Contamination of Surface-Water Bodies after Reactor Accidents by the Erosion of Atmospherically Deposited Radionuclides[1]

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Abstract

Reactor safety analyses usually do not consider the population risk which might result from the contamination of surface-water bodies after reactor accidents by the erosion of atmospherically deposited radionuclides. This paper is intended to provide perspective on the reasonableness of this omission. Data are presented which are suggestive of the rates at which atmospherically deposited radionuclides might erode into surface-water bodies. These rates are used in the calculation of potential health effects resulting from surface-water contamination due to such erosion. These health effects are compared with predicted health effects due to atmospheric and terrestrial pathways after reactor accidents. The presented results support the belief that the contamination of surface-water bodies after reactor accidents by the erosion of atmospherically deposited radionuclides is not a major contributor to the risk associated with such accidents.

1. Introduction. As indicated in several recent reports, reactor safety analyses usually do not consider the population risk which might result from the contamination of surface-water bodies after reactor accidents by the erosion of atmospherically deposited radionuclides (e.g., A182a, US82, US83). The following presentation is intended to provide perspective on the reasonableness of this omission. To this end, the paper has three primary purposes. The first is to present data which are suggestive of the rates at which atmospherically deposited radionuclides might erode into surface-water bodies. The second is to present calculations which are suggestive of the health effects which might result from such erosion. The third is to provide a comparison of the health effects associated with the erosion of atmospherically deposited radionuclides into surface-water bodies with the health effects associated with the atmospheric and terrestrial pathways after a reactor accident. The presented results support the belief that the contamination of surface-water bodies after reactor accidents by the erosion of atmospherically deposited radionuclides is not an important contributor to the risk associated with reactor accidents.

Section 2 presents models and data which have been developed to relate radionuclide concentrations in surface water to radionuclide concentrations on land due to fallout from nuclear weapons tests. It is suggested that these models and data can be used to infer the behavior of radionuclides

released in a reactor accident. Section 3 describes how the models and data presented in Section 2 can be used to predict health effects from the contamination of surface-water bodies due to the erosion of atmospherically deposited radionuclides. Then, Section 4 uses the computational procedures described in Section 3 to calculate health effects associated with reactor accidents at three different sites. Finally, Section 5 contains a comparison of the results calculated in Section 4 with results calculated for atmospheric and terrestrial pathways after reactor accidents.

2. Radioactive Fallout Data. Past testing of nuclear weapons has injected a large inventory of radionuclides into the atmosphere and thence into the terrestrial environment. The manner in which these fallout radionuclides erode from land to surface-water bodies provides a potential source of information with respect to the behavior of atmospherically deposited radionuclides after reactor accidents. A large effort has been devoted to gathering data on radioactive fallout and its relation to radionuclide levels in the environment (e.g., Ru69, Un77, Aa79). In particular, it has been found to be possible to relate fallout rates and accumulated fallout levels to radionuclide concentrations in various environmental components by relatively simple empirical relationships. The simplest of these relations is

$$C_i = a\, D_i + b\, A_{i-1(n)} \,, \qquad (2.1)$$

where C_i is the radionuclide concentration in the environmental component of interest in year i (units: Ci/l or Ci/kg,

as appropriate), D_i is the fallout rate of the radionuclide for year i (units: $Ci/km^2/yr$), $A_{i-1(n)}$ is the accumulated fallout from year 1 to year i-1 with an assumed effective half-life of n years (units: Ci/km^2), and each of a (units: km^2 yr/l or km^2 yr/kg) and b (units: km^2/l or km^2/kg) is a coefficient determined by regression analysis. The effective half-life n is at most the radioactive half-life of the radionuclide but can be less as it may also account for radionuclide loss by mechanisms other than decay (e.g., irreversible binding to soil materials).

More complex relations than (2.1) are also used. For example,

$$C_i = a\,D_i + b\,D_{i-1} + c\,A_{i-2(m)} + d\,A_{i-2(n)}, \qquad (2.2)$$

where C_i is the radionuclide concentration in the environmental component of interest in year i (units: Ci/l or Ci/kg, as appropriate), D_i and D_{i-1} are the fallout rates for years i and i-1 (units: $Ci/km^2/yr$), $A_{i-2(m)}$ and $A_{i-2(n)}$ are accumulated fallout from year 1 to year i-2 with assumed effective half-lives of m and n years (units: Ci/km^2), and each of a (units: km^2 yr/l or km^2 yr/kg), b (units: km^2 yr/l or km^2 yr/kg), c (units: km^2/l or km^2/kg), and d (units: km^2/l or km^2/kg) is a coefficient determined by regression analysis. Sometimes months and various other time increments are used in relations of the form of those in (2.1) and (2.2). Additional developments based on regression analysis (e.g., Ri77, Ly78) and other modeling approaches (e.g., Huf70, Hub80, He82) are also possible.

