu t$, where μ_c and μ are attenuation coefficients, cm^{-1} , in the source and in the shield respectively, t_c is the effective depth of the source in the volume and t is the shield thickness, cm. Ways of obtaining t_c are given by Etherington (1958).

μ_b is the attenuation coefficient in air, cm^{-1}

$$F(\theta, \mu t) = \int_0^\theta \exp(-\mu t \sec \theta') d\theta'$$

and the meaning of the other quantities is clear from Figure III-1 (all dimensions are in cm).

The integrated radiation dose from each radionuclide for different periods of time can be calculated by using the equation:

$$D_r = \frac{D}{\lambda_t} [1 - \exp(-\lambda_t \tau)] \quad (3-2)$$

where: D_r = integrated radiation dose (r)

D = radiation dose rate (r/sec) (from eq. (3-1))

FIGURE III-1 SOURCE GEOMETRY FOR EXTERNAL RADIATION DOSE CALCULATION

λ_t = total decay constant (sec^{-1}) = $\lambda_l + \lambda_r$, where

λ_l = leakage decay constant (sec^{-1}) =

$1.16 \times 10^{-8} \text{sec}^{-1}$ (0.1% per day), λ_r =

radioactive decay constant (sec^{-1}).

τ = exposure time (sec)

The integrated radiation dose from the containment building obtained by summing the values for each radionuclide (from equation (3-2)) is a function of distance z and time τ . The results of these dose calculations are given in Section III.8.1.

2. Maximum Ground-Level Concentration

The maximum ground-level concentration and the distance from the source at which it occurs are of major importance in an evaluation of environmental hazards. The integrated maximum ground-level concentration of sodium and the distance from the source to the point of maximum concentration, were calculated by differentiation of the Gaussian plume equation (Gifford, 1960). For the case of continuous leakage from the containment building, a volume source correction was introduced (Slade, 1968, 112) and a virtual distance of 300 m. was derived, representing the distance from the source required to produce the volume source concentration at the origin (0,0,0). From curves representing the ground level air concentration of a pollutant as a function of distance (Slade, 1968), it is seen that concentrations not very different from the maximum

concentration, are found over a relatively extended region. Therefore, the maximum concentration was used over an extended region.

The downwind distance for maximum concentration for each release condition considered, is given in Table III-1.

3. Dry Deposition

Since a mass median diameter of about 1μ was assumed for the sodium aerosol (see Table II-1), no local fallout due to the deposition of large sodium particles is considered. Dry deposition here refers instead to the process by which small particles are removed at the atmosphere-ground interface, by such mechanisms as impaction, sorption, electrical force attraction, chemical reactions, etc. The deposition process considered is not controlled by the terminal settling velocity, but may be considered as a mass transfer and diffusion process. The deposition effect may be felt at relatively large distances (Tadmor, 1971).

The velocity of deposition, v_g , (m/sec) of sodium was calculated using the equation (Chamberlain, 1960):

$$v_g = \frac{ku^*}{\ln(ku^*z_1D^{-1})} \quad (3-3)$$

Where: k = von Karman's constant = 0.4

u^* = friction velocity (assumed to be 40cm/sec)

z_1 = reference height above ground surface at which the pollutant concentration is measured (assumed to be 100cm)

TABLE III-1

REGIONS OF MAXIMUM CONCENTRATION

<u>Mode and Height of Release</u>	<u>Duration of Release (hr)</u>	<u>Meteorological Conditions</u>	<u>Distance to the point of Maximum Concentration</u>
Point source; Ground level	Instantaneous	Stable (F)	500*
Volume source; Ground level	2	Stable (F)	300*
Volume source; Ground level	150 hours or infinite time	Average**	300*
Volume source; 50 m.	2	Stable (F)	5,000'
Volume source; 50 m.	150 hours or infinite time	Average**	500
Point source; 100 m.	Instantaneous	Stable (F)	10,000

*Although the maximum concentration for a ground-level release occurs at the source itself, it is considered, for the purpose of the present evaluation, that the maximum occurs at a distance of 300 m. for the volume source, and 500 m. for the point source.

**Average refers to 30% stable and 70% unstable conditions.

D = molecular diffusivity (cm^2/sec)

The molecular diffusivity of sodium in air at 20°C was calculated as $0.12 \text{ cm}^2/\text{sec}$ using the data given (First et.al. 1971) for the diffusion of sodium in nitrogen. From these data a velocity of deposition of 1.7 cm/sec for sodium in air was computed. The integrated deposition was then obtained as the product of this velocity of deposition and the concentration of sodium. As with the ground-level concentration of sodium in air, maximum deposition is considered as existing over a relatively extended region. This assumption has been verified for deposition of ^{137}Cs (Slade, 1968, 384). The centerline concentration is also assumed to represent the crosswind concentrations over an extended region. It may be shown (Slade, 1968, 385) that this assumption does not introduce a significant error in the computations.

4. Washout

The washout case which maximizes deposition at a particular site is a rainfall which starts at the site itself washing out most of the activity on that particular region. The following expression was used to calculate w , the centerline maximum washout deposition, Ci/m^2 , (Slade, 1968, 218):

$$w = \frac{Q}{e \times \sigma_y (2\pi)^{1/2}} \quad (3-4)$$

where: Q = source strength (Ci)

e is the base of natural logarithms

x = distance at which washout occurs (m)

σ_y = crosswind dispersion coefficient at distance x (m)

5. Inhalation and Direct Ingestion Intake

The inhalation intake was obtained by multiplying the integrated concentration in air ($Ci \text{ sec m}^{-3}$) by the average breathing rate of $20 \text{ m}^3/\text{day}$ (ICRP, 1969, 152) for the longterm release or the "at work" breathing rate of $1.25 \text{ m}^3/\text{hr}$ (ICRP, 1959, 152) for the direct and 2-hour release cases. The direct water ingestion intake was calculated by multiplying the integrated concentration in water (surface or ground) by the drinking water intake of $0.0022 \text{ m}^3/\text{day}$ (ICRP, 1959, 152). For purposes of the calculations, it was assumed that a person drinks contaminated water for one day only.

a. Surface of Deposition

The surface of deposition, assumed circular, was calculated according to Slade, 1968, 405:

$$y_p = (2\sigma_y^2 \ln \frac{100}{p})^{1/2} \quad (3-5)$$

where: y_p is the radius (m) of the surface of deposition, and p is the percentage of the axial concentration arbitrarily selected as defining the limit of the contaminated area (p was taken as 50%).

Table III-2 shows the deposition areas as calculated for the different release cases which were studied.

TABLE III-2
DEPOSITION AREAS FOR SODIUM

Mode and Height of Release	Duration of Release (hrs)	Meteorological Conditions	Area of Deposition (m^2)
Volume source, Ground level	2	Stable (F)	10^3
Point Source, Ground level	Instantaneous	Stable (F)	1.7×10^3
Volume Source, Ground level	150 hours or infinite time	Average*	4.8×10^3
Volume Source, 50m	150 hours or infinite time	Average*	8.2×10^3
Volume Source, 50m	2	Stable (F)	10^5
Point Source, 100m	Instantaneous	Stable (F)	4×10^5

*Average refers to 30% stable and 70% unstable conditions.

The area of deposition was multiplied by the concentration of sodium deposited (from III.A.4) to obtain the total amount of sodium deposited.

b. Dilution

Dilution of sodium in surface water was calculated by the one-dimensional method of Carrit et al. (1959) (see

equation B-1 in Appendix B). Values used in the equation were:

$D = 100 \text{ m}$; $K = 0.4 \text{ m}^2/\text{sec}$ (Dawson et al., 1970);
 $u = 3 \text{ m/sec}$ (Pratt, 1968,15); $x = 100 \text{ m}$. The source term Q was obtained from III.A.5a.

For dilution in ground water, the same source term was used. As a conservative assumption, no credit was taken for sodium retardation compared to water flow through the water table. An inflow rate of $10^4 \text{ m}^3/\text{day}$ (Thomas, 1971), ground water table depth of 50 m, and a leaching velocity of 100 m/yr were assumed. On the basis of these conditions the time required for the sodium to reach to ground water table would be 180 days and the dilution volume would be 180×10^4 , or 1.8×10^6 , m^3 .

B. Results

1. External Radiation Dose from the Containment Building

Figure III-2 shows the integrated radiation dose from the containment building, as a function of distance, following a release of 1000 grams of sodium (0.18 Ci of ^{22}Na and 310 Ci of ^{24}Na) into the containment vessel and subsequent leakage of 0.1% per day. The integrated radiation doses are shown for two hour, 150-hour, and infinite time exposures.

2. Environmental Radioactivity Parameters

The four types of release listed in Section II (Reference Accident) were considered. For each type of release, environmental loadings were calculated using the methods and assumptions of Section III.A. Table III-3 gives these data for

FIGURE III-2 INTEGRATED RADIATION FROM CONTAINMENT BUILDING AS FUNCTION OF DISTANCE FROM BUILDING

TABLE III-3

SODIUM IN THE ENVIRONMENT
FOLLOWING A RELEASE FROM AN LMFBR

Mode of Release: Point Source Directly to Environment
Duration of Leakage: Instantaneous
Height of Release: Ground Level
Meteorological Conditions: Stable (F)
Location of Maximum Concentration: 0.5 km from point of release
Amount of Release: 1 gram of sodium (1.8×10^{-4} Ci of ^{22}Na and
0.31 Ci of ^{24}Na)

Factor Considered	Na	Data for ^{22}Na	^{24}Na
Amount Released to the Environment	1 gram	1.8×10^{-4} Ci	3.1×10^{-1} Ci
Maximum Integrated Concentration in Air	10^{-4} g-sec/m ³	1.8×10^{-8} Ci-sec/m ³	3.1×10^{-5} Ci-sec/m ³
Maximum Dry Deposition	1.8×10^{-6} g/m ²	3.1×10^{-10} Ci/m ²	5.3×10^{-7} Ci/m ²
Maximum Washout	1.6×10^{-5} g/m ²	2.7×10^{-9} Ci/m ²	4×10^{-6} Ci/m ²
Inhalation Intake	2.6×10^{-8} g	4×10^{-12} Ci	7.2×10^{-9} Ci
Amount Deposited on Surface Water (following Maximum washout)	2.7×10^{-2} g	4.6×10^{-6} Ci	6.9×10^{-3} Ci
Maximum Integrated Concentration in Surface Water	4.8×10^{-6} g-sec/m ³	8.2×10^{-10} Ci-sec/m ³	1.2×10^{-6} Ci-sec/m ³
Concentration in Ground-Water (180 days after deposition)	1.5×10^{-8} g/m ³	2.6×10^{-12} Ci/m ³	Negligible due to Radioactive Decay
Ingestion Intake from Surface Water	1.3×10^{-13} g	2.1×10^{-17} Ci	3.1×10^{-14} Ci
Ingestion Intake from Ground Water	3.3×10^{-11} g	5.6×10^{-15} Ci	Negligible due to Radioactive Decay

TABLE III-3 (Continued)

Height of Release: 100m

Location of Maximum Concentration: 10km from point of release

Factor Considered	Data for		
	Na	^{22}Na	^{24}Na
Amount Released to the Environment	1 gram	$1.8 \times 10^{-4} \text{ Ci}$	$3.1 \times 10^{-1} \text{ Ci}$
Maximum Integrated Concentration in Air	$4.3 \times 10^{-6} \text{ g-sec/m}^3$	$7.2 \times 10^{-10} \text{ Ci-sec/m}^3$	$1.3 \times 10^{-6} \text{ Ci-sec/m}^3$
Maximum Dry Deposition	$7.2 \times 10^{-8} \text{ g/m}^2$	$1.2 \times 10^{-11} \text{ Ci/m}^2$	$2.1 \times 10^{-8} \text{ Ci-m}^2$
Maximum Washout	$5.3 \times 10^{-8} \text{ g/m}^2$	$9 \times 10^{-12} \text{ Ci/m}^2$	$1.5 \times 10^{-8} \text{ Ci/m}^2$
Inhalation Intake	$9.8 \times 10^{-10} \text{ g}$	$1.7 \times 10^{-13} \text{ Ci}$	$3 \times 10^{-10} \text{ Ci}$
Amount Deposited on Surface Water (following Maximum washout)	$2.1 \times 10^{-2} \text{ g}$	$3.6 \times 10^{-6} \text{ Ci}$	$6 \times 10^{-3} \text{ Ci}$
Maximum Integrated Concentration in Surface Water	$3.8 \times 10^{-6} \text{ g-sec/m}^3$	$6.4 \times 10^{-10} \text{ Ci-sec/m}^3$	$1.1 \times 10^{-6} \text{ Ci-sec/m}^3$
Concentration in Ground-Water (180 days after deposition)	$1.2 \times 10^{-8} \text{ g/m}^3$	$2 \times 10^{-12} \text{ Ci/m}^3$	Negligible due to Radioactive Decay
Ingestion Intake from Surface Water	10^{-13} g	$1.7 \times 10^{-17} \text{ Ci}$	$2.8 \times 10^{-14} \text{ Ci}$
Ingestion Intake from Ground Water	$2.6 \times 10^{-11} \text{ g}$	$4.4 \times 10^{-15} \text{ Ci}$	Negligible due to Radioactive Decay

an instantaneous point source release of 1 g of sodium (containing 1.8×10^{-4} Ci of ^{22}Na and 0.31 Ci of ^{24}Na) during stable (F) conditions. Table III-4 presents the data for 0.1% per day leakage from the containment vessel for two hours during stable conditions following the release of 1000 grams of sodium into the containment. Tables III-5 and III-6 show the data for 0.1% per day release from the containment for 150 hours and infinite time during average atmospheric conditions, following the release of 1000 g of sodium into the containment vessel.

