The Human Health Effects of Radioactive Smoke from a Catastrophic Wildfire in the Chernobyl Exclusion Zone: A Worst Case Scenario[1]

Aaron Hohl[2], Andrew Niccolai[3], Chad Oliver[4], Sergiv Zibtsev[5], Johann Goldammer[6], Mykhaylo Petrenko[5], Vologymyr Gulidov[7]

June 17, 2011

[1] Please see Acknowledgements at end of text.
[2] Humboldt State University, Arcata, CA, USA
[3] Aviation Branch, U.S. Coast Guard R&D Center, USA
[4] Global Institute of Sustainable Forestry, School of Forestry and Environmental Studies, Yale University, New Haven, CT, USA
[5] National University of Life and Environmental Sciences of Ukraine, Kiev, Ukraine (NUBiP of Ukraine), Kiev, Ukraine
[6] Global Fire Monitoring Center, Freiburg University, Freiburg, Germany
[7] University of Edinburgh, Scotland

Table of Contents

Tables & Figures

ABSTRACT

The health implications of a potential catastrophic wildfire in the Ukrainian portion of the Chernobyl Exclusion Zone (CEZ) on populations living and working beyond the CEZ are assessed. The complete analysis consists of four linked sub-models: a source model, a transport model, an exposure model, and a cancer risk model. As a worst case scenario, it is assumed that a fire would consume the biomass of pine forests and former agricultural lands and release any associated radionuclides into the atmosphere. The transport model assumes that the wind would blow primarily towards Kiev throughout the fire event. The exposure model estimates adult and child (1 year old) external exposures and doses via the five exposure pathways: (1) external irradiation caused by immersion in a radioactive cloud during plume passage; (2) inhalation of radionuclides during plume passage; (3) external irradiation caused by deposited radionuclides on soil during the first year after wildfire; (4) ingestion of radionuclides in contaminated food during the first years after the wildfire, and (5) inhalation of resuspended radionuclides during the first year after the wildfire. Estimates of radionuclide releases, transport, exposures, and doses are based on conservative assumptions and consequently are likely to overestimate potential exposures to members of the general public during an actual wildfire event. Excluding the food ingestion pathways, calculated doses to populations at distances 30 km or greater from the release point are less than the critical thresholds that would require evacuations. However, Ukrainian law would require limiting ingestion of certain foodstuffs to avoid exposure through ingestion. The cancer risk model assumes that exposure through contaminated foodstuffs would be avoided.

INTRODUCTION

An accident occurred in reactor No. 4 of the Chernobyl nuclear power plant on April 26, 1986. The resulting explosions and subsequent fire in the plant released considerable quantities of radionuclides into the surrounding environment. Residents were permanently evacuated from a 30 km zone around the plant – the Chernobyl exclusion zone (CEZ) – which was determined to have especially high levels of contamination. This radioactive material has subsequently been incorporated into both the soil and the vegetation. Fires in the CEZ have been both frequent and widespread. From 1992 to 1994, 200 forest fires occurred in the CEZ (Budyka and Ogorodnikov 1995). Combustion of organic matter has been shown to lead to resuspension (Kashparov et al., 2000; Yoschenko et al., 2006a; Yoschenko et al., 2006b) and long range transport (Lujaniene et al., 2006) of radionuclides.

This paper analyses the potential adverse health effects that released radionuclides from a catastrophic wildfire within the CEZ would have on populations at different distances surrounding the exclusion zone.

BACKGROUND

A sample of the CEZ had been assessed for current and future potential fire risk using Ukrainian forest inventory, the LMS computer platform (Oliver et al. 2009), and both Ukrainian and United States forest fire risk assessments. Both the Ukrainian and U.S. fire risk assessments confirmed initial observations that much of the forest is in high danger of burning. Forest growth projections also confirmed that the fire risk would remain high without intervention, but could be reduced dramatically with appropriate silvicultural manipulations (McCarter et al. 2007).

The CEZ is 32% deforested and former agriculture areas, 38% Scots pine (*Pinus sylvestris*) forests, and 30% broadleaf (angiosperm) forests. It is largely on droughty glacial outwash, sandy soils. Seasonal droughts, overly crowded pine forests, and insects and pathogen infestations make the CEZ highly susceptible to wildfires. Insufficient forest management has also allowed the accumulation dead wood as fuel. Forest inventory data shows 15.3 thousand ha of forests in CEZ are damaged, including 5.3 thousand ha damaged by pests that are now very fire prone. An estimated 1.4 million cubic meters of dead wood has accumulated with the CEZ (State Forest Inventory). Within the forests are also contaminated machines and buried radioactive waste (Zibtsev et al. 2011).

There is concern that radionuclides in the smoke from a potential catastrophic fire could harm people directly from exposure and indirectly by contaminating food crops. Small fires have occurred within the CEZ; and there has been high concern of catastrophic fires there similar to fires that have occurred in the western United States during the past two decades and in Russia in the summer of 2010. Although few people work within the CEZ, villages and agriculture land surround it. The city of Kiev (population 2.7 million) is approximately 100 km southeast of Chernobyl, and Chernigiv (population 305,000) is approximately 100 km northeast of Chernobyl.

The analysis described in this paper is based on a generic screening model for use in assessing the impact of discharges of radioactive substances to the environment (IAEA, 2001). This generic model was selected because it offers a simplified and conservative assessment of the likely magnitude of a radioactive impact on a population. However, the model makes a number of simplifying assumptions which may not be appropriate for modeling transport of radionuclides during a wildfire. These assumptions are addressed in more detail in the discussion section of this report. The model accounts for all major pathways of radiation exposure and is purposefully conservative, reporting doses for cases that involve maximum exposure potential.

Transport of the discharged materials is considered through the atmosphere. Exposure pathways for external and internal mechanisms are systematically traced.

The nuclides of concern are: ^{90}Sr, ^{137}Cs, ^{154}Eu, ^{238}Pu, ^{239,240}Pu, and ^{241}Am. Independent estimates were not available for the inventory of ^{239}Pu and ^{240}Pu in the CEZ. The pooled inventory of $^{239, 240}$Pu is treated as a single isotope. ^{90}Sr and ^{137}Cs are the two most common radionuclides in the CEZ and, along with ^{154}Eu, have relatively high dose coefficients for external exposure pathways. Although they are less common, ^{238}Pu, ^{239,240}Pu, and ^{241}Am have high dose coefficients for internal exposure pathways (i.e. inhalation and ingestion). Standard dose coefficients for external exposure pathways have been adjusted to account for the ingrowth of daughters with a half-life of less than 30 minutes (IAEA 2001). Thus, the dose coefficient used for ^{90}Sr accounts for the contribution from ^{90}Y; the dose coefficient for ^{137}Cs accounts for the contribution from ^{137m}Ba.

The results are reported as the pathway-specific and total doses in Sieverts (Sv) exposed to an adult and child (1 y [1 year old]) during plume passage and for the first year after the event. Dose is a measure of energy deposited by radiation within a human target. The population of concern consists of the members of the public who share a relatively homogenous set of exposure pathways and typically are considered to receive the highest total dose from a given source of radioactivity. Individuals who are not in the direct centerline of the projected plume of radioactivity or who are impacted by fewer exposure pathways will likely receive lower doses. In this report, it is assumed that the total dose attributable to a catastrophic wildfire will be highest in the first year after the event. Consequently, exposure for subsequent years is not calculated.

The report does not directly address the potential exposure of personnel living and working within the CEZ itself. In particular, it does not address the exposure of fire fighters who might be called upon to contain a wildfire. Nor does the report address the consequences of Ukrainian and Belorussian portions of the CEZ burning simultaneously. Analysis of a broader catastrophic forest fire that would affect both countries is beyond the scope of this study.

