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Predicted cumulative dose to firefighters and the offsite public from natural and anthropogenic radionuclides in smoke from wildland fires at the Savannah River Site, South Carolina USA

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ABSTRACT

The contaminated ground surface at Savannah River Site (SRS) is a result of the decades of work that has been performed maintaining the country's nuclear stockpile and performing research and development on nuclear materials. The volatilization of radionuclides during wildfire results in airborne particles that are dispersed within the smoke plume and may result in doses to downwind firefighters and the public. To better understand the risk that these smoke plumes present, we have characterized four regions at SRS in terms of their fuel characteristics and radiological contamination on the ground. Combined with general meteorological conditions describing typical and extreme burn conditions, we have simulated potential fires in these regions and predicted the potential radiological dose that could be received by firefighting personnel and the public surrounding the SRS. In all cases, the predicted cumulative dose was a small percent of the US Department of Energy regulatory limit (0.25 mSv). These predictions were conservative and assumed that firefighters would be exposed for the duration of their shift and the public would be exposed for the entire day over the duration of the burn. Realistically, firefighters routinely rotate off the firefront during their shift and the public would likely remain indoors much of the day. However, we show that even under worst-case conditions the regulatory limits are not exceeded. We can infer that the risks associated with wildfires would not be expected to cause cumulative doses above the level of concern to either responding personnel or the offsite public.

1. Introduction

Wildfires can volatilize both natural and anthropogenic radionuclides. Man-made radionuclides in wildland fuels contaminated by nuclear releases are of great concern (Pazukhin et al., 2004; Hao et al., 2009). These radionuclides can be present in concentrated amounts due to aerial transport and deposition or as the result of releases to surface waters that are subsequently re-distributed in wetlands and forests downstream. The typical anthropogenic radionuclides of concern include radioisotopes of cesium (^{134,137}Cs), plutonium ($^{238/239}$Pu), uranium (^{234,235}U), and strontium ($^{89/90}$Sr). The most common anthropogenic contaminant found in the environment is usually ^{137}Cs (Paller et al., 2014). This is because ^{137}Cs has a relatively long physical half-life of 30.2 years, a high fission yield, and a high bioavailability-due to its physiological similarity to potassium. Certain natural radionuclides and their daughter products such as polonium (^{210}Po), radium (^{226,228}Ra), uranium (^{233,234,238}U), thorium (^{228,230,232}Th), lead (^{210,212}Pb), beryllium (^{7}Be), and potassium (^{4} K) are found in surface fuels throughout the world as a result of natural processes following cosmic ray interactions with particles in the atmosphere and decay of primordial geologic elements such as thorium and uranium (Sugihara et al., 1999; Persson and Holm, 2011; Hejl et al., 2013). Radionuclides can bio-accumulate in vegetation and then be re-cycled through litter fall to form components of the dead and live fire fuels. The natural radionuclides of greatest concern are those that are 1) common and long-lived, 2) easily volatilized, 3) have high energy particle emissions which can damage tissues, and 4) can accumulate in soft tissue (e.g. ^{210}Po) and bones (e.g. ^{226}Ra).

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In general, in most parts of the world, the individual natural radionuclides and man-made radionuclides from fall-out exist in the environment at concentrations too low to be a health risk relative to particulate matter (Paatero et al., 2009). However, during wildfires and prescribed fires, large quantities of radionuclides may be volatilized, or become attached to airborne particulates by the combustion process, and are detectable in smoke (Commodore et al., 2012; Volkerding, 2003). When radionuclides are considered individually, the concentration thresholds for a dose that exceeds the established worker limits are uncommon except for extreme values observed in wildland fuels contaminated in nuclear incidents (Viner et al., 2015). To date, no assessment has been made of cumulative dose to firefighters or to the public from natural radionuclides in conjunction with anthropogenic radionuclides from wildfires.

A method was recently developed to model emission, exposure, and dose to firefighters for any individual radionuclide during fires knowing only certain basic terms like fuel load and consumption, fire spread, the properties of the radionuclides, and their concentrations in the fuel (Viner et al., 2015). These components can then be coupled to a firefighter's physical location on the fire line, breathing rate, and shift length. The potential dose to a firefighter or a member of the public is a consequence of the cumulative dose from all radionuclides released during a fire, which can be calculated using this method. However, worldwide measurements contain very few observations of more than one radionuclide in wildland fuels either man-made or natural (Hejl et al., 2013; Viner et al., 2015). No published observations exist of a complement of man-made and dominant natural radionuclides in wildland fuels from which cumulative dose can be estimated. Our objective is to determine cumulative dose from both naturally occurring and man-made radionuclides and their relative contribution to the total dose to firefighters. From these results, we can 1) test the hypothesis that naturally occurring radionuclides are relatively small contributors to potential doses and 2) determine how fire dynamics may influence cumulative radiological concentrations in smoke and potential doses to firefighters and the public.

2. Methods

2.1. Forest and fire history

We estimate the potential doses to onsite firefighters and to the offsite public during wildfires for both anthropogenic and natural radionuclides present in forest surface fuels in areas contaminated from cooling water discharges at the Savannah River Site (SRS), South Carolina, USA. SRS is a large (800 km^2) U.S. Department of Energy nuclear facility located in the southeastern USA and it was built in the 1950's as part of the U.S. Cold War effort (Kilgo and Blake, 2005). During SRS's operational history, four major riparian or floodplain zones that flow through the Site were contaminated with various radionuclides as a result of discharges from nuclear processing facilities (Carlton, 1998). The forests that occupy the riparian areas of Fourmile Branch, Pen Branch, Steele Creek and Lower Three Runs were the focus of this study (Fig. 1).

The affected forest contains primarily loblolly pine (*Pinus taeda* L.), with some mixed hardwoods that include oaks (*Quercus* spp.), gums (*Nyssa* spp.), ash (*Fraxanus* sp.), and poplar (*Lirodendron* sp.). These riparian forests have been harvested sparingly since the Site was established in the early 1950's. However, from 1954 to 1988, portions of the flood plain forest vegetation were killed by hot water effluents from reactor facilities. These areas have either regenerated naturally over the last several decades or were planted (Barton et al., 2000). In these forests, the wildland fuels present to carry a fire are primarily litter, twigs, and branches. Live woody shrubs and grasses are present but are a small component of the available fuels to carry fires. No periodic prescribed burning has been conducted in these areas, except a single prescribed fire in 1994 in Pen Branch to clear selected areas prior to tree planting. The recent fire history of these areas is represented by occasional wildfires ignited by lightning. Fire control activities within these areas are limited because the soils are contaminated with radionuclides and disturbance is avoided to comply with the US Code of Federal Regulations (10 CFR 835). In the past, extensive wildfires have burned through the Savannah River swamp, adjacent flood plain and riparian forest during dry periods.

2.2. Radionuclide activity in fuels

The 1999 gamma spectrographic overflight (Aerial Radiological Survey by EG and G Energy Measurements, Bechtel Nevada, Remote Sensing Laboratory) for Fourmile Branch, Pen Branch, Steel Creek, and Lower Three Runs were overlaid with topographic information from a 2009 Light Detection and Ranging (LiDAR) overflight project and the most current aerial photos. The gamma overflight signal measures primarily surface ^{137}Cs contamination levels. We identified 48 paired sample points (contaminated, non-contaminated) in the study. Twelve systematically spaced sample locations along each of the four streams were located in February 2013 (Fig. 2). At each location, the contaminated sample point was confirmed with a surface radiological scanner using an Electra Plus portable survey instrument (NE Technology Model Electra Plus with Alpha-Beta-Gamma Detector), such that the observed gamma activity was the highest in the local vicinity. It was then marked and mapped using a global positioning system estimate. An adjacent non-contaminated paired sample point was similarly confirmed at the nearest point away from the stream such that the forest conditions, soils and topography were similar. It was then marked and mapped. For non-contaminated sampling locations, all field survey readings were less than ambient instrument background (about 2500 dpm). For the selected contaminated sampling locations, the field survey readings were between ambient background and up to 8000 dpm. We used sampling procedures similar to previous studies on SRS for wildland fuels to obtain a sufficient mass of surface fuel material of litter, small twigs, and branches (Maier et al., 2004). Beginning in March and ending in April 2013, four subsamples were obtained around each sample point from a 1 m^2 area plus an additional sample was taken at the central point. The material was placed in sample bags, labeled and sent to the SRS Environmental and Bioassay Laboratory for analysis. The composited vegetation fuel sample was dried in a 105 °C oven overnight until completely dry. The dried sample was then blended to create a homogeneous mixture. Each vegetation sample was analyzed for the reported radionuclides shown in Table 1 using the associated radioanalysis. An environmental laboratory method unique to SRS was instituted in analyzing the vegetation samples. This method has been published and recognized nationally (Maxwell et al., 2010).

