Risk-Cost Analysis of Strontium-90 Migration to Water Wells at the Chernobyl Nuclear Power Plant

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ABSTRACT

A decision-analysis methodology is applied to the problem of the migration of strontium-90 (^{90}Sr) in ground water to the Pripyat Town water wells located near the Chernobyl Nuclear Power Plant in the Ukraine. The source of the radiostrontium is the diffuse contamination of the ground surface by radioactive fallout, which was released from the damaged reactor core in 1986. The hydrostratigraphy at the site is idealized as a three-layer system; a marl layer separates an upper, unconfined aquifer from a lower confined aquifer which is used as a source of water. Management strategies are evaluated using a risk-cost objective function. The risk is defined as the expected costs of well-field failure due to exceedence of the ^{90}Sr regulatory standard for drinking water. The probability of well-field failure is estimated using Monte-Carlo simulation techniques. This probability is most dependent upon the ^{90}Sr regulatory standard and upon the hydraulic conductivity of the aquitard in a hypothesized annular facilitated-transport zone near the well casings. The risk-cost analysis indicates that a "no complex remedial action" alternative represents the preferred management strategy for the Pripyat Town water wells, but that the risk will increase in 25 years by about one order of magnitude. Therefore, efforts to obtain more reliable estimates of contaminant transport parameters, in particular ^{90}Sr concentrations in the unconfined aquifer, appear warranted.

INTRODUCTION

Ground water is a major source of drinking water in the Northern Polessje region of the Ukraine. This area has been contaminated by radioactive fallout released from the damaged #4 Reactor of the Chernobyl Nuclear Power Plant on April 26, 1986. The Chernobyl Nuclear Power Plant complex (Ch.NPP), with two reactor units continuing to operate, obtains potable water from the Pripyat Town well field (Figure 1). Before the accident, this well field supplied potable water to Pripyat Town and the Ch.NPP. After evacuation of the population of Pripyat Town at the end of April 1986, the water provided by the well field has been used mainly by the power plant. The potential threat of contamination of this water supply by radionuclides has been a subject of concern and controversy since the first days after the accident.

The Pripyat Town wells extract water from a 40–50 m deep confined aquifer (Figure 2). The well field is composed of about 20 wells distributed along two lines: the Yanov Line and Novo-Shepelichi Line. Usually from 12 to 17 wells operate simultaneously, pumping rates in individual wells are about 500 m^3/day (Skal'skij et al., 1994). The potential contaminant source is "hot" fuel particles (uranium oxide) from the #4 Reactor core which have been dispersed over the ground surface. Ground-surface contamination by ^{90}Sr in the area of the well field is estimated at $3 - 7x10^3$ kBq/m^2 (100–200 Ci/km^2). Contamination of the unconfined aquifer at the site may also be caused by radionuclide migration from radioactive waste burial grounds bordering the Yanov Line (the so-called "Yanov" radioactive waste burial site), which were created in 1986–1987 during cleanup of the area around the Ch.NPP (Dzhepo et al., 1994).

For the two months immediately following the Chernobyl accident, large-scale remediation measures were undertaken to mitigate radionuclide migration to groundwater at the Chernobyl site. Several drainage systems, containing approximately 300 drainage wells, were constructed. One of the aims of these drainage wells was to protect the Pripyat Town water wells by increasing the thickness of the unsaturated zone, and decreasing vertical hydraulic head gradients between the unconfined

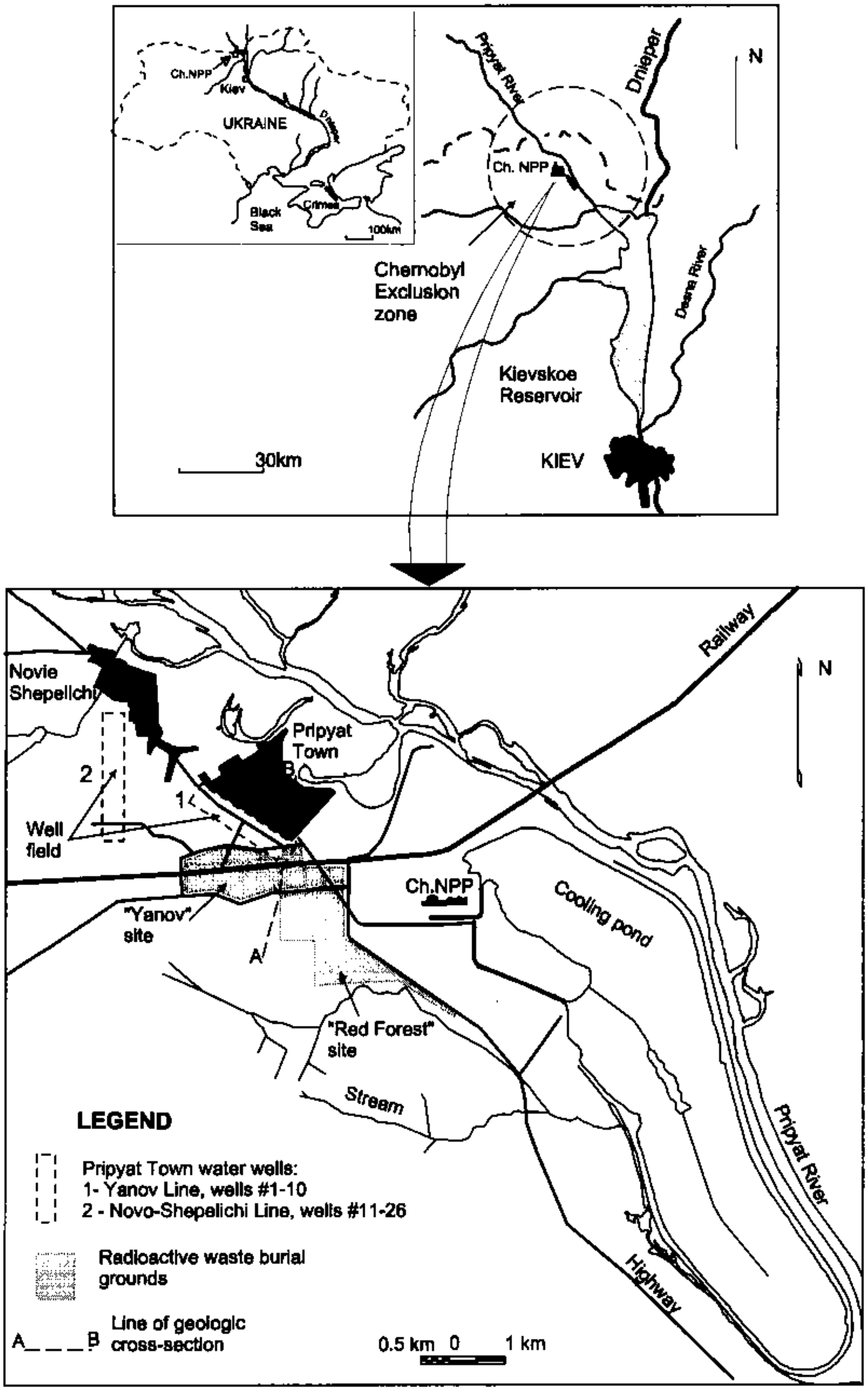

Figure 1. Map of the study area.