METHODS

The analysis of health effects from a catastrophic forest fire is described in this paper. It consists of four, linked sub-models in which the results from one sub-model are the inputs to the next. The sub-models are: source model, transport model, exposure model, and cancer incidence and mortality model. The source, transport, and exposure models are likely to over-estimate potential exposure. The source model assumes the entire CEZ is burned in a very hot fire that consumes all wood of the trees—a very unlikely scenario. The conventional approach to account for exposures from multiple pathways is to sum up the individual pathway contributions. In reality it is unlikely that any one individual would receive maximum exposure to all exposure pathways. Finally, the additional risk of cancer incidence and cancer mortality attributable to the exposure through inhalation, immersion, and ground deposition is estimated. For reasons explained below, ingestion is not considered in the calculation of cancer incidence and mortality.

Source model

The inventory of radionuclides in combustible material is estimated as a function of the inventories of radionuclides known to be in the soil of the CEZ (Table 1). Kashparov et al. (2003) estimated the total inventory of fuel component radionuclides for the six radionuclides used in this study. Their study estimated the inventory in the upper 30-cm soil level in the Ukrainian portion of the CEZ in 2000. Their analysis did not include radioactive waste storage sites and cooling ponds. The inventory of radionuclides expected to be in the soil in 2010 is estimated as:

$$N_{i,\ 2010} = N_{i,2000} e^{-\lambda_i t} \quad [1]$$

where

$N_{i,\ 2010}$ is the amount of radionuclide i in the soil in 2010 (Bq),

$N_{i,\ 2000}$ is the amount of radionuclide i in the soil in 2000 (Bq),

λ_i is the decay constant of radionuclide (d^{-1}),

t is the number of days between 2000 and 2010 (d).

No attempt is made to account for losses through processes other than radioactive decay. For the purposes of this report, it is assumed that the radionuclides are distributed uniformly in the soils of different cover types; for example, former agricultural lands are assumed to have the same average concentration of radionuclides as pine forests.

Radionuclides in the litter layer and in aboveground biomass are assumed to be potentially combustible. Concentration factors are used to estimate inventories of radionuclides in potentially combustible material as a function of soil concentration. Estimates of radionuclide concentrations in soil, vegetation, and litter in two grassland plots and one forest plot in the CEZ for ^{90}Sr, ^{137}Cs, ^{238}Pu, and $^{239,240}Pu$ (Yoschenko et al. 2006b) are used to estimate concentration factors for those four nuclides in grassland and pine forest. In the case of the grassland plots, the concentration factor for each nuclide is taken to be the higher of the two possible concentration factors. The upper 95th percentile value for each concentration factor, which is calculated based on propagated error terms, is used as the concentration factor for this analysis (Table 2). The concentration factor for ^{241}Am is assumed to be twice that for $^{239,240}Pu$ (Sokolik et al. 2004). The concentration factor for ^{154}Eu is assumed to be equal to that for $^{239,240}Pu$ (Lux et al. 1995).

It is assumed that the 32% of the CEZ classified as deforested/former agricultural areas and the 38% of the CEZ classified as pine forests could burn. Total inventory of radionuclide i in combustible material in 2010 is estimated as:

$$N_{i,comb2010} = \sum_{l=1}^{n} N_{i,2010}\, CF_{i,l} L_l \qquad [2]$$

where

$N_{i,comb2010}$ is the total inventory of radionuclide i in combustible material in the CEZ (Bq),

$N_{i,2010}$ is the inventory of radionuclide i in the soil in 2010 (Bq),

$CF_{i,l}$ is the concentration factor of nuclide i in land class l,

L_l is the proportion of the CEZ in landclass l.

Transport model

The primary means of transporting radioactive material through the environment in the event of a catastrophic wildfire would be atmospheric discharge. The discharged radioactive material would then be dispersed by means of a radioactive plume and finally be deposited on ground and water surfaces.

Atmospheric discharge

It is assumed that all vegetation and litter in both pine forests and former agricultural land in the Ukrainian portion of the CEZ would burn over a five day period. The total discharge of nuclide i to the atmosphere is assumed to be $N_{i,comb2010}$. The rate of atmospheric discharge (Qi), measured in Bq/s, is calculated as the total amount of the nuclide for the year 2010 divided by the time period of the wildfire event (sec). Because the model assumes steady state meteorological conditions for the duration of the fire, the length of time during which the fire burns does not affect the results. Thus, changing the duration of the fire from five days to 30 days would change the rate of discharge, but not the total discharge nor the pattern of dispersal.

The atmospheric discharge is treated as a point source and its trajectory is modeled using a Gaussian plume model. Treating the discharge as a point source is a simplifying assumption. Since it treats the full inventory of radionuclides as concentrated in a single point, it will tend to overestimate the air concentration both above that point and along the path of the plume. The wind is assumed to blow towards Kiev at 2 m/s for the entire duration of the wildfire. The windspeed is the default recommended by IAEA SRS-19 (2001).

As formulated in the IAEA-SRS19 model, dispersion or the average air concentration of a radionuclide during the event (C_A) at a given distance is independent of deposition velocity. Thus, the model does not take into account depletion of the plume because of deposition to the ground. C_A measured at a given distance from the source, is calculated as:

$$C_A = \frac{P_p F Q_i}{u_a} \quad [3]$$

where

C_A is the ground level air concentration at downwind distance *x* (Bq/m^3),

P_p is the fraction of time per event that the wind blows toward the target population,

Qi is the average discharge rate per event for radionuclide i (Bq/s),

u_a is the geometric wind speed average at the area of release representative of the duration of the event (m/s),

F is the Gaussian diffusion factor (m^{-2}).

The Gaussian diffusion factor assumes a neutral atmospheric stability class (Pasquill-Gifford stability class D) and is calculated as:

$$F = \frac{12}{\sqrt{2\pi^3}} * \frac{\exp\left[-\left(\frac{H^2}{2\sigma_z^2}\right)\right]}{x\sigma_z} \quad [4]$$

where

H is the release height (m)

x is the downwind distance (m),

σ_z is the vertical diffusion parameter (m)

Emission height is assumed to be 0 m. At the distances with which we are concerned, the release height has a negligible effect on dispersion pattern. The vertical diffusion parameter is calculated as:

$$\sigma_z \frac{=(0.06)(x)}{\sqrt{1+(0.0015)(x)}} \quad [5]$$

Ground concentration

For this model, it is assumed that the ground surface is represented by an infinite plane upon which all radionuclide deposition activity is uniformly distributed (IAEA 2001). The infinite plane model for estimating the dose from ground deposition is chosen because of the limited duration of the wildfire event for downward migration of radionuclides. Radionuclide concentration on the ground at a distance x from the source of emission is calculated as:

$$C_{gr} = \frac{d_i\left[1-e^{-\lambda_{E_i^s} t_b}\right]}{\lambda_{E_i^s}} \quad [6]$$

where

C_{gr} is the deposition density of radionuclide *i* (Bq/m^2)

t_b is the duration of the wildfire (d),

$\lambda_{E_i^s}$ is the effective rate constant for reduction of the activity in the top layer of the soil (d^{-1}), calculated by adding the radioactive decay constant for radionuclide *i* with the rate constant for reduction of soil activity owing to processes other than radioactive decay,

d_i is the total ground deposition rate (Bq/m^2/d), calculated as:

$$d_i = (V_d)C_A \qquad [7]$$

where

V_d is the deposition coefficient (deposition velocity) for a given radionuclide *i* (1000 m/d),

C_A is the radionuclide concentration in the air obtained from Equation [3] (Bq/m^3).

As recommended in IAEA (2001) deposition velocity is assumed to be 1000 m/d. The model assumes that deposition velocity does not vary with distance. In an experimental forest fire in the CEZ Yoschenko et al. (2006) found that total deposition velocity was high near the fire because of the rapid settling of large particles (e.g., partially burned pieces of organic matter). At distances of several hundred meters, deposition velocity was less than 1000 m/d. It is likely that 1000 m/d overestimates the deposition velocity one would encounter in a real fire. This depositional velocity analysis is a part of the model that could be refined.