For actinide analysis of vegetation fuel samples, an aliquot of 10–20 g of the dried and blended sample was measured into 250 mL zirconium crucibles. Standards were added to adjust for recovery efficiencies during processing. Crucibles were briefly dried on the hotplate and samples were then placed in a 200 °C muffle furnace and ramped to 600 °C for 2–4 h. Samples were removed from the furnace and allowed to cool. Concentrated nitric acid and hydrogen peroxide was added, 5 mLs each, and samples were carefully evaporated to dryness on a hotplate. Crucibles were placed back in the 600 °C furnace for 5–10 min or until the ash solids were white. Crucibles were then removed from the furnace and allowed to cool. Once cool, 15 g of sodium hydroxide was used for the fusion. Samples were covered and fused for 15 min. After fusion, the crucibles were removed from the furnace and allowed to cool. Water was then added to dissolve this fusion cake and the samples were transferred to 225 mL centrifuge tubes. All solids were dissolved by adding more water and heat to ensure complete dissolution. A final crucible rinse with 6 M nitric acid further removed any actinides from the crucible. Ferric nitrate (iron carrier) and lanthanum nitrate were added to the samples in the centrifuge tubes and the samples were diluted with DI water and cooled to room temperature.

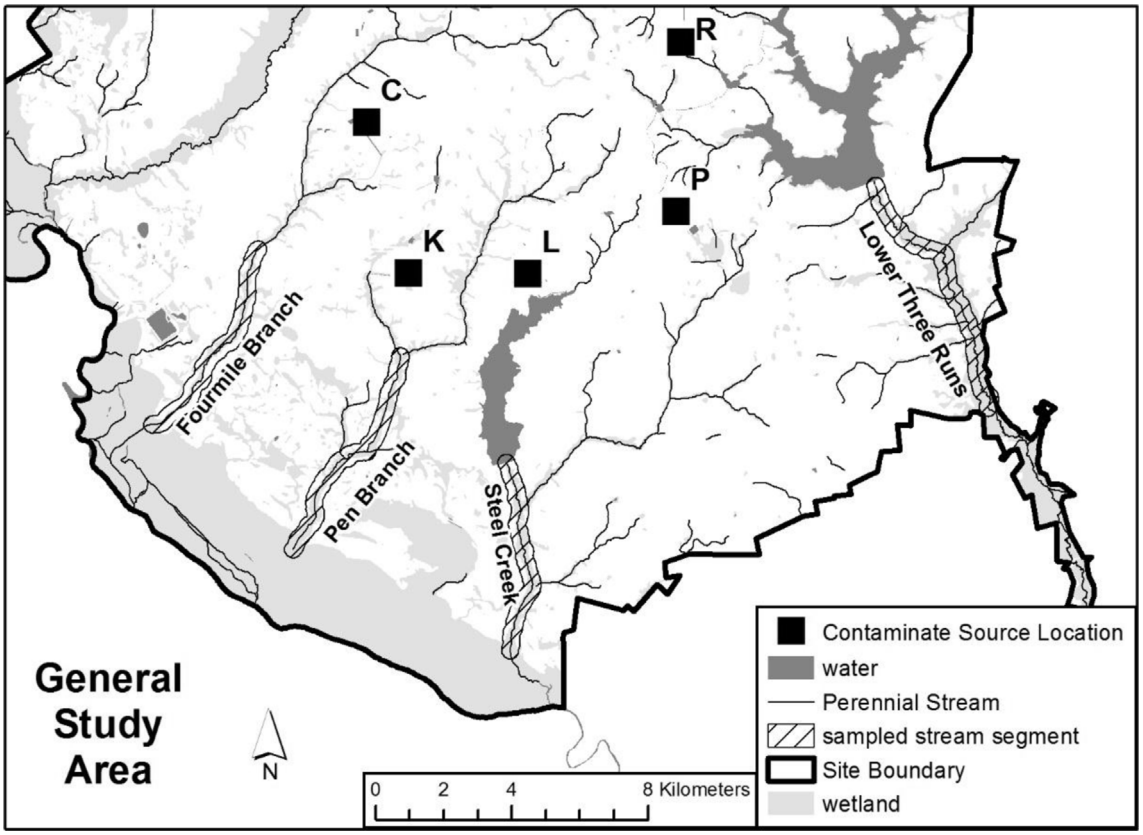

Fig. 1. Locations of four major stream systems and riparian forests with radiological contaminants at the SRS. Stream riparian forest sampled are Fourmile Branch, Pen Branch, Steele Creek and Lower Three Runs Creek. Locations of reactors and processing facilities are shown by letters.

Ammonium phosphate and calcium nitrate were next added to the samples and mixed well. The calcium nitrate enhances recovery for samples that are naturally low in calcium content and it is not needed for samples with high calcium content. Titanium chloride was added to aid in tracer-analyte valance equilibration. Samples were well mixed and cooled in ice bath to room temperature. The tubes were centrifuged and the supernatant was discarded. The iron hydroxide precipitate that formed was partially dissolved in 1.5 M HCL and then diluted with 0.01 M HCL. An additional amount of lanthanum nitrate and calcium nitrate was added, as well as more titanium chloride to ensure valance equilibration and facilitate complete precipitation. Hydrofluoric acid was added next, the samples were capped and mixed, then cooled briefly in ice bath and allowed to sit for 15 min. The hydrofluoric acid reduced calcium levels prior to further separation of actinides. Samples were centrifuged again and the supernatant was poured off. The remaining precipitate which contained the actinides was re-dissolved in nitric-boric, 7 M nitric, and aluminum nitrate and poured into 50 mL tubes. A valance adjustment was performed on this solution by adding ferric nitrate standard and ascorbic acid. Ascorbic acid ensured the reduction of Pu to Pu (III) and Np to Np (IV). After a 3 min wait to ensure this reduction, sodium nitrite was added to oxidize Pu to Pu (IV). Lastly, concentrated nitric acid was added to this sample load solution to reduce calcium retention on the DGA resin. After preparation of the load solution, actinides were then separated using TEVA, TRU, and DGA resins into three fractions: Pu/Np on TEVA, U and Am/Cm on TRU and DGA. Stacking of TEVA + TRU resin was then used to separate Th and U isotopes. Strontium resin was then used for the separation of $^{89/90}$Sr from the other elements for measurement by beta counting.

Gamma spectrometric determination of radionuclides present in samples were conducted using one of several HPGe detector systems consisting of Gamma Products copper-lined lead shields housing either GC 3918 or GC 4020 coaxial detectors, manufactured by Canberra Industries. Instrument calibrations and quality control checks were performed using NIST-traceable, mixed-gamma standards in geometry matched configurations following ANSI standards (Eckert & Ziegler, Atlanta GA). Reference background spectra were collected monthly for each instrument configuration and corrections were applied to spectral data provided. Spectral processing of gamma spectra was performed with Canberra's Procount software for VMS which takes advantage of Genie 2000's spectral processing algorithms. Decay corrected isotope activities were calculated from the empirical calibration data and activities were adjusted for recovery efficiencies. Sample count times were driven by required minimum detectable concentrations (MDC) required for analyses from specific locations. MDCs were calculated for each sample and isotope based on Currie's Equations. (Genie, 2000 3.3 Customization Tools Manual).

Alpha spectrometric determination of radionuclides present in samples were conducted using an Alpha Analyst system from Canberra Industries equipped with A450-18 AM PIPS detectors operated at 40 V. These systems are calibrated for efficiency at least annually and aligned for energy weekly using geometry matched, NIST traceable, electrodeposited standards in accordance with ANSI standards. Standard system checks are executed concurrent with weekly energy alignment. Background spectra are collected for each detector system at a minimum frequency of monthly and a maximum frequency of weekly. Spectral processing of alpha spectra was performed using Canberra's Alpha Management System (AMS) software for VMS. Decay corrected isotope activities were calculated from tracer recoveries. Sample count times were driven by required MDC for analyses from specific locations. MDCs were calculated for each sample and isotope based on Currie's Equations.