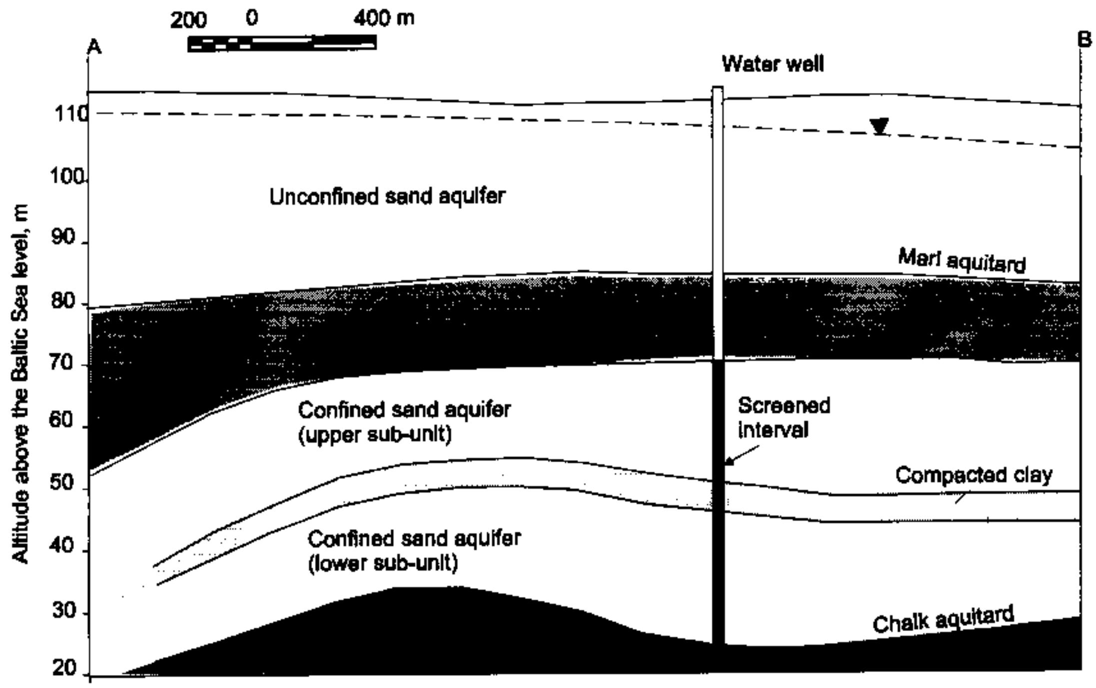

Figure 2. Geologic cross section through the study site. The water table location is shown as measured in April 1991.

and confined aquifer. These remedial measures were based on a "worst case" analysis. By the end of 1986, however, these remedial measures were abandoned. Three reasons can be cited: actual levels of radioactive contamination of the ground surface at the Ch.NPP site were found to be about 100 times less than "worst case" estimates, "hot" fuel particles from the reactor core were found to be rather insoluble, and early monitoring data had shown no breakthrough of radionuclides to the ground-water system. The drainage wells have never been put into operation, and some pumps and drainage collectors have been dismantled and removed (Afonin et al., 1992; Waters et al., 1994).

The optimistic predictions of 1987 about the stability of the fuel particles, which were based on short-term laboratory studies, were not confirmed by subsequent field observations (Borzilov, 1990). Beginning in 1988, an increase in the mobility of some radionuclides at the Chernobyl site was observed, due to weathering and leaching of "hot" fuel particles (Bobovnikova et al., 1990). From a hydrogeologic perspective, of all the long-lived Chernobyl radionuclides (cesium-137, strontium-90, plutonium-239 and plutonium-240), migration of ^{90}Sr is the primary concern (Dzhepo et al., 1994; Bugai et al., 1995b). This radionuclide is characterized by rapid release from the matrix of the "hot" particles, and by relatively low sorption onto local soils and geological materials. Recent studies by the Institute of Geological Sciences (IGS) have shown that in the "close" zone (5–10 km radius) around the Ch.NPP, the upper portion of the unconfined aquifer (to a depth of 0.5–1 m below the water table) is contaminated by ^{90}Sr in concentrations exceeding the Ukrainian drinking water standard (DWS) of 3.7 Bq/L (100 pCi/L). The exceedence level ranges from a factor of two to several orders of magnitude (Institute of Geological Sciences, 1994). The increased ^{90}Sr mobility potentially threatens the Pripyat Town well field, and has caused new concerns among the Chernobyl site authorities. Several recent studies propose to implement remedial measures to protect the ground-water resources at Chernobyl from further radioactive contamination. The Ukrainian Water Project Institute (UWPI) proposes to restore and put into operation the drainage system constructed in 1986 (Ukrainian Water Project Institute, 1993). Ivanushkina and Ryabtseva (1991) proposed to mitigate ground-water contamination through excavation and safe disposal of the contaminated upper soil layer.

The purpose of this paper is to evaluate the necessity of proposed protective measures for the Pripyat Town well field. To accomplish this objective a decision analysis framework similar to that developed by Massman and Freeze (1987), Freeze and others (1990), and Massman and others (1991) was applied. Management alternatives are compared using a risk-cost objective function. A well-field failure is considered to occur if the concentration of ^{90}Sr at a well head exceeds the Ukranian drinking water standard at any time during a defined planning horizon. The risk is defined as the net present value of the expected costs of well-field failure. The main technical issue to be evaluated is the probability of well-field failure. To estimate this probability, we develop a simple hydrogeological model, which is coupled in a probabilistic Monte-Carlo assessment, to models describing the uncertainties in hydrogeological parameters, geological boundaries on the local hydrogeological system, and possible changes in regulatory standards.

HISTORY OF GROUND-WATER CONTAMINATION

A discussion of the hydrogeology near the Chernobyl site can be found in Bugai and others (1994) and Vovk and others (1994). Only the key features of this system are highlighted here. The uppermost geologic unit is a Quaternary alluvial deposit, represented mostly by fine to medium-grained quartz/feldspar sands (Figure 2). This unit forms an unconfined aquifer with a saturated thickness of 20–30 m, and a regional hydraulic gradient that ranges from 0.001 to 0.005. The depth to the water table ranges from 1 to 5 m across the site. Precipitation averages 550–600 mm/year. Ground-water recharge is estimated by various methods to be from 50–200 mm/year. Hydraulic conductivity of the sands range from 1 to 20 m/day. The Quaternary deposits overlie a Paleogene marl, which has a low hydraulic conductivity (0.001–0.01 m/day). Beneath the marl, Eocene sands form a confined aquifer that varies in thickness from 20 to 40 m. This confined aquifer consists of two sub-units, separated by a clay layer. Transmissivities for this aquifer are estimated to range from 50 to 250 m^2/day (Dzhepo et al., 1994).

Because neither a monitoring well network has been installed, nor has there been a consistent monitoring program for existing wells, only limited information is available on ground-water contamination in the area of the well field. Auger-sampling by the Institute of Geological Sciences (1994) showed that the upper portion of the unconfined aquifer within 50 m of the water wells #1 and #26 is contaminated by ^{90}Sr with concentrations ranging from 10 to 40 Bq/L (3 sampling points). Table 1 presents results of a radiological survey of the well field in 1993, at which time most water wells were sampled for ^{90}Sr and tritium (^{3}H). The tritium data for wells #1, 8 and 15 suggest that a significant portion of the water withdrawn from the confined aquifer is formed by ground water with an age that is less than 35–40 years (see Table 5.4 of Davis and Murphy, 1987). Interpretation of sampling results for other wells is difficult because of the analytical detection limit of the tritium measurements. Water vapor released from the destroyed Ch.NPP #4 reactor caused tritium concentrations in the local precipitation in May, 1986 to increase by 2–4 times the pre-accident levels (Katrich, 1990). However, it is reasonable to assume that the tritium observed in the water wells originates from pre-accident global sources.