Conservative assumptions were used throughout. Thus, the calculations of the concentration of sodium in surface and ground water were based on deposition by washout, which usually gives the highest amount of sodium deposited. The ingestion intake from water, following a contained accident with subsequent leakage into the environment, is calculated on the basis that a man drinks for one day water contaminated by the time-integrated activity over the entire duration of leakage.

The data from Tables III-3 to III-6 were used as input data for the computation of radiation doses resulting from the postulated releases from an LMFBR. Two representative cases out of the four were chosen for further evaluation. They were the instantaneous ("direct") release of 1 g of sodium and the release of 1000 g of sodium into the containment, followed by leakage at a rate of 0.1% per day for an infinite time, under average meteorological conditions ("contained" or "indirect"

TABLE III-4

SODIUM IN THE ENVIRONMENT
FOLLOWING A RELEASE FROM AN LMFBR

Mode of Release: Direct Release into Containment Vessel followed
by 0.1% per day Leakage from Containment Vessel
into Environment

Duration of Leakage: 2 hours

Height of Release: Ground Level

Meteorological Conditions: Stable (F)

Location of Maximum Concentration: 0.3 km from point of release

Amount of Release: 1,000 grams of Sodium into the Containment
Vessel (0.18 Ci of ^{22}Na and 310 Ci of ^{24}Na)

Factor Considered	Data for		
	Na	^{22}Na	^{24}Na
Amount Released to the Environment	$8.3 \times 10^{-2} \text{ g}$	$1.5 \times 10^{-5} \text{ Ci}$	$2.5 \times 10^{-2} \text{ Ci}$
Maximum Integrated Concentration in Air	$4.1 \times 10^{-5} \text{ g-sec/m}^3$	$7.3 \times 10^{-9} \text{ Ci-sec/m}^3$	$1.3 \times 10^{-5} \text{ Ci-sec/m}^3$
Maximum Dry Deposition	$7 \times 10^{-7} \text{ g/m}^2$	$1.2 \times 10^{-10} \text{ Ci/m}^2$	$2.2 \times 10^{-7} \text{ Ci/m}^2$
Maximum Washout	$4.2 \times 10^{-6} \text{ g/m}^2$	$7.5 \times 10^{-10} \text{ Ci/m}^2$	$1.3 \times 10^{-6} \text{ Ci/m}^2$
Inhalation Intake	$1.4 \times 10^{-8} \text{ g}$	$2.4 \times 10^{-12} \text{ Ci}$	$4.3 \times 10^{-9} \text{ Ci}$
Amount Deposited on Surface Water (following Maximum washout)	$4.2 \times 10^{-3} \text{ g}$	$7.5 \times 10^{-7} \text{ Ci}$	$1.3 \times 10^{-3} \text{ Ci}$
Maximum Integrated Concentration in Surface Water	$7.7 \times 10^{-7} \text{ g-sec/m}^3$	$1.4 \times 10^{-10} \text{ Ci-sec/m}^3$	$2.2 \times 10^{-7} \text{ Ci-sec/m}^3$
Concentration in Ground-Water (180 days after deposition)	$2.3 \times 10^{-9} \text{ g/m}^3$	$4.2 \times 10^{-13} \text{ Ci/m}^3$	Negligible due to Radioactive Decay
Ingestion Intake from Surface Water	$2.1 \times 10^{-14} \text{ g}$	$3.8 \times 10^{-13} \text{ Ci}$	$6.1 \times 10^{-15} \text{ Ci}$
Ingestion Intake from Ground Water	$5.1 \times 10^{-12} \text{ g}$	10^{-15} Ci	Negligible due to Radioactive Decay

TABLE III-4 (Continued)

SODIUM IN THE ENVIRONMENT
FOLLOWING A RELEASE FROM AN LMFBR

Height of Release: 50 m.

Location of Maximum Concentration: 5 km from point of release

Factor Considered	Data for		
	Na	²² Na	²⁴ Na
Amount Released to the Environment	$8.3 \times 10^{-2} \text{ g}$	$1.5 \times 10^{-5} \text{ Ci}$	$2.5 \times 10^{-2} \text{ Ci}$
Maximum Integrated Concentration in Air	$1.7 \times 10^{-6} \text{ g-sec/m}^3$	$2.9 \times 10^{-10} \text{ Ci-sec/m}^3$	$5 \times 10^{-7} \text{ Ci-sec/m}^3$
Maximum Dry Deposition	$2.9 \times 10^{-8} \text{ g/m}^2$	$5 \times 10^{-12} \text{ Ci/m}^2$	$8.5 \times 10^{-9} \text{ Ci/m}^2$
Maximum Washout	$2.1 \times 10^{-8} \text{ g/m}^2$	$3.7 \times 10^{-12} \text{ Ci/m}^2$	$6.3 \times 10^{-9} \text{ Ci/m}^2$
Inhalation Intake	$5.6 \times 10^{-10} \text{ g}$	$9.6 \times 10^{-14} \text{ Ci}$	$1.7 \times 10^{-10} \text{ Ci}$
Amount Deposited on Surface Water (following Maximum washout)	$2.1 \times 10^{-3} \text{ g}$	$3.7 \times 10^{-7} \text{ Ci}$	$6.3 \times 10^{-4} \text{ Ci}$
Maximum Integrated Concentration in Surface Water	$3.8 \times 10^{-7} \text{ g-sec/m}^3$	$6.7 \times 10^{-11} \text{ Ci-sec/m}^3$	$1.1 \times 10^{-7} \text{ Ci-sec/m}^3$
Concentration in Ground-Water (180 days after deposition)	$1.2 \times 10^{-9} \text{ g/m}^3$	$2.1 \times 10^{-13} \text{ Ci/m}^3$	Negligible due to Radioactive Decay
Ingestion Intake from Surface Water	10^{-14} g	$1.8 \times 10^{-18} \text{ Ci}$	$3 \times 10^{-15} \text{ Ci}$
Ingestion Intake from Ground Water	$2.6 \times 10^{-12} \text{ g}$	$4.6 \times 10^{-16} \text{ Ci}$	Negligible due to Radioactive Decay

TABLE III-5

SODIUM IN THE ENVIRONMENT
FOLLOWING A RELEASE FROM AN LMFBR

Mode of Release: Direct Release into Containment Vessel followed by 0.1% per day Leakage from Containment Vessel into Environment

Duration of Release: 150 hours

Height of Release: Ground Level

Meteorological Conditions: Average*

Location of Maximum Concentration 0.3 km from point of release

Amount of Release: 1,000 g of Sodium (0.18 Ci of ^{22}Na and 310 Ci of ^{24}Na)

Factor Considered	Na	Data for ^{22}Na	^{24}Na
Amount Released to the Environment	6.3 g	$1.1 \times 10^{-3} \text{ Ci}$	$2.8 \times 10^{-1} \text{ Ci}$
Maximum Integrated Concentration in Air	$1.9 \times 10^{-4} \text{ g-sec/m}^3$	$3.3 \times 10^{-8} \text{ Ci-sec/m}^3$	$8.4 \times 10^{-6} \text{ Ci-sec/m}^3$
Maximum Dry Deposition	$3.2 \times 10^{-6} \text{ g/m}^2$	$5.6 \times 10^{-10} \text{ Ci/m}^2$	$1.4 \times 10^{-7} \text{ Ci/m}^2$
Maximum Washout	$9.5 \times 10^{-5} \text{ g/m}^2$	$1.6 \times 10^{-8} \text{ Ci/m}^2$	$4.2 \times 10^{-6} \text{ Ci/m}^2$
Inhalation Intake	$4.4 \times 10^{-8} \text{ g}$	$7.6 \times 10^{-12} \text{ Ci}$	$1.9 \times 10^{-9} \text{ Ci}$
Amount Deposited on Surface Water (following maximum washout)	$4.6 \times 10^{-1} \text{ g}$	$7.7 \times 10^{-5} \text{ Ci}$	$2 \times 10^{-2} \text{ Ci}$
Maximum Integrated Concentration in Surface Water	$7.7 \times 10^{-5} \text{ g-sec/m}^3$	$1.4 \times 10^{-8} \text{ Ci-sec/m}^3$	$3.6 \times 10^{-6} \text{ Ci-sec/m}^3$
Concentration in Ground-Water (180 days after deposition)	$2.5 \times 10^{-7} \text{ g/m}^3$	$4.4 \times 10^{-11} \text{ Ci/m}^3$	Negligible due to Radioactive Decay
Ingestion Intake from Surface Water	$2.1 \times 10^{-12} \text{ g}$	$3.8 \times 10^{-16} \text{ Ci}$	$9.7 \times 10^{-14} \text{ Ci}$
Ingestion Intake from Ground Water	$5.5 \times 10^{-10} \text{ g}$	$9.7 \times 10^{-14} \text{ Ci}$	Negligible due to Radioactive Decay

*Average refers to 30% stable and 70% unstable conditions

TABLE III-5 (Continued)

SODIUM IN THE ENVIRONMENT
FOLLOWING A RELEASE FROM AN LMFBR

Height of Release: 50 m.

Location of Maximum Concentration: 0.5 km. from point of release

Factor Considered	Data for		
	Na	²² Na	²⁴ Na
Amount Released to the Environment	6.3 g	1.1×10^{-3} Ci	2.8×10^{-1} Ci
Maximum Integrated Concentration in Air	1.3×10^{-5} g-sec/m ³	2.2×10^{-9} Ci-sec/m ³	5.6×10^{-7} Ci-sec/m ³
Maximum Dry Deposition	2.2×10^{-7} g/m ²	3.7×10^{-11} Ci/m ²	9.5×10^{-9} Ci/m ²
Maximum Washout	4.8×10^{-5} g/m ²	8.1×10^{-9} Ci/m ²	2.1×10^{-6} Ci/m ²
Inhalation Intake	3×10^{-9} g	5.1×10^{-13} Ci	1.3×10^{-10} Ci
Amount Deposited on Surface Water (following maximum washout)	3.9×10^{-1} g	6.6×10^{-5} Ci	1.7×10^{-2} Ci
Maximum Integrated Concentration in Surface Water	7×10^{-5} g-sec/m ³	1.2×10^{-8} Ci-sec/m ³	3.1×10^{-6} Ci-sec/m ³
Concentration in Ground-Water (180 days after deposition)	2.2×10^{-7} g/m ³	3.7×10^{-11} Ci/m ³	Negligible due to Radioactive Decay
Ingestion Intake from Surface Water	1.9×10^{-12} g	3.2×10^{-16} Ci	8.3×10^{-14} Ci
Ingestion Intake from Ground Water	4.9×10^{-10} g	8.2×10^{-14} Ci	Negligible due to Radioactive Decay

TABLE III-6

**SODIUM IN THE ENVIRONMENT
FOLLOWING A RELEASE FROM AN LMFBR**

Mode of Release: Direct Release into Containment Vessel followed by 0.1% per day Leakage from Containment Vessel into Environment

Duration of Release: Infinite Time

Height of Release: Ground Level

Meteorological Conditions: Average*

Location of Maximum Concentration: 0.3 km. from point of release

Amount of Release: 1,000 g of Sodium (0.18 Ci of ^{22}Na and 310 Ci of ^{24}Na)

Factors Considered	Na	Data for ^{22}Na	^{24}Na
Amount Released to the Environment	1,000g	10^{-1}Ci	$2.8 \times 10^{-1}\text{Ci}$
Maximum Integrated Concentration in Air	3×10^{-2} g-sec/m ³	3×10^{-6} Ci-sec/m ³	8.4×10^{-6} Ci-sec/m ³
Maximum Dry Deposition	5.1×10^{-4} g/m ²	5.1×10^{-8} Ci/m ²	1.4×10^{-7} Ci/m ²
Maximum Washout	1.5×10^{-2} g/m ²	1.5×10^{-6} Ci/m ²	4.2×10^{-6} Ci/m ²
Inhalation Intake	6.9×10^{-6} g	7.1×10^{-10} Ci	2×10^{-9} Ci
Amount Deposited on Surface Water (following maximum washout)	7.2×10 g	7.3×10^{-3} Ci	2.1×10^{-2} Ci
Maximum Integrated Concentration in Surface Water	1.3×10^{-2} g-sec/m ³	1.3×10^{-6} Ci-sec/m ³	3.8×10^{-6} Ci-sec/m ³
Concentration in Ground-Water (180 days after deposition)	4×10^{-5} g/m ³	4×10^{-9} Ci/m ³	Negligible due to Radioactive Decay
Ingestion Intake from Surface Water	3.5×10^{-10} g	3.5×10^{-14} Ci	10^{-13} Ci
Ingestion Intake from Ground Water	8.8×10^{-8} g	8.8×10^{-12} Ci	Negligible due to Radioactive Decay

*Average refers to 30% stable and 70% unstable conditions.