Air concentration of resuspended material

Resuspension of radionuclides previously deposited on ground surfaces can be an additional source of exposure through inhalation even after the initial release has stopped. Airborne concentration of radionuclides in the year after the fire is calculated as:

$$C_{AR} = KC_{gr} \qquad [8]$$

where

C_{AR} is the concentration in the air attributable to resuspension (Bq/m^3)

K is an the resuspension factor (Bq/m^3 per Bq/m^2)

C_{gr} is the deposition density of radionuclide i (Bq/m^2)

Data collected after the initial Chernobyl accident indicated that: 1) the resuspension factor tended to decline over time (Garger et al 1999); and, 2) there was a negative correlation between the initial deposited concentration and the local resuspension factor (IAEA 1992). However, Garger et al. (1997) found that estimating the initial value for K following the Chernobyl release was dependent on a substantial amount of subjective estimation and was associated with a high level of uncertainty. The uncertainty in the value of K could be decreased by averaging experimental data over time. Following the initial release of radioactivity from Chernobyl in April and May of 1986, the resuspension factor for areas throughout Europe corresponding to mid-June 1986 ranged from $3.6*10^{-9}$ in highly contaminated areas to $4.9*10^{-8}$ in more lightly contaminated areas (IAEA 1992). Although higher resuspension factors were recorded for certain locations for brief periods of time, for this analysis, it is assumed that the average resuspension factor was $4.9*10^{-8}$ for the entire year following the fire.

Exposure model

Five exposure pathways are modeled for six nuclides: (1) external irradiation caused by immersion in a radioactive cloud during plume passage (plume immersion); (2) inhalation of radionuclides during plume passage (plume inhalation); (3) external irradiation caused by deposited radionuclides on soil during the first year after wildfire (groundshine); (4) ingestion of radionuclides in contaminated food during the first years after the wildfire (ingestion), and (5) inhalation of resuspended radionuclides during the first year after the wildfire (resuspension inhalation).

Exposures via inhalation and immersion during plume passage are transient; they cease to be factors after the plume has passed. Exposures via the other three pathways are assumed to occur for the full year following the wildfire.

Plume inhalation

The internal dose from an intake of radioactive material into the body following inhalation depends in part on the age and metabolism of the individual as well as the physicochemical behavior of the radionuclide under consideration. For most radionuclides, dose coefficients are available for materials with three different types of absorption characteristics (fast, medium, slow). The maximum dose coefficient is used to calculate the committed dose (ICRP 1996). Additionally, this study differentiates between children at one year of age and adults in terms of differences in dose coefficients and inhalation rates. The dose coefficients assume a 50-year dose commitment for adults and a 70-year dose commitment for children. The model assumes that both groups will be exposed to the ambient air concentration for the full duration of the wildfire event.

The committed effective dose from inhalation for both adults and children after exposure to radionuclide transportation from a catastrophic wildfire in the CEZ are calculated as:

$$E_{inh} = C_A R_{inh} DF_{inh} \quad [9]$$

where

E_{inh} is the committed effective dose (Sv),

C_A is the radionuclide concentration in the air obtained from Equation [3] (Bq/m^3),

R_{inh} is the inhalation volume during the wildfire event (m^3),

DF_{inh} is the inhalation dose coefficient (Table 3; Sv/Bq).

For adults, R_{inh} is 115 m^3 or $\frac{8400\ m^3/y}{365\ d/y} * 5\ d$. For children, R_{inh} is 19 m^3 or $\frac{1400\ m^3/y}{365\ d/y} * 5\ d$ (IAEA 2001).

Plume immersion

Calculations of the effective dose from immersion in the discharge plume are based on the semi-infinite cloud model which assumes that radiation from the plume cloud is in a state of radiative equilibrium. This assumption implies that the energy absorbed by a given volume within the cloud is the equivalent of that energy emitted by the same cloud volume. This model has been widely used and includes provisions for partial shielding of the plume cloud by impervious surfaces such as the side of a building. However, the instantiation of the model presented here does not incorporate the effect of buildings. As with inhalation, the model assumes that both groups will be exposed to the ambient air concentration for the full duration of the wildfire event and that ambient air concentration will return to normal immediately following the event. In practice, most individuals will not remain exposed to the plume cloud for the duration of the wildfire event.

The effective dose from immersion in the atmospheric plume is calculated as:

$$E_{im} = C_A DF_{im} O_f \qquad [10]$$

where

E_{im} is the effective dose from immersion (Sv),

C_A is the radionuclide concentration in the air obtained from Equation [3] (Bq/m^3),

DF_{im} is the effective dose coefficient for immersion (Table 3; Sv/y per Bq/m^3),

O_f is the fraction of the year for which the population is exposed to this plume (O_f=0.014y^{-1} or 5d/365d/y).

Groundshine

The radioactive material deposited to the ground is assumed to linger for the entire year. Individuals are assumed to be exposed to surface deposits for the entire year. In practice, individuals may be exposed to a lower level during the time they spend indoors or outside of the region contaminated by the plume.

The effective dose from ground deposition is calculated as follows:

$$E_{gr} = C_{gr} DF_{gr} O_f \quad [11]$$

where

E_{gr} is the effective dose from ground deposition (Sv),

DF_{gr} is the dose coefficient for exposure to ground deposits (Table 3; Sv/y per Bq/m^2),

O_f is the fraction of the year for which the population is exposed to this pathway ($Of=1y^{-1}$ or 365d/365d/y),

Cgr is the deposition density of radionuclide *i* (Bq/m^2), obtained from Equation [6].

Ingestion

The food chain models assume that the population is exposed to radionuclides through ingestion of crops, meat, and milk products that have been exposed to atmospheric discharges. Much like the rates of atmospheric inhalation, the ingestion of vegetation, meat, and milk is highly variable within a population; however conservative estimates of annual consumption rates for adults and children are available (Table 4). The general calculation of the committed effective dose from consumption of radionuclide *i* in foodstuff *p* is:

$$E_{ing,p} = C_{p,i} H_p DF_{ing} \quad [12]$$

where

$E_{ing,p}$ is the committed effective dose from consumption of radionuclide *i* in foodstuff *p* (Sv),

H_p is the total amount on an individual foodstuff consumed in the first year following the wildfire event (kg), calculated as the product of the consumption rate (kg/y; Table 4) and one year of intake (y),

DF_{ing} is the dose coefficient for ingestion of radionuclide *i* (Sv/Bq),

$C_{p,i}$ is the concentration of radionuclide *i* in foodstuff *p* at the moment of consumption (Bq/kg).

The calculation for $C_{p,i}$ is a function of discharge method; radionuclide characteristics; and methods of cultivation, irrigation, foraging, and grazing. As such, separate models for calculating radionuclide concentration are needed for vegetation, meat, and milk. The models are outlined here. Details of the individual $C_{p,i}$ models can be found in *Section 5* of IAEA SRS No. 19.

Radionuclides intercepted and preserved by vegetation may result from deposition from atmospheric fallout, precipitation rainout, or irrigation with contaminated water. A percentage of these external deposits become incorporated into vegetation through foliar absorption or root uptake. Radioactive decay, growth dilution, non-contaminated water wash-off, and soil fixation can eventually lead to reductions in the radionuclide concentration within vegetation. The model estimates the exposure that would occur over the course of the year following the wildfire if one were to eat only crops grown on soil contaminated as the radioactive plume passed by. Element-specific transfer factors are used which take into account both uptake from soil and soil adhesion to the surface of plants (Table 5).

