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Radiological risk from consuming fish and wildlife to Native Americans on the Hanford Site (USA)

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ABSTRACT

Historical operations at the Hanford Site (Washington State, USA) have released a wide array of non-radionuclide and radionuclide contaminants into the environment. As a result of stakeholder concerns, Native American exposure scenarios have been integrated into Hanford risk assessments. Because its contribution to radiological risk to Native Americans is culturally and geographically specific but quantitatively uncertain, a fish and wildlife ingestion pathway was examined in this study. Adult consumption rates were derived from 20 Native American scenarios (based on 12 studies) at Hanford, and tissue concentrations of key radionuclides in fish, game birds, and game mammals were compiled from the Hanford Environmental Information System (HEIS) database for a recent time interval (1995–2007) during the post-operational period. It was assumed that skeletal muscle comprised 90% of intake, while other tissues accounted for the remainder. Acknowledging data gaps, median concentrations of eight radionuclides (i.e., Co-60, Cs-137, Sr-90, Tc-99, U-234, U-238, Pu-238, and Pu-239/240) in skeletal muscle and other tissues were below 0.01 and 1 pCi/g wet wt, respectively. These radionuclide concentrations were not significantly different (Bonferroni $P > 0.05$) on and off the Hanford Site. Despite no observed difference between onsite and offsite tissue concentrations, radiation dose and risk were calculated for the fish and wildlife ingestion pathway using onsite data. With median consumption rates and radionuclide tissue concentrations, skeletal muscle provided 42% of the dose, while other tissues (primarily bone and carcass) accounted for 58%. In terms of biota, fish ingestion was the largest contributor to dose (64%). Among radionuclides, Sr-90 was dominant, accounting for 47% of the dose. At median intake and radionuclide levels, estimated annual dose (0.36 mrem/yr) was below a dose limit of 15 mrem/yr recommended by the United States Environmental Protection Agency (USEPA), as well as below a dose limit of 100 mrem/yr proposed by the International Commission on Radiation Protection (ICRP). Similarly, lifetime cancer risk ($1.7E-5$), calculated with median inputs, was below risk levels corresponding to these dose limits. However, our dose and risk estimates apply to only one pathway within a multidimensional exposure scenario for Native Americans. On the other hand, radiation dose and risk corresponding to onsite tissue concentrations were not significantly different from those corresponding to offsite (background) concentrations. Recognizing uncertainties in exposure and toxicity assessments, our results may facilitate informed decision making and optimize resource allocation within a risk assessment framework at the Hanford Site.

1. Introduction

The Hanford Site, located in southeastern Washington State (USA), was established in 1943 to produce plutonium for nuclear weapons tested and used in World War II (Poston et al., 2008). The last plutonium production reactor (N Reactor) was shut down in January 1987 (PNNL, 1998). These historic operations resulted in the production of both radiological and non-radiological chemical wastes. Four areas of the Hanford Site (i.e., 100, 200, 300, and 1100 Areas) have been placed on the National Priorities List for environmental cleanup by the United States Environmental Protection Agency (USEPA) (PNNL, 1998). Due to these past activities, numerous risk assessments have been conducted at Hanford to support site cleanup.

As a result of tribal concerns, evaluation of a Native American exposure scenario has become common practice in risk

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assessments (e.g., USDOE, 2008a). Native American exposure scenarios at the Hanford Site represent a subsistence lifestyle and include a wide range of exposure pathways, reflecting elements of both traditional practices (e.g., fishing, hunting, gathering, and sweat lodge use), as well as more contemporary activities (e.g., irrigated farming, pasturing livestock, and well water use). Risk assessments require exposure factors that accurately reflect the tribal lifestyle, along with representative contaminant data in exposure media (Harper and Harris, 2008).

In particular, the fish and wildlife consumption pathway has been the subject of recent interest in Native American exposure scenarios (Bridgen, 2005; Donatuto and Harper, 2008). Concurrently, methods employed in radiological risk assessment have been steadily evolving (Till and Grogan, 2008). As such, we examined radiological risk, associated with the fish and wildlife ingestion pathway.