TABLE III-6 (Continued)

SODIUM IN THE ENVIRONMENT
FOLLOWING A RELEASE FROM AN LMFBR

Height of Release: 50 m.

Location of Maximum Concentration: 0.5 km from point of release

Factors Considered	Na	Data for ^{22}Na	^{24}Na
Amount Released to the Environment	1,000g	10^{-1}Ci	$2.8 \times 10^{-1}\text{Ci}$
Maximum Integrated Concentration in Air	$2.1 \times 10^{-3}\text{g-sec/m}^3$	$2 \times 10^{-7}\text{Ci-sec/m}^3$	$5.5 \times 10^{-7}\text{Ci-sec/m}^3$
Maximum Dry Deposition	$3.6 \times 10^{-5}\text{g/m}^2$	$3.4 \times 10^{-9}\text{Ci/m}^2$	$9.4 \times 10^{-9}\text{Ci/m}^2$
Maximum Washout	$8.1 \times 10^{-3}\text{g/m}^2$	$7.6 \times 10^{-7}\text{Ci/m}^2$	$2.1 \times 10^{-6}\text{Ci/m}^2$
Inhalation Intake	$5 \times 10^{-7}\text{g}$	$4.8 \times 10^{-11}\text{Ci}$	$1.3 \times 10^{-10}\text{Ci}$
Amount Deposited on Surface Water (following Maximum washout)	$6.5 \times 10\text{g}$	$6.2 \times 10^{-3}\text{Ci}$	$1.7 \times 10^{-2}\text{Ci}$
Maximum Integrated Concentration in Surface Water	$1.2 \times 10^{-2}\text{g-sec/m}^3$	$1.1 \times 10^{-6}\text{Ci-sec/m}^3$	$3 \times 10^{-6}\text{Ci-sec/m}^3$
Concentration in Ground-Water (180 days after deposition)	$3.6 \times 10^{-5}\text{g/m}^3$	$3.5 \times 10^{-9}\text{Ci/m}^3$	Negligible due to Radioactive Decay
Ingestion Intake from Surface Water	$3.3 \times 10^{-10}\text{g}$	$2.9 \times 10^{-14}\text{Ci}$	$8.2 \times 10^{-14}\text{Ci}$
Ingestion Intake from Ground Water	$8 \times 10^{-8}\text{g}$	$7.7 \times 10^{-12}\text{Ci}$	Negligible due to Radioactive Decay

release). The data and parameters appropriate to these cases are in Tables III-3 and III-6. Computer codes EXREM and INREM (Turner, 1968) were used in these calculations. In selecting inputs for these codes, the most conservative data were taken. Thus, whichever height of release (ground level or elevated height) and whichever mechanism of deposition (dry or washout) resulted in higher environmental concentrations were chosen as input data for further calculations.

3. External Radiation Dose from Environmental Sodium

Tables III-7 and III-8 show the external doses at the maximum location, resulting from the "direct" and "indirect" release of sodium, respectively. The integrated concentrations in air and in water were used to obtain integrated dose equivalent (rems) from submersion in the respective media. The dose equivalents from contaminated ground were calculated on the basis of a man standing on the ground without shielding, for a period of 24 hours following contamination. For the indirect release, it was assumed that the ground had been contaminated by the time-integrated deposition over the entire leakage duration.

4. Internal Radiation Dose from Air and Water

The internal radiation dose equivalents to selected tissues resulting from inhalation of contaminated air or from ingestion of contaminated water are presented in Tables III-9 to III-12. The doses resulting from soluble sodium are the subject of Tables III-9 and III-10, for the direct and indirect cases, respectively, while Tables

TABLE III-7

EXTERNAL RADIATION DOSES RESULTING FROM A
"DIRECT" RELEASE OF RADIOACTIVE SODIUM
FROM LMFBR

Mechanism of Irradiation	Type of Radiation	Radiation Dose (rem)	
		^{22}Na	^{24}Na
Submersion in contaminated air	β	9.2×10^{-10}	1.3×10^{-6}
	γ	6.5×10^{-9}	3.7×10^{-5}
	Total	7.4×10^{-9}	3.8×10^{-5}
Submersion in contaminated water	β	4.5×10^{-14}	5×10^{-11}
	γ	6.3×10^{-13}	2.9×10^{-9}
	Total	6.8×10^{-13}	3×10^{-9}
Accumulated dose from contaminated ground (during 24 hours, immediately after deposition)	β	1.6×10^{-7}	2×10^{-4}
	γ	1.4×10^{-6}	3.5×10^{-3}
	Total	1.5×10^{-6}	3.7×10^{-3}

NOTE: The beta dose given is that at the skin surface; the dose decreases rapidly with tissue depth.

TABLE III-8
EXTERNAL RADIATION DOSES RESULTING
FROM AN "INDIRECT" RELEASE OF
RADIOACTIVE SODIUM FROM LMFBR

<u>Mechanism of Irradiation</u>	<u>Type of Radiation</u>	<u>Radiation Dose (rem) from:</u>	
		<u>^{22}Na</u>	<u>^{24}Na</u>
Submersion in contaminated air	β	1.5×10^{-7}	3.5×10^{-7}
	γ	1.1×10^{-6}	10^{-5}
	Total	1.2×10^{-6}	10^{-5}
Submersion in contaminated water	β	7.2×10^{-11}	1.6×10^{-10}
	γ	10^{-9}	9.1×10^{-9}
	Total	1.1×10^{-9}	9.5×10^{-9}
Accumulated dose from contaminated ground(during 24 hours, immedi- ately after deposition)	β	9×10^{-5}	2.1×10^{-4}
	γ	7.8×10^{-4}	3.7×10^{-3}
	Total	8.4×10^{-4}	3.9×10^{-3}

NOTE: The beta dose given is that at the skin surface; the dose decreases rapidly with tissue depth.

TABLE III-9
INTERNAL RADIATION DOSES FROM INTAKE OF
SOLUBLE RADIOACTIVE SODIUM--"DIRECT" RELEASE

Organ	Radiation Doses (rem) from: Mode of Intake					
	Inhalation		Ingestion from Surface Water		Ingestion from Ground Water	
	^{22}Na	^{24}Na	^{22}Na	^{24}Na	^{22}Na	^{24}Na
All Organs (except G.I.)	5.6×10^{-8}	9.5×10^{-6}	4×10^{-13}	5.3×10^{-11}	1.1×10^{-10}	Negligible Due to Radioactive Decay
G.I. Tract	1.3×10^{-8}	3.8×10^{-5}	1.4×10^{-13}	3×10^{-10}	3.7×10^{-11}	Negligible Due to Radioactive Decay

TABLE III-10
INTERNAL RADIATION DOSES FROM INTAKE OF
SOLUBLE RADIOACTIVE SODIUM--"INDIRECT" RELEASE

Organ	Radiation Doses (rem) from: Mode of Intake					
	Inhalation		Ingestion from Surface Water		Ingestion from Ground Water	
	^{22}Na	^{24}Na	^{22}Na	^{24}Na	^{22}Na	^{24}Na
All Organs (except G.I.)	10^{-5}	2.6×10^{-6}	6.6×10^{-10}	1.7×10^{-10}	1.7×10^{-7}	Negligible due to Radioactive Decay
G.I. Tract	2.2×10^{-6}	1.1×10^{-5}	2.3×10^{-11}	9.7×10^{-10}	5.8×10^{-8}	Negligible due to Radioactive Decay

TABLE III-11

INTERNAL RADIATION DOSES FROM INTAKE OF INSOLUBLE
RADIOACTIVE SODIUM--"DIRECT" RELEASE

Radiation Doses (rem) from:
Mode of Intake:

Organ	Inhalation		Ingestion from Surface Water		Ingestion from Ground Water	
	^{22}Na	^{24}Na	^{22}Na	^{24}Na	^{22}Na	^{24}Na
G. I. Tract	1.7×10^{-7}	3.1×10^{-4}	1.4×10^{-12}	2×10^{-9}	3.6×10^{-10}	Negli- gible Due to Radio- active Decay
Lungs	3.1×10^{-6}	6.2×10^{-5}				

TABLE III-12

INTERNAL RADIATION DOSES FROM INTAKE OF INSOLUBLE
RADIOACTIVE SODIUM--"INDIRECT" RELEASE

Radiation Doses (rem) from:
Mode of Intake:

Organ	Innhalation		Ingestion from Surface Water		Ingestion from Ground water	
	^{22}Na	^{24}Na	^{22}Na	^{24}Na	^{22}Na	^{24}Na
G. I. Tract	3×10^{-5}	8.7×10^{-5}	2.3×10^{-9}	6.5×10^{-9}	5.8×10^{-7}	Negli- gible Due to Radio- active Decay
Lungs	5.5×10^{-4}	1.7×10^{-5}				

III-11 and III-12 present the doses from insoluble sodium for the two cases. In each case the doses are computed on the basis of the most conservative values from Tables III-3 and III-6.

5. Internal Radiation Doses from Ecological Pathways

Internal doses received via terrestrial or aquatic ecological pathways, through consumption of contaminated food, were also calculated, for both adults and children. It was assumed that the total food supply comes from the contaminated area. In calculations of dose via aquatic ecological pathways, it was assumed that the entire food supply was of aquatic origin.

Two main terrestrial pathways, soil and milk, and two main aquatic pathways, fresh water and seawater, were included in the models (Ng et.al., 1968; Pratt, 1968). (See Appendix C for further description of the models.) Pratt's models were used for calculating the radiation doses resulting from a direct instantaneous release although they were developed for continuous contamination and equilibrium conditions. Integrated concentrations ($C_i \text{ sec m}^{-3}$) were used rather than continuous concentrations ($C_i \text{ m}^{-3}$). The results are accordingly expressed in integrated dose equivalent (rem) rather than in dose equivalent rates (rem/hr). It was assumed that the ecological behavior and transfer coefficients appropriate for sodium in a continuous contamination situation were also applicable to a transiently contaminated environment. This is believed to be reasonable because the overwhelmingly greater mass of natural sodium in the environment would control the behavior rather than the small mass involved in

the release. Tables C-5 and C-6 show that the half-time to equilibrium is short for ^{24}Na , 0.45 to 1.5 days (except for the soil pathway for adults in which the half-time is 80.7 days). The half time to equilibrium for ^{22}Na varies from 1.6 to 32.5 days, except for the soil pathway for which it is of the order of 1000 days.

Tables III-13 (direct release) and III-14 (indirect release) give the integrated internal radiation dose equivalents from terrestrial ecological pathways, and Tables III-15 and III-16 give them via aquatic ecological pathways, for the two types of release. Once again conservative results were assured by using the highest concentration data from Tables III-3 and III-6 and the equilibrium radiation dose equivalent from Tables C-5 and C-6.