The particular environmental component of interest in this analysis is surface water. Thus, for our purposes, C_i is concentration in surface water. There are numerous studies which provide information on the coefficients a and b in (2.1) for water draining from regions varying in size from large watersheds to small experimental plots. For this reason, the following investigation will use the model presented in (2.1) rather than one of the more complicated models indicated in the preceding paragraph for which less data are available. For a given region, the coefficients a and b depend on the size of the region, the amount of runoff from the region, and the fractions of recently deposited and accumulated radionuclide removed in runoff. In particular, a and b are given by

$$a = L\,\lambda_a/R \text{ and } b = L\,\lambda_b/R, \tag{2.3}$$

where L is the area of the watershed (units: km^2), R is the annual runoff rate (units: 1/yr), λ_a is the fraction of recently deposited radionuclide removed by runoff (units: dimensionless) and λ_b is the rate constant for the removal of accumulated radionuclide (units: yr^{-1}). Then, 100 λ_a and 100 λ_b are the percent of recently deposited radionuclide removed in runoff and the annual percentage of accumulated radionuclide removed in runoff, respectively. The time periods for which λ_a has been calculated vary but are typically less than or equal to a year. Due to the varying conditions under which radionuclide washoff data have been recorded, it is more informative to present such information in terms of λ_a and λ_b than in terms of the regression coefficients a and b.

Tables 1 and 2 present values for λ_a and λ_b that have been determined in field investigations. The values in Table 1 are for larger regions while the values in Table 2 are for experimental plots. There is a tendency for higher washoff rates from the smaller experimental plots. The washoff rates from the larger regions are the ones most relevant to this analysis. The fact that initial washoff rates are generally higher than subsequent washoff rates is probably due to time-dependent processes such as the movement of radionuclides into the soil column and the fixation of radionuclides to soil materials. The higher washoff rates for small plots probably result from factors such as short travel paths to collection points and use of experimental plots with little or no plant cover.

Other substances which have been dispersed in the environment can also be used in the estimation of washoff rates. Two such substances are pesticides and herbicides. These are often spread on the surfaces of plants and fields, and considerable data exist on their washoff rates (e.g., Pio73, Caro76, Wau78, Leo79, We80). The pattern of data here is similar to that for fallout. The overall washoff rates tend to be small with the initial washoff rates greater than subsequent washoff rates. As another example, Andren et al. (An75) studied a watershed in Tennessee and found that 2-3% of the annual atmospheric lead input is transferred out of the watershed by streamflow.

3. Modeling Approach. For the prediction of radionuclide washoff from land surfaces to surface-water bodies after reactor accidents, it is assumed that radionuclides behave in a manner

consistent with the relationship in (2.1) and the data in Table 1. There exist various reviews on both the properties of radioactive fallout (e.g., Ad60, Bj63, Cr66, No71, Pa73) and the properties of fission products released in reactor accidents (e.g., Nu79, Ca81, Mo81, US81). There also exist a number of reviews on the behavior of fallout radionuclides in soils (e.g., Sc65, Po73, Pr73). However, the authors are unaware of any reviews which directly compare the properties of radioactive fallout from weapons tests with the chemical and physical properties of fission products potentially released in reactor accidents. This is unfortunate as the behavior of fallout radionuclides is often used to infer the behavior of radionuclide releases associated with such accidents.

In the present study, the practice of using fallout radionuclides as analogues for the same radionuclides released in reactor accidents is continued. If a given element is either soluble in both types of release, forming the same aqueous species, or is insoluble in both, due to the physical nature of the deposited particles or to rapid sorption after deposition, then the behavior of that element in fallout probably provides a good analogy for the behavior of that element in a reactor accident release. The thermodynamically most stable forms of both Sr and Cs released in a reactor accident will probably be soluble. Such stable oxides and hydroxides are indicated in several studies of reactor accident source terms (e.g., Mo81; US75, Table VI 8-2). Also, thermochemical equilibrium calculations indicate that the soluble strontium hydroxides reported

by Francis (Fra78) and by Huff and Kruger (Huf67) for weapons fallout are the most stable forms of strontium, in preference to similarly soluble oxides. The observed behavior of Cs in fallout is consistent with initially highly soluble solid phases, speciating to Cs^+ which rapidly undergoes ion exchange and is sorbed onto soil materials. Thus, since the behavior of Sr and Cs appears to be controlled by rapidly formed, thermodynamically stable phases and species independent of the origin of the material, the use of fallout as an analogue for these elements after reactor accidents seems justified.

Since Sr and Cs tend to form soluble phases in aqueous environments near the earth's surface, any differences in the actual solubility of the forms released in a reactor accident will probably have minor effect on the general appropriateness of the analogy made. Even if some differences occur, the removal rates assumed for the following calculations should be acceptable for use in a preliminary assessment of the effects associated with radionuclide washoff after a reactor accident. This statement is made because the following pattern of behavior seems to occur for a wide range of materials deposited on surfaces in the environment (e.g., see the data on different radionuclides in Table 1 and the previously cited data on herbicides and pesticides): a few percent or less of the deposited material washes off over a relatively short time period after deposition and then a smaller percentage of the remaining material is removed each year thereafter.