More specifically, the purpose of this study was to quantify radiological dose and risk for current conditions in the post-operational period at the Hanford Site, as a result of consuming fish and wildlife, within the context of an adult Native American exposure scenario. In particular, consumption rates for fish and wildlife, along with radiological contaminant concentrations in these biota, require cultural and site-specific inputs, respectively. Therefore, our study utilized results in the literature on relevant tribal consumption rates and employed measured (rather than modeled) radionuclide data in Hanford biota to more accurately assess dose and risk.

2. Methods

2.1. Consumption rates for fish and wildlife

Consumption rates for adults from 20 Native American scenarios, applied at the Hanford Site, were compiled from 12 studies (Table 1). Not all of these studies are independent. For example, USDOE (2008a) cites Harris and Harper (1997) as a source of information, Rittmann (2004) is based on PNNL (1998), and USEPA (2002a) extends results of CRITFC (1994). Child-specific values were excluded in our compilation. Reported consumption rates for these scenarios predominately represent reasonable maximum exposure (RME) estimates. Consumption rates from these 20 scenarios were statistically summarized by identifying selected points within the distribution of consumption rates (i.e., 0th, 25th, 50th, 75th, 95th, and 100th percentiles).

For our study, fish and wildlife were defined as finfish, game birds, and game mammals. Because some exposure scenarios did not distinguish between wildlife and domesticated animals (e.g., livestock, poultry), several assumptions were made. Where consumption rates were listed for meat, meat and game, or meat and

Table 1
Studies describing Native American scenarios geographically relevant to the Hanford Site.

Reference	Number of scenarios[a]	Tribe	Adult receptor	Data collection method	CTE or RME approach[b]	Diet focus	Traditional and contemporary lifestyle activities
Harper et al. (2002)	2	Spokane	Subsistence resident	Ethnographic information, interviews	RME	High fish, high game	Yes
Harris and Harper (1997)	1	CTUIR[c]	Subsistence resident	Expert elicitation, survey	CTE	More game, less domestic meat	Yes
Harris and Harper (2004)	1	CTUIR	Subsistence resident	Ethnographic information, interviews	RME	High fish	Yes
PNNL (1998)	5	NS[d]	Subsistence resident, upland hunter, river focused hunter/ fisher, gatherer of plant materials, Columbia River Island user	Interviews	RME	Receptor dependent	Yes
Ridolfi Inc., 2007	1	Yakama Nation	Subsistence resident	Literature review, ethnographic interviews	RME	NS	Yes
Rittmann (2004)	1	NS	Subsistent resident	NS	RME	NS	Yes
Thatcher (2003)	4	CTUIR, Yakama, Nez Perce	Offsite critical population, intruder, upland hunter, river resident	Literature review, interviews	RME	Mammals and birds but no fish	Yes
USDOE (2008a)	2	CTUIR	Subsistent resident	Interviews	NS	High fish, high game	NS
USDOE (2008b)	2	NS	Resident farmer, hunter gatherer	NS	CTE	More contemporary (RF), more traditional (HG)[e]	Yes
USDOE/WDOE, 1996	1	NS	Subsistent resident	Literature review, interviews	RME	NS	Yes
CRITFC (1994) and USEPA (2002a)	2	Umatilla, Nez Perce, Yakama, Warm Springs	Fish consumer	Survey	CTE (mean), RME (99th percentile)	Fish only	NS

[a] No estimates of consumption rates for fish, game birds, and game mammals were provided by two scenarios, including the gatherer of plant materials (PNNL, 1998) and the Columbia River Island user (PNNL, 1998).
[b] CTE=central tendency exposure, RME=reasonable maximum exposure.
[c] CTUIR=Confederated Tribes of the Umatilla Indian Reservation.
[d] NS=not specified.
[e] RF=resident farmer, HG=hunter gatherer.

poultry, the intake was divided evenly among game mammals, game birds, livestock, and poultry (Thatcher, 2003; Ridolfi Inc., 2007; USDOE, 2008b). In other scenarios, when consumption rates were specified for meat, meat and game, or animal protein, along with a separate intake for game birds, the non-game bird consumption rate was divided evenly among game mammals, livestock, and poultry (USDOE/WDOE, 1996; Harris and Harper, 1997; PNNL, 1998; Thatcher, 2003). Finally, in one scenario where a consumption rate was listed for game and fowl, this intake was split evenly between game mammals and game birds (Harris and Harper, 2004).