Table 1. *^{90}Sr and tritium concentrations in the Pripyat Town water wells, August 1993 (adopted from Skal'skij et al., 1994).*

Well Number	^{90}Sr, Bq/L	^{3}H, T.U.
1	< 0.0146	28±3
5–7, 9	< 0.0146	< 15
8	< 0.0146	16±2
11, 13, 17, 20, 23–25	< 0.0146	< 25
14	0.013±0.002	< 25
15	< 0.0146	31±3
16	0.018±0.001	< 25
26	0.015±0.001	< 25

In wells #14, 16 and 26, ^{90}Sr has been detected in concentrations of about 10^{-2} Bq/L. The ^{90}Sr detected in these wells is not necessarily related to the Chernobyl accident; it could have originated from fallout due to global nuclear testing. According to a few sampling data, ^{90}Sr concentrations in shallow ground water in the Polessje region before the Chernobyl accident were about 10^{-2} – 10^{-1} Bq/L (Ivanushkina and Ryabtseva, 1991; Sobotovich et al., 1992).

The mechanism which may have lead to the relatively fast migration of "bomb" ^{90}Sr from the unconfined aquifer to the water wells in less than two decades of well field operation is a topic of considerable speculation. One hypothesis to explain the ^{90}Sr migration to the confined aquifer is the existence of an annular, more permeable zone in the aquitard adjacent to the well casings, created during well installation (Figure 3). We refer to this pathway as a "weak zone." S. Dzhepo (1995) and A. Skal'skij (1995) from the IGS have suggested that it is likely that percussion drilling techniques were used to install the water wells, with little or no attempt made to seal the annular space around the well casing. A second hypothesis is that ^{90}Sr penetrated the water wells directly from the contaminated unconfined aquifer through leaks in improperly sealed joints of well casings (V. Shestopalov, 1995). This last hypothesis is not evaluated in detail here, this mechanism has a limited potential to impair the quality of extracted ground water and, in this case, remediation of the confined aquifer is not an issue. However, because this mechanism may lead to misinterpretation of water quality data, this second hypothesis deserves further evaluation.

RISK-COST-BENEFIT FRAMEWORK

Alternative management strategies to deal with the potential contamination of the well field are compared using a risk-cost-benefit objective function:

$$\Phi = \sum_{t=0}^{T} \left[B(t) - C(t) - R(t)\right] / \left(1 + i\right)^t \qquad \text{Eq. 1}$$

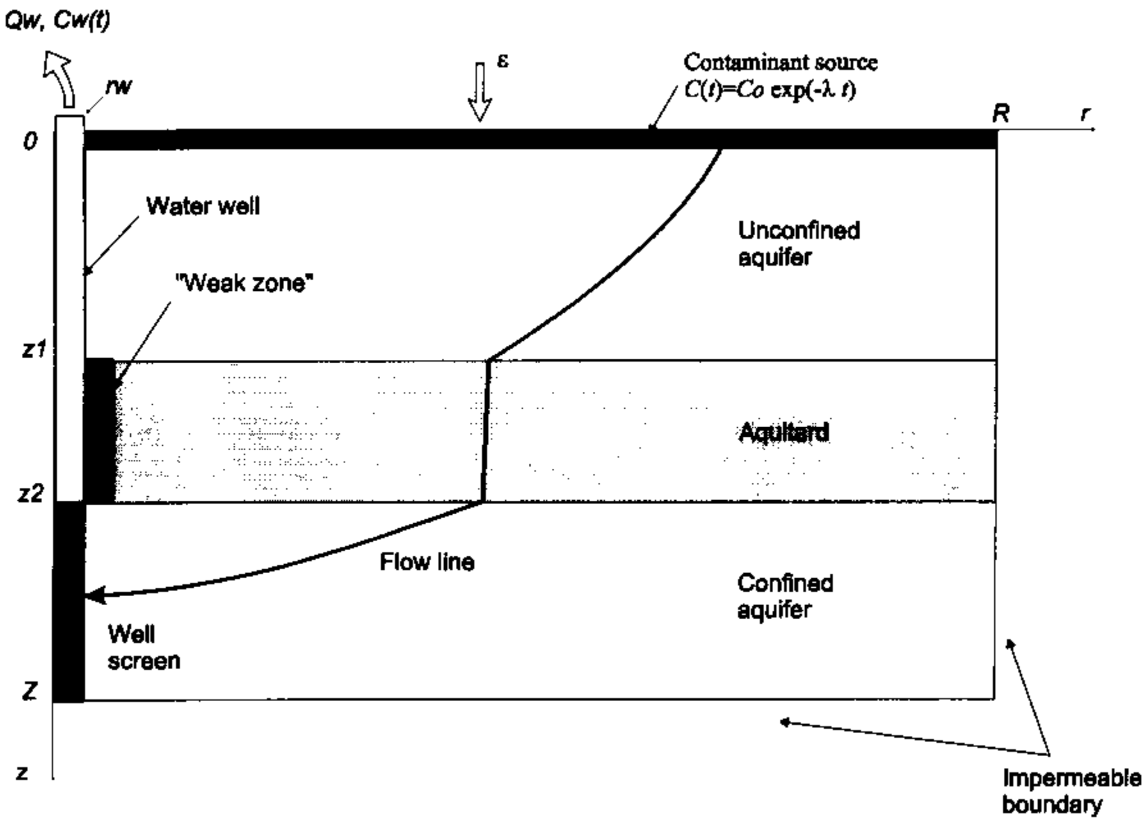

Figure 3. Flow domain for the simulation model.

where Φ is the value of the objective function for a given management strategy, $B(t)$ are the annual benefits, $C(t)$ are annual costs, $R(t)$ are the annual risks, T is planning time horizon, and i is discount rate. The risk term is given by

$$R(t) = P_f(t)\ C_f(t) \qquad \text{Eq. 2}$$

where $P_f(t)$ is the probability of failure of a given management strategy to meet the technical design requirements and $C_f(t)$ is the cost of failure in year t, respectively. For our purposes, failure of the well field will occur if the ^{90}Sr concentration at the well head exceeds the Ukranian drinking water standard. The goal is to select the management alternative that maximizes the objective function given by Equations 1 and 2. Freeze and others (1990) noted that the pertinent time horizon for decision analysis is on the order of 20–50 years, because for a discount rate of 10 percent, the net present value of future dollars approaches zero for periods more than 50 years into the future. In the present analysis a longer time horizon of 70 years is used (i.e., an average human life span), which is a time period commonly utilized in radiological risk assessment studies.

The probability of failure of the well field is estimated using a hydrogeological simulation model, coupled to an uncertainty model. The hydrogeological model simulates the performance of the well field. The uncertainty model describes uncertainties in geological boundaries, hydrogeologic parameters, and a possible change in the regulatory standards. The models are coupled in a probabilistic Monte-Carlo performance assessment.

A number of simplifications can be adopted for this analysis. First, direct revenues from preventive or remedial measures are zero. Second, because the goal of this study is to evaluate the necessity of implementing a ground-water protection strategy at the present time, the cost of failure of the well field C_f can be assigned a single value, independent of time (while still discounting future costs of failure to the present value in the calculation of Φ). Third, if we assume that capital costs of ground-water remediation dominate the operating costs, then the objective function reduces to:

$$\Phi = -C - \sum_{t=0}^{T} \left[P_f(t)\, C_f\right] / \left(1+i\right)^t \qquad \text{Eq. 3}$$

where C are the capital costs for the preventive or remedial measures. We can remove for convenience the minus sign from Equation 3 and treat our problem as one of cost-risk minimization.