For the following, it is assumed that an initial fraction λ_a (units: dimensionless) of a deposited radionuclide washes off in a relatively short time period after an accident and that thereafter the radionuclide washes off with a rate constant λ_b (units: yr^{-1}). This assumption is that described by (2.1) with the effective half-life conservatively taken to be the same as the radioactive half-life. If x_0 (units: Ci) denotes the total amount of a radionuclide with a radioactive decay constant λ(units: yr^{-1}) initially released to a land surface and x(t) denotes the amount present on the land surface at time t, then the change of x(t) subsequent to the initial washoff is described by the differential equation

$$dx/dt = -(\lambda + \lambda_b)\, x, \quad x(0) = (1 - \lambda_a)\, x_0 . \tag{3.1}$$

which has the solution

$$x(t) = (1 - \lambda_a)\, x_0 \exp\,[-(\lambda + \lambda_b)t] . \tag{3.2}$$

Hence, the total amount W (units: Ci) of the radionuclide washing into the surface-water body is given by

$$\begin{aligned} W &= \lambda_a x_0 + \int_0^\infty \lambda_b\, x(t)\, dt \\ &= \lambda_a x_0 + (1 - \lambda_a)\, \lambda_b\, x_0/(\lambda + \lambda_b) \\ &= (\lambda_a\lambda + \lambda_b)\, x_0\, /(\lambda + \lambda_b) . \end{aligned} \tag{3.3}$$

For subsequent calculations to examine the effects of radionuclide washoff, the preceding relation will be used to estimate the total amount of a radionuclide washing off of a land surface and into a surface-water body. The assumption that

λ_b remains constant rather than decreasing with time is probably conservative.

The determination of potential health effects which might result from radionuclide washoff after reactor accidents to surface-water bodies is now considered. Three pathways are included: drinking water, fish and irrigated food. The models used are similar to those presented in NUREG 1.109 (US77); a more detailed development of the use of these models to determine population exposure and risk from long-term radionuclide releases to the surface environment is available elsewhere (He83).

Exposure from drinking water is considered first. Suppose W_i (units: Ci) represents the amount of a radionuclide washing off a land surface during the i^{th} year after an accident and into a river system with an average annual discharge D (units: l). Then, the average radionuclide concentration in the river for the ith year can be approximated by W_i/D, and the future population health effects HE_{wi} (units: lcf*) from water ingestion during the ith year after the accident are given by

$$HE_{wi} = (W_i/D)\ WT\ WC\ POP\ RF\ , \qquad (3.4)$$

where WT is a water treatment factor (units: dimensionless), WC is annual individual water consumption (units: l/ind**), POP is the size of the population receiving drinking water from the contaminated system (units: ind), and RF is a risk factor (units:

* latent cancer fatalities
**individual

lcf/Ci). Thus, if total washoff W (units: Ci) and total population health effects HE_w (units: lcf) are given by

$$W = \Sigma_i W_i \quad \text{and} \quad HE_w = \Sigma_i HE_{wi} \quad , \tag{3.5}$$

respectively, then

$$HE_w = (W/D)\ WT\ WC\ POP\ RF \quad . \tag{3.6}$$

In a similar manner, population health effects HE_f (units: lcf) from fish consumption can be approximated by

$$HE_f = (W/D)\ CR\ FP\ RF \quad , \tag{3.7}$$

where CR is a concentration ratio from water to fish (units: l/kg) and FP is the annual production of edible fish products from the water body (units: kg). Also, the population health effects HE_p from the direct contamination of plant material for human consumption by sprinkler irrigation can be approximated by

$$HE_p = (W/D)\ IR\ FR\ (WHL/LN(2.))\ RF, \tag{3.8}$$

where IR is the annual rate at which water used in the sprinkler irrigation of plant material for direct human consumption (units: l/yr), FR is the fraction of radionuclide in sprinkler irrigation water which is retained on edible plant material (units: dimensionless), WHL is the weathering half- life for radionuclides retained on edible plant materials (units: yr), and LN(2.) is the natural logarithm of 2. The preceding calculation ignores potential exposure due to root uptake. However, it is felt that this omission is unlikely to seriously

affect the overall predicted consequences (Ru67).

For each radionuclide, the risk factor RF (units: lcf/Ci) is based on the data and methodology used in the WASH-1400 analysis (US75). Each risk factor is actually a sum of the form

$$RF = \Sigma_j \Sigma_k RF_{jk} DF_{jk} EF_j \quad , \tag{3.9}$$

where RF_{jk} is the risk factor (units: lcf/rem) for the j^{th} cancer and k^{th} time period after exposure, DF_{jk} is the ingestion dose factor (units: rem/Ci) for the organ associated with the j^{th} cancer and the k^{th} time period after exposure and EF_j is the effectiveness factor (units: dimensionless) associated with the j^{th} cancer. For this analysis, the following cancers are considered: leukemia, lung, gastrointestinal, pancreatic, breast, bone and "other". The values for RF_{jk} and DF_{jk} are derived from Tables VI 9-5 and VI 8-4, respectively, of WASH-1400 (US75). The central estimate risk model described in WASH-1400 is used. In WASH-1400, whole-body dose factors are used for pancreatic, breast and "other" cancers; this same pairing of factors is employed in the CRAC2 computer model (Ri83) for the calculation of reactor accident consequences. As exposures are at very low levels, an effectiveness factor of .2 (US75, Table VI 9-7) is applied in this study for all cancers except breast cancer (US75, p. 9-25). Half-lives and risk factors for selected radionuclides are given in Table 3; these are the radionuclides considered in the WASH-1400 analysis for long-term ingestion exposure (US75, Sections 8.3.1.3 and E3.2). The risk factors are calculated as indicated in this paragraph. The authors are aware of the

uncertainties associated with the prediction of health effects due to very low exposures; however, the approach outlined in this paragraph was selected so that health effects would be calculated in the same manner as in the WASH-1400 analysis.