The intake of radionuclides by animals depends on the size, species, age, feed material, and milk yield. Element-specific transfer factors are used to account for the transfer from feed to milk and meat products (Table 5). For this study, it is assumed that the meat from animals originated as cattle byproducts and that the cattle grazed on pasture with soil contaminated by the plume during the grazing season. The concentration of radionuclides in the milk is dependent upon the radioactivity concentration in the feed consumed by the milk-producing animals. This study uses values specific to dairy cows; however, the values are also applicable to other lactating animals without significantly underestimating the radioactive concentration in those milk products.

Resuspension inhalation

The committed effective dose from inhalation of materials resuspended after plume passage is calculated in a similar manner to the committed effective dose from inhalation during plume passage. The same dose coefficients are used as for inhalation during plume passage. For both adults and children the committed dose is calculated as:

$$E_{inhR} = C_{AR} R_{inh} DF_{inh} \quad [13]$$

where

E_{inhR} is the committed effective dose (Sv),

C_{AR} is the concentration in the air attributable to resuspension obtained from Equation [8],

R_{inhR} is the inhalation volume for the year following the wildfire event (m^3),

DF_{inh} is the inhalation dose coefficient (Table 3; Sv/Bq).

Total Dose

The total dose of the population (Sv) for a given radionuclide *i* is finally calculated as the sum of the potential dose pathways given in Equations [9, 10, 11, 12 and 13]:

$$E_{tot,i} = E_{inh} + E_{im} + E_{gr} + E_{ing,p} \qquad [14]$$

Then the total dose for all radionuclides considered is calculated as follows:

$$\sum E_{tot,i} \qquad \text{for all } i \text{ radionuclides} \qquad [15]$$

Cancer incidence and mortality model

The risk of developing cancer and the risk of dying from cancer as a result of exposure to the radionuclides of concern through the five modeled pathways are estimated. For these calculations, it is assumed that highly contaminated food would not be consumed. Lifetime attributable risk of cancer incidence and cancer mortality is modeled as a function of age at time of exposure, sex, and dose. The estimated number of additional cancer cases per 100,000 population exposed to 0.1 Sv was reported by the Committee to Assess Health Risks from Exposure to Low Levels of Ionizing Radiation (2006; Table 6). The Committee's preferred model assumes a linear relationship of risk between the actual exposure and the calculated exposure values. Thus, additional cancer incidence can be calculated as:

$$M_{D,a,s} = \frac{LAR_{a,s}}{0.1/D} \qquad [16]$$

where,

$M_{D,a,s}$ is the additional risk of mortality per 100,000 people of a given sex (s) who are a given age (a) at the time of exposure to an expected dose (D).

$LAR_{a,s}$ is the Lifetime attributable risk for 100,000 people of a given sex (s) who are a given age (a) at the time of exposure to a one time dose of .1 Sv, and

D is the estimated total dose from all exposure pathways.

RESULTS

A catastrophic wildfire event in the Exclusion Zone surrounding Chernobyl would release airborne radioactive materials that may adversely impact the health of people living downwind of the contaminated smoke plume. Table 2 shows the estimated inventories (in Bq) of ^{90}Sr, ^{137}Cs, ^{154}Eu, ^{238}Pu, ^{239,240}Pu, and ^{241}Am in potentially combustible materials within the Ukrainian portion of the CEZ for the year 2010. The total amount of radioactivity that could potentially be released into the environment in the event of a catastrophic wildfire is estimated to be 4×10^{14} Bq in the vegetation and forest floor litter layer.

Table 7 presents the estimated activity concentrations of each radionuclide in the air, ground, and food products at 30, 50, 100 and 150 km downwind of the release point. As expected based on the Gaussian plume model, the estimated activity concentrations of all radionuclides at the plume centerline decrease with increasing downwind distances. Table 8 presents estimates of the radionuclide specific activity concentrations in contaminated crops as a function of downwind distance. It shows that, for all radionuclides at all distances, direct deposition of airborne radionuclides is the primary mode of crop (and forage) contamination by a very large margin. In this study, it is assumed that crops and forage exposed directly to the plume would not be consumed. Consumption of crops directly exposed to the plume could have large health consequences.

Figure 1 shows the pathway-specific doses (in Sv) summed across all radionuclides as a function of distance from the center of the CEZ along the plume centerline. For children (1 y [1 year old]), ingestion is the exposure pathway that contributes most to the total dose, followed by plume inhalation. For adults plume inhalation contributes slightly more than ingestion. Figure 2 shows the total doses with and without ingestion for children (1 y) and adults. At 100 km (i.e., the approximate distance to Kiev), the adult exposure though pathways other than ingestion during the first year after the event is $3.5x10^{-3}$ Sv (3.5 mSv). Ingestion is responsible for an additional Sv $5.9x10^{-3}$ Sv (5.9 mSv) during that first year. For children, the equivalent figures are $1.6x10^{-3}$ Sv (1.6 mSv) and $5.5x10^{-3}$ Sv (5.5 mSv).

The additional risk of cancer incidence and mortality for males and females exposed through pathways other than ingestion at distances of 30, 50, 100 and 150 km are given in Table 9. If we assume that children would not be permitted inside of the CEZ itself, the highest calculated risk is to 20 year old women residing at 30 km from the center of the CEZ. Their additional lifetime risk of dying from cancer would be 170 per 100,000. The additional lifetime risk of dying of cancer for 20 year old men residing at 30 km from the center of the CEZ would be 110 per 100, 000. The additional lifetime risk for a 20 year old adult women residing in Kiev would be 27 per 100,000; for men it would be 18 per 100,000.

DISCUSSION

Calculated doses and safety context

According to the United Nations Scientific Committee on the Effects of Atomic Radiation, the worldwide average background dose is 2.4 mSv/y, but ranges from 1-10 mSv/y (UNSCEAR, 2000). For a limited number of people living in known high background radiation areas of the world, doses can exceed 20 mSv/y; and there is no evidence that this poses a health risk. Bennett et al. (2000) estimated that, between 1986 and 1995, the total arithmetic mean effective dose (excluding thyroid doses) received by the population of areas of Ukraine contaminated by the Chernobyl was 11 mSv. The International Commission on Radiological Protection's current dose limits for occupational and public exposures for application to regulated sources in planned exposure situations are 20 mSv/y, when averaged over five years, and 1 mSv/y, respectively (ICRP, 2007).

The Ukrainian government has adopted safety norms to govern the level of intervention as a function of the prevented dose (Law of Ukraine, 1991). Populations should be evacuated if the prevented dose in the first two weeks exceeds 50 mSv. Time spent outdoors should be limited if the prevented dose in the first two weeks exceeds 1 mSv for children and 2 mSv for adults. Resettlement should occur if the prevented dose for the first 12 months exceeds 50 mSv or if the prevented dose during the resettlement exceeds 200 mSv. Temporary resettlement should occur if the average prevented dose exceeds 100 mSv or if the average monthly dose for the resettlement period exceeds 5 mSv per person.

Total doses from pathways other than ingestion at locations outside of the CEZ are moderately high, but do not rise to the level that mandatory evacuation or temporary resettlement would be required under Ukrainian law. For adults, the estimated total dose from plume immersion and inhalation during the fire itself plus resuspension inhalation and ground exposure in the year subsequent to the fire ranges from 22 mSv for those residing at the edge of the exclusion zone (30 km) to 3.5 mSv for people residing in Kiev (100 km) to 1.9 mSv for those residing 150 km from the center of the CEZ (Figure 2). For children (1 y) the equivalent figures are 10 mSv, 1.7 mSv and 0.9 mSv. These doses generally exceed the ICRP dose limits for public exposures in planned exposure situations but are generally less than the limits set for occupational exposure. Since a large proportion of the dose is attributable to plume inhalation, efforts to avoid direct exposure to the plume would be prudent.