The simplest management alternative for a ground-water contamination problem is a "no action" alternative. In this case $C = 0$, and the value of the objective function is given by:

$$\Phi^{(*)} = \sum_{t=0}^{T} \left[P_f(t)\, C_f\right] / (1+i)^t \qquad \text{Eq. 4}$$

Correspondingly, in the case of a "perfect" performance of a remediation design, the term $P_f(t)$ in Equation 3 is reduced to zero. Therefore, the objective function for any remedial design involving an "action" will be equal to or greater than:

$$\Phi^{(**)} = C \qquad \text{Eq. 5}$$

where C are capital costs for the remedial alternative under analysis. Thus, as a first approximation, the necessity of ground-water protective measures can be evaluated by comparing the expected no-action costs (Equation 4), with the action costs (Equation 5), and choosing an alternative that minimizes cost-risk. This approach is essentially a formal restatement of the basic premise of the cost-benefit assessment framework adopted here, that the value of avoiding a ground-water contamination incident (i.e., cost of ground-water remediation measures) is not higher than expected (probabilistic) costs that would be incurred if the incident were allowed to occur.

In the first step of this framework, $\Phi^{(*)}$ is estimated by determining the risk of well-field failure assuming no ground-water remediation measures are undertaken. The probability of well-field failure in year t, $P_f(t)$, is equal to the probability that the ^{90}Sr concentration in the water well in year t ($C_w[t]$) exceeds the value allowed by the regulatory agency, that is, the ^{90}Sr drinking water standard C_{DWS}:

$$P_f(t) = P(C_W[t] > C_{DWS}) \qquad \text{Eq. 6}$$

The methods used to estimate $P_f(t)$ are described in the following sections.

HYDROGEOLOGICAL MODEL

The screening character of this decision analysis, the sparse data on hydrogeological parameters, and limited radiological monitoring justify the use of a relatively simple assessment model. We use simple models to represent the ground-water flow system (steady state, saturated, radially-symmetric flow to a production well) and to characterize radionuclide migration (advective transport, reversible equilibrium sorption and decay). The effects of dispersion on the ^{90}Sr concentration at the well are neglected. We use the estimated concentration of ^{90}Sr in a single production well as a surrogate for the failure of the well field. Ground-water flow to a production well is described by the equation:

$$\frac{1}{r}\frac{\partial}{\partial r}\left(Kr\frac{\partial h}{\partial r}\right) + \frac{\partial}{\partial z}\left(K\frac{\partial h}{\partial z}\right) = 0 \qquad \text{Eq. 7}$$

where $h(r,z)$ is hydraulic head, $K(r,z)$ is hydraulic conductivity, and (r,z) are cylindrical coordinates. The flow domain is shown in Figure 3. The boundary $z = 0$ corresponds to the upper boundary of the unconfined aquifer (water table), and the boundary $z = Z$ corresponds to the base of the confined aquifer. The boundary conditions for Equation 7 are prescribed as follows:

$$K\frac{\partial h}{\partial z}\bigg|_{(z=0)} = \varepsilon,\; K\frac{\partial h}{\partial z}\bigg|_{(z=Z)} = 0, \qquad \text{Eq. 8}$$

$$K\frac{\partial h}{\partial r}\bigg|_{(r=r_w,\, 0<z<z_2)} = 0,\; h\big|_{(r=r_w,\, z_2<z<Z)} = 0,\; K\frac{\partial h}{\partial r}\bigg|_{(r=R)} = 0,$$

where ε is the ground-water recharge rate, r_w is the well radius, R is the radial extent of the flow domain, and z_1, z_2 are the vertical coordinates defining the location of the aquitard. The well screen is located over the depth interval z_2 to Z.

Each model layer is homogeneous and isotropic, with values K_1, K_2 and K_3 in the unconfined aquifer, aquitard and confined aquifer, respectively. The radius of the flow domain R is calculated from the water balance condition:

$$\pi \times R^2 \times \varepsilon = Q_w$$

where Q_w is the prescribed pumping rate for the water well.

A two-dimensional, finite difference model is used to obtain the hydraulic head solution. The grid spacing was logarithmic in the r dimension and uniform in the z direction. Once $h(r,z)$ is determined, it is a simple matter to calculate the velocity field $v(r,z)$ using the associated stream function, $\psi(r,z)$, for the flow domain (Frind and Matanga, 1985). Isolines of $\psi(r,z)$ are constructed, which define a set of streamtubes that originate at the water table and terminate at the well screen.

The next step is the simulation of advective transport of ^{90}Sr from the source zone to the water well. It is assumed that the contaminant source is the contaminated upper portion of the unconfined aquifer (i.e., the flow domain boundary $z = 0$ in Figure 3). The ^{90}Sr concentration at $z = 0$ is prescribed as:

$$C(t) = C_0 \exp(-\lambda t) \qquad \text{Eq. 9}$$

where C_0 is the initial concentration at time $t = 0$, and λ is the ^{90}Sr radioactive decay constant. Specific mechanisms of contaminant migration to the water table are not modeled. In fact, these mechanisms are not well understood at the study site. The most probable contaminant source is radioactive fallout on the ground surface. However ^{90}Sr migration from unknown radioactive waste burial grounds may occur as well. The boundary condition (Equation 9) does not distinguish between migration mechanisms to the unconfined aquifer, which simplifies and adds flexibility to the subsequent analysis. The C_0 parameter in Equation 9 can be assigned a range of values based on monitoring data and/or judgment.

Given the boundary condition (Equation 9), the ^{90}Sr concentration in the water well is computed as:

$$C_w(t) = Q_{con}(t)\ C_0 \exp(-\lambda t) \qquad \text{Eq. 10}$$

where $Q_{con}(t)$ denotes the fraction of well discharge at time t, formed by contaminated water. $Q_{con}(t)$ is calculated as a sum of the discharges of flow tubes with contaminant travel times from source to the well equal or less than t:

$$Q_{con}(t) = \sum_{t_j \leq t} Q_j \qquad \text{Eq. 11}$$

The sum in the above formula is taken over the set of flow lines forming the flow net, Q_j is the portion of the total well discharge attributed to flow line j, and t_j is the contaminant travel time for flow line j from the contaminant source to the water well. Contaminant travel times are estimated by numerical evaluation of the following integral:

$$t_j = \int \frac{n(l) + \rho(l)\, K_d(l)\, dl}{n(l)\, v(l)} \qquad \text{Eq. 12}$$

where n is porosity, ρ is the bulk density of the sediment, K_d is the distribution coefficient for ^{90}Sr, and $v(l)$ is the advective velocity defined along the flow line. According to the model assumptions, each hydrogeologic unit possesses different hydrogeologic properties, therefore, the parameters n, ρ and K_d change in a stepwise manner when a flow line exits one hydrogeological unit and enters another unit. Equation 10 is based on the assumption that the pre-accident concentration of ^{90}Sr in ground water is negligible compared to the contaminant concentration in the source zone.

UNCERTAINTY MODELS

Hydrogeological Parameter Uncertainty

It is convenient to divide the hydrogeological parameters into two groups. The first group, listed in Table 2, includes parameters which are better characterized and/or expected to be less important. These parameters are treated in a deterministic manner, with the "best guess" estimates listed in Table 2. The second group, listed in Table 3, includes the "critical" parameters which are judged to be a major source of uncertainty in estimating the probability of well-field failure. These parameters are the ^{90}Sr concentration at the water table (i.e., C_0 in Equation 9), and the ^{90}Sr distribution coefficients. These parameters are assigned uniform probability distributions within the bounds specified in Table 3. Our level of knowledge does not provide sufficient data to support the use of a more complex and "informative" probability distribution than the uniform distribution. The rationale for the values of the upper and lower bounds is explained below.