4. Potential Consequences. Consequences are now estimated for the potential contamination of surface-water bodies after reactor accidents by the erosion of atmospherically deposited radionuclides. An example involving direct deposition on a large lake is also given. Accidents involving reactors located near the following areas are considered: the middle and lower Mississippi valley, Lake Michigan, and the Rhine-Meuse valley. These locations were selected due to the convenience of using data in their analyses which were already compiled for use in other studies. Due to their potentially large release quantities, relatively long radiological halflives, and recognized radiotoxicity, effects associated with the following radionuclides are considered: ^{89}Sr, ^{90}Sr, ^{134}Cs and ^{137}Cs. As already indicated, these are the radionuclides considered in the WASH-1400 analysis for long-term ingestion exposure. The preceding radionuclides are assumed to be released due to an SST1 accident (i.e., a core melt with loss of all installed safety features followed by a large overpressure failure of the containment building (A182b)). The radionuclide releases assumed to take place in association with such an accident are presented in Table 4. For each example, the entire release is deposited on the watershed or waterbody under consideration. As the models used to predict health effects are linear with

respect to the size of this deposition, the presented results can be scaled to represent the effects of a fractional deposition.

Mississippi Valley. The release involving the middle and lower Mississippi valley is considered first. This region was selected due to the convenience of using the data compiled on the area by Niemczyk et al. (Nie81). Of the studies on radionuclide washoff compiled in Table 1, the one by Menzel (Me74) is selected for guidance with respect to the present analysis. This study estimates λ_a and λ_b for various regions of the United States, of which the Southern Plains is the region which contains the lower Mississippi valley. For this region, the estimated values of λ_a and λ_b for ^{90}Sr are

$$\lambda_a = 1.79E\text{-}2 \text{ and } \lambda_b = 0.51E\text{-}2 \text{ yr}^{-1} \quad (4.1)$$

(Me74, Table 2). These values will be used for ^{89}Sr and ^{90}Sr. Adjoining regions have values which are both larger and smaller but the overall variation is not large. There is not as much information available on ^{137}Cs as on ^{90}Sr. Examination of Tables 1 and 2 suggests that a smaller percentage of deposited ^{137}Cs than of deposited ^{90}Sr washes into surface-water bodies. Aarkrog (Aa79) indicates that the washoff rate for ^{137}Cs is perhaps 20% that for ^{90}Sr. Further, measurements involving New York City tap water (Hea76) indicate that the ratio of ^{137}Cs to ^{90}Sr is about .1 even though there is about 50% more ^{137}Cs than ^{90}Sr in fallout and their half-lives are approximately equal. However, for the following calculations, it is conservatively assumed that the wash-off rates for Cs and Sr

are the same. Thus, the values for λ_a and λ_b in (4.1) are also used for ^{134}Cs ^{137}Cs. If desired, the results contained in this paper can be easily scaled to indicate the effects of other wash-off rates. As indicated in (3.3), the values for λ_a and λ_b can be used to estimate the fraction of a radionuclide release eventually transported into the river. The results of this calculation are given in Table 5.

The calculated health effects for an SST1 accident due to radionuclide washoff into the middle and lower Mississippi river are presented in Table 5. The value for D is taken to be 5.7E14 1 (Nie81, Table B7). Further, for the other values in (3.6) for water consumption, WT = .87 for Sr and WT = .53 for Cs (Nie81, Table D2.1), WC = 370 L/yr (US77, Table D-1), and POP = 2.9E6 ind (Nie81, Table E7). For the additional values in (3.7) for fish consumption, CR = 30 1/kg for Sr and 2000 1/kg for Cs (US77, Table A-8), and FP = 2.0E6 kg/yr from an assumed catch of 6.5E6 kg/yr (Nie81, Table E35) and an edible fraction of .31 (Nie81, p. 229). Finally, for the additional values in (3.8) for the direct contamination of plant material for human consumption by sprinkler irrigation, IR = 8.1E8 1/yr (Nie81, Table E42 with the conservative assumption that all plant material raised with sprinkler irrigation is used for human consumption), FR = .2 (Bo81), and WHL = .038 yr (Bo81).