The integrated internal radiation dose from terrestrial ecological pathways was also calculated using the unit-rad-deposition factors of Table C-3 (Ng et al., 1968), which were based on short-term radionuclide releases. These results are presented in Tables III-17 and III-18. Comparison of the terrestrial pathway dose equivalents calculated in the two ways (Tables III-13 and III-14 vs. III-17 and III-18) show that the values calculated by the unit-rad-deposition formula are higher than those based on equilibrium data by about an order of magnitude. This difference arises because the unit-rad-deposition calculations are done with maximum washout concentrations while the equilibrium concentration calculations use values for dry deposition. The dry deposition data are about an order of

TABLE III-13

INTEGRATED INTERNAL RADIATION DOSES FROM TERRESTRIAL
ECOLOGICAL PATHWAYS--"DIRECT" RELEASE
(Based on Equilibrium Data)

Organ	Pathway	Radiation doses (rem) to:			
		ADULT		CHILD	
		^{22}Na	^{24}Na	^{22}Na	^{24}Na
Bone	Soil	1.6×10^{-4}		1.6×10^{-4}	
Whole Body	Soil	3.2×10^{-5}		3.2×10^{-5}	
Bone	Milk	3.1×10^{-4}	2.6×10^{-3}	2.1×10^{-3}	1.8×10^{-2}
Whole Body	Milk	3.1×10^{-4}	2.6×10^{-3}	2.1×10^{-3}	1.8×10^{-2}

TABLE III-14

INTEGRATED INTERNAL RADIATION DOSES FROM TERRESTRIAL
ECOLOGICAL PATHWAYS--"INDIRECT" RELEASE
(Based on Equilibrium Data)

Organ	Pathway	Radiation Doses (rem) to:			
		ADULT		CHILD	
		^{22}Na	^{24}Na	^{22}Na	^{24}Na
Bone	Soil	2.7×10^{-2}		2.7×10^{-2}	
Whole Body	Soil	5.3×10^{-3}		5.3×10^{-3}	
Bone	Milk	5.1×10^{-2}	7×10^{-4}	3.5×10^{-1}	4.9×10^{-3}
Whole Body	Milk	5.1×10^{-2}	7×10^{-4}	3.5×10^{-1}	4.9×10^{-3}

TABLE III-15
INTEGRATED INTERNAL RADIATION DOSES FROM AQUATIC
ECOLOGICAL PATHWAYS--"DIRECT" RELEASE
(Based on Equilibrium Data)

		Radiation doses (rem) to:			
		ADULT		CHILD	
Organ	Pathway	^{22}Na	^{24}Na	^{22}Na	^{24}Na
Bone	Seawater	1.2×10^{-12}	1.9×10^{-10}	1.2×10^{-12}	9.8×10^{-10}
Whole Body	Seawater	2.1×10^{-13}	3.5×10^{-11}	2.2×10^{-13}	1.9×10^{-10}
Bone	Freshwater	1.8×10^{-9}	3.2×10^{-7}	1.8×10^{-9}	1.6×10^{-6}
Whole Body	Freshwater	3.7×10^{-10}	5.9×10^{-8}	3.7×10^{-10}	3.2×10^{-7}

TABLE III-16
INTEGRATED INTERNAL RADIATION DOSES FROM AQUATIC
ECOLOGICAL PATHWAYS--"INDIRECT" RELEASE
(Based on Equilibrium Data)

		Radiation doses (rem) to:			
		ADULT		CHILD	
Organ	Pathway	^{22}Na	^{24}Na	^{22}Na	^{24}Na
Bone	Seawater	1.9×10^{-9}	6.3×10^{-10}	1.9×10^{-9}	3.2×10^{-9}
Whole Body	Seawater	3.4×10^{-10}	1.2×10^{-10}	3.6×10^{-10}	6.3×10^{-10}
Bone	Freshwater	2.9×10^{-6}	1.1×10^{-6}	2.9×10^{-6}	5.3×10^{-6}
Whole Body	Freshwater	6×10^{-7}	1.9×10^{-7}	6×10^{-7}	10^{-6}

TABLE III-17
 INTEGRATED INTERNAL RADIATION DOSES FROM
 TERRESTRIAL ECOLOGICAL PATHWAYS-
 "DIRECT" RELEASE
 (Based on the Unit-Rad-Deposition Factor)

		Radiation doses (rad) to:			
Organ	Pathway	ADULT		CHILD	
		^{22}Na	^{24}Na	^{22}Na	^{24}Na
Bone	Soil	1.8×10^{-3}		1.9×10^{-3}	
Whole Body	Soil	3.4×10^{-4}		3.5×10^{-4}	
Bone	Milk	3.2×10^{-3}	2.3×10^{-2}	2.2×10^{-2}	1.7×10^{-1}
Whole Body	Milk	3.2×10^{-3}	2.3×10^{-2}	2.2×10^{-2}	1.7×10^{-1}

TABLE III-18
 INTEGRATED INTERNAL RADIATION DOSES FROM
 TERRESTRIAL ECOLOGICAL PATHWAYS
 "INDIRECT" RELEASE
 (Based on the Unit-Rad-Deposition Factor)

Radiation doses (rad) to:					
Organ	Pathway	ADULT		CHILD	
		^{22}Na	^{24}Na	^{22}Na	^{24}Na
Bone	Soil	1		1.1	
Whole Body	Soil	1.9×10^{-1}		2×10^{-1}	
Bone	Milk	1.8	2.4×10^{-2}	1.3×10	1.8×10^{-1}
Whole Body	Milk	1.8	2.4×10^{-2}	1.3×10	1.8×10^{-1}

magnitude lower than the washout values (Tables III-3 and III-6). This shows that the two models would give the same results with the same input data and, therefore, the equilibrium data (Pratt, 1968) can be used for calculation of the integrated radiation dose after an instantaneous release.

Tables III-7 through III-18 show clearly that many of the possible mechanisms by which radioactive sodium releases can result in dose to man are actually insignificant. Tables III-19 and III-20 summarize the significant vectors for the cases of direct release of 1 g of sodium and indirect release, continued for an infinite time, of 1000 g of sodium. It is clear that the milk pathway is most significant for both types of release.

TABLE III-19
SIGNIFICANT DOSE PATHWAYS FOR RADIOACTIVE SODIUM IN
DIRECT RELEASE

Based on 1 g of Na discharged directly to environment

	Doses in millirems							
	ADULT				CHILD			
	<u>Whole Body</u>	<u>G-I</u>	<u>Lung</u>	<u>Bone</u>	<u>Whole Body</u>	<u>G-I</u>	<u>Lung</u>	<u>Bone</u>
Contaminated Ground	3.7				3.7			
Inhalation (insoluble)		0.3				0.3	0.1	
Soil-Plant*				0.2				0.2
Forage-Cow- Milk*	2.9			2.9	20			20
Soil-Plant**	0.3			1.8	0.4			1.9
Forage-Cow- Milk**	26			26	190			190

*Calculated from equilibrium data

**Calculated from Unit-Rad-Deposition Factor

TABLE III-20
SIGNIFICANT DOSE PATHWAYS FOR RADIOACTIVE SODIUM IN
INDIRECT RELEASE

Based on 1000 g Na released, to containment, leakage to environment 0.1% per day, to infinite time.

	Doses in millirems							
	ADULT				CHILD			
	<u>Whole Body</u>	<u>G-I</u>	<u>Lung</u>	<u>Bone</u>	<u>Whole Body</u>	<u>G-I</u>	<u>Lung</u>	<u>Bone</u>
Contaminated Ground	4.7				4.7			
Inhalation (insoluble)		0.1	0.6			0.1	0.6	
Soil -Plant*	5			27	5			27
Forage-Cow-Milk*	52			52	355			355
Soil-Plant**	190			1000	200			1100
Forage-Cow-Milk**	1820			1820	13,200			130200

*Calculated from equilibrium data

**Calculated from Unit-Rad-Deposition Factor

IV. Possible Countermeasures

It appears that in accidents of the size postulated, the release of radioactive sodium, for the most part, would not result in any appreciable radiation exposure. There is one notable exception, the entry of sodium into the food chain through the milk supply. This exposure, although even here relatively low for an accident situation, does result in the young, or most sensitive portion of the population having a potential for exposure. Thus it is important to review the countermeasures which might be available in the case of an accident. Such countermeasures can be divided into those applicable to the potentially exposed population and those applicable to the released sodium.

A. Limitation of Potential Exposure

The most obvious countermeasure available is to remove the population from the source of exposure. For a remotely situated reactor or with accident conditions in which only a small number of persons are potentially exposed, this type of action is most appropriate. Under this approach, nothing would be done about the radioactive sodium; the natural radioactive decay would remove that part of the potential hazard due to sodium-24. This is clearly the optimal countermeasure with little or no potential for exposure. Total population dose is kept to a minimum, although the expense and turmoil from such an action could be large.

A second approach would be to permit the exposed population to remain in the area and to suggest a high sodium diet for those potentially exposed, relying on isotopic dilution and

natural radioactive and biological decay to eventually solve the problem. Because of the possible medical implications of this approach (particularly with respect to people on low sodium diets), it would probably not be either acceptable or practical. The National Advisory Committee on Radiation (1962), for example, has stated that diet supplementation "should usually be reserved for limited application due to dangers inherent in the administration of food additives and medicants to large population groups." With respect to the possibility of reducing strontium-90 uptake by increasing calcium intake, the Committee stated that "many uncertainties exist concerning the effects on nutrition and health of high calcium intakes over long periods of time in both humans and dairy herds" and that much research must be done before this type of countermeasure "can be used effectively in a radiation control program." However, supply of compounds of stable iodine to the population to reduce uptake of radioactive iodine into the human thyroid has been recommended (Adams and Bonnell, 1962; Cronquist et al., 1971).

Sullivan (1970) has suggested the use of a commercial cation exchange resin to bind sodium during its passage through the alimentary tract. Compounds which can inhibit the absorption of sodium in animals may be useful (e.g. Maizels, 1954, Mudge and Taggart, 1950, Strickler and Kessler, 1963) as might be peritoneal dialysis (Deichman and Gerade, 1969) but these techniques would have to be carried out under close medical

supervision and would thus be applicable only to the severely exposed individual.

B. Countermeasures against the released sodium

Decontamination of sodium-contaminated foodstuffs is in general impractical as a countermeasure, except in a very few cases. Thus contaminated foods must be destroyed or else stored until their radioactivity can decay. For the case of ^{24}Na contaminated material, this would be feasible; where the contamination is principally ^{22}Na , storage would not be useful.

Methods for the decontamination of water contaminated with sodium do exist although the cost is such that these techniques would be applicable only for the decontamination of small emergency drinking water supplies. It has long been recognized that the removal of sodium from water is a difficult task, principally because of the high solubility of most sodium compounds. Thus conventional water treatment methods would not be particularly applicable for decontamination, although many desalination techniques might be. These techniques would include distillation, vacuum distillation, permeation, etc. and might be used with some expectation of success. The processes could be further enhanced with the addition of stable sodium to the contaminated water prior to treatment, effecting a form of isotopic dilution during treatment.

Conventional water treatment processes may reduce sodium contamination levels somewhat. This was shown, for example, in measurements of ^{24}Na in surface water, performed at Richland,

Pasco and Kennewick, Washington, near the site of the AEC Hanford Plant. Although Pasco and Kennewick are situated at about the same distance from the radioactivity source release, it was found (Woolridge, 1969, 36) that the ^{24}Na concentration in the Kennewick water was lower than that of Pasco, by about an order of magnitude. This difference could be interpreted as resulting from the water treatment and decontamination system of Kennewick. At Pasco a flocculation-filtration water treatment system is used, whereas the Kennewick water is pumped from collectors (infiltration pipes) laid in the riverbank.

For the time period in question for such an accidental release, it would appear much more feasible to utilize non-contaminated sources (such as wells) as a drinking water source or simply to transport non-contaminated water to the area for culinary and drinking purposes.

Soil contamination by radioactive sodium may also be deleterious to man through the soil-plant ecological pathway. Although they would be expensive, countermeasures might also be sought for the treatment of soil following sodium contamination. Displacement of sodium from ion-exchange sites in soil by the addition of a source of soluble calcium has been proposed (Pearson and Bernstein, 1968) as one possible approach. It has also been indicated (Menzel, 1969) that fertilizers may reduce the availability of cesium-137 from the soil, due to the competition with the large amount of potassium present in the fertilizer. It may be expected that similar competition may exist between sodium and potassium. Again, however, as in the case of contaminated

drinking water, it is quite possible that the safest and most feasible approach for limiting the presence of radioactive sodium in dairy products might simply be to place cows on stored feed for a specified period of time following the incident. (National Advisory Committee on Radiation, 1962) Federal Radiation Council, 1964). Such an approach is well known as a successful countermeasure for limiting the concentrations of radioactive iodine in milk and it would undoubtedly be equally successful and practical in the case of the shorter lived ^{24}Na . The longer lived ^{22}Na could presumably be removed from milk by replacing the radioactive sodium atoms by stable atoms through application of an ion exchange process such as has been developed on a pilot plant scale for ^{90}Sr removal. (Easterly, Brooks, Hasuike, and Weaver, 1971)

Removal of sodium from soil by washing with fresh water is a recognized method for the desalination of soils which have accumulated Na by over irrigation. Although moderately expensive, a two-stage system could be envisioned in which salty water is used for isotopic dilution followed by fresh water leaching to lower the Na content.