The potential dose derived from the consumption of contaminated foodstuffs could exceed acceptable levels. The Ukrainian government calls for limitations on the consumption of foodstuff if the prevented internal irradiation dose exceeds 5 mSv or if the prevented average annual dose exceeds 1 mSv. For both adults and children these levels could be exceeded by consuming food produced at distances up to 150 km from the center of the CEZ. Limitations on the consumption of milk is called for if the radioactive contamination by ^{137}Cs exceeds 100 Bq/l or if the contamination by ^{90}Sr exceeds 20 Bq/l for adults or 5 Bq/l for children. The limits for other foodstuffs are 200 Bq/kg for ^{137}Cs and 40 Bq/kg (adults) or 10 Bq/kg (children) for ^{90}Sr. Foodstuffs produced on land directly along the trace of the plume could exceed the acceptable level of ^{90}Sr at distances as great as 150 km (Table 7). Thus, consumption of certain foodstuffs would be banned by the government. For this reason, the dose attributable to ingestion was not used to calculate cancer incidence or mortality.

It is important to note that the highest levels of contamination would occur directly along the trace of the plume. As one moved away from the trace, contamination levels would decline. Consequently, the actual amount of agricultural land that would need to be taken out of production would be limited. An analysis of the area of land that could be affected is important but beyond the scope of this study.

Model assumptions and limitations

All models represent abstractions of reality and cannot capture the full complexity of natural systems. Simplifying assumptions must be made both when data is not available and when the dynamics of the system being studied are not fully understood. The model used here consists of four linked sub-models in which the results from one sub-model are the inputs to the next.

The model that forms the basis for the estimates presented here, IAEA-SRS-19 (IAEA 2001) is a screening model for estimating the release, transport, exposure, and doses from radionuclides released into the environment. It is intended to run without a lot of site specific data. Instead, most parameter values given in the IAEA report are intentionally very conservative and the model is designed to over-estimate the dose that is likely to be received. If the estimated total doses contributed by all radionuclides through all exposure pathways is less than the acceptable numerical dose limit, one may conclude that the actual total dose will likely be lower. On the other hand, if the estimated dose is greater than the level of concern, then a more refined model may be needed to determine whether actual total dose is likely to exceed an acceptable level.

This analysis made a number of additional conservative assumptions that are likely to lead to an over-estimation of the dose that would be received in the event of a wildfire in the CEZ. The most important of these are outlined here. First, instead of using the inhalation dose coefficients contained in IAEA-SRS20, which are ICRP-recommended default values for inhalation dose coefficients, this analysis used the most conservative inhalation dose coefficient given in ICRP publication 72 (ICRP 1996). As a result, the calculated inhalation doses reported here are more than twice what they would be if the default inhalation dose coefficients had been used.

Second, upper 95th percentile concentration factors were used to calculate the inventory of radionuclides in combustible material. As a result, the calculated inventory is twice what it would have been had mean concentration factors been used.

Third, it was assumed that the all pine forests and former agricultural land in the CEZ would burn in a single year and that the entire inventory of radionuclides in combustible material would be released. Assuming complete combustion of all potentially combustible products in both forest and agricultural lands is extremely conservative and is unlikely to occur in reality.

First, fires tend to be patchy and do not consume all vegetation or litter in their path (Madoui et al 2010). Second, tree trunks are unlikely to be completely consumed by even high-severity wildfires (North and Hurteau 2011). This incomplete combustion is important because in a study on the resuspension and redistribution of radionuclides during forest fires in the CEZ, Yoschenko et al (2006b) found that more than 40% of the ^{137}Cs in combustible material was contained in timber. Approximately 8% of the ^{90}Sr was located in timber. Finally, the entire CEZ is unlikely to burn completely in any one year. However, large fires are possible; in 1992, 17,000 ha within the CEZ burned over a two week period (Zibtsev et al. 2011). The assumption of complete combustion that was done in this analysis is consistent with a worst case scenario.

Finally, the ingestion model makes one assumption that is conservative and one assumption that is not. The calculation of the dose attributable to ingestion assumes that all food consumed by a person at a given distance from the center of the CEZ would be produced in that location. It is unlikely that an individual would consume only food produced on land lying directly along the trace of the plume. To the extent that foodstuffs produced away from the trace of the plume were consumed, committed dose from ingestion would be lower than reported here. On the other hand, it is also assumed that vegetation directly exposed to deposition from the plume would not to be consumed at all. Instead, the doses reported here are based on soil uptake and adhesion rather than deposition. Consuming crops exposed to direct deposition could lead to a much higher committed dose than is reported here (Table 8). The analysis presented here assumes that the Ukrainian government would be able to move quickly to restrict consumption of vegetation contaminated through direct deposition.

The Gaussian plume model, which was used to model atmospheric transport, makes several simplifying assumptions which may not hold during a wildfire. It assumes steady-state meteorological conditions over long distances; continuous and uniform emissions of radionuclides; and plume geometry in which lateral and vertical concentrations profiles follow a normal distribution. Although IAEA SRS-19 does not recommend using the Gaussian plume model at distances greater than 20 km, Lutman et al. (2004) compared a simple Gaussian dispersion model for predicting long-range dispersion (up to 1700 km) to a more physically realistic, but computationally complex, Lagrangian dispersion model. They found that the differences between the two models were small compared to the expected precision of the models and that the Gaussian plume model over-estimated, rather than underestimated, environmental concentrations. A review by Miller and Hively (1987) found that a Gaussian plume model was widely used to estimate airborne radionuclide exposures within 80 km of a release point and could be used to predict annual average air concentrations over flat terrain within a factor of 2 to 4. That said, a more refined analysis could be conducted using a Lagrangian puff model (e.g., CALPUFF, Scire et al 2000) or a Eulerian grid model. Such models can take into account time- and space-varying meteorological conditions. Notably, they may be more appropriate for modeling short duration releases of radionuclides than the Gaussian plume model. However, these models are demanding of computer resources; and parameterizing such a model was beyond the scope of this project.

Given the assumptions and model limitations discussed here, there are several areas in which further analyses may be warranted. First, the point source model presented here could be replaced with a two-dimensional model that accounts for the distribution of radionculides across the landscape. At the same time, the analysis could be expanded to include contaminated zones within Belarus. Second, the likely absorption characteristics of materials released during a fire could be investigated and that information could be incorporated into the selection of inhalation dose coefficients. Third, the Gaussian dispersion model could be replaced with a Lagrangian dispersion model. These refinements would result in more realistic estimates of total dose that are likely to be less than the estimates of total dose reported here. Finally, additional analysis should be conducted to assess the likely health effects of a fire in the CEZ on those working to control the fire.

CONCLUSIONS

A catastrophic wildfire in the Ukrainian portion of the CEZ which completely consumed the vegetation and litter in former agricultural lands and pine forests could release approximately 4×10^{14} Bq of radioactive material. A screening model using conservative assumptions was used to estimate exposure through plume immersion and plume inhalation during the fire itself and resuspension inhalation and ground exposure in the year following the fire. The estimated exposure of populations 30 or more kilometers from the source of the fire through these three pathways (22 mSv) is below the critical thresholds that would require evacuations. Since the estimated total ingestion doses to a child (1 y) and adult were found to exceed acceptable levels, it is likely that the Ukrainian government would restrict intakes of contaminated vegetation, meat, and milk indefinitely. Although uncalculated, it is likely that doses to people living and working in the CEZ would exceed acceptable levels.

ACKNOWLEDGEMENTS

We thank Dr. V.A. Kashparov and Dr. V.I. Yoschenko of the Ukrainian Institute of Agricultural Radiology, and Dr. Yeter Goksu, Adiyaman University, Turkey, for critically reviewing previous drafts of this report.

We also thank Dr. Goksu also for her advice throughout the project and during preparation of the report.

This report would not have been possible without the support of Dr. Dmytro Melnychuk, Rector, National University of Life and Environmental Sciences of Ukraine (NUBiP of Ukraine) and Mr. George Chopivsky, Jr., President, Chopivsky Family Foundation for his advice, encouragement, and financial support.