Due to its low temperature of vaporization, relatively short half-life (138.4 d) and long (1–1.5 y) hold up time between sampling and analyses, ^{210}Po was not analyzed for in the samples. We chose to use

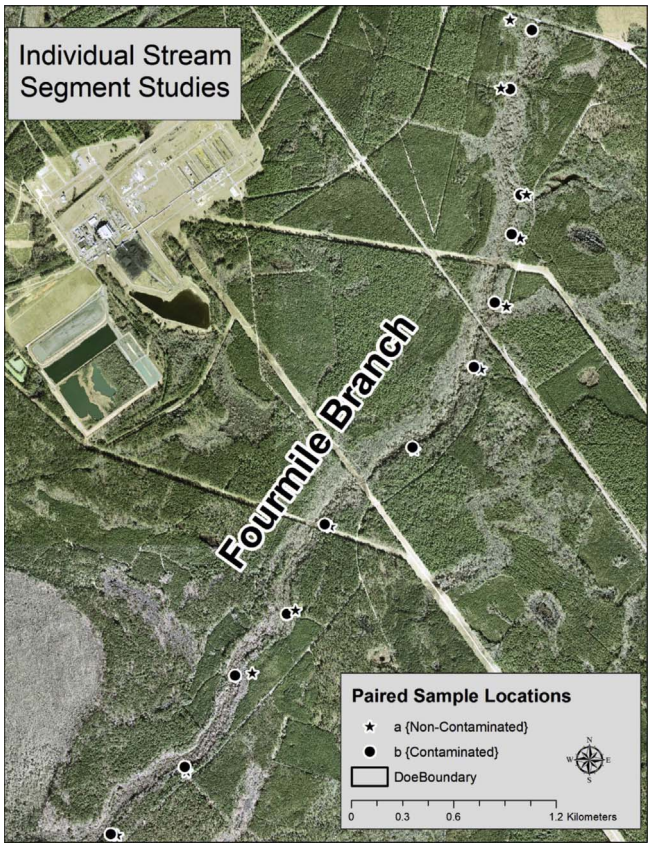

Fig. 2. Sample locations in riparian forest systems. Sample locations were identified in pairs, with one location representing contaminated and a second representing a co-located non-contaminated forest locations.

Table 1
Radioanalysis performed for each reported radionuclide.

Radionuclide Grouping	Radioanalyses Performed
Actinides (Alpha Emitters)	
^{234}U, ^{235}U, ^{238}U, ^{238}Pu, ^{239,240}Pu, ^{241}Am, and ^{244}Cm	Chemical separation followed by alpha spectroscopy
Pure Beta Emitters	
^{89,90}Sr	Chemical separation followed by beta counting
Gamma Emitters	
^{228}Ac, ^{7}Be, ^{212}Bi, ^{214}Bi, ^{60}Co, ^{137}Cs, ^{131}I, ^{4} K, ^{234m}Pa, ^{212}Pb, ^{214}Pb, ^{224}Ra, ^{226}Ra, ^{228}Th, ^{232}Th, and ^{208}Tl	Direct gamma spectroscopy with HPGe detector

published literature data for ^{210}Po to set lower and upper bounds for activity due to the high natural variability expected. ^{210}Po is the last radionuclide generated as part of the complex particle decay sequence of the ^{238}U series, it shows extremely large seasonal variation, and is found on surfaces of vegetation fuels from direct atmospheric deposition (LeCloarec et al., 1995). We used published ranges reported for ^{210}Po in surface organic matter and plants to establish the sensitivity of our results to the possible assumptions of the background concentrations of this radionuclide. The African data from LeCloarec et al. (1995) for grass fuels gave a mean of 3.25×10^{2} Bq kg^{-1} during the peak season and a low value of 14.5×10^{0} during the low season. Gjelsvik et al. (2012) observed a mean of 9.9×10^{1} Bq kg^{-1} for soil humus samples from Norway with a range from 0 to 3.63×10^{2} Bq kg^{-1}. Most mosses and lichens that are known accumulators have higher values but the riparian forest at SRS does not support lichens or mosses near the soil surface. Persson and Holm (2011) reported values from Scandinavian countries in the surface organic peat of about 2.5×10^{2} Bq kg^{-1}.

Table 2
Forest fuel loads and modeled fire environmental inputs to the Fire Emission Production Simulator (Anderson et al., 2004), BehavePlus (Andrews et al., 2005) and FARSITE (Noonan-Wright et al., 2011) under most likely and upper range. The corresponding U.S. National Fire Danger System indices (Burning Index, Energy Release Component) were generated from SRS from daily records for fuel model 'G' (Cohen and Deeming, 1985).

Fuel Component	Most Likely	Upper Range
	Fuel Loading (t ha^{-1})	
Litter	5.74	11.2
Duff	6.62	14.88
Woody	2.31	5.47
Shrub/Vines	0.38	0.94
Grasses/Forbs	0.28	0.94
Total	15.3	33.4
	Fuel Initial Moisture Content (%)	
Percentile Range	70th	97th
Litter (1 h)	6	4
Duff (partially decayed)	40	25
Litter (10 h)	8	6
Live foliage	90	60
	24-H Range of Atmospheric Conditions	
Relative Humidity (%)	34–95	19–61
Temperature (°C)	18–29	23–36
Wind Speed (m s^{-1})	0.9–2.2	0.9–4.5
	National Fire Danger Rating System Indices	
Energy Release Component	28	38
Burning Index	20	32
	Exposure Time per 16-h Shift	
Time of Exposure (hours)	4	14
Days to Burn ~50 ha	5	2

We chose 1.0×10^{1} Bq kg^{-1} and 3.5×10^{2} Bq kg^{-1} as the median and maximum expected concentrations in the fuel for ^{210}Po when assessing cumulative dose. Our maximum value is less than ^{210}Po maximum concentrations (1.15×10^{4} Bq kg^{-1}) back calculated from emissions factors for prescribed fires at high elevations in New Mexico, but the our minimums are similar (1.55×10^{1} Bq kg^{-1}) (Reinhardt et al., 2004).

2.3. Fuel loads, fire spread and fuel consumption

The predominant fuel to carry a fire are litter, small twigs, and branches, which also contribute the largest fraction to smoke emissions at the Site (Goodrick et al., 2010). Duff or decaying litter also contributes to emissions, but not fire spread. We used the average (50%) and high (upper 95% confidence interval) levels of fuel loads for the specific riparian forests at SRS from the forest inventory (Parresol et al., 2012). The bulk or composite samples collected for radionuclide analysis did not allow for characterization of the fuel components required for modeling. Since the forest are similar, we pooled the measured fuel loads to provide a mean load estimate for fire spread emissions modeling (Table 2). Small quantities of live woody shrubs and grasses occur, but are not significant components for fire spread or emissions.

SRS fire weather records accessible at the Western Regional Climate Center database (SAVRIV Station ID:383101) were used to model most likely (70th percentile), and upper range (97th percentile) environmental conditions and initial 1-h and 10-h fuel moisture content (Table 2). These conditions represent moderate and extreme wildfire potential for the area. Model default values were used for live fuel moisture and duff moisture. In order to run multi-day periods (> 24 h) to achieve a comparable fire size for most likely and upper range conditions, we used the identical 24-h weather scenarios corresponding to most likely and upper range conditions for each subsequent 24-h period.

We simulated the initial rate of spread and area for the first hour for each environmental condition and fuel load by using a head fire and standard forest fire model (Timber/Litter TL6) fuel model in BehavePlus software (Andrews et al., 2005). BehavePlus simulations

are limited to 8-h and could not be used for a full firefighter shift. The 1-h conditions were input into Wildland Decision Support System (WDSS) and the imbedded spread model simulator called Near Term Fire Behavior Analysis (NTFB), which is similar to FARSITE but models fire spread on a 2-D surface of zero slope and uniform fuel conditions (Finney, 2004; Noonan-Wright et al., 2011). Fire spread under the70th and 97th environmental conditions and the average and high fuel loads were them simulated. BehavePlus was used for the first hour because the NTFB would not spread from the point source.

Landscape edits within the NTFB model were applied to create uniform slope and fuel conditions. We constrained the geometry of the fire spread to the riparian forests based on the LiDAR topography and conducted multiple ignition point modeling scenarios along each stream. However, the geometry had a minimal effect on fire spread for a given fuel and environmental condition (within ± 5%), so we elected to use a single location for all model runs. The total area burned in the first two 24-h periods for the upper range conditions was determined (~50 ha) and the same area for the most likely conditions was then simulated. The latter required five 24-h periods. Fuel consumption rates for fires were calculated using the Fire Emission Production Simulator (Anderson et al., 2004). The model simulates total emissions in response to the estimated rate of spread. The modeled fuel consumption is similar to empirically measured results under similar environmental and fuel load condition at SRS (Goodrick et al., 2010). For the 70th and 97th percentile environmental conditions, the fuel consumption averaged 72.5% and 85.0% respectively.