V. Discussion and Conclusions

In evaluating the accident potential of the release of radioactive sodium to the environment from a liquid metal fast breeder reactor a number of conservative assumptions have been made. Even with these assumptions and for the cases which were studied there appears to be only a slight possibility of significant population exposure from the released radioactive sodium.

For purposes of this report it was assumed that the sodium aerosol released into the containment would remain entirely airborne for the duration of leakage from the containment into the environment. This, of course, would never happen; during leakage some of the material would be deposited within the containment vessel. Thus this certainly is a conservative assumption.

Conservative environmental conditions were also chosen. For example, a stable atmosphere was used, washout conditions were assumed to prevail both during and after the release, no allowance was made for cross-wind dilution (which of course could be quite significant) and finally, in evaluating the radiation dose caused by terrestrial or aquatic ecological pathways it was assumed that the exposed population obtains their entire food supply from the contaminated area. This last assumption is especially conservative since in the case of an accident steps could readily be taken to prevent the population from receiving food from the contaminated area.

Therefore, while a wide spectrum of accidents and sodium

releases might be assumed, the accident and subsequent release which was analyzed in this report can be considered as indicative of all similar or larger accidents to at least within an order of magnitude. Thus it should be possible to use the data presented here as an indication of the degree and duration of restrictions which might have to be imposed on contaminated areas, agricultural land and food supplies, as well as of the countermeasures which could be implemented to diminish loss and damage.

The most significant exposure is to a child, from the forage-cow-milk route. For the instantaneous release, a total dose of 0.19 rem is estimated from this source while from the 1000 gm. release (infinite time leakage) a child could receive a maximum dose of 13.2 rems. In these calculations, it was conservatively assumed that all the activity brought down by rain was retained on grass, and none was washed onto the soil. Limitations of the leakage from the containment vessel to 150 hours would reduce the child dose from 13.2 to 0.3 rem. Adult doses are about one-seventh as much, reflecting the smaller amount of milk consumed by adults. Estimates of these exposures based on less conservative equilibrium data calculations give values for children of 0.02 and 0.355 rem respectively for the two cases. No matter how the number is calculated, however, it is clear that this exposure could be limited by restricting the use of milk coming from the contaminated area, which is certainly a minimal step to take and a most conservative one.

Another important source of radiation exposure appears to be direct radiation from contaminated ground. For the postulated accident an average dose of about 4 to 5 mrem is

estimated. Since natural background radiation contributes about 150 mrem per year population exposure, ground contamination does not appear to be a significant radiation source. In a somewhat larger accident, prudence might dictate removal of the affected population until the hazard abates.

Inhalation of the released radioactive cloud appears to be among the least significant routes of exposure. This is important since such inhalation following an accident is the most difficult exposure route to control. For example, with the instantaneous release it is probable that the exposure could take place in many cases before the exposed persons would be notified, let alone moved.

There are certain reservations associated with these conclusions which should be recognized. The state of current research on the ecological behavior of sodium does not make possible a sound quantitative analysis of its transfer and concentration factors in the different environmental compartments. Concentration factors for sodium in animals are one of the highest among different cations and this results in the largest exposure coming from the food chain pathway.

On the other hand, about 75% of the sodium in the human body is available for rapid exchange and therefore countermeasures may be developed to eliminate part of the radioactive sodium from the human body, and thus diminish considerably the radiation doses to the different tissues.

Countermeasures may also be sought and developed for the

pre-human-intake stage, to cope with the environmental hazards caused by a release of sodium from an LMFBR.

Thus we can conclude that although the preliminary present hazards evaluation has been made on a conservative basis, the results indicate that for the cases which were studied, both the internal and external radiation doses to man would be small.

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APPENDIX A

SODIUM AND ITS COMPOUNDS

Important physical, chemical and toxicological properties of stable and radioactive sodium and its compounds are included here for reference purposes.

1. Distribution of Sodium in Nature

Because of its high reactivity, sodium is always found combined in nature as NaCl , Na_2CO_3 , Na_2SO_4 , NaNO_3 , etc. The earth's crust contains 2.63% of sodium in combined form (Sittig, 1965,1). In igneous rocks, sodium abundance varies between 1.8 and 4.0% whereas in sedimentary rocks, the variation is between 4.0 and 9.6%, (Levin et al., 1970). The average concentration of sodium in sea water is 10.56 g/kg (Junge, 1963,165).

The mean concentration of sodium in air in coastal areas, measured at Round Hill, some 50 miles south of Boston (Junge, 1963, 171) is $0.1 \mu\text{g}/\text{m}^3$ in aerosol particles having radii between 0.2 and $1 \mu\text{m}$ and increasing to $1.6 \mu\text{g}/\text{m}^3$ in airborne particles larger than $1 \mu\text{m}$ radii. An increase in sea salt concentration in air was found (Woodcock; 1953; Junge, 1963, 162) with an increase in wind velocity over the sea and on coastal areas. A steady state concentration of sea salt particles is suggested for a certain wind velocity, balanced by the production at the sea surface on one hand, and by deposition and washout on the other hand. The NaCl content of rainwater at Quillayate site, Washington, U.S.A. (marine atmosphere) was found to be about 1 mg/liter (Perkins et al., 1970).

Concerning the radioactive species of sodium in the atmosphere, ^{22}Na and ^{24}Na were found to be produced continuously in the atmosphere by cosmic ray spallation reactions (e.g. of argon). They are also formed by induced activation reactions during nuclear weapon tests. A concentration of ^{22}Na of 0.017 dis/min per liter was found in rainwater at Rio de Janeiro, Brazil (Marquez et al., 1957). A mean concentration of ^{22}Na in air, of about 0.01 dis/min per thousand cu. ft. was measured in Richland, Washington, during the period 1962-1964 (Perkins et al., 1965).

Table A-1 shows the average content of stable sodium (ppm) in the different aquatic systems as computed by Chapman et al. (1968).

TABLE A-1
CONTENT OF STABLE SODIUM IN AQUATIC SYSTEMS

<u>Aquatic System</u>	<u>Concentration* (ppm) in:</u>	
	<u>Marine Water*</u>	<u>Fresh Water</u>
Water	1.05×10^4	6.3**
Plants	6×10^3	10^3
Invertebrates	2×10^3	1.7×10^2
Fish	7×10^2	2×10^2

*in the continental shelf or estuarine water

**mean values ranging from 1.9 to 17 ppm were found in different rivers in the world.

The sodium content for different soil depths is shown in Table A-2 for tropical moist and the Premontane wet forests (Golley et al., 1969).

TABLE A-2
SODIUM LEVEL (g/m^2) IN TROPICAL MOIST AND PREMONTANE WET FORESTS

<u>Soil Depth (cm)</u>	<u>Site</u>	<u>Sodium Level (g/m^2)</u>
0 - 7.6	Tropical Moist	44.194
	Premontane	17.067
30 - 41	Tropical Moist	88.540
	Premontane	16.125
0 - 91	Tropical Moist	652.466
	Premontane	100.352

Ng et al., (1968) reported a concentration of sodium of 6,300 ppm in typical agricultural soil.

Table A-3 shows the concentration of sodium in different parts of vegetation found in Panama forests and reported by Golley et al. (1969). Comparison of these data with those shown in Table A-2 for soil content of sodium reveals the concentration factor of sodium in the vegetation of the Premontane area as compared to the tropical moist area.

TABLE A-3

A COMPARISON OF SODIUM IN STANDING CROPS IN TROPICAL
MOIST AND PREMONTANE FOREST IN PANAMA

Compartment	Site	Sodium Content (mg/m ²)	Biomass kg/hectare
Overstory leaves	Tropical moist	491	7,349
	Premontane	2,469	10,576
Understory leaves	Tropical moist	67	580
	Premontane	91	348
Overstory stems	Tropical moist	25,281	252,126
	Premontane	79,294	258,434
Understory stems	Tropical moist	317	3,446
	Premontane	216	564
Fruits and flowers	Tropical moist	2	8
	Premontane	2	7
Epiphytes	Premontane	484	1,440
Litter	Tropical moist	449	6,200
	Premontane	1,185	4,820
Roots	Tropical moist	496	12,633
	Premontane	3,417	12,707

2. Properties of Elemental Sodium

a. Physical Properties

Table A-4 (Handbook, 1962; Sittig, 1956,2) shows some of the physical constants of elemental sodium.

TABLE A-4

PROPERTIES OF ELEMENTAL SODIUM

Molecular Weight	22.991
Crystalline Form	cubical, silver-metal
Density at 20°C*	0.97
Index of Refraction	4.22
Melting Point (°C)	97.5
Boiling Point (°C)	880
Heat of Fusion (cal/g)	27.1
Heat of Vaporization (cal/g)	902

*The density of solid sodium decreases from 0.97 g/cc at 0°C to 0.94 g/cc at 100°C, while that of liquid sodium decreases from 0.92 g/cc at 101°C to 0.8 g/cc at 626°C (Sittig, 1956, 363).

Sodium has a relatively low vapor pressure (about 3 mm at 500°C) compared to other liquid heat transfer media (Sittig, 1956, 388), and to other alkali metals. For example, vapor pressure data (Clough et al., 1970) show that cesium at 500°C is about 70 times as volatile as sodium at the same temperature. Sodium vapor at room temperature is monoatomic; as it is heated it dimerizes.

b. Sodium Fires.

The ignition point of sodium in air is 230°C but, because of its high thermal conductivity, ignition does not occur until the entire mass of sodium has been heated to this temperature. (Sittig, 1956, 201). In sodium spray fire experiments performed in oxygen environments of 2 to 21 vol.% it was found (Granger et al., 1970) that the major reaction product is sodium peroxide (Na_2O_2). Granger et al. hypothesized that the monoxide may be formed close to the sodium droplet but it is likely to be converted to peroxide in the cooler regions surrounding the burning zone. In sodium pool fire experiments, the initial burning rate of sodium in 21 vol. % oxygen is 5 lb/hr/sq. ft of exposed surface, (Chase 1966), whereas in an oxygen depleted atmosphere the burning rate is lower.

The size of aerosol particles measured immediately after sodium burning experiments was found to be about 0.5 μm . In high concentrations, due to agglomeration, the oxide aerosol reaches in a short time a median radius of the order to 4-5 μm (Greenfield et al., 1970) and may agglomerate to particles of much larger size.

c. Reactions of Sodium in the Atmosphere.

Metallic sodium oxidizes rapidly when exposed to moist air and then absorbs H_2O and CO_2 to form NaOH and Na_2CO_3 . The following reactions occur between metallic sodium and atmospheric components:

(1) In the presence of an excess of water (Beusman, 1959, Sittig, 1956, 187):

(2) In the presence of an excess of sodium (Reist, 1969):

(3) In absence of water vapor (Sittig, 1956, 190-191):

(4) In presence of CO_2 :

Most of the reactions are exothermic and the formation of sodium compounds occurs with an increase in temperature and pressure.

d. Other reactions of Sodium.

In addition to high reactivity in the presence of oxygen, hydrogen, and water, metallic sodium also reacts vigorously with halogens and with most inorganic halides. Thus, a decontamination factor for iodine of at least 1000 was found (Cave and Crickmer, 1967) when iodine was passed through a significant depth of liquid sodium and only 10% of the iodine which reacted with the liquid metal was released when the sodium was burned in air.

Hydrocarbons are generally unreactive in the presence of sodium metal (Beusman, 1959).

Sodium absorbs hydrogen at about 200°C and more rapidly at 300 - 400°C (Sittig, 1956, 201) to form sodium hydride (NaH) which covers the liquid sodium as a film, leaving the reaction incomplete. Sodium hydride reacts with water even more violently than sodium with water.