We also thank the following people for critical reviews of a previous draft of this paper. Their reviews will be posted on the web site of the Yale University Global Institute of Sustainable Forestry (http://environment.yale.edu/gisf/):

--Dr. Albert Lee, Director of Nuclear Study and Analysis, Office of Chief Engineer, Atomic Energy of Canada Limited, Mississauga, Ontario, Canada

--Dr. Akira Osawa, Professor, Division of Forest and Biomaterials Sciences, Graduate School of Agriculture, Kyoto University, Sakyo-Ku, Kyoto, Japan

--Dr. Jerome S. Puskin, Director, Center for Science and Technology, Radiation Protection Division, Environmental Protection Agency, Washington, D.C., USA

--Dr. A.N. Razdayvodin, Head, Department of Radioactive Ecology of Forests;

--Dr. A.I. Radin, Radiation Control Laboratory Chief;

--Dr. D.Y. Romashkin, Radioecology Laboratory Chief;

Forest Radioecology Department of Federal State Institution, Russian Research Institute for Silviculture and Mechanization of Forestry (VNIILM), Moscow Region, Russia

--Dr. Chris G. Whipple, ENVIRON International Corporation, Emeryville, CA, USA

--Dr. David J. Winfield, Vienna, Austria

Table 1. Estimated fuel component radionuclides in soil and vegetation of the 30-km Chernobyl exclusion zone in Ukraine in 2000 and 2010 outside the ChNPP industrial site and also excluding activity in the radioactive waste storages and the cooling pond. Fuel component radionuclides in 2000 in upper 30-cm soil layer are from Kashparov et al. (2003).

Radionuclide	Radionuclide Inventory (Bq)			Concentration Factor	
	Soil in 2000	Soil in 2010	Combustible in 2010	Forest	Grassland
^{90}Sr	7.7E+14	6.1E+14	1.8E+14	0.69	0.10
^{137}Cs	2.8E+15	2.2E+15	2.1E+14	0.20	0.062
^{154}Eu	1.4E+13	6.4E+12	2.4E+11	0.060	0.048
^{238}Pu	7.2E+12	6.7E+12	2.3E+11	0.060	0.038
^{239,240}Pu	1.5E+13	1.5E+13	5.7E+11	0.060	0.048
^{241}Am	1.8E+13	1.8E+13	6.2E+12	0.12	0.96

Table 2. Estimated mean and upper 95th percentile concentration factors for forests and grasslands in the CEZ. Inventory data and concentration factors for ^{90}Sr, ^{137}Cs, ^{238}Pu and ^{239,240}Pu were calculated based on Yoschenko et al. (2006b). Concentration factors for ^{154}Eu and ^{241}Am were derived from Lux et al. (1995), Sokolik et al. (2004).

	Total inventory (GBq)	Total combustible (Gbq)	Concentration Factor	Upper 95th percentile
		Forest		
^{90}Sr	14.8±4.5	5.2±1.9	0.35±0.17	0.69
^{137}Cs	16.7±3.3	1.8±0.7	0.11±.047	0.20
^{154}Eu				0.060
^{238}Pu	89±21	2.7±1.2	0.030±.015	0.060
^{239,240}Pu	190±46	6.0±2.3	0.032±.014	0.060
^{241}Am				0.12
		Grassland		
^{90}Sr	16±12	0.57±0.30	0.035±0.033	0.10
^{137}Cs	28±17	0.64±0.39	0.023±0.020	0.062
^{154}Eu				0.048
^{238}Pu	180±110	2.6±1.5	0.014±0.012	0.038
^{239,240}Pu	370±210	5.8±4.8	0.016±0.016	0.048
^{241}Am				0.96

Table 3. Effective immersion, surface, inhalation, and ingestion dose coefficients for various radionuclides (IAEA 2001).

Radionuclide	Immersion (Sv/y per Bq/m^3)	Surface (Sv/y per Bq/m^2)	Inhalation (Sv/Bq)		Ingestion (Sv/Bq)	
			Adult	Child (1-2 y)	Adult	Child (1-2 y)
^{90}Sr	3.1E-09	3.5E-09	1.6E-07	4.0E-07	2.8E-08	7.3E-08
^{137}Cs	8.7E-07	1.8E-08	3.9E-08	1.0E-07	1.3E-08	1.2E-08
^{154}Eu	2.0E-06	3.8E-08	5.3E-08	1.5E-07	2.0E-09	1.2E-08
^{238}Pu	1.7E-10	2.9E-11	1.1E-04	1.9E-04	2.3E-07	4.0E-07
^{239,240}Pu	1.6E-10	2.8E-11	1.2E-04	2.0E-04	2.5E-07	4.2E-07
^{241}Am	2.6E-08	8.9E-10	9.6E-05	1.8E-04	2.0E-07	3.7E-07

Table 4. Ingestion of food stuffs per year (IAEA 2001).

	Intake per person	
Ingestion	Adult	Child (1 y)
Fruit, vegetables and grain (kg/y)	410	150
Milk (L/y)	250	300
Meat (kg/y)	100	40

Table 5. Element-specific transfer factors for terrestrial foods for screening purposes. The values for milk and meat represent the fraction of the animal's daily intake of the radionuclide that appears in each liter of milk or kg of meat (IAEA 2001).

Element	Forage (Bq/kg plant dry weight)/ (Bq/kg soil dry weight)	Crops (Bq/kg plant fresh weight)/ (Bq/kg soil dry weight)	Milk (d/L)	Meat (d/kg)
Sr	10	0.3	0.003	0.01
Cs	1	0.04	0.01	0.05
Eu	0.1	2.0E-03	6.0E-05	2.0E-03
Pu	0.1	1.0E-03	3.0E-06	2.0E-04
Am	0.1	2.0E-03	2.0E-05	1.0E-04

Table 6. Lifetime attributable risk of cancer incidence and cancer mortality per 100,000 people exposed to a single dose of 0.1 Sv (Committee to Assess Health Risks from Exposure to Low Levels of Ionizing Radiation, 2006).

Age at time of exposure	Incidence (occurrences/ 100,000 people)		Mortality (occurrences/100,000 people)	
	Female	Male	Female	Male
0	4777	2563	1770	1099
20	1646	977	762	511
40	886	648	507	377
60	586	489	409	319
80	214	174	190	153

Table 7. Estimated concentrations of radioactive materials in the environment after a catastrophic wildfire.

Radionuclide	Distance (km)	Air Concentration (Plume) (Bq/m^3)	Ground Concentration (Bq/m^2)	Air Concentration (Resupension) (Bq/m^3)	Food Concentration (Bq/kg)		
					Vegetation	Meat	Milk
^{90}Sr	30	36	1.8E+05	8.7E-03	210	1600	660
	50	16	8.2E+04	4.0 E-03	95	760	300
	100	5.8	2.9E+04	1.4 E-03	33	270	110
	150	3.2	1.6E+04	7.7 E-04	18	150	58
^{37}Cs	30	47	2.4E+05	1.2 E-02	36	1100	290
	50	22	1.1E+05	5.3 E-03	17	500	130
	100	7.7	3.8E+4	1.9 E-03	5.9	180	47
	150	4.2	2.1E+4	1.0 E-03	3.2	96	26
^{154}Eu	30	4.8E-02	240	1.2E-05	1.9 E-03	4.4 E-03	1.8 E-04
	50	2.2E-02	110	5.5E-06	8.6 E-04	2.1E-03	8.3E-05
	100	7.9E-03	39	1.9E-06	3.0E-04	7.3E-04	2.9E-05
	150	4.3E-03	21	1.1E-06	1.7 E-04	3.9E-04	1.6E-05
^{238}Pu	30	4.6E-02	230	1.1E-05	8.9 E-04	4.3E-04	8.6E-06
	50	2.1E-02	110	5.3E-06	4.10E-04	2.0E-04	4.0E-06
	100	7.6E-03	38	1.9E-06	1.50E-04	7.0E-05	1.4E-06
	150	4.1E-03	21	1.0E-06	7.90E-05	3.8E-05	7.6E-07
^{239,240}Pu	30	0.11	570	2.8E-05	2.2 E-03	1.1E-03	2.1E-05
	50	5.3E-02	260	1.3E-05	1.0 E-03	4.9E-04	9.7E-06
	100	1.9E-02	93	4.6E-06	3.6 E-04	1.7E-04	3.4E-06
	150	1.0E-02	51	2.5E-06	1.90E-04	9.3E-05	1.9E-06
^{241}Am	30	1.2	6200	3.1E-04	4.8 E-02	20	5.3
	50	0.58	2900	1.4E-04	2.2 E-02	9.2	2.4
	100	0.20	1000	5.00E-05	7.8 E-03	3.2	0.86
	150	0.11	550	2.70E-05	4.2 E-03	1.8	0.47

Table 8. Estimated concentration of radioactive material in crops. Deposition is the concentration on plant surfaces estimated immediately after a catastrophic wildfire. Soil uptake and adhesion is estimated for the growing season immediately following a catastrophic wildfire.