2.4. Radionuclide emissions and dispersion

The methodology we employ here follows that of Viner et al. (2015) which generates a base scenario for radionuclide emissions of 1 × 10^{7} Bq ha^{-1} for the fire spread under 70th percentile environmental conditions (EC_{70th}) and fuel consumption for the average fuel load ($FC_{ave,70th}$). The radionuclide activity in the base scenario (RA_b) fuel is set at exactly 911.0 Bq kg^{-1} and the fraction loss (FL_b) is 1.0 to achieve a base emission of 1·10^{7} Bq ha^{-1} from which we can scale the emissions of all other radionuclides. The emission rate per ha of the ith radionuclide (ER_i) can be scaled to the base case by knowing only the measured activity in the fuel (RA_i), the FL_i of each element under EC_{70th} or EC_{97th} (Table 3), and FC for average or high loads under either 70th or 97th percentile conditions relative to the base case (Eq. (1)). The result leads to four emission scenarios for each stream.

$$ER_i = (1 \times 10^{7}\ \text{Bq ha}^{-1}) \text{ x } (RA_i / RA_b) \text{ x } (FL_i \text{ for } EC_{70th \text{ or } 97th}) \text{ x } (FC / FC_{ave,70th}) \quad (1)$$

Atmospheric dispersion relative to the base scenario source term was also scaled because the relationship between source strength and downwind concentration at any point is linear. The PUFF-PLUME model used for this study is a Gaussian dispersion model that was validated at SRS for radiological emissions and found to give good agreement with measured values (Garrett and Murphy, 1981). The model was modified to incorporate multiple sources and used to estimate downwind dispersion of radiological contaminates based on the rate of radionuclide emission. We assume that the radionuclides are incorporated within fine particulate matter below 10 μm (PM_{10}) based on previous studies (Reinhardt et al., 2004; Commodore et al., 2012). While the radionuclides are assumed to be attached to PM_{10} particles in the smoke, no deposition was modeled as deposition was assumed to be negligible due to the low deposition velocity of particles within smoke plumes which has been measured to be on the order of 1e-4 m s^{-1} (Newman et al., 2011). This also maintains conservatism in the dose calculations. PUFF-PLUME uses the traditional Gaussian dispersion equation to predicted downwind concentrations:

$$X(x, y, z) = \frac{Q}{2\pi U \sigma_y \sigma_z}\left(e^{\frac{-y^2}{2\sigma_y}}\right)\left(e^{\frac{-(z-H(x))^2}{2\sigma_z}}\right)\left(e^{\frac{-(z+H(x))^2}{2\sigma_z}}\right) \quad (2)$$

where $X(x, y, z)$ is the atmospheric concentration in (Bq m^{-3}) at a certain distance downwind (x; m), crosswind distance from the plume centerline (y; m) and elevation (z; m), Q is the magnitude of the source term (Bq s^{-1}), U is the wind speed in (m s^{-1}), σ_y and σ_z are the standard deviations of the plume concentration distribution (m) in the lateral and vertical directions, respectively, and H is the source height (m). Wind speeds were determined by the most-likely and upper-range environmental conditions. The turbulent diffusion terms were calculated using guidance from the U.S. Environmental Protection Agency and assuming an extremely unstable environment (U.S. Environmental Protection Agency, 2000).

Smoke plumes would be expected to rise due to buoyancy induced by the heat from the fire (Achtemeier et al., 2011). Therefore, model runs were conducted to ensure that plume rise effects were included. We used a regression model, which relates the final plume height to meteorological and fire parameters of surface wind speed, atmospheric temperature and fuel moisture (Table 6 in Liu, 2014). Deposition processes were not modeled due to uncertainties in the dry deposition parameters and to ensure conservatism in the model predictions. The model was run at hourly intervals with a new fire front configuration specified for each h. The fire front was simulated as an elliptical arc stretching to approximately 45° on either side of the wind direction to the width of the riparian zone. Sources were placed at 3 m intervals along the arc and the source strength was determined using the fire's rate of consumption. The total radionuclide activity was then spread evenly across the distance of the arc during the modeled hour. Downwind concentrations were predicted in the atmosphere from zero to several thousand meters downwind of the expanding fire ellipse.

Table 3
Fraction loss values for various elements detected in wildland fuel samples based upon temperature of vaporization and fraction of the fuel consumed following the method of Viner et al. (2015). For elements with a temperature of vaporization ≥ Calcium (1484 °C), the fraction loss is 0.11 × fuel consumption fraction of either 72.5% or 85% for the 70th and 97% percentile conditions.

Elements	Temperature of vaporization °C	Environmental Condition 70th Percentile	Environmental Condition 97th Percentile
Ac, Am, Be, Bi, Cm, Co, Mn, Nb, Pa, Pb Pu, Ra, Th, Tl, U	≥1484	0.08	0.093
Sr	1384	0.117	0.136
K	774	0.343	0.402
Po	962	0.268	0.320
Cs	768	0.346	0.404
I	184	0.551	0.659

2.5. Firefighter exposure and dose

Mean hourly dose to emergency personnel responding to combat the fire were calculated for the range of condition combinations from the expected upper range (high fuel load, 97th percentile meteorology) scenario to the mostly likely (average fuel load, 70th percentile meteorology). The length of the shift per day over the total number of days of the burn to achieve a size of about 50 ha was used to determine dose under various scenarios. Firefighter exposure was directly related to the concentration of radionuclides in the air/smoke at the work location. Since the concentration in air varies with the dynamic meteorological conditions and the rate of fire spread, firefighter exposure to radionuclides in air was defined as the mean hourly value over the exposure time. The radionuclide dose was influenced by the position of the individual relative to the active fire line, the breathing rate, and the exposure time. From previous analysis the maximum exposure is along

Table 4
Median, maximum and minimum values of radionuclide activity concentrations (Bq kg^{-1}) in non-contaminated and contaminated sites among vegetation samples (Adetona et al., 2011). ^{210}Po* values are estimated from references in text. " < RL" in the table indicates that the value was below the reporting level for that radionuclide.