It was also assumed that consumption rates for fish and wildlife in the literature generally represent ingestion of skeletal muscle tissue, unless otherwise specified. In this paper, we use the terms "muscle" to refer to skeletal muscle and "non-muscle" to indicate other tissues (e.g., internal organs). We implemented the recommendation by Harris and Harper (1997) that intake of muscle comprises 90% of total animal intake, while non-muscle represents the remaining 10%. In some cases, a consumption rate for muscle tissue was calculated by subtracting 10% of total animal intake to account for non-muscle tissues (USDOE/WDOE, 1996; Harper et al., 2002; Harris and Harper, 2004; Thatcher, 2003). For non-muscle tissue in mammals where data were available for a particular radionuclide in two internal organs, non-muscle tissue intake was split evenly between these organs.

2.2. Radionuclide concentrations in fish and wildlife

Based on a preliminary evaluation of radionuclide time series data in fish and wildlife, we selected a time interval from 1995 to 2007 to best represent current conditions, while maintaining sufficient data for statistical analysis. Radiological concentrations for both onsite and offsite biota (i.e., fish, game birds, game mammals) were obtained from the Hanford Environmental Information System (HEIS) database (HEIS, 1989; PNNL, 2008). We evaluated radionuclide data in both muscle (skeletal) and non-muscle tissues (i.e., liver, heart, kidney, intestine, bone, and carcass). Onsite vs. offsite comparisons of radionuclide concentrations were performed to evaluate if Hanford is contributing to dose and risk above background.

Results in the HEIS database are reported primarily in muscle tissue of fish and wildlife, bone of wildlife, and carcass of fish. Carcass samples consisted of the head, skeleton, and skin with viscera removed. A much smaller amount of data exists for a variety of internal organs. For the time period 1995–2006, only concentrations of Co-60, Cs-137, Pu-238, Pu-239/240, Sr-90, Tc-99, U-234, and U-238 were reported. These particular radionuclides are monitored in the environment based on historical operations and waste streams produced at the Hanford Site that continue to persist due to relatively long half-lives. For Co-60 and Cs-137, most of the data correspond to muscle tissue samples. For Pu-238 and Pu-239/240, data were predominately in liver. For Sr-90, most of the data correspond to bone and carcass. Finally, the U-234 and U-238 results pertain mainly to muscle and carcass samples. Data from 2007 contained several other gamma emitting radionuclides, but these data were sparse and most concentrations were below laboratory detection limits. Therefore, these additional radionuclides were not included in our analysis.

Descriptive statistics were calculated for radionuclide concentrations in fish and wildlife tissues when sufficient sample numbers (i.e., $N \geq 5$) were available. A unique characteristic of radionuclide analytical data is the possible occurrence of numerically negative concentrations. The net radionuclide concentration in an environmental sample is determined by subtracting an instrument background measurement (i.e., instrument blank) from a gross environmental sample measurement. Because of the statistical nature of radioactive decay and other measurement uncertainties, the net result may be negative. USEPA and others have recommended including all reported data in statistical analysis, including negative values (USEPA, 1980; Gilbert, 1987; USEPA, 2002a). We have followed this recommendation and use data as reported in calculating descriptive statistics.

In addition to this statistical summary, inferential methods were also applied. Specifically, onsite vs. offsite median radionuclide concentrations in fish and wildlife tissues were compared with a nonparametric Mann–Whitney test (SGC, 1998). The Bonferroni inequality was employed to maintain the overall alpha level at 0.05 in these tests (Stevens, 1986; Suter, 1996).

2.3. Dose and risk from fish and wildlife consumption

Dose and risk were calculated from consumption rates and radiological concentrations in muscle and non-muscle tissues of onsite fish and wildlife. Radionuclide concentrations in offsite biota were not subtracted from onsite concentrations for dose and risk calculations. Because radionuclide concentrations can be negative (as described above), it is possible that descriptive statistics (e.g., median radionuclide concentration) can also be negative. In these cases, a value of zero was assigned to the statistic in dose and risk calculations.