		Crop Contamination (Bq/kg)	
Radionuclide	Distance	Deposition	Soil Uptake and Adhesion
^{90}Sr	30	47000	210
	50	22000	95
	100	7700	33
	150	4200	18
^{137}Cs	30	63000	36
	50	29000	17
	100	10000	5.9
	150	5600	3.2
^{154}Eu	30	64	1.9 E-03
	50	30	8.6 E-04
	100	10	3.0E-04
	150	5.7	1.7 E-04
^{238}Pu	30	62	8.9 E-04
	50	28	4.1 E-04
	100	10	1.5 E-04
	150	5.5	7.9E-05
$^{239,240}Pu$	30	150	2.2 E-03
	50	70	1.0 E-04
	100	25	3.6 E-04
	150	13	1.9 E-04
^{241}Am	30	1700	4.8 E-02
	50	770	2.2 E-02
	100	270	7.8 E-03
	150	150	4.2 E-03

Table 9. Lifetime attributable risk of cancer incidence and mortality per 100,000 people for various levels of exposure.

Distance (km)	Dose (mSv)	Age at time of exposure	Incidence (occurrences/100,000 people)		Mortality (occurrences/100,000 people)	
			Female	male	female	male
25	10	0	490	260	180	110
	22	20	370	220	170	110
	22	40	200	140	110	84
	22	60	130	110	91	71
	22	80	48	39	42	34
50	4.8	0	230	120	85	53
	10	20	170	100	78	52
	10	40	91	66	52	39
	10	60	60	50	42	33
	10	80	22	18	19	16
100	1.7	0	80	43	30	18
	3.5	20	58	35	27	18
	3.5	40	31	23	18	13
	3.5	60	21	17	15	11
	3.5	80	8	6	7	5
150	0.91	0	44	23	16	10
	1.9	20	32	19	15	10
	1.9	40	17	13	10	7
	1.9	60	11	9	8	6
	1.9	80	4	3	4	3

Figure 1. Estimated dose from individual exposure pathways as a function of distance from the center of the CEZ. Doses for plume inhalation, resuspension inhalation, and ingestion are differentiated between adult and child (1y [1 year old]).

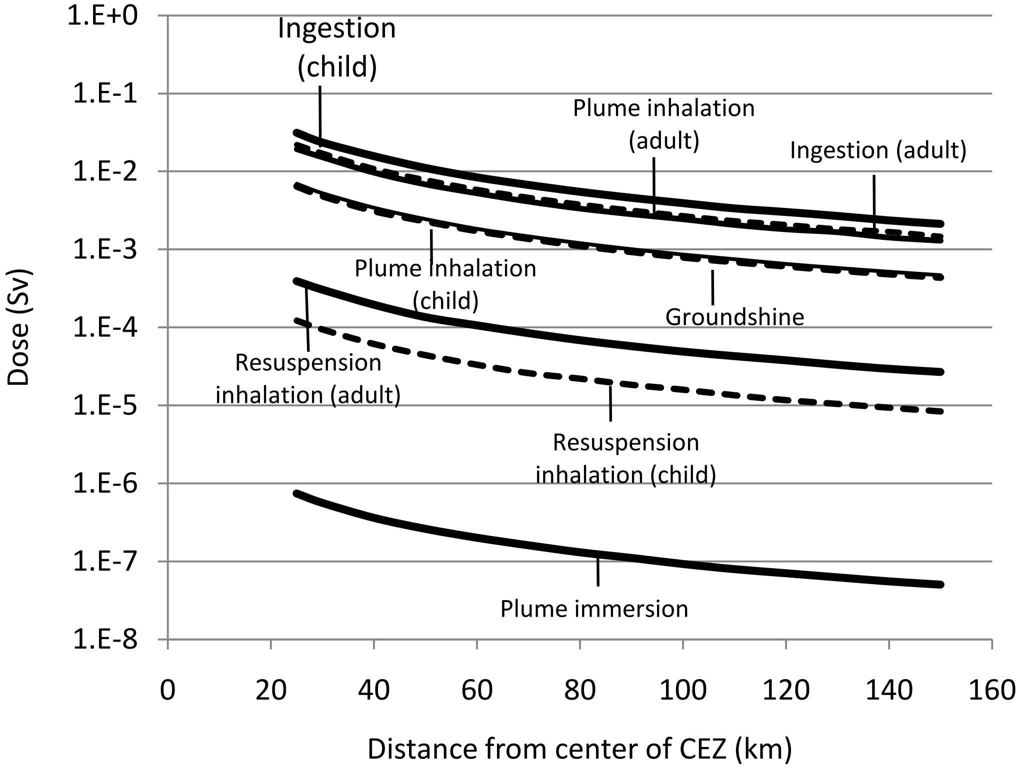
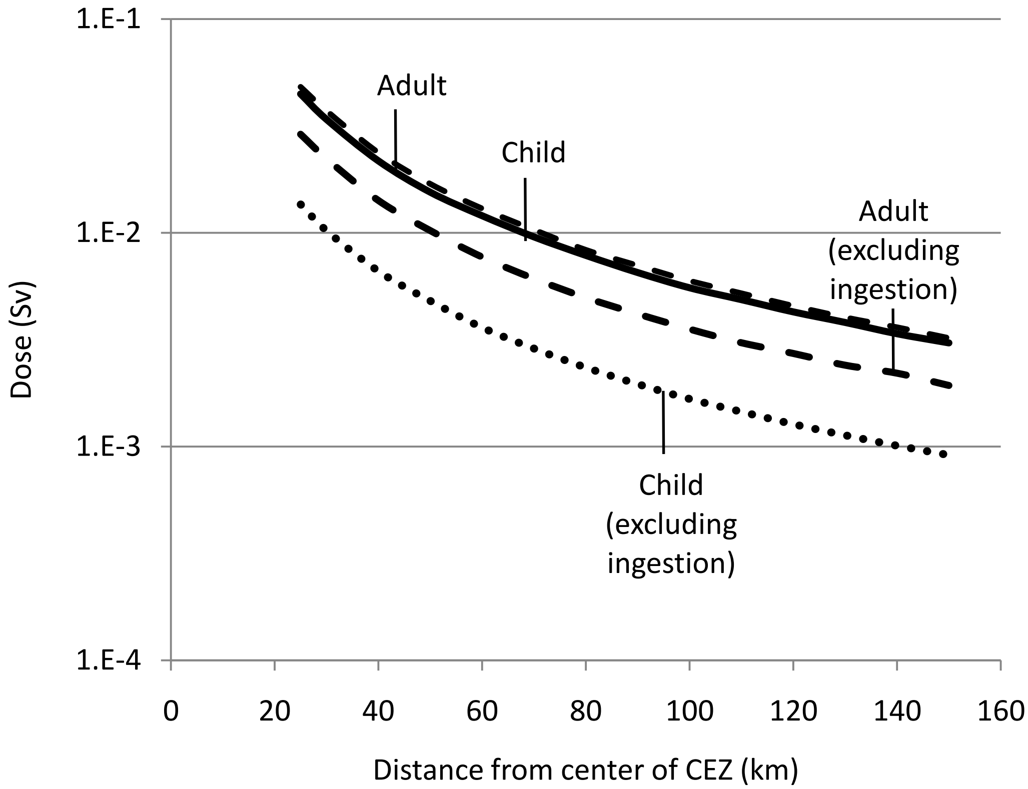


Figure 2. Estimated total dose (with and without ingestion), as a function of distance from the center of the CEZ, that could be received by children (1 y [1 year old]) and adults during the year following a catastrophic wildfire.