Lake Michigan. As a second example, radionuclide washoff around Lake Michigan is considered. This situation is approached by first determining the effects of an SST1 accident with release directly to Lake Michigan. Again, the following

radionuclides are considered: ^{89}Sr, ^{90}Sr, ^{134}Cs and ^{137}Cs. The rate of radionuclide removal from the lake will be based on empirical rate constants derived from radioactive fallout data. The radionuclide ^{90}Sr seems to be little removed by incorporation into lake sediments; for example, Lerman and Tanaguichi (Ler72a, Ler72b) estimate that the annual loss of ^{90}Sr from Lake Michigan by outflow and incorporation into sediments to be near 2%. Klein (Kle75) also estimates a small loss for Sr due to sedimentation. In this analysis, it is assumed for both ^{89}Sr and ^{90}Sr that the annual loss from Lake Michigan is that due to radioactive decay plus a 2% loss due to outflow and incorporation into sediments. The physical removal rate for ^{137}Cs appears to be much higher. Wahlgren et al.(Wah74, Wah75a, Wah75b) estimate the effective half-life of ^{137}Cs in Lake Michigan to be on the order of 3 to 4 years and suggest that it may be shorter (i.e., 1 year). Similar estimates are also given by Klein (Kle75) and Lerman and Taniguchi (Ler72b). In a study of Lake Huron, Barry (Bar73) estimated an effective half-life of .97 years for ^{137}Cs. For this study, an effective half-life of 3.5 years is assumed for ^{137}Cs in Lake Michigan. For ^{134}Cs, it is assumed that loss from Lake Michigan is due to radioactive decay plus a physical removal with the same effective half-life of 3.5 years that was assumed for ^{137}Cs. The resulting effective half-lives for the four radionuclides under consideration are summarized in Table 6.

For the following, it is assumed that Lake Michigan can be treated as a single uniformly-mixed cell with the already indicated effective half-lives for ^{89}Sr, ^{90}Sr, ^{134}Cs and ^{137}Cs. Thus, with the assumption that the lake has a volume of 4.87E15 l (Nie81, Table B4), it follows that the resultant concentration c(t) (units: Ci/l) in the lake at a time t after an initial release of size x_0 is

$$c(t) = x_0 \, e^{-\lambda t}/4.87E15, \qquad (4.2)$$

where λ = LN(2.)/HL and HL is the effective half-life of the particular radionuclide under consideration. In turn, the integrated concentration (units: Ci yr/l) is given by $\int_0^\infty c(t)dt$ and can be used in the calculation of total health effects. For an SST1 accident with complete deposition on Lake Michigan, the resultant concentration integrals are given in Table 6. The initial releases (i.e., the x_0) are given in Table 4.

The integrated concentrations can be used in conjunction with (3.6), (3.7) and (3.8) to predict the consequences of an SST1 accident with complete deposition of the released radio- activity onto Lake Michigan. This event might occur if the contents of the plume were washed out over the lake by a rainstorm. In such a calculation, the integrated concentration is used instead of the ratio W/D. (Note: For the units to come out correctly in this computation, the water consumption represented by WC should be treated as a rate with the units of 1/yr.) The results of this calculation are given in Table 6. Except as indicated in

the following, the parameter values used in the generation of Table 6 are the same as those used in the generation of Table 5. For (3.6), WT = 1. is conservatively assumed for all radionuclides and POP = 1.1E7 i d (Nie81, Table E3). For (3.7), FP = 5.1E6 kg/yr (Nie81, Table E16). For (3.8), IR = 6.8E9 L, which is obtained from a withdrawal of 7.E9 L/yr from Lake Michigan for irrigation (Nie81, Table E45) and an application of 97% of this withdrawal by sprinkler irrigation (Nie81, Table E42). It is conservatively assumed that all sprinkler-irrigated foodstuffs are used for human consumption (Nie81, Table E42).

The results presented in Table 6 are for Lake Michigan only. For at least ^{90}Sr, the possibility exists for significant population exposures in the lower Great Lakes. However, inclusion of these lower lakes would probably not increase the results in Table 6 for ^{90}Sr by much more than a factor of 2. This conclusion was arrived at by using the integrated concentrations for the lower lakes following a release to Lake Michigan (Nie81, Table B17) and appropriate usage parameters for these lakes (Nie81, Tables E3, E16 and E45). For ^{137}Cs, Lake Michigan is almost a closed system (Wah75b) and so exposure in the lower lakes following a release to Lake Michigan would be small relative to the exposure in Lake Michigan itself. Due to their short half-lives, Lake Michigan is in effect a closed system for ^{89}Sr and ^{134}Cs.

The effects of washoff into Lake Michigan are now considered. If the data for the north central United States in Menzel's study (Me74, Table 2) are used as a guide, then 2.02%

of recently deposited ^{89}Sr and ^{90}Sr is removed, and thereafter, .63% per year is removed. As indicated in (3.3), this results in an eventual removal to Lake Michigan of 2.1% of the deposited ^{89}Sr and 21.9% of the deposited ^{90}Sr. As before, the removal rates for ^{134}Cs and ^{137}Cs are assumed to be the same as those for ^{90}Sr. This results in an eventual removal of 2.5% of the deposited ^{134}Cs and 23.1% of the deposited ^{137}Cs. Due to the linearity of the underlying models, the results of such washoff for an SST1 accident can be obtained by multiplying the health effects for ^{89}Sr, ^{90}Sr, ^{134}Cs and ^{137}Cs in Table 6 by .021, .219, .025 and .231, respectively. The results of this calculation are presented in Table 7. The watershed for Lake Michigan is only twice the area of the lake; thus, a release entirely to the lake or entirely to the watershed may be unlikely. However, the values in Tables 6 and 7 provide an indication of the potential magnitude of the effects.

Rhine-Meuse Valley. As a final example, radionuclide washoff in the Rhine-Meuse valley is considered. As in the two previous examples, an SST1 accident is modeled (i.e., an FK1 accident in the terminology of the German Reactor Safety Study).