Dose calculations were performed for median consumption rates and median onsite radionuclide concentrations for each radionuclide evaluated in fish, birds, and mammals. In addition, dose and risk were tabulated at two consumption rates (50th and 95th percentiles) and two onsite radionuclide concentrations in fish and wildlife (50th and 95th percentiles), summed over both radionuclides and biota. For lifetime risk estimates, adult consumption rates for fish and wildlife were assumed (for simplicity) over a 70 year lifetime.

Annual dose was estimated with committed (50 year period) effective dose conversion factors for ingestion in adults (ICRP, 1996), while lifetime risk was calculated with morbidity risk coefficients for ingestion of food (USEPA, 1999). Risk coefficients include the contribution to dose from production of decay chain members in the body after intake of the parent radionuclide. Dose conversion factors and risk coefficients were consistent with RESRAD modeling software (version 6.4) (Yu et al., 2001), a dose estimation tool commonly used at radiological cleanup sites. Dose conversion factors and risk coefficients, along with their associated gastrointestinal absorption fraction ($f1$), are listed in Table 2.

Table 2
Committed effective dose conversion factors for ingestion and morbidity risk coefficients for ingestion of food, along with gastrointestinal absorption fractions ($f1$), for radionuclides evaluated in fish and wildlife consumption.

Radionuclide	$f1$	Dose conversion factor[a] (mrem/pCi)	Risk coefficient[b] (risk/pCi)
Co-60	0.1	1.26E−5	2.23E−11
Cs-137	1	4.81E−5	3.74E−11
Pu-238	0.0005	8.51E−4	1.69E−10
Pu-239/240	0.0005	9.25E−4	1.74E−10
Sr-90	0.3	1.04E−4	6.88E−11
Tc-99	0.5	2.37E−6	4.00E−12
U-234	0.02	1.81E−4	9.55E−11
U-238	0.02	1.67E−4	8.66E−11

[a] ICRP (1996).
[b] USEPA (1999).

3. Results

3.1. Consumption rates for fish and wildlife

Selected percentiles of consumption rates of muscle tissue are shown in Table 3. The number of scenarios underlying derivation of these percentiles is also indicated in the table. Consumption rates of muscle tissue reported in these scenarios span an order of magnitude or more. Ingestion of non-muscle tissues was derived from total animal intake (Harris and Harper, 1997).

3.2. Radionuclide concentrations in fish and wildlife

Statistics for radionuclide concentrations in muscle tissue of fish, birds, and mammals, collected over the period 1995–2007, are displayed in Table 4. Represented by their 50th and 95th percentiles, radionuclide concentrations in muscle were below 0.01 and 0.1 pCi/g, respectively. For cases where data were available, no significant differences (Bonferroni $P > 0.05$) were observed between onsite and offsite median radionuclide concentrations in muscle.

Similar statistics for radionuclide concentrations in non-muscle tissues for the same time period are shown in Table 5. With the exception of Sr-90 in mammal bone, these concentrations were below 0.1 and 1 pCi/g at their 50th and 95th percentiles, respectively. Again, for cases where data were available, no significant differences (Bonferroni $P > 0.05$) were observed between onsite and offsite median radionuclide concentrations in non-muscle tissues.

3.3. Dose and risk from fish and wildlife consumption

Annual dose estimates at median consumption rates and radionuclide concentrations in muscle and non-muscle tissues,

Table 3
Distribution percentiles of fish and wildlife consumption rates of muscle tissue (g/day wet wt) for adult Native American scenarios at the Hanford Site[a].

Biota	Number of	Percentile						References
Group	scenarios	0th	25th	50th	75th	95th	100th	(see Table 1)
Fish	15	42	170	540	550	849	972	All but Thatcher (2003)
Game birds	18	18	37	63	88	116	176	All but CRITFC (1994) and USEPA (2002a)
Game mammals	17	50	63	90	150	859	910	All but CRITFC (1994) and USEPA (2002a)

[a] Derived from 12 studies. Ingestion of non-muscle tissues (i.e., internal organs, carcass) is estimated as 10% of total animal intake (Harris and Harper, 1997).