3. Radioactive Sodium

Sodium - 23 constitutes 100% of natural sodium. Two major radioactive isotopes (Hungerford, 1966), are produced from it in nuclear reactors:

^{24}Na , produced by the (n,γ) reaction:

and, ^{22}Na , produced by the (n,2n) reaction with a neutron energy threshold of 11.7 MeV:

About 10% of the ^{22}Na disintegrations are by electron capture rather than positron emission. The cross section for the ^{23}Na (n,2n) ^{22}Na reaction is very low (of the order of 6 microbarns, averaged over the fission spectrum) and therefore the amount produced in nuclear reactors is relatively small as compared to the amount of ^{24}Na , the cross section of which is 0.53 barn for thermal neutrons. However, due to its long half-life, ^{22}Na may be more important as a radiological hazard.

Table A-5 shows some of the principal nuclear properties of ^{24}Na and ^{22}Na (from: Hungerford, 1966, 425; Radiological Health Handbook, 1970; and Turner et al., 1968) and Table A-6 (from Hungerford, 1966, 435, Fies, 1971) shows the specific activity of sodium primary coolant, as measured or estimated, in different fast reactors.

TABLE A-5

NUCLEAR PROPERTIES OF RADIOACTIVE SODIUM ISOTOPES

Isotope	^{22}Na	^{24}Na
Half-life	2.6 yr	15.1 hr
Decay constant	$8.5 \times 10^{-9} \text{ sec}^{-1}$	$1.28 \times 10^{-5} \text{ sec}^{-1}$
Photons per disintegration		
Nuclear gamma	1	2
Annihilation	1.8	
Energy per photon	γ_1 1.28 MeV (100%) γ_{an} 0.51 MeV (180%)	γ_1 1.37 MeV (100%) γ_2 2.75 MeV (100%)
Energy absorption Coef. (Soft Tissue)	γ_1 $3.3 \times 10^{-5} \text{ cm}^{-1}$ γ_{an} $3.7 \times 10^{-5} \text{ cm}^{-1}$	γ_1 $3.3 \times 10^{-5} \text{ cm}^{-1}$ γ_2 $2.68 \times 10^{-5} \text{ cm}^{-1}$
Particulate radiation		
β^- particles/disinteg.	0	1
β^+ particles/disinteg.	0.90	0
Electron cap/disinteg.	0.10	
Beta particle energy, MeV	0.545 (89.8%) 1.82 (0.05%)	1.39 (~100%) 4.17 (0.003%)
Specific activity (Ci/g) of pure nuclide	6.25×10^3	8.71×10^6

TABLE A-6

SPECIFIC ACTIVITY OF PRIMARY SODIUM COOLANT
FOR VARIOUS FAST REACTORS

Reactor	Reactor Power (Mwt)	Specific Activity			
		Ci/cm ³		Ci/cm ³ /Mwt	
		²² Na	²⁴ Na	²² Na	²⁴ Na
Fermi	300	1.0×10^{-6}	3.0×10^{-2}	3.3×10^{-9}	1.0×10^{-4}
Dounreay	60	3.5×10^{-6}	3.5×10^{-3}	58.0×10^{-9}	0.58×10^{-4}
EBR-I	1.4		2.5×10^{-6}		0.02×10^{-4}
EBR-II	62.5		1.4×10^{-3}		0.23×10^{-4}
FFTF	400	3.2×10^{-7}	4.2×10^{-2}	8×10^{-10}	1.05×10^{-4}

APPENDIX B

BASIC DATA ON ENVIRONMENTAL BEHAVIOR AND TRANSPORT OF SODIUM

Most of the sodium released from a reactor containment vessel will combine to form an oxide, hydroxide, or carbonate by reaction within the containment vessel or in the outside atmosphere. Following an accidental release of sodium to the atmosphere, it will be transported some distance downwind from the source by the wind and return to earth by a number of deposition mechanisms (Gifford, 1968) which include fallout, dry deposition, and washout. At this point sodium enters various pathways through which it may ultimately reach man by mechanisms other than direct inhalation, i.e., through water, soil, vegetation, fish, and animals.

1. Transport in the Hydrosphere

After dispersion through the atmosphere, sodium may deposit directly on surface waters (river, lake, and ocean) or first on soil and then reach surface waters by runoff or it may leach through the ground to reach subsurface waters. This will contaminate drinking water and provide a means of uptake by plants, fish, and other water biota. These mechanisms have been studied by Fisher (1971) and others. In a review of the transport of liquid waste effluents in surface water, Magno et al. (1971) defined an Effective Dilution Factor (EDF) as the product of the Water Dilution Factor (DF) and the Stream Absorption Factor (AF). All processes which remove a radionuclide from the water are included in AF. Several processes participate in the dilution or concentration of radionuclides being transported as particulate matter or solute in surface water, (Rice, 1963).

The major processes influencing dilution and dispersion are currents and turbulent diffusion. Dispersion of a pollutant in turbulent surface water is governed by two main mechanisms: (a) convection, which depends on the current velocity patterns, and (b) eddy diffusion, which depends on the natural turbulence of the water. Several equations including the following have been proposed for the estimation of the diffusion and dispersion of pollutants in surface water. Considering the centerline concentration of the pollutant and diffusion along the x direction only, Carrit et al., (1959) calculated the maximum concentration, C_{max} , at a distance, $x(m)$ of a pollutant released from a continuous source with the following equation:

$$C_{max} = \frac{Q}{2D(2\pi Kux)^{1/2}} \quad (B-1)$$

where

Q = rate of release of pollutant (Ci or g per second)

D = Depth of water (m)

K = diffusion coefficient (m^2/sec)

u = current velocity (m/sec)

x = distance (m)

For an instantaneous source, assuming no current to aid dispersal, the concentration, C , at time, t , at the point of release, is given by the equation:

$$C = \frac{C_0}{4} \frac{V}{(\pi K t)^{3/2}} \quad (B-2)$$

where:

C_0 = initial concentration

V = volume of source

A three dimensional solution to the mass conservation equation assuming that lateral and vertical velocities are negligible, yields the following equation (Dawson, et al., 1970)

$$C(x,y,z,t) = \frac{k}{t^{3/2}} \exp\left\{-\frac{1}{4}t\left[\frac{(x-ut)^2}{K_x} + \frac{y^2}{K_y} + \frac{z^2}{K_z}\right]\right\} \quad (B-3)$$

where:

k =a constant evaluated from boundary conditions

y, z =lateral and vertical coordinates, respectively

K_x, K_y, K_z =diffusion coefficients in x, y , and z direction, respectively.

Additional processes influencing dilution and dispersion are isotopic dilution by the corresponding stable element in water, and biological transport. Radioactive decay reduces the concentration of radionuclides in water. Sorption on sediments or suspended material, ion-exchange, co-precipitation, flocculation and sedimentation influence physico-chemical concentration whereas uptake directly from water or passage through food webs influences biological concentration. For example, the degree of concentration of ^{24}Na by lower water organisms and plants has given rise to a proposal to reduce by a factor of 100 the MPC established by NCRP for this radionuclide (Thompson et al., 1956).

2. Transport through soil and underground water

Soil deposition of radionuclides results in external radiation doses to people standing on the contaminated ground. Resuspension of the deposited material by wind erosion or other disturbance may cause inhalation hazards and consequently internal radiation doses. Uptake of the radionuclides from soil by plant roots may transfer the radionuclides through the different food chains to man. Run-off, erosion and leaching of contaminated soil may transfer the radionuclides to a potable water supply or to aquatic food chains and thus, ultimately to man.

Leaching of a radionuclide within the ground, its transport through the underground water table and contamination of drinking water are influenced by the amount of deposited pollutant, amount of rainfall on the site where the radionuclide deposits, half-life of the radionuclide, its sorption (chemical or physical) and ion exchange properties in soil and underground materials, structure of the underground material (porosity, density, etc.), residence time of water, distance of water pumping site from source of deposited radionuclide, and amount of water pumped from the site. Studies by Kelley (1951) showed that leaching of sodium from the ground surface to the underground water table yielded a highly concentrated leachate of (a) sodium carbonate when CO_2 is present from the air, and (b) sodium hydroxide when CO_2 is absent. It was found (Barshad, 1960) that about 50% of the absorbed sodium was replaced following the leaching of Na-clays with distilled water. In a study of pos-

sible contamination of ground water systems by accelerator-induced activity, an evaluation of ^{22}Na transport in underground water was made by Thomas (1971). He concluded that the specific activity of ^{22}Na decreased very slowly for residence times up to about 1000 days for the residence and transport times he assumed.

Based on Thomas' (1971) estimations, and those of Mawson (1969), it can be concluded that adsorption, chemical sorption, and exchange processes between radionuclides and constituents of the soil decrease the leachability of radionuclides and therefore, the concentration of radionuclides in ground water should be reduced considerably. Once in the ground water table, radionuclides move faster, but still at a lower rate (by orders of magnitude) than the ground water movement. The movement of a radionuclide through an idealized water-shed system was analyzed by Werner (1969) who used a retardation factor to express the ratio between the flow rate of water to that of the radionuclide.

A variety of other models have been proposed for the redistribution of radioactive material from the ground surface to the underground water table (e.g., Kaufman, 1963; Nelson, 1965; Huff and Kruger, 1967) and a simplified model was proposed by Charnell et al. (1969). A conclusion drawn from these studies is that measured data on the exchange and sorption properties of sodium in underground materials are required to make a sound estimation of the transport of sodium from the ground surface

to the underground water table. Movement of a radionuclide through soil is influenced by the self-diffusion behavior of the radionuclide. This, in turn, depends on the concentration difference between two layers of soil. Diffusion mechanisms may control the rate of ion movement through soil when the evapo-transpiration processes are low but in a saturated soil system, diffusion becomes less important relative to liquid flow (Fried et al., 1960). Even at temperatures ranging between about 1200° and 1400°C, the diffusion coefficient of sodium measured in molten silicates was only about 10^{-5} cm²/sec (Freiling et al., 1965).

3. Concentration factors in aquatic systems

A major source of data on concentration factors of sodium in aquatic systems is a comprehensive and detailed study performed at Lawrence Radiation Laboratory on the environmental consequences of Plowshare Projects, especially the Atlantic-Pacific Interoceanic Canal (Chapman et al., 1968). The concentration factors compiled in this study are based mainly on the behavior of stable elements, but they are useful for evaluating hazards from a release of radioisotopes of the same element. Table B-1 shows concentration factors of stable sodium in marine and fresh water as compiled by Chapman et al. (1968). The authors warn that these data should be considered to have only an order of magnitude reliability because some radionuclides may have greater biological availability than stable elements. However, sodium isotopes exchange rapidly and the reliability should be better.

Table B-1 indicates that marine plants and animals contain less sodium than their water environment whereas those in fresh waters contain considerably more. Casarett (1968)

found a concentration factor of 0.07 to 1 in marine water for sodium accumulations in algae, vertebrates, and invertebrates. Considering the uncertainties in the estimation and the large amount of sodium in the biosphere, it appears likely that no significant concentration of radio-sodium relative to normal sodium will occur.

TABLE B-1

CONCENTRATION FACTORS* OF SODIUM IN AQUATIC SYSTEMS

<u>Aquatic System</u>	<u>Concentration Factor** in:</u>	
	<u>Marine Water</u>	<u>Fresh Water</u>
Plants	5.7×10^{-1}	1.6×10^2
Invertebrates	1.9×10^{-1}	2.7×10
Fish	6.7×10^{-2}	3.2×10

*The concentration factor is defined as:

$$\frac{\text{Concentration of an element in aquatic organism (ppm-wet weight)}}{\text{Concentration of the element in water (ppm)}}$$

**The concentration factors given by Chapman et al. (1968) were rounded to the first decimal figure.

4. Concentration in sediments

Underground water table sediments, which have a high-ion-exchange capacity, may hinder the transport of pollutants, compared to the water flow, and delay their appearance at a site where the water is consumed. Considering short-lived radionuclides, this effect is beneficial since they would decay during this period. However, high ion-exchange capacity of the sediment may result in storage of long-lived radionuclides that can be released suddenly at a later time.

In a study of the behavior of radionuclides in the Columbia River, it was found (Nielsen, 1963) that half the ^{24}Na released failed to appear 38 miles downstream. Radioactive decay and sorption by the sediments are responsible for the depletion. Cremers and Thomas (1966) found the sodium exchange capacity of sodium montmorillonite clay to be about 1 meq/g. It increased with an increase in concentration of dissolved sodium. Similar results were found by Mokady et al. (1962) on sand; however, they found a drastic reduction in the ion exchange capacity when the moisture content of the soil was reduced. Much lower exchange capacities of cesium (down to 0.2 meq/100g) were determined by Kennedy (1963) for some fractions of stream sand (e.g., quartz-feldspar fraction).