Table 2. *Values assigned to deterministic parameters of the hydrogeological simulation model.*

Parameter	Unit	Value
Ground-water recharge	mm/y	200
Well diameter	mm	320
Well pumping rate	m^3/day	500
Unconfined Aquifer		
Saturated thickness	m	20
Hydraulic conductivity	m/day	10
Porosity	%	30
Sediment bulk density	g/m^3	1.65
Aquitard		
Thickness	m	10
Hydraulic conductivity	m/day	0.01
Porosity	%	5
Sediment bulk density	g/m^3	2.0
Confined Aquifer		
Thickness	m	20
Hydraulic conductivity	m/day	5
Porosity	%	30
Sediment bulk density	g/m^3	1.65

The probability distribution for the ^{90}Sr concentration at the water table, C_0, is derived from both onsite data and data from adjacent areas. The lower bound for C_0 of 10 Bq/L is the minimum ground-water concentration reported by Skal'skij and others (1994). The upper bound of 1,000 Bq/L is a conservative estimate, derived by extrapolating ground-water concentration data from the better characterized "Red Forest" site located to the southwest of the well field (Figure 1). At the Red Forest site, where ^{90}Sr surface contamination is about 10 times higher than that in the area of the well field, ^{90}Sr concentrations in ground water reported by Dzhepo and others (1995) are about 104 Bq/L, or 10 times higher then the upper limit for C_0 specified in Table 3. At the Red Forest site, ground water is permanently or seasonally contacting radioactive materials which were disposed of in shallow trenches constructed in the local sandy soil (Dzhepo et al., 1994, 1995). The hydrogeological conditions at this site correspond to a 'worst case' radionuclide migration scenario.

Balance calculations suggest that if the ^{90}Sr concentration in ground-water recharge is 1,000 Bq/L, then the surface inventory of contaminants of 7 x 10^3 kBq/m^2 (i.e., upper estimate of the surface contamination for the well field area) will be exhausted in 35 years (see

Table 3. *Uniform probability distribution ranges assigned to the uncertain parameters of the hydrogeological simulation model.*

Parameter	Unit	Minimum Value	Maximum Value
^{90}Sr concentration in contaminant source zone, C_0	Bq/L	10	1000
Sorption distribution coefficient for the unconfined aquifer, $K_{d,1}$	L/kg	0.5	5
Sorption distribution coefficient for the aquitard, $K_{d,2}$	L/kg	0.5	20
Sorption distribution coefficient for the confined aquifer, $K_{d,3}$	L/kg	0.5	5

Appendix). This estimated contaminant pulse duration is of the same order of magnitude as the time horizon of the analysis, which confirms that the upper limit of 1,000 Bq/L for the initial radionuclide concentration in the unconfined aquifer is a reasonably conservative estimate.

The probability distribution assigned to the ^{90}Sr sorption parameter (K_d) for the unconfined aquifer is based on site-specific sorption data (Table 4). Conversely, the K_d distribution for the marl is based on literature data for ^{90}Sr sorption on carbonate rocks and calcite (Table 5), because no data from the Chernobyl site are available. Because sorption parameters for carbonate sediments reported in the literature vary over a wide range, uncertainty in the aquitard sorption parameters is significant. Lastly, in the absence of specific data, it is assumed that the sorption properties of quartz sands of the confined aquifer are similar to those of the unconfined aquifer.

Regulatory Parameter Uncertainty

At the present time the permissible ^{90}Sr concentration in drinking water is 3.7 Bq/L (100 pCi/L), as defined by the 'Temporary Permissible Levels' established in 1991 by the Ukrainian National Commission on Radiation Protection (TPL-91). The title of the document reflects the temporary character of this regulation. The TPL-91 concentration level has replaced the previous Soviet drinking water standard for ^{90}Sr of 14.8 Bq/L (400 pCi/L). It is quite possible that the drinking water standard will be further lowered in the future, but this change may be jeopardized by economic difficulties in the Ukraine. For comparison, the U. S. Environmental Protection Agency's drinking water standard for ^{90}Sr is 0.30 Bq/L (8 pCi/L). Given these considerations, the ^{90}Sr drinking water standard is treated as an uncertain parameter. A 50 percent probability is assigned to the event that the present Ukrainian DWS for ^{90}Sr of 3.7 Bq/L will be changed during the planning horizon to 0.30 Bq/L. An equal probability is given to the scenario that the present regulation will not be changed.

Geological Uncertainty

Our analysis of ^{90}Sr migration to the water well recognizes the possibility of an annular zone of higher permeability (a so-called "weak zone") in the aquitard adjacent to the well casing, created during installation of the water well (Figure 3). The existence of this facilitated transport zone may explain, as discussed earlier, trace amounts of ^{90}Sr in the water wells reported by Skal'skij and others (1994). The radius of this 'weak zone' is set to 0.5 m, and its hydraulic conductivity is assumed to be the same as that of the unconfined aquifer (i.e., equal to the highest hydraulic conductivity value in the domain). We estimate on a subjective basis that the probability is 50 percent that a weak zone is present in the aquitard adjacent to the well casing.

A significant simplification in the hydrogeological model is that it does not account for possible hydraulic interactions between individual water wells. Such an interaction may create non-symmetric flow patterns, greater drawdown in the confined aquifer, and consequently larger downward ground-water fluxes into the confined aquifer compared to the case of a single water well. This non-conservatism is compensated by a number of conservative assumptions. In particular, several 'known' parameters of Table 2, that is, the thickness and hydraulic conductivity of the aquitard and the ground-water recharge rate, are assigned values which are conservative from the perspective of the water well

Table 4. *^{90}Sr distribution coefficients for the sandy Quaternary sediments at the Ch.NPP site.*

Sediment	Site	K_d, L/kg	Reference
sands	"Red Forest" site	2–4	Olkhovik et al., 1992
loams		4–10	
sands	Ch.NPP site	< 1	Radium Institute, 1992
sand	cooling pond site	1–2	Bugai et al., 1995a
sand	"Red Forest" site	0.5–2	Dzhepo et al., 1995

Table 5. *Literature data on ^{90}Sr distribution coefficients for sorption on argillaceous sediments and carbonate minerals (reducing conditions).*

	$^{90}SrK_d$, L/kg			
Sediment	Conservative (Minimal)	"Best Guess"	Maximal	Reference
Marl	1	5		Allard, 1985
Consolidated	0.5	10	50	Lever and Woodwark, 1990
Calcite	1		7	Torstenfelt et al., 1982
Calcite	0			Vandergraaf and Ticknor, 1994

failure. Also, the confined aquifer is modeled as a single homogeneous unit, whereas in reality it consists of two sub-units, separated by the clay layer. Site characterization data indicate that this layer is discontinuous in some areas near the Chernobyl site. Though several modeling assumptions tend to be conservative, every effort was made to avoid an unreasonable 'worst case' analysis. The next section demonstrates that despite these simplifications, model predictions are in qualitative agreement with some important hydrologic characteristics of the Pripyat Town water well system.

EVALUATION OF THE HYDROGEOLOGICAL MODEL

The simulated hydraulic head pattern and ground-water flow lines for the "no weak zone" scenario are shown in Figure 4. The flow lines are designated by the corresponding value of the normalized stream function $\Psi = 2\pi r_w \psi / Q_w (0 \leq \Psi \leq 1)$. For the "weak zone" scenario the flow pattern is altered only in a small zone near the well casing. In this latter scenario, because of the large hydraulic gradient between the aquifers, and the high permeability of the "weak zone," the ground-water flux from the unconfined aquifer to the confined aquifer is three orders of magnitude higher compared to that in the region beyond the weak zone (Figure 5). However, because of the extremely small area of the "weak zone" compared to the recharge area for the water well, the total flux through the "weak zone" constitutes only 0.6 percent of the well production rate.