Table 4
Descriptive statistics and onsite vs. offsite comparisons of radionuclide concentrations in fish and wildlife muscle tissue (pCi/g wet wt) on and off the Hanford Site from 1995 to 2007[a].

Radionuclide	Statistic[b]	Fish[c]		Birds[d]		Mammals[e]	
		Onsite	Offsite	Onsite	Offsite	Onsite	Offsite
Co-60	*N*	135	55	106	26	119	20
	Mean	0.0047	0.0042	0.0032	0.00098	0.00089	−0.0012
	SD	0.012	0.013	0.012	0.0045	0.0087	0.0095
	50th percentile	0.0026	0.0015	0.0015	0.0011	0.00087	−0.00084
	95th percentile	0.028	0.031	0.027	0.0097	0.016	0.0071
	P	0.74		0.62		0.67	
Cs-137	*N*	135	55	106	26	119	20
	Mean	0.0079	0.0089	0.0053	0.010	1.7	0.010
	SD	0.011	0.021	0.012	0.032	18	0.028
	50th percentile	0.0064	0.0082	0.0041	0.0015	0.0025	0.0036
	95th percentile	0.028	0.048	0.022	0.066	0.091	0.075
	P	0.81		0.15		0.61	
Sr-90	*N*	18	6	20	ID	ID	ID
	Mean	0.00080	−0.00054	0.0019			
	SD	0.0017	0.0024	0.0027			
	50th percentile	0.00078	−0.00094	0.0012			
	95th percentile	0.0048	0.0035	0.0075			
	P	0.077					
Tc-99	*N*	ID	ID	ID	ID	ID	ID
U-234	*N*	22	17	ID	ID	5	ID
	Mean	0.0023	0.0019			−0.00068	
	SD	0.0076	0.0021			0.0028	
	50th percentile	0.00038	0.0016			−0.0017	
	95th percentile	0.0095	0.0067			0.0041	
	P	0.34					
U-238	*N*	22	17	ID	ID	5	ID
	Mean	0.0042	0.00030			−0.0013	
	SD	0.010	0.0016			0.0033	
	50th percentile	0.0013	0			−0.0013	
	95th percentile	0.017	0.0042			0.0020	
	P	0.068					

[a] Radionuclide data are from the Hanford Environmental Information System (HEIS) database (HEIS, 1989; PNNL, 2008). Cases with insufficient data (ID, i.e., $N < 5$) were excluded from analysis.
[b] Statistics include all data (including U-qualified data). SD=standard deviation, P=P value associated with Mann–Whitney test comparing onsite vs. offsite medians (50th percentiles). No significant differences were observed in these comparisons with significance set at $P < 0.0031$ (Bonferroni $P < 0.05$).
[c] Fish include bass, carp, sucker, and whitefish.
[d] Birds include goose, pheasant, and quail.
[e] Mammals include, deer, elk, and rabbit.

determined for each radionuclide and biota group, are shown in Table 6. Total annual dose for the fish and wildlife consumption pathway was estimated to be 0.36 mrem/yr. In terms of tissue, muscle provided 42% of this dose, while non-muscle tissue contributed 58% (primarily bone and carcass). In descending order, dose contributions were muscle > bone > carcass > intestine > heart > kidney > liver. With respect to biota, fish ingestion provided the majority of the dose (64%), followed by game mammals (30%) and game birds (6%). Among radionuclides, Sr-90 was dominant, accounting for 47% of the dose. More specifically, Sr-90 in fish and mammal bone and carcass (38%), followed by Cs-137 in fish muscle (17%) and U-238 in fish muscle (12%), were the primary dose contributors.

Annual dose and lifetime cancer risk at two consumption rates and two radionuclide concentrations, summed over radionuclides and biota, are presented in Table 7. Annual dose ranged from 0.36 to 23 mrem/yr, while lifetime cancer risk ranged from 1.7E−5 to

Table 5
Descriptive statistics and onsite vs. offsite comparisons of radionuclide concentrations in fish and wildlife non-muscle tissues (pCi/g wet wt) on and off the Hanford Site from 1995 to 2007[a].