Literature cited

Bennett, B., Boubille, A., Hall, P., Sabkin, M. and Storm, H. Chernobyl accident: exposures and effects. Procedings of the 10th International Congress of the International Radiation Protection Associatation (IRPA-10). Paper T-12-1. Hiroshima, Japan: May 14-19. 2000.

Budyka, A.K., Ogorodnikov, B.I., 1995. Radioactive aerosols formed owing to fire in regions contaminated by products of Chernobyl accident. Radiation Biology. Radioecology 35:100-112 (in Russian).

Committee to Assess Health Risks from Exposure to Low Levels of Ionizing Radiation. (2006). Health risks from exposure to low levels of ionizing radiation: Beir VII, Phase 2. Washington: National Academies Press.

Garger, E.K., Hoffman, F.O., and Thiessen, K.M. (1997). Uncertainty of the long-term resuspension factor. Atmospheric Environment 31:1647-1656.

Garger, E.K., Hoffman, F.O., and Thiessen, K.M., Galeriu, D., Kryshev, A.I. et al. (1999). Test of exisiting mathematical models for atmospheric resuspension of radionuclides. Journal of Environmental Radioactivity 42:157-175.

IAEA. (1992). Modelling of resuspension, seasonality and losses during food processing. First report of the VAMP Terrestrial Working Group. International Atomic Energy Agency, IAEA-TECDOC-647. Vienna, Austria: International Atomic Energy Agency.

IAEA. (2001). Generic models for use in assessing the impact of discharges of radioactive substances to the environment (No. SRS-19). Vienna, Austria: International Atomic Energy Agency.

ICRP. (1996). Age-dependent doses to members of the public from intake of radionculides. Part 5: Compilation of ingestion dose coefficients. ICRP publication 72. Annals of the ICRP 26 (1).

ICRP. (2007). The 2007 recommendations of the International Commission on Radiological Protection. ICRP Publication 103. Annuals of the ICRP 37 (2-4).

Kashparov, V. A., Lundin, S. M., Kadygrib, A. M., Protsak, V. P., Levtchuk, S. E., Yoschenko, V. I., et al. (2000). Forest fires in the territory contaminated as a result of the Chernobyl accident: radioactive aerosol resuspension and exposure of fire-fighters. Journal of Environmental Radioactivity, 51(3), 281-298.

Kashparov, V. A., Lundin, S. M., Zvarych, S. I., Yoshchenko, V. I., Levchuk, S. E., Khomutinin, Y. V., et al. (2003). Territory contamination with the radionuclides representing the fuel component of Chernobyl fallout. Science of the Total Environment, 317(1-3), 105-119.

Law of Ukraine. (1991). On legal regime of the territories contaminated by the Chernobyl disaster (N 795-XII). Kiev: Ukraine.

Lujaniene, G., Sapolaite, J., Remeikis, V., Lujanas, V., Jermolajev, A., & Aninkevicius, V. (2006). Cesium, Americium and Plutonium Isotopes in Ground Level Air of Vilnius. Czechoslovak Journal of Physics, 56, D55-D61.

Lutman, E.R., Jones, S.R., Hill, R.A., McDonald, P. and Lamgers, B. (2004). Comparison between the prediction of a Gaussian plume model and a Lagrandian partile dispersion model for annual average calculations of long-range dispersion of radionuclides. Journal of Environmental Radioactivity 75: 339-355.

Lux, D., Kammerer, L., Ruhm, W., & Wirth, E. (1995). Cycling of Pu, Sr, Cs, and other longliving radionuclides in forest ecosystems of the 30-km zone around Chernobyl. Science of the Total Environment, 173(1-6), 375-384.

Madoui, A, Luduc, A, Gauthier, S. and Bergeron, Y. (2010). Spatial pattern analyses of post-fire residual stands in the black spruce boreal forest of western Quebec. International Journal of Wildland Fire 19:1110-1126.

McCarter, James, Sergiy Zibtsev, Aaron Hohl, Mykhaylo Petrenko. 2007. Analysis of Wildfire Likelihood and Opportunities for Mitigation in the Chernobyl Irradiated Forests. Working Report. Global Institute of Sustainable Forestry, School of Forestry and Environmental Studies, Yale University, New Haven, CT, U.S.A. 12 pp.

Miller, C.W.; Hively, L.M (1987). A review of validation studies for the Gaussian plume atmospheric dispersion model. Nuclear Safety. 28: 522-531.

North, M.P and Hurteau, M.D. 2011. High-severity wildfire effects on carbon stocks and emissions in fuels treated and untreated forest. Forest Ecology and Management 261:1115-1120.

Oliver, C.D., J. B. McCarter, J. M. Comnick, K. Ceder, C. S. Nelson, (2009) in *Models for Planning Wildlife Conservation in Large Landscapes*. Academic Press, San Diego: 339-366.

Sansone, U., Belli, M., Voitsekovitch, O. V., & Kanivets, V. V. (1996). Cs-137 and Sr-90 in water and suspended particulate matter of the Dnieper River reservoirs system (Ukraine). Science of the Total Environment, 186(3), 257-271.

Scire, J.S., Strimaitis, D.G., and Yamartino, R.J. (2000). A User's Guide for the CALPUFF Dispersion Model (Version 5). Concord, MA: Earth Tech, Inc.

Sokolik, G. A., Ovsiannikova, S. V., Ivanova, T. G., & Leinova, S. L. (2004). Soil-plant transfer of plutonium and americium in contaminated regions of Belarus after the Chernobyl catastrophe. Environment International, 30(7), 939-947.

UNSCEAR. (2000). Sources and effects of ionizing radiation (Vol. Volume I: Sources). New York: United Nations.

WHO. (2005). WHO Mortality Database. Retrieved from http://www.who.int/healthinfo/morttables/en/

Yoschenko, V. I., Kashparov, V. A., Levchuk, S. E., Glukhovskiy, A. S., Khomutinin, Y. V., Protsak, V. P., et al. (2006a). Resuspension and redistribution of radionuclides during grassland and forest fires in the Chernobyl exclusion zone: part II. Modeling the transport process. Journal of Environmental Radioactivity, 87(3), 260-278.

Yoschenko, V. I., Kashparov, V. A., Protsak, V. P., Lundin, S. M., Levchuk, S. E., Kadygrib, A. M., et al. (2006b). Resuspension and redistribution of radionuclides during grassland and forest fires in the Chernobyl exclusion zone: part I. Fire experiments. Journal of Environmental Radioactivity,86(2), 143-163.

Zibtsev, Sergiy, Chadwick Dearing Oliver, Johann Georg Goldammer, Aaron Hohl, James McCarter, Andrew Niccolai, Mykhaylo Petrenko, Olexandr Borsuk. 2011. Wildfires Risk Reduction From Forests Contaminated by Radionuclides: A Case Study of the Chernobyl Nuclear Power Plant Exclusion Zone. Proceedings of the 5th International Wildland Fire Conference - 'Wildfire 2011', Sun City / Pilanesberg National Park, South Africa, 9-13 May 2011.