To check the adequacy of the ground-water flow model, water travel times were calculated using Equation 12 (with $K_d = 0$) and compared to ground-water age estimates based on tritium measurements (Table 1). Results of the simulation (water travel times from the top of the unconfined aquifer to the water well for different flow lines) are shown in Figure 6. The horizontal axis, plotted in terms of the normalized stream function, can be translated into a radial distance from the well by comparison to Figure 4. Ground-water travel time to the well is represented as a sum of the travel times in the separate hydrogeological units. The data of Figure 6 do not account for the residence time in the unsaturated zone, which is estimated at about 1–2 years, and therefore is relatively unimportant. According to this model, one half of the discharge at the water well would be composed of water that entered the saturated zone from 12 to 50 years ago. This result is in qualitative agreement with tritium ground-water age estimates for wells #1, 8 and 15. Average water travel times (i.e., water travel time for the flow line Ψ

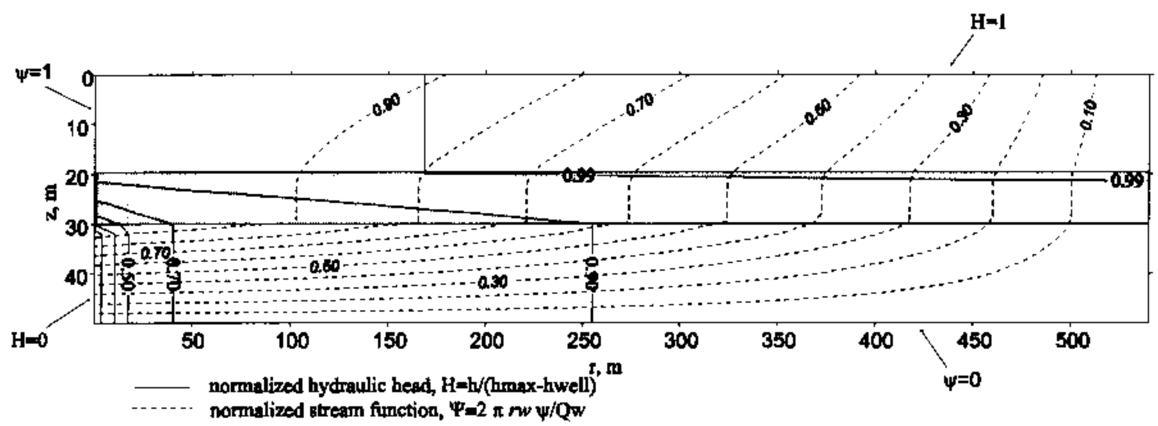

Figure 4. Ground-water flownet showing normalized values of hydraulic head and stream function, with no weak zone considered in the aquitard.

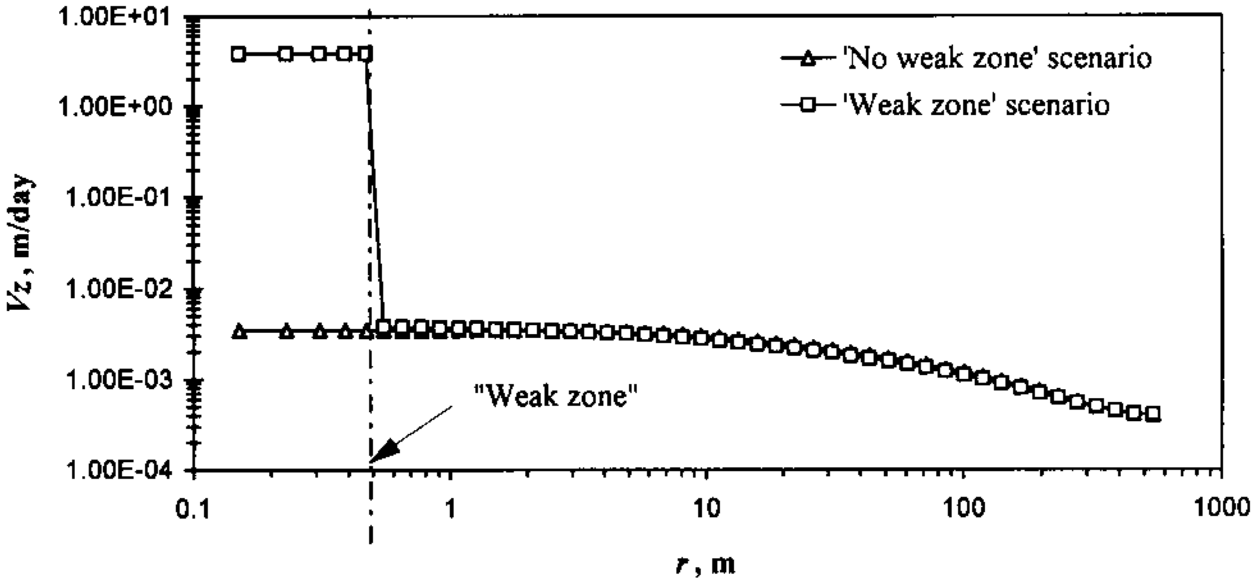

Figure 5. Vertical flux through the aquitard.

= 0.5) in the three units are: in the unconfined aquifer –30 years, aquitard –3 years, confined aquifer –17 years (totalling to 50 years).

Water travel time from the water table through the "weak zone" to the well is estimated from modeling to be about 6 years. This estimate indicates that a "weak zone" may explain the breakthrough of a minor amount of global ^{90}Sr fallout from the unconfined aquifer to the water wells, in case of a very low sorption of this radionuclide by sediments of the unconfined aquifer (i.e., assuming the ^{90}Sr K_d is equal to the lower value in Table 3). However, no conclusion about validity of the "weak zone" hypothesis can be made based on this result because no site-specific data on