	Fourmile Branch			Lower Three Runs			Pen Branch			Steel Creek		
	Median	Max	Min	Median	Max	Min	Median	Max	Min	Median	Max	Min
Non-contaminated sites												
^{228}Ac	< RL	< RL	< RL	13.209	17.205	9.213	20.3315	22.718	17.945	< RL	< RL	< RL
^{241}Am	0.01554	0.07955	0	0.02997	0.04995	0.00444	0.04181	0.19536	0.01776	0.02442	0.0592	−0.00259
^{7}Be	355.94	606.8	290.08	440.3	625.3	211.27	431.05	632.7	193.14	407	614.2	243.09
^{212}Bi	< RL	< RL	< RL	19.055	21.978	16.132	9.694	9.694	9.694	< RL	< RL	< RL
^{214}Bi	3.45728	13.283	2.40093	2.40093	19.055	2.40093	4.32678	44.03	2.40093	2.40093	11.322	2.40093
^{244}Cm	0	0.01332	−0.00851	0	0.00703	−0.00444	0.00111	0.05143	−0.00444	0.00185	0.02738	−0.00888
^{60}Co	−0.31598	1.0582	−1.0471	0.39442	3.737	−1.8352	0.46102	1.9129	−2.2015	−0.30525	1.3764	−1.7612
^{137}Cs	7.511	16.428	1.6946	15.614	40.33	2.3347	11.2295	26.714	5.513	6.031	19.906	1.9388
^{131}I	< RL	< RL	< RL	125.06	125.06	125.06	< RL	< RL	< RL	< RL	< RL	< RL
^{4} K	31.82	117.29	14.874	32.301	73.63	−14.171	39.22	145.78	8.88	25.4005	61.05	−13.283
^{54}Mn	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL
^{95}Nb	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL
^{210}Po*	10	350		10	350		10	350		10	350	
^{234M}Pa	< RL	< RL	< RL	< RL	< RL	< RL	351.13	351.13	351.13	274.54	274.54	274.54
^{212}Pb	5.846	10.471	2.20594	4.0367	16.428	2.20594	6.1605	20.239	2.20594	5.069	9.139	2.20594
^{214}Pb	7.77	14.43	2.66622	2.66622	22.903	2.66622	6.475	42.55	2.66622	2.66622	14.245	2.66622
^{238}Pu	0	0.02516	−0.00888	0.00444	0.02479	−0.00333	0.00962	0.11914	−0.01591	0	0.01258	−0.00518
^{239}Pu	0	0.02183	−0.02294	0.01665	0.05291	−0.00629	0.03589	0.3034	−0.00518	0.00555	0.03885	−0.00629
^{224}Ra	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL
^{226}Ra	< RL	56.24	< RL	< RL	< RL	< RL	< RL	124.69	< RL	< RL	101.75	< RL
^{106}Ru	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL
$^{89,\ 90}$Sr	19.24	36.445	0.01036	18.426	51.06	10.397	18.537	29.415	6.512	26.418	36.778	12.025
^{228}Th	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL
^{229}Th	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL
^{231}Th	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL
^{232}Th	10.952	10.952	10.952	< RL	< RL	< RL	16.909	16.909	16.909	< RL	< RL	< RL
^{234}Th	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL
^{208}Tl	1.79154	4.255	1.26688	1.26688	5.957	1.26688	1.26688	5.439	1.26688	2.06534	5.106	1.26688
^{234}U	0.49395	1.9388	0.00037	0.6438	1.8278	0.4773	0.91575	6.623	0.21682	0.4995	1.2876	0.20202
^{235}U	0.00666	0.17612	0	0.09213	0.24938	0.00666	0.04847	0.4995	0	0.03737	0.07622	0
^{238}U	0.55685	2.1497	0	0.5365	1.8204	0.3774	0.86395	6.697	0.21793	0.42735	1.3283	0.20017
Contaminated sites												
^{228}Ac	18.4445	29.896	6.993	< RL	< RL	< RL	24.827	37.74	11.211	< RL	< RL	< RL
^{241}Am	0.00037	0.32967	0	0.04958	0.16465	0.02072	0.07622	0.17279	#VALUE!	0.11174	0.31524	0.03478
^{7}Be	444	603.1	263.81	370	621.6	329.67	341.51	510.6	241.24	369.075	629	291.19
^{212}Bi	10.064	10.064	10.064	10.545	10.545	10.545	14.282	21.534	7.104	< RL	< RL	< RL
^{214}Bi	28.971	49.95	17.168	11.137	19.647	2.40093	14.393	38.11	2.40093	17.797	48.47	2.40093
^{244}Cm	0	0.16021	0	0	0.00666	−0.01258	0.00814	0.04366	0	0.00407	0.06586	−0.00259
^{60}Co	1.00085	2.6529	−0.0259	−0.23606	1.7649	−0.9213	0.7474	1.7723	−0.7696	0.8066	7.659	−0.24975
^{137}Cs	40.145	96.94	22.792	244.2	392.2	9.509	8.3435	17.686	2.8046	154.66	344.1	83.99
^{131}I	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL
^{4} K	41.07	93.24	20.979	46.065	96.2	22.977	50.875	98.05	23.717	51.245	105.08	4.07
^{54}Mn	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL
^{95}Nb	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL
^{210}Po*	10	350		10	350		10	350		10	350	
^{234M}Pa	< RL	< RL	< RL	189.81	242.35	137.27	237.91	237.91	237.91	< RL	< RL	< RL
^{212}Pb	8.88	14.615	6.253	7.326	15.725	2.20594	9.7125	20.831	2.20594	9.694	12.913	2.20594
^{214}Pb	25.7335	52.54	20.683	11.2665	23.125	2.66622	16.7425	48.1	2.66622	18.8885	47.36	2.66622
^{238}Pu	0.00037	0.2072	0	0.00592	0.04958	−0.0037	0.00629	0.11729	−0.00777	0.05513	0.14874	0.00777
^{239}Pu	0.00037	0.16428	0	0.0407	0.13209	0.00999	0.05328	0.12654	−0.00888	0.1258	0.26677	0.01776

(*continued on next page*)

Table 4 (*continued*)

	Fourmile Branch			Lower Three Runs			Pen Branch			Steel Creek		
	Median	Max	Min	Median	Max	Min	Median	Max	Min	Median	Max	Min
^{224}Ra	< RL	< RL	< RL	< RL	< RL	< RL	42.55	55.13	29.933	55.685	62.16	49.21
^{226}Ra	84.175	230.51	< RL	< RL	142.45	< RL	38.57879	95.09	< RL	< RL	175.38	< RL
^{106}Ru	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL
$^{89,\ 90}$Sr	0.07955	135.05	0.02516	22.6995	125.06	3.811	11.396	25.863	6.327	21.5895	34.336	5.846
^{228}Th	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL
^{229}Th	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL
^{231}Th	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL
^{232}Th	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL
^{234}Th	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL	< RL
^{208}Tl	2.97295	5.106	1.26688	3.0636	5.735	1.26688	3.45765	9.028	1.26688	1.26688	6.216	1.26688
^{234}U	0.00629	5.513	0.00111	2.8194	10.471	0.5143	1.295	11.84	0.33374	1.68905	4.477	0.4107
^{235}U	0.00037	0.21682	0	0.16095	0.33707	0.0185	0.09176	7.252	0.02146	0.09805	0.222	0.04403
^{238}U	0.00629	4.551	0.00111	2.47715	8.991	0.3885	1.2025	11.766	0.29563	1.8315	4.662	0.4884

the flank of the fire (Viner at al. 2015). The maximum shift exposure time is 14-h per 24-h period, but firefighters typically shift locations to minimize exposure, so the most likely scenario is 4-h per 24-h period (Adetona et al., 2011).

The committed effective dose received can then be calculated using the specific radionuclide(s) inhalation dose coefficient for an adult worker (U.S. Environmental Protection Agency, 1988). For this assessment, radionuclide-specific dose conversion factors were developed for a unit concentration of 1 Bq m^{-3} and a unit exposure time of 1 h using:

$$DCF_i = BR*DC_i*AC*ET \quad (3)$$

- DCF_i = radionuclide *i* dose conversion factor, Sv m^3 $(Bq\ h)^{-1}$.
- BR = breathing rate for an industrial worker, 1.3 $m^3\ h^{-1}$ (Yu et al., 2001).
- DC_i = radionuclide *i* inhalation dose coefficient, Sv Bq^{-1}(U.S. EPA, 1988).
- AC = unit air concentration 1.0 Bq m^{-3}.
- ET = unit exposure time, 1 h.

Because air concentrations and exposure times have linear effects on the dose, the effective dose per radionuclide *i* is determined by multiplying the DCF_i by the modeled air concentration, which is directly based on the actual measured concentration in fuel, of radionuclide *i* and by the applicable scenario exposure time (effective shift exposure time · number of shifts). The total dose is the summation of all doses from the measured radionuclides in the source term. To calculate the dose from naturally occurring radionuclides in fuel emissions, we used the dose determined from the maximum activities obtained from all radionuclides on non-contaminated sites for each stream, as well as estimated lower and upper bounds for ^{210}Po concentration. We used the same procedure to calculate total dose from each of the contaminated sites for firefighters.

2.6. Off site representative person exposure and dose

For determining potential doses to an offsite representative person, we used equation (3) to develop radionuclide-specific dose conversion factors based on an age and gender averaged reference person. For SRS, the reference breathing rate for an affected person was determined to be 0.73 $m^3\ h^{-1}$ (Stone et al., 2014). The reference person inhalation dose coefficients were taken from U.S. Department of Energy (2011). We calculated dose for the public at various locations downwind as a function of burn conditions and exposure time for contaminated and non-contaminated sites for the longest burn period and the highest fuel loading or maximum emission and exposure scenarios. It was assumed that the offsite person continuously breathes the air for the length of the burns, either 48-h or 120-h. As with the firefighter dose calculations, only the maximum measured concentration was used to predict dose to maintain conservatism in our results.

3. Results

3.1. Radionuclide activities in forest fuels

Median values for the measured radionuclides show that contaminated locations have substantially higher values than non-contaminated locations for ^{137}Cs, as expected from previous contaminated cooling water releases (Table 4). Although $^{89/90}$Sr was released to the streams, the median values are very similar between contaminated and non-contaminated sites. Most other man-made radionuclides showed no increase or only a small increase in activity. Activities of radionuclides that are distributed by atmospheric processes or are generated locally through the decay of primordial radionuclides and cosmic particle interactions are similar at both contaminated and non-contaminated

Table 5
Base 1-h cumulative dose to firefighters at the maximally exposed flanking position in each stream system and for the control sites at the SRS. Dose estimates are the mean dose hr^{-1} over the period of the fire required to reach 50 ha, which was 2-days for the 97% environmental conditions and 5-days for 70th percentile environmental conditions. Detailed exposure, fuel and environmental conditions are listed in Table 2.