Radionuclide	Biota Group[b]	Tissue	Location[c]	*N*	Percentiles[d]		*P*[e]
					50th	95th	
Co-60	M	Heart	On	5	0.0017	0.0038	
Co-60	M	Intestine	On	5	0.0035	0.020	
Cs-137	M	Heart	On	5	−0.00016	0.0017	
Cs-137	M	Intestine	On	5	−0.0064	0.0018	
Pu-238	M	Bone	On	5	0.00016	0.00035	
Pu-238	M	Liver	On	23	1.4E−8	0.00076	0.47
Pu-238	M	Liver	Off	6	0.000015	0.00024	
Pu-238	F	Carcass	On	7	0.000012	0.00011	
Pu-239/240	M	Bone	On	5	−0.000067	−0.0000071	
Pu-239/240	M	Liver	On	23	−0.0000066	0.00047	0.11
Pu-239/240	M	Liver	Off	6	−0.000064	0	
Pu-239/240	F	Carcass	On	7	0.000022	0.000061	
Sr-90	B	Bone	On	134	0.049	0.27	0.29
Sr-90	B	Bone	Off	36	0.065	0.36	
Sr-90	M	Bone	On	135	0.27	4.8	0.072
Sr-90	M	Bone	Off	20	0.52	2.0	
Sr-90	F	Carcass	On	162	0.016	0.18	0.078
Sr-90	F	Carcass	Off	75	0.022	0.091	
U-234	M	Bone	On	7	−0.0023	0.073	
U-234	M	Kidney	On	5	−0.00038	0.0021	
U-234	F	Carcass	On	38	0.0075	0.056	0.66
U-234	F	Carcass	Off	11	0.0071	0.060	
U-238	M	Bone	On	7	0.00031	0.11	
U-238	M	Kidney	On	5	0.00012	0.0074	
U-238	F	Carcass	On	38	0.0072	0.039	0.99
U-238	F	Carcass	Off	11	0.0049	0.050	

[a] Radionuclide data are from the Hanford Environmental Information System (HEIS) database (HEIS, 1989; PNNL, 2008). Cases with insufficient data ($N<5$) were excluded from analysis.
[b] M=mammal (deer, elk, rabbit), F=fish (carp, sucker, whitefish), B=bird (goose, pheasant, quail).
[c] On=onsite, Off=offsite.
[d] Percentiles derived with all data (including U-qualified data).
[e] *P*=*P* value associated with Mann–Whitney test comparing onsite vs. offsite medians (50th percentiles). No significant differences were observed in these comparisons with significance set at $P<0.0031$ (Bonferroni $P<0.05$).

Table 6
Annual effective dose from consumption of muscle and non-muscle tissues in fish, game birds, and game mammals at the Hanford Site for eight radionuclides, calculated at median consumption rates and median radionuclide tissue levels.

Radionuclide	Tissue	Dose (mrem/yr)				Contribution (%)[b]
		Fish	Game birds	Game mammals	Sum	
Co-60	Muscle	0.0064	0.00043	0.00036	0.0072	2
Cs-137	Muscle	0.061	0.0045	0.0040	0.070	19
Sr-90	Muscle	0.016	0.0029	ID[a]	0.019	5
Tc-99	Muscle	ID	ID	ID	0	0
U-234	Muscle	0.014	ID	0	0.014	4
U-238	Muscle	0.043	ID	0	0.043	12
Co-60	Non-muscle	ID	ID	0.00012	0.00012	0
Cs-137	Non-muscle	ID	ID	0	0	0
Pu-238	Non-muscle	0.00022	ID	0.00025	0.00047	0
Pu-239/240	Non-muscle	0.00045	ID	0	0.00045	0
Sr-90	Non-muscle	0.036	0.013	0.10	0.15	42
U-234	Non-muscle	0.030	ID	0	0.030	9
U-238	Non-muscle	0.026	ID	0.00013	0.026	7
Sum		0.23	0.021	0.11	0.36	
Contribution (%)[c]		64	6	30		

[a] ID=insufficient data ($N<5$).
[b] Contribution of dose from each radionuclide in muscle or non-muscle tissue (summed over biota) to total dose from fish and wildlife consumption.
[c] Contribution of dose from fish, game birds, or game mammals in muscle and non-muscle tissues (summed over radionuclides) to total dose from fish and wildlife consumption.