Data describing the release are taken from the German Reactor Safety Study (Bu79, Tables F8, 3-1 and F8, 3-2) and are presented in Table 8. The washoff rates are taken from the extensive study by Jacobi et al. (Ja69a) for fallout ß-emitters in Germany. Although the results are not specific for ^{90}Sr and ^{137}Cs, it is indicated that the individual washoff rates for these nuclides are probably similar to those obtained for ß-emitters collectively

(Ja71, p. 1154). The particular washoff rates selected are those for the Rhine watershed above Wesel-Wittlaer/Beckum (Ja69a, Table 3). Thus,

$$\lambda_a = 1.05E\text{-}2 \text{ and } \lambda_b = 0.63E\text{-}2 \text{ yr}^{-1} . \qquad (4.3)$$

These values are used for both Sr and Cs; this probably overestimates the washoff rate for Cs. The resulting washoff fractions can be obtained with use of (3.3) and are presented in Table 9.

The calculated health effects for this case are presented in Table 9. The same calculational procedures are used as for the Mississippi river. The value for D is taken to be 8.5E13 l/yr (Bay78, p. 80). Further, for the other values in (3.6) for water consumption, WT = 1. for Sr and WT = .1 for Cs (Bay82a, p. (A) T-20), WC = 440 L/ind-yr (Bay82a, p. (A) T-30), and POP = 1.3E7 ind/yr (Bay82a, p. T-3). For fish consumption, the concentration ratios are the same as used earlier, and FP = 7.8E5 kg/yr from an assumed catch of 2.5E6 kg/yr (Bay82a, p. T-14) and an edible fraction of .31. For exposure from sprinkler irrigation, it is possible to derive a value for IR from assumptions in Bayer (Bay82a). It is assumed that (1) 1.E6 individuals are supplied with foodstuffs grown on irrigated land (Bay82a, p. 32), (2) a population of 60.4E6 individuals is supported on an area of 2.2E5 km^2 (Bay82a, p. T1), (3) 20% of all land is used to grow food for direct human consumption (Bay82a, p. Ti1), and (4) an irrigation rate of .1 m/yr is used (Bay82a, p. 32). From the preceding, it follows that IR = 7.4E10 l/yr.

5. Discussion. The purpose of this paper is to provide perspective with respect to the potential importance of radionuclide washoff from land surfaces into surface-water bodies as a contributor to the risk associated with reactor accidents. The preceding section presents estimates of consequences associated with such washoff for three sites: two in the United States and one in central Europe. To interpret the significance of these consequences as contributors to risk, it is necessary to compare them with the consequences associated with other aspects of reactor accidents. For the two sites in the United States, it is informative to use the consequences calculated in a recent study of reactor accidents in the United States which was performed to assist in the development of reactor siting criteria (A182b). For the German site, it is informative to use consequences calculated in the German Reactor Safety Study (Bu79, Bay81, Bay82b).

The siting criteria study (A182b) considered four reactor sites in the middle and lower Mississippi valley and five reactor sites at Lake Michigan. The study analyzed the consequences associated with accidents involving a standard 1120 MWe reactor at each site; the results of this analysis are summarized in Appendix C of the study's documentation (A182b). For the four sites in the middle and lower Mississippi valley, the number of latent cancer fatalities associated with an SST1 accident ranged from a few tens to a few thousands. The mean number of such fatalities ranged from 700 to 950. For the five sites at Lake Michigan, the number of latent cancer fatalities

associated with an SST1 accident ranged from a few tens to a few ten thousands. The mean number of such fatalities ranged from 1400 to 4000.

Tables 5 and 7 present predicted consequences associated with radionuclide washoff into surface-water bodies after SST1 accidents in the two regions in the United States. These accidents have the same assumed radionuclide releases as the SST1 accidents considered in the Siting Study (A182b). As comparison with the results indicated in the preceding paragraph shows, the consequences are much smaller than predicted mean consequences associated with atmospheric and terrestrial exposure pathways after reactor accidents. Indeed, it is only when the entire release of the radionuclides under consideration is assumed to go into the receiving surface-water body that the predicted consequences from aquatic pathways start to exceed the smaller of the consequences associated with the atmospheric and terrestrial pathways (see distribution of latent cancer fatalities in Appendix C of (A182b)). Such results for a complete deposition on Lake Michigan are given in Table 6. Further, due to the linearity of relations used, the results in Table 5 can be scaled to provide an estimate of the effects associated with the entire release of the radionuclides under consideration going into the river. The calculations performed for the present study are conservative in the sense that no mitigating actions are assumed to be taken to reduce the effects of any radionuclides which may enter a surface-water body. In contrast, various interdiction and decontamination procedures

are assumed in the generation of the results presented in the Siting Study (A182b).