Stream System	Fuel Load	Environmental Condition	Dose (mSv)	Additional Dose from ^{210}Po at	
				10 Bq kg^{-1}	350 Bq kg^{-1}
Fourmile Branch	Average	70th Percentile	4.11e-5	1.83e-7	6.41e-6
	Average	97th Percentile	7.47e-5	3.41e-7	1.19e-5
	High	70th Percentile	8.87e-5	3.95e-7	1.38e-5
	High	97th Percentile	1.61e-4	7.37e-7	2.58e-5
Pen Branch	Average	70th Percentile	2.92e-5	1.83e-7	6.41e-6
	Average	97th Percentile	5.31e-5	3.41e-7	1.19e-5
	High	70th Percentile	6.31e-5	3.95e-7	1.38e-5
	High	97th Percentile	1.15e-4	7.37e-7	2.58e-5
Steele Creek	Average	70th Percentile	4.04e-5	1.83e-7	6.41e-6
	Average	97th Percentile	7.35e-5	3.41e-7	1.19e-5
	High	70th Percentile	8.72e-5	3.95e-7	1.38e-5
	High	97th Percentile	1.59e-4	7.37e-7	2.58e-5
Lower Three Runs	Average	70th Percentile	2.81e-5	1.83e-7	6.41e-6
	Average	97th Percentile	5.10e-5	3.41e-7	1.19e-5
	High	70th Percentile	6.06e-5	3.95e-7	1.38e-5
	High	97th Percentile	1.10e-4	7.37e-7	2.58e-5
Control Sites	Average	70th Percentile	2.30e-5	1.83e-7	6.41e-6
	Average	97th Percentile	4.48e-5	3.41e-7	1.19e-5
	High	70th Percentile	5.03e-5	3.95e-7	1.38e-5
	High	97th Percentile	9.78e-5	7.37e-7	2.58e-5

Table 6
Total dose to firefighters in the maximally exposed position for a 50 ha fire in Fourmile Branch and Control Sites for all streams as a function of exposure time per day. Dose was determined for two levels of ^{210}Po in fuels. The 70th percentile environmental conditions resulted in 20 total h and 70 total h of exposure for 4 and 14 h per day exposure. The 97th percentile environmental conditions resulted in 8 total h and 28 total h of exposure for 4 and 14 h per day exposure.

Stream System	Fuel Load	Environmental Condition	Firefighter Dose (mSv)			
			Base plus 10 Bq kg^{-1} ^{210}Po		Base plus 350 Bq kg^{-1} ^{210}Po	
			4-hr day^{-1}	14-hr day^{-1}	4-hr day^{-1}	14-hr day^{-1}
Fourmile	Average	70th Percentile	8.26E-04	2.89E-03	9.50E-04	3.33E-03
	Average	97th Percentile	6.00E-04	2.10E-03	6.93E-04	2.42E-03
	High	70th Percentile	1.78E-03	6.24E-03	2.05E-03	7.18E-03
	High	97th Percentile	1.29E-03	4.53E-03	1.49E-03	5.23E-03
Control Sites	Average	70th Percentile	4.64E-04	1.62E-03	5.88E-04	2.06E-03
	Average	97th Percentile	3.61E-04	1.26E-03	4.54E-04	1.59E-03
	High	70th Percentile	1.01E-03	3.55E-03	1.28E-03	4.49E-03
	High	97th Percentile	7.88E-04	2.76E-03	9.89E-04	3.46E-03

locations. Thus, we would expect the potential dose from naturally occurring radionuclides present in general forest areas (including ^{7}Be, ^{4} K, ^{226}Ra, and ^{234}U) to be similar on contaminated and non-contaminated sites. There are few background measurements of these radionuclides in comparable vegetation fuels (Hejl et al., 2013; Viner et al., 2015). The non-contaminated levels of ^{4} K, ^{212}Bi, ^{7}Be and ^{137}Cs are similar to values reported for upland pine forest fuels on SRS, but the maximum levels of ^{226}Ra, and ^{238}U in non-contaminated sites are less than mean values previously reported (1.95×10^{2} for ^{226}Ra, and 2.59×10^{2} Bq kg^{-1} for ^{238}U) on non-contaminated forest areas by Hejl et al. (2013).

3.2. Impacts to firefighting personnel

The maximum potential dose received by an individual varied across the four sites because of variations in the types and amounts of radionuclides measured in the dead vegetation fuels. However, the greatest range in average dose was a result of the fuel load and environmental conditions (Table 5). In the absence of consideration for shift length and the number of days of active burning the average dose is always greatest under the high fuel load and 97% environmental conditions followed by the high fuel load and 70% environmental conditions. When the number of days of active burning are included, the relationship changes so that the highest dose is always under the high fuel load and 70% environmental conditions because the total exposure time is greater under the latter scenario (5 days for the 70% environmental conditions vs. 2 days for the 97% environmental conditions)Table 6). For the control locations and the Fourmile Branch system, which represented the highest average dose, none of the dose amounts were above the U.S. DOE (2012) regulatory guidance limit of 0.25 mSv. The latter value is one-fourth of the maximum annual dose limit. This relationship is true even at the upper range of ^{210}Po activity in fuels. The highest dose for the Fourmile Branch system is less than 3% of the 0.25 mSv level. In the control sites, the dose is about 2% or less of the 0.25 mSv level.

3.3. Impacts to downwind communities

At SRS, a number of communities and individual members of the public live close to the boundaries of the site, so there is ample concern that any volatilization of the radionuclides onsite could potentially cause exposures offsite. The simulations run for each of the four regions at SRS extended to 20 km to capture the potential downwind concentrations and doses that would affect nearby communities using the maximum measured concentration of each radionuclide in the fuel in order to represent worst-case scenarios (Fig. 3).

In each of the four cases, the predicted dose dropped rapidly with distance in the first 5 km. At 5 km, the modeled dose was to be 2–3 orders of magnitude lower than was predicted for firefighting personnel within 0.5 km of the fire. At greater distances approaching 20 km, the hourly dose proceeded to drop another order of magnitude. Along three of the four waterways (Fourmile, Lower Three Runs, and Steel Creek), the predicted dose was 2–4 times higher using the measurements from the contaminated sample sites than the corresponding uncontaminated sample sites which would be expected. However, along the fourth (Pen Branch), the predicted dose was actually slightly higher using the uncontaminated sample site values than the contaminated sample sites. The inclusion of plume rise acts to decrease the predicted cumulative dose in each scenario at distances out to around 5 km.

4. Discussion

Cumulative doses calculated for both firefighters close to a burn and for communities farther away did not approach the dose limits established for individuals even under the most conservative scenario. The meteorological conditions we selected simulated typical (70th percentile) and more extreme (97th percentile) wildfire conditions at SRS. In both scenarios, the cumulative dose to firefighters and communities was sufficiently low that it would not exceed the dose limits set by DOE ($< 3\%$). We conclude that the additional effort to quantify a large array of radionuclides may not warrant the expense because activities of natural and background and anthropogenic radionuclides are too low,