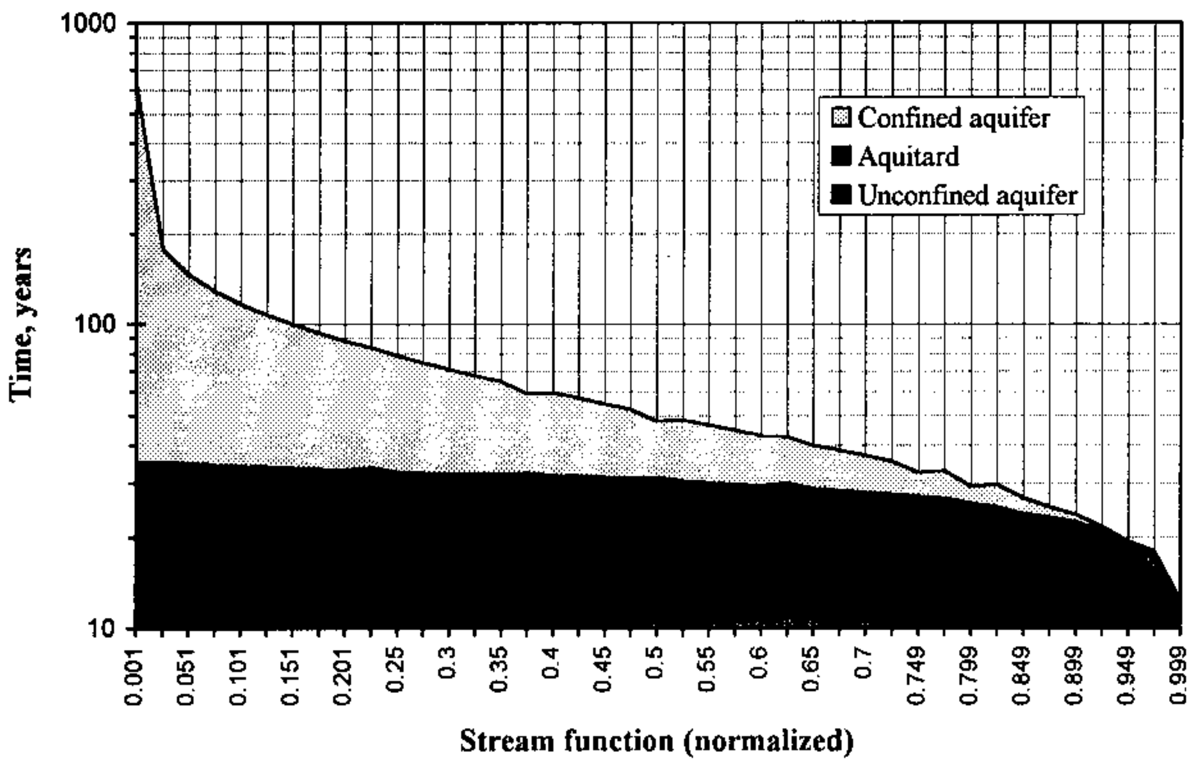

Figure 6. Ground-water travel time from the water table to the well, shown as a function of the different hydrogeologic units. Stream functions are shown on Figure 4.

pre-accident ^{90}Sr concentration in the unconfined aquifer are available.

SIMULATION OF ^{90}Sr MIGRATION TO THE WATER WELLS

A deterministic analysis of ^{90}Sr migration to the water wells using conservative parameters was carried out to first determine whether a more complex probabilistic assessment was warranted (eg. Parker, 1991). For this conservative analysis, the K_d's for ^{90}Sr were set to 0.5 L/kg (the minimum value from the probable range; see Table 5) for all hydrogeological units. The simulated ^{90}Sr concentration in the water well as a function of time is shown in Figure 7. The maximum ^{90}Sr concentration in the water well within the next 70 years is approximately $0.0025C_o$. As the maximum possible C_o value is estimated as 1,000 Bq/L, a well-field failure will not occur in the next 70 years if the present Ukrainian DWS for ^{90}Sr of 3.7 Bq/L remains unchanged. However, if the ^{90}Sr DWS were reduced to 0.3 Bq/L (a 50 percent probability scenario), this last standard may be exceeded by about one order of magnitude. An immediately useful estimate from this analysis is the identification of a 'threshold' value for the ^{90}Sr concentration in the source zone: it follows that for $C_0 < C_0^{(threshold)} = 120$ Bq/L, well-field failure will not occur for any combination of assumed hydrogeological and regulatory parameters.

Another important estimate obtained from the deterministic simulations are 'threshold' K_d's for the hydrogeological units listed in Table 6. If the K_d is greater than the threshold value, contaminant travel times to the water well will be longer than the 70 year time horizon of this study, and therefore the well field will not fail.

The probabilistic analysis was carried out using Monte-Carlo simulation techniques. A total of 50,000 realizations were generated, using the probability distributions assigned to the parameters listed in Table 3. Concentration-time histories at the well head were calculated for each of these realizations. For each realization, a failure occurred in year t if the concentration in the well exceeded the drinking water standard. The probability of failure in year t is calculated as the ratio of the number of realizations in which failure occurred in that year, to the total number of realizations generated. Figure 8 shows the conditional probabilities of well-field failure when $C_{DWS} = 0.3$ Bq/L, for the case where no weak zone exists through the aquitard, and the case where such a zone exists. The cumulative conditional probability of well failure during the next 70 years for the weak zone scenario is estimated to be approximately 30 percent, while for the scenario where the possibility of a weak zone is not considered, the probability of well-field failure is about 0.05 percent. The overall probability of well-field failure at any time during the 70 year time horizon is obtained by summing the conditional probabilities for well failure for each year (Figure 8), with each conditional probability multiplied by the respective probabilities assigned to the presence or absence of a weak zone around the wellbore. A similar procedure is applied to account for the uncertainty assigned to the value of the drinking water standard. As shown earlier in the deterministic analysis, the conditional probability of well failure is zero in the case where $C_{DWS} = 3.7$ Bq/L. Using the data in Figure 8, it would be simple to recalculate the probability of well-field failure for alternate estimates assigned to the probability that a weak zone is present in the aquitard.

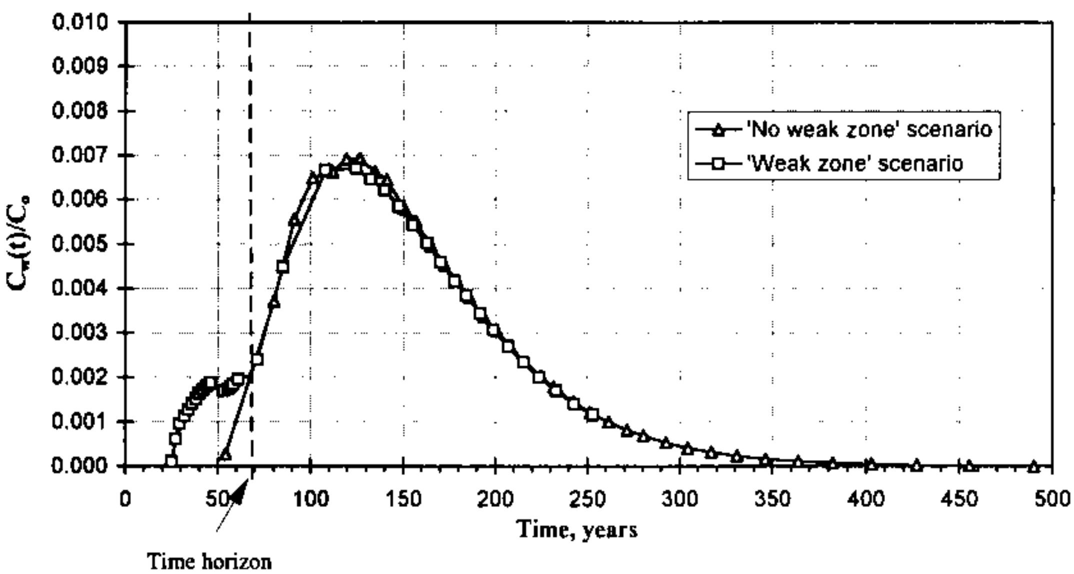

Figure 7. Conservative prediction of ^{90}Sr concentration in the water well.

Table 6. *Estimated 'threshold' K_d values (L/kg) the exceeding of which results in ^{90}Sr travel times to the water well greater than $T_{horizon}$ = 70 years.*

Hydrogeological Unit	"No Weak Zone" Scenario	"Weak Zone" Scenario
Unconfined aquifer	1	2.5
Aquitard	2	-

RISK OF WELL-FIELD FAILURE

Estimates for the probability of the well-field failure permit an evaluation of the risk-cost objective function for the 'no action' alternative (i.e., $\Phi^{(*)}$ given by Equation 4). The other important parameter necessary for developing an estimate for $\Phi^{(*)}$ is the cost of the well-field failure, C_f. Development of the firm estimates of failure costs in hydrogeological applications is a difficult task (Massman et al., 1991). In our analysis, we use an order of magnitude estimate for the failure costs. If we conservatively assume that failure of the Pripyat Town well field will require development of an equivalent ground-water supply consisting of 20 new wells exploiting the same confined aquifer, costs can be estimated as:

$$C_f = 20 \times \$40{,}000 \times 2 = \$1{,}600{,}000$$

where the first multiplier represents the number of water wells, the second term ($40,000) represents an estimate of the cost of installing one 80 m deep water well, and the last multiplier accounts for costs related to exploratory investigations, installation of new water supply pipelines, and similar additional costs.