A mass conservation equation was developed by Reynolds (1963) to express the radioactivity budget in a stream, taking into consideration sorption by sediments, and other stream systems but the equation does not give a straight-forward solution because some of the parameters included in it are interdependent.

5. Concentration in food

Contamination of vegetation by radionuclides results in hazards to man from direct consumption and from consumption of contaminated animal food products (e.g., milk, eggs, and meat). Leaching of radionuclides from contaminated vegetation results in contamination of the ground and surface and underground water.

Uptake of radionuclides from contaminated soil by plant roots occurs by sorption, penetration, and ion-exchange processes and accumulation is specific for various ions (Collander, 1941). Differences of up to three orders of magnitude between the absorption of Na in various plants have been found by Bertrand (1951), Larson and Pierre (1953) and others. Although differences of more than an order of magnitude have been observed between the ^{22}Na content of different plant samples in the Western United States, an average value for the concentration of sodium in terrestrial plants was assumed by Ng et al. (1968) in their calculations of radiation doses. The behavior of radioactive sodium in the soil-root pathway is of interest with respect to ^{22}Na only, since Ng and Thompson (1966) have indicated that this pathway cannot be significant for short lived radionuclides.

The half-residence time of radionuclides on forage was found to vary from 9 to 14 days regardless of the nature of the element (Ng and Thompson, 1966). Thus, this range of half-residence life on forage may be considered as applicable to both

^{22}Na and ^{24}Na .

Aquatic food systems play an important role in the transport of radionuclides to man. In a study made on freshwater ecology in eastern Panama and northwestern Colombia (Templeton et al., 1969), the distribution of freshwater fish and the chemical composition of the aquatic food items were described to predict accumulation of radionuclides based on the logical assumption that the biological behavior of radionuclides will be identical to that of the corresponding stable element. Table B-2 shows the mean sodium concentration factors in representative aquatic organisms.

TABLE B-2

MEAN SODIUM CONCENTRATION FACTORS IN REPRESENTATIVE
SAMPLES FROM PANAMA AND COLOMBIA
(Chemical Analysis Based on Wet Weight)

<u>Sample</u>	<u>Na Concentration Factor</u>
Fish	
Fish muscle	234
Fish bone	598
Fish, remainder	372
Crustaceans	
Shrimp muscle	145
Shrimp, entire	
Rio Sabana	173
Rio Cuadi	109
Crayfish muscle	170
Reptile	
Turtle muscle	202
Cayman-tail muscle	148
Gastropod	
Snail, soft parts	96.5

It may be seen from Table B-2 that the fish bone has the highest sodium concentration factor. Measurements performed in the Hanford area in the U.S.A. (Bustad, 1960) also showed that the ^{24}Na concentration factor in fish bone is the highest of all fish tissues. He found the Hanford fish bone concentration of ^{24}Na to be $2 \times 10^{-3} \mu\text{Ci/g}$ wet weight. ^{24}Na measured in Columbia River water at Richland, Washington (Woodridge, 1969) was found to be in the range of 2600 to 3100 pCi/l (measurements of ^{24}Na in water and fish do not refer to the same year).

Templeton et al. (1969) showed that adult organisms probably never reach 100% equilibration obtained in laboratory experiments for fish and macro-invertebrates may lead to misinterpretation because the low rate of accumulation related to growth has not been taken into account in these experiments. This is also evident from experiments reported by Peterson (1971) in which ^{134}Cs uptake in marine pelagic fish and most marine shellfish was characterized by an equilibrium time of 0.5 to 4 days, whereas for bottom feeding organisms, the equilibrium time was 6 to 17 days. Equilibrium times for freshwater fish living in the lowest salinity water were found to be 30 to 120 days.

Frøke (1967) used a parameter, R , ($\mu\text{Ci}/\text{cm}^3 \text{ yr}$) equal to the product of a derived working limit for sea water ($\mu\text{Ci}/\text{cm}^3$) and the radioactive decay constant (yr^{-1}) to express the limiting replacement value with respect to return of activity of different

radionuclides via various routes in a marine environment.

Table B-3 shows the R values for ^{22}Na and ^{24}Na , as calculated by Freke (1967) for different marine organisms.

TABLE B-3
R VALUES FOR ^{22}Na AND ^{24}Na WITH RESPECT TO RETURN OF ACTIVITY
VIA VARIOUS ROUTES

<u>Radionuclide</u>	<u>R Value (pCi/cm³-yr)</u>				
	<u>Fish</u>	<u>Crustacea</u>	<u>Molluscs</u>	<u>Algae</u>	<u>Silt(external)</u>
^{22}Na	4×10^{-5}	4×10^{-5}	2×10^{-5}	1×10^{-4}	1×10^{-4}
^{24}Na	7×10^{-2}	6×10^{-2}	2×10^{-2}	2×10^{-1}	2×10^{-1}

Dunson (1969) found that uptake of sodium by freshwater turtles is similar to the ion transport in fish. A concentration of about 75 μ -moles/g, most of which is exchangeable, was measured in turtles. This represents a concentration factor of about 300.

Cattle grazing on contaminated vegetation, drinking contaminated water, or breathing contaminated air pose a problem for man because of the food pathway. For short and intermediate-lived radionuclides, Ng and Thompson (1966) considered the forage-cow-milk pathway the worst because the cow may graze large contaminated areas, ingest about 10 Kg forage/day, with a plant retention factor of 0.67, and promptly transfer this radioactivity to man. Blue et al. (1969) have determined the

stable sodium content in different bovine tissues in Panama. Ng et al. (1968) estimated the average concentration of sodium in meat to be 500 ppm and the fraction of sodium reaching the milk to be from about 50% up to 100% of the daily ingestion. Reichle et al. (1970) suggested that the concentration factor of sodium in herbivores and carnivores was found to be 21 as compared to 0.1 for strontium and 0.1 to 0.5 for cesium. The biological half-life of sodium in terrestrial vertebrates and invertebrates is in the range of 1 to 100 days indicating a relatively rapid equilibration.

APPENDIX C
MODELS FOR EVALUATING ENVIRONMENTAL TRANSFER
OF RADIONUCLIDES OF SODIUM

1. Sodium in Man

Sodium content in the adult human body is 105g (0.15% by wt.) Spiers (1968), of which more than one third is contained in the skeleton (on the crystal surface of the bone). The remainder is contained in the body fluids and cells. As approximately half of the bone sodium is readily exchangeable, about 3/4 of the sodium in the human body is available for relatively rapid exchange. Similar data are reported by Bauer (1960). It was found that injected radioactive sodium equilibrates with the exchangeable sodium in the human body in about 12 hours. Excretion occurs afterwards exponentially with a biological half-life of 11 days (Ng et.al., 1968). Fixation of sodium in bone is a much slower process. Experiments show that only about 1% of a single administration of sodium is fixed in the bone during the time it is present in plasma. Veall et.al. (1955) found that, following administration of a single dose of ^{22}Na , the ratio of its concentration in serial blood samples to the fraction retained in the body remained approximately constant over a period of 30 days. Sodium uptake in bone appears to be governed by a power law (Spiers, 1968).

The concentration of ^{24}Na in man, per unit weight, measured in the Western United States and Alaska is higher by orders of magnitude than in most other environmental compartments. Table C-1 shows the f_B values (fraction of ingested isotope deposited in the specific organ) and the concentration C_B of Na in various human body organs (Ng et al., 1968). Table C-2 shows the radiation dose from ingestion of radio-sodium (Wang, 1969). From the standpoint of external radiation caused by ^{24}Na deposited on the ground, it was estimated (Brodsky, 1965) that when ^{24}Na is spread over a wide, smooth area, the activity to produce 1 mR/hr at 3 ft. above ground is 120 Ci ^{24}Na /sq. mile.

2. Models for Biological Transfer and Concentration of a Radionuclide through the Environment

The hazards to man from a release of a radionuclide to the environment depend, in addition to the characteristics of the radionuclide itself, on the modes of transfer and concentration in the different environmental compartments and pathways until it is made available to man. Simplified models are reviewed which take into account the main environmental pathways of radionuclides through the environment.

Analog computers were used initially to simulate the movement of radionuclides in ecological systems (Neel and Olson, 1962) but more recently, mathematical models have been developed. Ng et al. (1966, 1968) used the "biological exchangeable pool" model in their analysis of the environmental transfer of a radionuclide in which transport of a radionuclide is assumed to be identical to that of the corresponding stable element in the pool.

TABLE C-1

FRACTION OF INGESTED SODIUM DEPOSITED (f_B) AND
CONCENTRATION (C_B) OF SODIUM IN HUMAN BODY ORGANS

Organ	f_B	C_B (ppm)
Kidney	4.4×10^{-3}	2.2×10^3
Liver	2.6×10^{-2}	1.4×10^3
Ovaries	1.1×10^{-4}	2.0×10^3
Spleen	2.5×10^{-3}	1.2×10^3
Testes	8.6×10^{-4}	2.1×10^3
Whole Body	1	2.0×10^3
G.I. Tract	10^{-1}	3.1×10^3
Bone		1.06×10^4

TABLE C-2

RADIATION DOSES FROM INGESTION OF
RADIOACTIVE SODIUM

<u>Radionuclide</u>	<u>Solubility</u>	<u>Organ of Reference</u>	<u>Dose (rem) per μCi Ingested</u>
^{22}Na	Soluble	Total Body GI(LLI)	1.9×10^{-2} 3.4×10^{-3}
	Insoluble	GI(LLI)	6.8×10^{-2}
^{24}Na	Soluble	GI(SI) Total Body	6.2×10^{-3} 1.7×10^{-3}
	Insoluble	GI(LLI)	4.8×10^{-2}

Two main modes of transfer were assumed in this model: 1) direct deposition on vegetation (leading to the forage-cow-milk pathway) and 2) deposition on the soil surface (leading to the soil-root-plant pathway). Equations were developed (Ng et al. 1968) to describe the concentration of radionuclides in plants, cattle, meat, and man. For a conservative estimate, instantaneous equilibration was assumed in the agricultural land, vegetation, and human diet. (1)

A coupled compartment model and corresponding transfer equations were developed by Cowser et al. (1967) for a tropical environment. This model included as initial compartments, highland forest foliage, lowland forest foliage, agricultural crop foliage, grasses and herbs, bare surface, and sea and surface water plus additional compartments derived from the above. Data required (Cowser et al., 1967, appendix) for the coupled compartment model include source term (physical and chemical characteristics, etc.); population (dietary and domestic habits); land and water use (drinking water, agriculture, recreation, etc.); geology and geophysics (porosity, permeability, etc.); ground and surface water (composition, flow rate and pattern, etc.); meteorology and climatology (wind and turbulence characteristics, rainfall, etc.); soil (type, moisture, composition, etc.); vegetation (productivity, surface coverage, etc.); animals (food values, productivity, etc.);

An aquatic transfer model was developed by Tamplin et al. (1968) and by Pratt (1968) which also assumed instantaneous

equilibration between the contaminant and aquatic ecosystem. Transfer based on a human diet totally or partially of aquatic origin was evaluated.

Werner (1969) summarizes a number of hydrologic models of radionuclide transport including the following cycles: precipitation, infiltration, surface runoff, percolation, ground water flow, evaporation, transpiration and penetration into springs, lakes, rivers, and seas. Uptake in the vegetation compartment either from soil moisture or ground water, and plant storage were included in some of the models.

A simplified mathematical model for predicting the transport of radionuclides in a marine environment was developed by Bloom and Raines (1971). This model included both a minimized physical dilution mechanism and a biological transport process.

An ecological model based on a concept called "Mass Extraction Rate" was developed by Fleming (1969) for the estimation of the environmental contamination following a nuclear explosion. This model makes possible a calculation of the mass of earth material from which a radionuclide must be extracted by natural processes to yield the daily permissible dose from ingestion.

Martin et al. (1969) developed mathematical models which describe the transport of radionuclides in the environment, including the various biogeochemical cycles. The main ecological systems (e.g., forest, agricultural land, food chains) by which a radionuclide may reach man were included in these models.