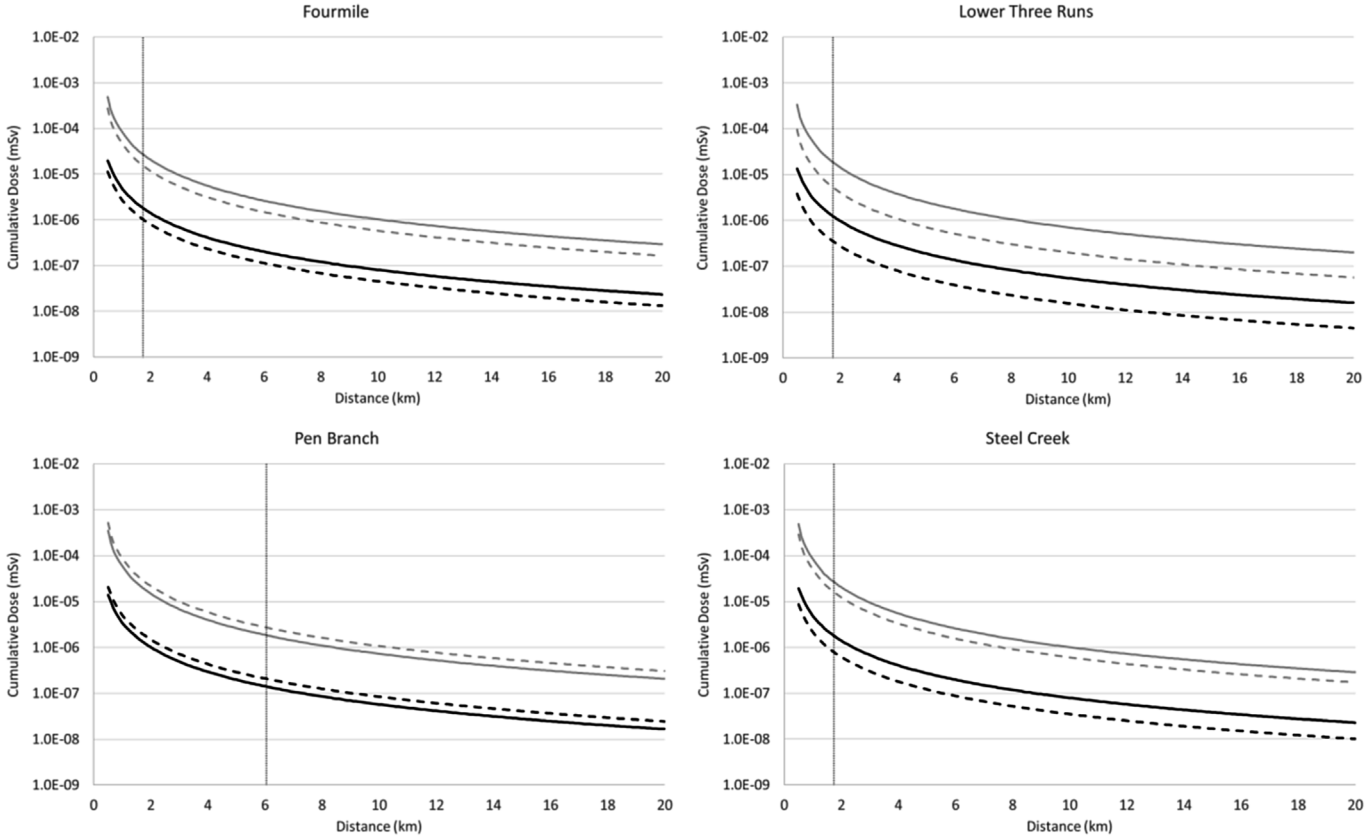

Fig. 3. Comparison of modeled maximum cumulative dose from contaminated (solid line) vs. non-contaminated (dashed line) sites for the 70th Percentile Meteorological and Average Fuel Loading Conditions (black lines) and the 97th Percentile Meteorological and High Fuel Loading Conditions (gray lines) for off-site communities. Doses were calculated assuming that a person was exposed to the plume for 48 hr (97th High Conditions) or for 120 hr (70th Average Conditions). These conditions are the bounding conditions for the scenarios run in this study and all other simulations would fall between these. No plume rise effects are included, leading to additional conservatism.

Table 7
Percent contribution of each radionuclide to the cumulative dose for the Pen Branch burn and an average of the all control sites from each of the four sample regions. The 97th percentile meteorological conditions and high fuel load case was used in these calculations. Cases for using the median and maximum estimated values for ^{210}Po were also used.

	Control Sites		Pen Branch	
	10 Bq kg^{-1} ^{210}Po	350 Bq kg^{-1} ^{210}Po	10 Bq kg^{-1} ^{210}Po	350 Bq kg^{-1} ^{210}Po
^{228}Ac	0.03	0.03	0.03	0.03
^{241}Am	0.78	0.68	1.21	1.02
^{7}Be	0.00	0.00	0.00	0.00
^{212}Bi	0.03	0.03	0.05	0.04
^{214}Bi	0.07	0.06	0.08	0.07
^{244}Cm	0.13	0.12	0.22	0.19
^{60}Co	0.00	0.00	0.00	0.00
^{137}Cs	0.05	0.04	0.50	0.42
^{40}K	5.29	4.61	4.41	3.73
^{234M}Pa	0.01	0.01	0.01	0.00
^{212}Pb	0.36	0.31	0.35	0.30
^{214}Pb	0.06	0.05	0.07	0.06
^{210}Po	0.43	13.24	0.54	15.91
^{238}Pu	0.52	0.46	0.71	0.60
^{239}Pu	1.45	1.26	1.02	0.86
^{224}Ra	N/A	N/A	11.07	9.36
^{226}Ra	45.09	39.29	72.03	60.90
$^{89,\ 90}$Sr	0.16	0.14	0.54	0.45
^{232}Th	40.98	35.71	0	0
^{234}U	2.39	2.05	3.61	3.05
^{235}U	0.16	0.14	0.80	0.67
^{238}U	1.99	1.73	2.76	2.33

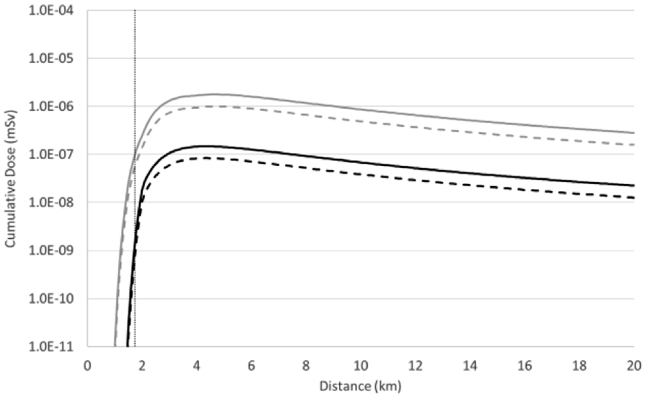

Fig. 4. Modeled cumulative dose from fires along the Fourmile branch with the effects of plume rise included in the simulation. Lines indicate doses from contaminated (solid line) vs. non-contaminated (dashed line) sites for the 70th Percentile Meteorological and Average Fuel Loading Conditions (black lines) and the 97th Percentile Meteorological and High Fuel Loading Conditions (gray lines) for off-site communities.

volatilization temperatures too high and atmospheric dispersion is able to sufficiently lower concentrations. This conclusion assumes that suite of radionuclides can be representative of other locations with similar geological, biological and climatic conditions.

The relative contribution of each radionuclide to the cumulative dose in each scenario demonstrates that only a few contribute substantially to dose (Table 7). The greatest contributors to dose for the Pen Branch indicator sites were the naturally occurring radionuclides

^{226}Ra and ^{232}Th. For the average of the Control Sites, the naturally occurring radionuclides ^{226}Ra and ^{224}Ra were the largest contributors. Uranium isotopes in total contributed about 4–7% to the dose in each case while other isotopes consistently had little overall impact to the cumulative dose. Exposure time has a major effect on potential dose as reflected in the fact that the greatest dose occurred under the 70th percentile conditions and high fuel load because the burn lasted many more days and therefore has a longer cumulative exposure time. Firefighters are assumed to be limited to 14 h of exposure each day as they rotate on and off shifts. The public on the other hand is potentially exposed 24 h each day. However, the cumulative dose was actually lower for the public than for the firefighting personnel because of downwind dispersion.

The one surprising result in our predictions of dose was that the uncontaminated sites produced a higher predicted dose than the contaminated sites for Pen Branch. To explain this discrepancy, we examined the concentration of each radionuclide and their relative contribution to the total dose. We found that the maximum ^{226}Ra concentration measurement at the uncontaminated sites was slightly greater than the maximum ^{226}Ra measurement at the contaminated sites. Because we were using maximum values to ensure conservativism, this led to the unexpected result. Had we used the median values rather than maximum values for concentrations among the site, the predictions from the contaminated sites would have been higher as expected, so the unexpected result is an outlier rather than indicative of the entire sample area.

The inclusion of plume rise in the models was intended to show whether the height of the plume had an impact on the predicted doses to local communities. We found that while the inclusion of plume rise did act to lower the predicted radionuclide surface concentration slightly, this difference diminished with distance until there was a nominal difference of less than 10% in the near-surface atmospheric concentrations and dose predictions at distances greater than 6 km as the plumes became mixed within the atmosphere (Fig. 4). Therefore, assumption of the initial plume release height is not of particular importance in estimating dose to the public from these systems and the distances between the fire and the closest downwind communities. Because of the size of the SRS, this distance between a fire and sensitive locations in the local communities would generally be about 5 km and often would be greater than 10–15 km.

The scenarios we assessed simulates a moderate rate of spread of the burn to achieve a specified area. The latter results from the flat topography, moderate winds and limited fuel loads. Smoke plume dynamics and subsequent exposure could change dramatically under steep slopes and high wind conditions that create greater plume rise Continued smoldering and burning during the night can create additional concentrations of smoke due to the presence of an inversion layer, usually within a few hundred meters of the surface that acts to trap smoke. Firefighters generally avoid being present under these conditions since the rate of fire spread and fire activity is low under these conditions.

5. Conclusions

The contaminated ground surface at SRS is a result of the decades of work that has been performed maintaining the country's nuclear stockpile and performing research and development on nuclear materials. To better understand the risk that these smoke plumes present, we have characterized four regions at SRS in terms of their fuel characteristics and radiological contamination in the ground. Combined with general meteorological conditions describing typical and extreme burn conditions, we simulated potential fires in these regions and predicted the potential radiological dose that could be received by firefighting personnel and the public surrounding SRS. We conclude that cumulative dose to firefighters and the public is strongly influenced by fire and fuel conditions and smoke plume dynamics, but that inclusion of a wider array of natural and anthropogenic radionuclides to provide an exhaustive cumulative dose estimate from all possible radioactive materials in the environment is not warranted.