Combining this C_f estimate with $P_f(t)$ estimates, and assuming a discount rate i of 10 percent results in:

$$\Phi^{(*)} \approx \$5{,}000$$

Obviously, any complex ground-water remediation measures at the Pripyat Town well field (e.g., reconstruction of drainage systems of 1986, excavation of contaminated soils), will require financial investments orders of magnitude greater than this conservatively estimated value of the objective function of $5,000. This result implies that a 'no complex remedial action' alternative represents the preferable management strategy for the Pripyat Town well field.

The conditional probability estimates shown in Figure 8 indicate that failure will not occur for at least 25 years. It is of interest to investigate the time dependence of the risk, which because of the effect of the discount rate, will vary with the extent of the period between t = 0, and the time when the probability of failure will no longer be negligible. The first column in Table 7 gives the risks of well-field failure if the calculation were to be performed 10 years in the future, and 25 years in the future. We assume that the uncertainty in model parameters remains at the present level. Risk in 25 years increases by about one order of magnitude compared to present value, reaching $51,000. Because the risk shows a tendency to increase significantly with time, it is recommended that the next several years be used to obtain more reliable

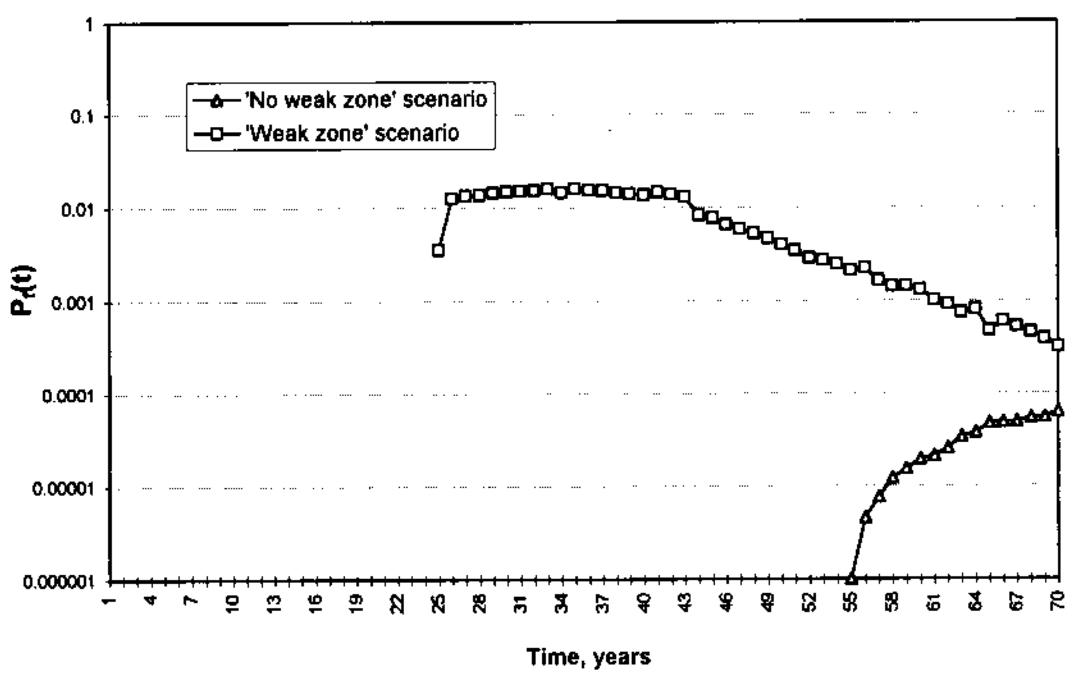

Figure 8. Results of the Monte Carlo simulation, showing the conditional probabilities of well-field failure for C_{DWS} = 0.3 Bq/L.

Table 7. *Current ($t = 0$) and projected (for $t = 10$ y and $t = 25$ y) values of the objective function for well-field failure.*

	Upper Limit for ^{90}Sr Concentration in the Unconfined Aquifer, C_0, Bq/L		
	1000	500	200
$t = 0$	$5,000	$2,000	$100
$t = 10$ y	$12,000	$5,000	$400
$t = 25$ y	$51,000	$23,000	$1,500

information on the parameters governing radionuclide migration to water wells. In particular, estimates of the ^{90}Sr concentration at the contaminant source, C_0, can be relatively easily refined based on ground-water monitoring studies. The upper limit of 1,000 Bq/L used in this analysis is probably too conservative because, as discussed earlier, it assumes the "worst case" hydrogeological conditions of radionuclide migration to ground water.

Columns two and three of Table 7 show the risk of well-field failure assuming the upper limit for C_0 is restricted through the ground-water monitoring studies to 500 Bq/L and 200 Bq/L respectively. A sharp decrease in the projected 25 year risk from $51,000 for $C_0 = 1{,}000$ Bq/L to $1,500 for $C_0 = 200$ Bq/L clearly indicates that there is some value in more precisely knowing the contaminant concentration in the unconfined aquifer.

CONCLUSIONS

The results indicate that the risk of contamination of the Pripyat Town well field near the Chernobyl Nuclear Power plant is low. The residence time of radiostrontium within the ground-water flow system appears long enough to permit decay of the ^{90}Sr concentrations to levels that do not lead to exceedence of the regulatory standard at the well head. A Monte-Carlo assessment indicates, however, that a well-field failure is possible within a 70 year planning horizon if several hydrogeological and regulatory conditions are satisfied simultaneously:

1. The sediments have a limited ability to sorb ^{90}Sr (in particular, the ^{90}Sr distribution coefficient for the unconfined aquifer is less than 2.5 mL/g).
2. The ^{90}Sr concentration in the contaminant source zone exceeds 120 Bq/L.
3. A facilitated transport zone exists in the aquitard around the wellbores.
4. The current Ukrainian ^{90}Sr drinking water standard is lowered by approximately one order of magnitude.

Because the above scenario is speculative (which is reflected in the value of the objective function for the "no-action" alternative), we believe that there is little technical basis at the present time to initiate any complex ground-water remediation measures for the Pripyat Town well field. Sensitivity analysis indicates that the risk will increase in 25 years by about one order of magnitude, assuming that uncertainties in model parameters remain at the present level. Therefore, it would be prudent to develop a monitoring system at the site, and to obtain more reliable estimates of the key contaminant transport parameters, in particular the ^{90}Sr concentration in the unconfined aquifer.

ACKNOWLEDGMENTS

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APPENDIX

The change in time of the surface radionuclide inventory, $A(t)$, is described by the equation:

$$\frac{dA}{dt} = -\lambda A - C(t)\,\varepsilon, \quad A(0) = A_0 \qquad \text{Eq. A1}$$

where the first term on the right hand side accounts for radioactive decay, and the second term accounts for the downward migration of the radionuclide with recharging water. Substituting Equation 9 for $C(t)$ in Equation A1 and solving the resulting ordinary differential equation for $A(t)$ gives:

$$A(t) = A_0\, e^{-\lambda t} - \varepsilon C_0\, e^{-\lambda t}\, t, \qquad \text{Eq. A2}$$

The duration of contaminant release T_s, which an initial radionuclide inventory A_0 is capable of sustaining, is determined from condition $A(T_s) = 0$, which yields:

$$T_s = \frac{A_0}{\varepsilon\, C_0} \qquad \text{Eq. A3}$$