They also presented a generalized diagram describing the major ecological pathways and their interrelationships, as well as a simple crop model diagram. The forest cycle compartments are leaves, litter, soil, roots, stems, fruits, herbivores, and carnivores. The agricultural cycle components are leaves, fruits, stems, soil and roots, forage, herbivores and food products, streams, fish and stream-suspended and bottom material. A description of the marine ecosystem by Martin et al. includes sea water, phytoplankton, zooplankton, carnivores, omnivores, shell fish, and sea-suspended and bottom material. In both terrestrial and marine ecosystems, transfer of a radionuclide from one compartment to another is assumed to be governed mainly by movement of water and organic material. Because of the complexity of the model, simpler models were considered (e.g., a crop model) in which different compartments were coupled. For instance, the eight-compartment model includes: fallout, soil, plant leaves, plant edibles, surface water, fish, animals, and man. A ten compartment model was used by Bloom et al. (1970) in which surface water was subdivided into fresh and sea, and fish into fresh and marine, and sets of equations were developed to describe the transport of a radionuclide through each of the different compartments (Martin et al., 1969; Levin et al., 1970; Bloom et al., 1970).

An important parameter required in these models is the transfer coefficient of an element from one compartment to the other. Some transfer coefficients for sodium have been estimated but for a large number of pathways they are lacking.

One of the few studies which present general data on the ecological transfer of elements through the different environmental pathways is an investigation of elemental and hydrologic budgets performed at the Panamanian tropical moist forest by McGinnis and Golley (1969). They found that the elemental movement is lower than that of water by up to several orders of magnitude. The soil compartment is characterized by the longest turnover time (40.2 years) and is the most effective ecological sink for radionuclides. The fruits and flowers compartment has a relatively long turnover time (180 days) and it appears that forest fruits may be practically free of short-lived radionuclides. The animal food compartment is characterized as the most rapid terrestrial pathway to man. Other studies have been made by Levin et al. (1970) and by Perkins et al. (1965) but no attempt has been made to correlate distribution of ^{22}Na in ecological pathways with the concentration of ^{22}Na measured in air or deposited on the ground to make an estimation of the transfer coefficients of ^{22}Na through the various food chains to man. However, it may be assumed that a mean concentration of ^{22}Na in air of 0.01 dis/min per thousand cu. ft. (measured in Richland, Washington, by Perkins, et al., 1965) is representative of the concentration in the air in the Western U.S. area, where ^{22}Na was determined for the different food chains. It may also be assumed that about 0.005 dis/min, per thousand cu. ft. of air is representative of the ^{22}Na concentration in Alaska (Perkins, 1971). On this basis, some of the required transfer coefficients

and concentration factors of sodium through the different environmental compartments and food chains may be estimated.

Another simplified system analysis model for the terrestrial movement of radionuclides is based on equations which include the transfer coefficients from one compartment to another (Booth and Kaye, 1971). Derivation of transfer coefficients is also described in this study. The model is intended to predict the cumulative intake of a radionuclide by man through consumption of contaminated milk, beef, and plants. It includes as compartments fallout, food crops, pasture grass, soil, beef, milk, and man. In this model, the radionuclide is assumed to be transferred from grass directly to milk and meat. In a sample system analysis on various radionuclides exemplifying their model, Booth and Kaye (1971) used the same transfer coefficient for a number of radioelements because of the unavailability of more relevant data and also because some of the transfer coefficients have a negligible effect on the cumulative intake of a radionuclide by man. For instance, a 14-day environmental half-time was assumed for the transfer from grass to soil and from soil surface to food crop for all radionuclides. The following conclusions were drawn from Booth and Kaye's analysis: in the first few days after deposition of the radionuclide, milk and directly contaminated vegetation will be the main pathways. After about 30 days, the intake via beef will be about 2.5 times larger than from directly contaminated vegetation; after 500 days, the intake from vegetation, contaminated via soil, will be the most important pathway of contamination.

Additional mechanisms contributing to human radiation dose after release of radionuclides into the environment include external radiation from immersion in contaminated air or water; external radiation from standing on contaminated ground; internal radiation following inhalation of contaminated air; and internal radiation following ingestion of contaminated water. Most of the models developed for estimating radiation doses resulting from these mechanisms were developed for the Flowshare Atlantic-Pacific interoceanic canal project (Cowser et al., 1967). Cowser's et al. (1967) models included external submersion exposure to beta and gamma radiation from contaminated surfaces and internal radiation to different organs of the body, including the age dependence effect. The specific activity concept was applied to determine the concentration of a radionuclide which could be tolerated in the biological environment. The model takes into consideration radioactive decay but does not include atmospheric or aquatic dispersion and dilution. Later, this model was elaborated into EXREM (external radiation doses) and INREM (internal radiation doses) computer codes to evaluate the dose rates and cumulative doses for several hypothetical releases of radionuclides including the contribution of nuclide chains to the dose commitment (Turner et al., 1968; Turner, 1969). Sensitivity analysis is proposed in the generalized equations of INREM and EXREM models and computer codes (Kaye et al., 1969) (Kaye and Rohwer, 1969) as a method for identifying critical pathways and critical population groups.

In a series of studies performed at Lawrence Radiation Laboratories and summarized by Shore et al. (1969), the biological availability of radionuclides in different ecological pathways was analyzed and the contribution of individual radionuclides to the total internal radiation dose and the dose to different human organs was estimated.

A unit-rad deposition factor, F_1 ($\mu\text{Ci}/\text{m}^2/\text{rad}$) was defined by Ng and Thompson (1966) as the minimum ground deposition of a radionuclide that could result in a 30 year internal dose of 1 rad to a given tissue of man assuming either one of the two main modes of intake, i.e., the forage-cow-milk pathway or the soil-plant-root pathway. An equation for the unit-rad deposition factor was developed and F_1 was estimated for various radionuclides. For ^{24}Na , the value of F_1 to the whole body of a child of 10 kg weight via milk, (assuming a daily consumption of 1 liter milk) was estimated to be $24 \mu\text{Ci}/\text{m}^2/\text{rad}$ and the 30-year dose from ^{24}Na having a half life of 15.1 hours would be delivered in a few days.

The (meter)²-rad unit used by Ng and Thompson (1966) is defined as the quotient of the activity released (μCi) and the unit-rad deposition factor, F_1 ($\mu\text{Ci}/\text{m}^2/\text{rad}$). They concluded that the m^2 -rad values via soil are lower than those via milk by two or more orders of magnitude and that the total internal dose to the whole body via milk would exceed considerably that via soil-plant-root. They also indicated that the soil-root

pathways can be significant only for relatively long-lived radionuclides which can equilibrate within the biological exchangeable pool before their disappearance by radioactive decay. Table C-3 (compiled from Ng et al., 1968) summarizes these conclusions with respect to ^{22}Na and ^{24}Na .

Referring to water contamination, a unit-rad contamination factor, F_A was defined by Ng et al. (1968) as the initial concentration of a radionuclide in water ($\mu\text{Ci}/\text{m}^3$) that could result in a 30 year integrated dose of 1 rad to a certain tissue of the human body. Table C-4 shows F_A values for ^{22}Na and ^{24}Na as compiled from Ng et al. (1968).

Air and water concentrations of radionuclides capable of producing an equilibrium dose rate of 1 rad/yr were estimated by Pratt (1968). For the aquatic system, it was assumed that the human diet is completely aquatic in origin. Table C-5 shows the unit dose rate air concentration and the corresponding half time to equilibrium, via soil and milk. Table C-6 shows the unit dose-rate water concentration via sea and fresh water and the corresponding half time to equilibrium for ^{22}Na and ^{24}Na .

TABLE C-3

UNIT-RAD-DEPOSITION FACTORS (F_1)
FOR ^{22}Na AND ^{24}Na

Unit-Rad-Deposition Factor (F_1 -in $\mu\text{Ci}/\text{m}^2/\text{rad}$)

Organ	Via Milk				Via Soil-Root (Typical Agricultural Soil)	
	ADULT		CHILD		ADULT	CHILD
	^{22}Na	^{24}Na	^{22}Na	^{24}Na	^{22}Na	^{22}Na
Bone	0.82	170	0.12	24	1.5	1.4
Kidneys	0.82	170	0.16	32	7.0	6.9
Liver	0.81	160	0.13	26	11	11
Ovaries	0.85	170	0.08	16	7.7	7.6
Spleen	0.82	170	0.13	26	13	13
Testes	0.82	170	0.019	3.8	7.4	7.3
Whole Body	0.82	170	0.12	24	7.7	7.6
GI Tract	0.85	32	0.12	4.5	4.9	4.9

TABLE C-4
UNIT RAD CONTAMINATION FACTORS (F_A)
FOR ^{22}Na AND ^{24}Na

Unit-Rad-Contamination Factor (F_A in $\mu\text{Ci}/\text{m}^3/\text{rad}$)

<u>Via Seawater</u>				
<u>Organ</u>	ADULT		CHILD	
	<u>^{22}Na</u>	<u>^{24}Na</u>	<u>^{22}Na</u>	<u>^{24}Na</u>
Bone	6.1	8.0×10^4	6	4.3×10^3
Kidneys	29	3.9×10^5	29	2.1×10^4
Liver	46	6.1×10^5	45	3.3×10^4
Ovaries	32	4.3×10^5	32	2.3×10^4
Spleen	54	7.1×10^5	53	3.8×10^4
Testes	31	4.0×10^5	30	2.2×10^4
Whole Body	32	4.3×10^5	32	2.3×10^4
GI Tract	21	5.0×10^4	21	1.5×10^4

<u>Via Freshwater</u>				
<u>Organ</u>	ADULT		CHILD	
	<u>^{22}Na</u>	<u>^{24}Na</u>	<u>^{22}Na</u>	<u>^{24}Na</u>
Bone	0.0036	48	0.0036	8.6
Kidneys	0.018	230	0.017	12
Liver	0.028	360	0.027	20
Ovaries	0.019	260	0.019	14
Spleen	0.032	430	0.032	23
Testes	0.018	240	0.018	13
Whole Body	0.019	260	0.019	14
GI Tract	0.012	30	0.012	8.8

TABLE C-5

UNIT-DOSE-RATE AIR CONCENTRATIONS AND EQUILIBRIUM

HALF-TIMES FOR ^{22}Na and ^{24}Na

<u>Organ</u>	<u>Air Concentration ($\mu\text{Ci}/\text{m}^3$)</u>			
	<u>per rem/yr</u>			
	ADULT		CHILD	
	<u>^{22}Na</u>	<u>^{24}Na</u>	<u>^{22}Na</u>	<u>^{24}Na</u>
Bone (via soil)	3.3×10^{-6}	4.4×10^{-2}	3.3×10^{-6}	8.3×10^{-3}
Whole Body (via soil)	1.8×10^{-5}	2.3×10^{-1}	1.8×10^{-5}	4.4×10^{-2}
Bone (via milk)	1.9×10^{-6}	3.8×10^{-4}	2.7×10^{-7}	5.5×10^{-5}
Whole Body (via Milk)	1.9×10^{-6}	3.8×10^{-4}	2.7×10^{-7}	5.4×10^{-5}

<u>Organ</u>	<u>Half-Time to Equilibrium</u>			
	<u>(days)</u>			
	ADULT		CHILD	
	<u>^{22}Na</u>	<u>^{24}Na</u>	<u>^{22}Na</u>	<u>^{24}Na</u>
Bone (via soil)	1,100	80.7	940	0.62
Whole Body (via soil)	1,100	80.7	940	0.62
Bone (via milk)	32.5	1.5	13.8	0.6
Whole Body (via milk)	32.5	1.5	13.8	0.6

TABLE C-6

UNIT-DOSE-RATE WATER CONCENTRATION AND EQUILIBRIUM
HALF-TIMES FOR ^{22}Na AND ^{24}Na

Organ	<u>Water Concentration ($\mu\text{Ci}/\text{m}^3$)</u> <u>per rem/yr</u>				<u>Half-Time to Equilibrium</u> <u>(days)</u>			
	ADULT		CHILD		ADULT		CHILD	
	<u>^{22}Na</u>	<u>^{24}Na</u>	<u>^{22}Na</u>	<u>^{24}Na</u>	<u>^{22}Na</u>	<u>^{24}Na</u>	<u>^{22}Na</u>	<u>^{24}Na</u>
Bone (via seawater)	22.8	200	22.6	37.8	10.9	0.6	1.6	0.45
Whole Body (via seawater)	121	1,060	120	200	10.9	0.6	1.6	0.45
Bone (via fresh-water)	0.014	0.12	0.014	0.023	10.9	0.6	1.6	0.45
Whole Body (via fresh-water)	0.072	0.64	0.072	0.12	10.9	0.6	1.6	0.45

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