Potential cumulative dose to the firefighters or public, even under worst-case conditions, did not exceed the regulatory limits. We can infer that the risks associated wildfires at these locations would not be expected to cause doses above the level of concern to either responding personnel or the offsite public.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jenvrad.2017.10.017.

References

U.S. Environmental Protection Agency, 2000. Meteorological Monitoring Guidance for Regulatory Modeling Applications. Environmental Protection Agency, Washington, D.C EPA-454/R-99–9005.

Achtemeier, G.L., Goodrick, S.A., Liu, Y., Garcia-Menendez, F., Hu, Y., Odman, M.T., 2011. Modeling smoke plume-rise and dispersion from southern United States prescribed burns with Daysmoke. Atmosphere 2, 358–388.

Adetona, O., Dunn, K., Hall, D.B., Achtemeier, G., Stock, A., Naeher, L.P., 2011. Personal PM2.5 Exposure among wildland firefighters working at prescribed forest burns in southeastern United States. J. Occup. Environ. Hyg. 8, 503–511.

Anderson, G.K., Sandberg, D.V., Norheim, R.A., January 2004. 2004. Fire Emission Production Simulator User's Guide Version 1.0. USDA Forest Service, Pacific Northwest Forest Exp. Stn, Portland, OR.

Andrews, P.R., Bevins, C.D., Seli, R.C., 2005. BehavePlus Fire Modeling System Version 3.0 User's Guide. USDA Forest Service Rocky Mountain Res. Stn., Gen. Tech. Rep. RMRS-GTR-106WWW Revised (Fort Collins, CO).

Barton, C., Nelson, E.A., Kolka, R.K., McLeod, K.W., Conner, W.H., Lakly, M., Martin, D., Wigginton, J., Trettin, C.C., Wisniewski, J., 2000. Restoration of a severely impacted riparian wetland system - the pen branch project. Ecol. Eng. 15, 3–15.

Carlton, W.H., 1998. Assessment of Radionuclides in the Savannah River Site Environment-summary. WSRC-TR-98-00162. Savannah River Site, Aiken, SC.

Cohen, J.D., Deeming, J.E., 1985. The National Fire-danger Rating System. USDA Forest Service Pacific Southwest Forest and Range Exp. Stn, Gen. Tech. Rep. GTR-PSW-82 (Berkley, CA).

Commodore, A.A., Jannik, G.T., Eddy, T.P., Rathbun, S.L., Hejl, A.M., Pearce, J.L., Irvin-Barnwell, E.A., Naeher, L.P., 2012. Radionuclide concentrations in smoke from prescribed burns at the Savannah River Site and forest lands in Southeastern United States. Atmos. Environ. 54, 643–656.

Finney, M.A., 2004. FARSITE: fire area simulator – model development and evaluation. USDA Forest Service, Rocky Mountain Res Stn., Res. Pap. RMRS-RP-4 Revised. (Fort Collins, CO).

Garrett, A.J., Murphy, C.E., 1981. A PUFF/PLUME Atmospheric Deposition Model for Use at SRP in Emergency Response Situations. Savannah River Laboratory DP-1595. (Aiken, SC).

Gjelsvik, R., Brown, J., Holm, E., Roos, P., Saxen, R., Outola, I., 2012. Strålevern Rapport. Polonium-210 and Other Radionuclides in Terrestrial, Freshwater and Brackish Environments, 2012, (Results from the NKS project GAPRAD, Filling knowledge gaps in radiation protection methodologies for non-human biota), vol. 3 Norwegian Radiation Protection Authority, Østerås.

Goodrick, S.L., Shea, D., Blake, J., 2010. Estimating fuel consumption for the upper coastal plain of South Carolina. South. J. Appl. For 34, 5–12.

Hao, W., Bondarenko, O.O., Zibtsev, S., Hutton, D., 2009. Vegetation fires, smoke emissions, and dispersion of radionuclides in the Chernobyl exclusion zone. In: Bytnerowicz, M., Arbaugh, A., Riebau, M., Andersen, C. (Eds.), Wildland Fires and Air Pollution: Developments in Environmental Science. Elsevier, Amsterdam, Netherlands, pp. 265–275.

Hejl, A.M., Ottmar, R.D., Jannik, G.T., Eddy, T.P., Rathbun, S.L., Agyepong, A.D., Pearce, J.L., Naeher, L.P., 2013. Radionuclide concentrations in forest surface fuels at the Savannah River Site. J. Environ. Manage 115, 217–226.

Kilgo, J., Blake, J.I., 2005. Ecology and Management of a Forested Landscape: Fifty Years on the Savannah River Site. Island Press, Washington, D.C.

LeCloarec, M.F., Ardouin, B., Cachier, H., Liousse, C., Neveu, S., Nho, E.-Y., 1995. ^{210}Po in savanna burning plumes. J. Atmos. Chem. 23, 111–122.
Liu, Y., 2014. A regression model for smoke plume rise of prescribed fires using meteorological conditions. J. Appl. Met. Clim. 53, 1961–1975.
Maier, B., Ottmar, R., Wright, C., 2004. Forest Floor Bulk Density and Depth at Savannah River –Final Report. Fire and Environmental Applications Team, USDA Forest Service Pacific Northwest Research Station. Pacific Wildland Fire Sciences Laboratory, Seattle, WA.
Maxwell III, S.L., Culligan, B.K., Noyes, G.W., 2010. Rapid separation of actinides and radiostrontium in vegetation samples. J. Radioanal. Nucl. Chem. 2010. http://dx.doi.org/10.1007/s10967-010-0653-y.
Newman, J.S., Su, P., Yee, G., 2011. Smoke deposition velocity in industrial fire environments. Fire Saf. Sci. 10, 655–668.
Noonan-Wright, E.K., Opperman, T.S., Finney, M.A., 2011. Developing the US wildland fire decision support system. J. Combust. 2011 (14), 168473. http://dx.doi.org/10.1155/2011/168473. 2011.
Paatero, J., Vesterbacka, K., Makkonen, U., Kyllönen, K., Hellen, H., Hatakka, J., Anttila, P., 2009. Resuspension of radionuclides into the atmosphere due to forest fires. J. Radioanal. Nucl. Chem. 282, 473–476.
Paller, M.H., Jannik, G.T., Baker, R.A., 2014. Effective half-life of Cesium-137 in various environmental media at the Savannah River site. J. Environ. Rad. 131, 81–88.
Parresol, B.R., Blake, J.I., Thompson, A., 2012. Effects of overstory composition and prescribed fire on fuel loading across a heterogeneous managed landscape in the southeastern USA. For. Ecol. Mgt 273, 29–42.
Pazukhin, E.M., Borovoi, A.A., Ogorodnikov, B.I., 2004. Forest fire as a factor of environmental redistribution of radionuclides originating from Chernobyl accident. Radiochem 46, 102–106.
Persson, B.R., Holm, E., 2011. Polonium-210 and lead-210 in the terrestrial environment: a historical review. J. Environ. Radioact. 102, 420–429.
Reinhardt, T., Wrobel, C., Eberhart, C., 2014. Radionuclide emission factors from prescribed burns in northern New Mexico. Technical Report LA-14113. Los Alamos National Laboratory.
Stone, D.K., Higley, K.A., Jannik, G.T., 2014. Site-specific reference person parameters and derived concentration standards for the Savannah River Site. Health Phys. 106 (2), S59–S64.
Sugihara, S., Osaki, S., Baba, T., Tagawa, Y., Maeda, Y., Inokura, Y., 1999. Distribution and mean residence time of natural radionuclides in forest ecosystems. J. Radioanal. Nucl. Chem. 239, 549–554.
U.S. Department of Energy, 2011. Derived Concentration Technical Standard. pp. 1196–2011 Washington DC: U.S. DOE; DOE-STD.
U.S. Environmental Protection Agency, 1988. Limiting Values of Radionuclide Intake and Air Concentration and Dose Conversion Factors for Inhalation, Submersion, and Ingestion. Federal Guidance Report#11, EPA 520/1-88-020 (Washington, D.C.).
Viner, B.J., Jannik, T., Stone, D., Hepworth, A., Naeher, L., Adetona, O., Blake, J., Eddy, T., 2015. Modelling and mitigating dose to firefighters from inhalation of radionuclides in wildland fire smoke. Int. J. Wildl. Fire 24, 723–733.
Volkerding, J.M., 2003. Comparison of the radiological dose from the Cerro grande fire to a natural wildfire. Environ. Intern 29, 987–993.
Yu, C., Zielen, A.J., Cheng, J.J., Lepoire, D.J., Gnanapragasam, E., Kamboj, S., Arnish, J., Wallo III, A., Williams, W.A., Peterson, H., 2001. Users's Manual for RESRAD Version 6. Argonne National Laboratory; Argonne, IL: ANL/EAD-4.