For the 25 reactors considered in the German Reactor Safety Study, the mean, minimum and maximum numbers of predicted latent cancer fatalities subsequent to an SST1 accident are 43,100 lcf, 160 lcf and 107,800 lcf respectively (Bu79, Table F8,8-6). In the American Reactor Safety Study (US75), the previously referenced siting study (A182b) and the calculations performed in this paper, a dose effectiveness factor of .2 is assumed in the calculation of cancer induction due to low levels of radiation exposure for all cancers except breast cancer. Such a factor was not used in the German Reactor Safety Study. Thus, for comparison with the number of predicted latent cancer fatalities from the German Reactor Safety Study, it may be best to use five times the estimates presented in Table 9. However, even then the estimates derived from Table 9 are in the lower range of the consequences predicted in the German Reactor Safety Study for exposure from atmospheric and terrestrial pathways and are far below the mean estimate of 43,100 latent cancer fatalities. Indeed, this is even the case if the entire release of the four radionuclides considered is assumed to go into the river. As with the American sites, no mitigating actions are assumed to be taken to reduce the effects of the radionuclide releases to the Rhine. Also, the calculated results in Table 9 may be high due to conservative assumptions with respect to the size of the exposed population, the individual water consumption rate, and the washoff rate for Cs.

Although the authors' recognize that radiation exposure and resultant cancer induction after a reactor accident are highly sensitive to both site and release characteristics, it is felt that the contamination of surface-water bodies after reactor accidents by the erosion of atmospherically deposited radionuclides is not a major contributor to the risk associated with such accidents. This conclusion is drawn because of the use of conservative modeling techniques to determine the effects of the erosion of radionuclides into surface-water bodies and the small size of such effects in comparison with the effects predicted for atmospheric and terrestrial pathways. Although this study and the studies to which it is compared contain large uncertainties, it is felt that inclusion of these uncertainties probably would not change the conclusion which has been reached. Further, this conclusion is consistent with the results of various other analyses of widespread radioactive depositions (e.g., Gu69, Har69, Ja69b, Aa71, Ng73, Da81, Klu81, Gj82, US82). In each of the preceding analyses, various ingestion pathways were considered after a radionuclide release (primarily fallout from weapons tests); the water-related pathways were consistently found to be among the smallest contributors to exposure. For the water-exposure pathway after a reactor accident to be important, an extensive entry directly into a public water supply would be necessary (e.g., Nic81). However, this seems unlikely and would generally be amenable to interdiction.

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Table 1. Removal Rates for Fallout Radionuclides Determined on a Regional Basis.

Reference	Nuclide	100 λ_a	100 λ_b	Location	Comment
Straub et al. (St60)	^{90}Sr	1.7*		Continental United States	Three month period
	β-emitters	1.6		Continental United States	Nine month period
	^{90}Sr	3.9-12.2		Five Watersheds in Ohio Valley	Three month period

*The values for λ_a may be overestimated as λ_a is defined by λ_a=R/D, where R and D are the removal in surface water and total deposition, respectively, for each time period considered. No credit is taken for removal of accumulated fallout.

Miyake and Tsubota (Mi63)	^{90}Sr		.7-3.3	10 watersheds in Japan	Mean value for ^{90}Sr is 1.5. Value for ^{137}Cs is estimated as that due to direct deposition on water bodies.
	^{137}Cs	1.7			
Yamagata et al. (Ya63)	^{90}Sr	7.2	.31	Japan	1 month period for λ_a
	^{137}Cs	1.3	.06		
Jacobi et al. (Ja69a)	β-emitters	.20-1.05	.12-.97	Twenty watersheds in West Germany	1 month period for λ_a. Extensive study.
Jordan et al. (Jo73, Jo76)	^{90}Sr		.036	Tropical rain forest, Puerto Rico	Derives λ_b for stable Sr and uses to predict behavior of ^{90}Sr.
Kamada et al. (Ka73)	^{90}Sr		.2-2.*	Three watersheds in Japan	

*The values for λ_b may be overestimated as λ_b is defined by λ_b=R/D, where R and D are the annual radionuclide removal to surface water and total accumulated deposition, respectively.

Table 1. (Continued)

Reference	Nuclide	100 λ_a	100 λ_b	Location	Comment
Miyake et al. (Mi73)	^{239}Pu		.12	Japan	
Kamada et al. (Ka74)	^{90}Sr		.2-2.	Four watersheds in Japan	
Menzel (Me74)	^{90}Sr	.59-2.17	.17-.75	Eight regions in United States	Two month period for λ_a. Extensive investigation.
Simpson et al. (Si76)	^{137}Cs		.1	Hudson River Watershed	
Carlsson (Ca78)	^{137}Cs	1.9	.56	Small watershed in Sweden	
Sprugel and Bartelt (Sp78)	$^{239,240}Pu$		.05	Ohio	
Aarkrog (Aa79)	^{90}Sr	.5	.1	Denmark	Extensive investigation.
Linsley et al. (Li82)	^{90}Sr	10.	1.	Upland lake water supplies, England	^{90}Sr model for upland lake water slightly different from (2.1).
	^{90}Sr	1.	.067	River Thames, England	
	^{137}Cs	.1	.0067	River Thames, England	

Table 2. Removal Rates for Fallout Radionuclides Determined From Experimental Plots.

Reference	Nuclide	100 λ_a	100 λ_b	Location	Comment
Menzel (Me60)	^{90}Sr			La Crosse, Wi. and Tifton, Ga.	Various experimental plots. Removal measured for each rainfall event. Considerable variability in amount of ^{90}Sr removed in runoff after raintall event. Amount removed over 25% in one case.
Frere and Roberts (Fre63)	^{90}Sr			Coshocton, Oh.	One-third to two-thirds of ^{90}Sr deposited by fallout up to 1960 lost from cultivated experimental plots.
Graham (Gr63)	^{85}Sr	.009-.98		McCredie, Mo.	Three month periods, variety of experimental plots.
Rogowski and Tamura (Ro65)	^{137}Cs	2.6-11.9		Oak Ridge, Tn.	Three month period; bare, clipped meadow and tall meadow plots.
Haghire (Hag69)	^{90}Sr		.23-1.02	Wooster, Oh.	Five experimental plots. Results based on five year period for loss by runoff water and runoff sediment.
Rogowski and Tamura (Ro70a, Ro70b)	^{137}Cs ^{137}Cs ^{137}Cs	30.7* 18.9 6.7	12.3 .57 .11	Oak Ridge, Tn. Oak Ridge, Tn. Oak Ridge, Tn.	Bare plot. Clipped meadow plot. Tall meadow plot.
Pisarev et al. (Pis72)	^{90}Sr	.62	.16		Experimental plots.

*λ_a and λ_b determined from first and second year, respectively, of two year experiment.

Table 3. Half-lives and Risk Factors for Selected Radionuclides.

	^{134}Cs	^{137}Cs	^{89}Sr	^{90}Sr
Half-life (units: yr)	2.05	30.2	0.14	28.1
Risk Factor (units: 1cf/Ci)	3.3	2.5	0.30	2.8

Table 4. Radionuclide Release Used in Siting Criteria Development for an SST1 Accident.

Nuclide	Initial[a] Inventory	Release[b] Fraction	Initial Release
^{89}Sr	9.6E7[c]Ci	0.07	6.7E6 Ci
^{90}Sr	5.2E6 Ci	0.07	3.6E5 Ci
^{134}Cs	1.3E7 Ci	0.67	8.7E6 Ci
^{137}Cs	6.5E6 Ci	0.67	4.4E6 Ci

[a]From Table B.1-1 (A182b)
[b]From Table 2.3.1-2 (A182b)
[c]9.6E7 = 9.6×10^7

Table 5. Calculated Latent Cancer Fatalities for an SST1 Accident Due to Radionuclide Washoff into the Middle and Lower Mississippi River.

Nuclide	Washoff Fraction	Drinking Water	Fish	Sprinkler Irrigation
^{89}Sr	.02	.06 lcf	<.01 lcf	<.01 lcf
^{90}Sr	.19	.3 lcf	.02 lcf	<.01 lcf
^{134}Cs	.03	.9 lcf	6.4 lcf	.01 lcf
^{137}Cs	.20	2.2 lcf	15.0 lcf	.03 lcf

Table 6. Calculated Latent Cancers for an SST1 Accident due to Complete Deposition on Lake Michigan.

Nuclide	Effective Half-life	Integrated Concentration	Drinking Water	Fish	Sprinkler Irrigation
^{89}Sr	.14 yr	2.8E-10 Ci yr/l	.34 lcf	.01 lcf	.01 lcf
^{90}Sr	15.4 yr	1.6E-9 Ci yr/l	18.2 lcf	.69 lcf	.34 lcf
^{134}Cs	1.3 yr	3.4E-9 Ci yr/l	45.7 lcf	114.4 lcf	.86 lcf
^{137}Cs	3.5 yr	4.5E-9 Ci yr/l	45.8 lcf	114.7 lcf	.87 lcf

Table 7. Calculated Latent Cancer Fatalities for an SST1 Accident due to Radionuclide Washoff into Lake Michigan.

Nuclide	Drinking Water	Fish	Sprinkler Irrigation
^{89}Sr	<.01 lcf	<.01 lcf	<.01 lcf
^{90}Sr	4.0 lcf	.15 lcf	.07 lcf
^{134}Cs	1.1 lcf	2.9 lcf	.02 lcf
^{137}Cs	10.6 lcf	26.5 lcf	.20 lcf

Table 8. Radionuclide Release Used in German Reactor Safety Study for an SST1 Accident.

Nuclide	Initial[a] Inventory	Release[b] Fraction	Initial Release
^{89}Sr	1.05E8 Ci	.067	7.0E6 Ci
^{90}Sr	5.30E6 Ci	.067	3.6E5 Ci
^{134}Cs	1.38E7 Ci	.5	6.3E6 Ci
^{137}Cs	7.06E6 Ci	.5	3.5E6 Ci

[a]From Table F8, 3-1 (Bu79)
[b]From Table F8, 3-2 (Bu79)

Table 9. Calculated Latent Cancer Fatalities for an SST1 Accident Due to Radionuclide Washoff into the Rhine River.

Nuclide	Washoff Fraction	Drinking Water	Fish	Sprinkler Irrigation
^{89}Sr	.012	1.7 lcf	<.01 lcf	.24 lcf
^{90}Sr	.212	14.3 lcf	.06 lcf	2.0 lcf
^{134}Cs	.029	22.2 lcf	12.1 lcf	6.3 lcf
^{137}Cs	.223	65.6 lcf	35.8 lcf	18.6 lcf