1.1E−3. At 95th percentiles of consumption rates and radionuclide concentrations, 74% of the total dose came from Sr-90 in mammal bone, followed by 8% from Cs-137 in fish and mammal muscle and 4% from U-238 in fish muscle. When comparing dose or risk, corresponding to 50th vs. 95th percentiles of consumption rates and radionuclide tissue concentrations, radionuclide concentration had about a three-fold greater influence than consumption rate.

Table 7
Annual effective dose and lifetime cancer morbidity risk estimates from consumption of fish and wildlife at the Hanford Site, summed over radionuclides and biota, and calculated at 50th and 95th percentiles of consumption rates and radionuclide tissue levels.

Consumption rate percentile	Dose (mrem/yr)		Risk	
	Radionuclide level percentile		Radionuclide level percentile	
	50th	95th	50th	95th
50th	0.36	4.3	1.7E−5	2.0E−4
95th	1.4	23	6.6E−5	1.1E−3

4. Discussion

4.1. Consumption rates for fish and wildlife

Fish consumption rates listed in Table 3 fall within the range of rates compiled by Harper and Harris (2008) but are much higher than many of the fish consumption rates recommended in USEPA's Exposure Factors Handbook for various population groups (USEPA, 1997b). For Native American subsistence fishing populations, USEPA recommended a mean intake of 70 g/day (170 g/day at the 95th percentile). For the general population, USEPA listed a mean intake of 6.6 g/day for freshwater/estuarine fish. In a more recent compilation, USEPA (2002b) reported a similar mean intake (6.3 g/day) for freshwater/estuarine finfish and shellfish for the United States population (41 g/day at the 95th percentile). However, for the fish consuming portion of the United States population, USEPA (2002b) listed a mean intake (78 g/day), comparable to their mean intake for Native Americans (USEPA, 1997b).

Similarly, consumption rates for game birds and game mammals (Table 3) are much higher than game intakes listed by USEPA (1997b). For example, USEPA presented mean intakes for both Native American and White populations as <1 g/day with only 0.6% of Native Americans and 1.4% of Whites consuming game. These data included both children and adults in survey populations.

4.2. Radionuclide concentrations in fish and wildlife

Radionuclide concentrations in fish in our study (Tables 4 and 5) were well below levels of concern identified by Scott et al. (2005) for Cs-137, Sr-90, U-238, and Pu-239/240. Similarly, ATSDR (2006) found that current radionuclide concentrations in muscle tissue of Hanford game animals (e.g., deer, elk, rabbits) were below their screening levels. These Hanford biota data (ATSDR, 2006) were also below comparison values reported by PNNL (1998).

Relative to radionuclide biota concentrations in Table 4, higher concentrations have been reported historically at the Hanford Site for the operational period (ATSDR, 2006). However, from 1971 to 1988, decreasing trends at Hanford have been documented for Cs-137 in rabbits, mule deer, and upland game birds, as well as for Zn-65 and Co-60 in mountain whitefish (Eberhardt et al., 1989). More recently, Poston (1994) reported a gradual reduction for Cs-137 in bass muscle tissue and Sr-90 in bass and whitefish carcass from 1982 to 1992 in the Hanford Reach. With respect to muscle tissue, Poston (1994) also found no difference in onsite vs. offsite concentrations for Sr-90 and Cs-137 in bass and for Cs-137 in whitefish, but found higher onsite concentrations for Cs-137 in carp and for Sr-90 in whitefish. These higher onsite radionuclide concentrations in muscle were not observed for fish in our study (Table 4), likely due to the more recent time period evaluated.

4.3. Dose and risk from fish and wildlife consumption

4.3.1. Background dose and risk

Given that the purpose of this study was to determine current dose and risk for a fish and wildlife consumption pathway at Hanford, dose and risk (Table 7) were calculated from onsite radionuclide concentrations in biota. However, because onsite vs. offsite radionuclide concentrations in biota were not statistically different for the nuclides examined (Tables 4 and 5), it follows that corresponding onsite dose and risk are not significantly different from background. The primary radionuclides contributing to dose and risk are found not only in Hanford waste streams but are also globally present from fallout, due to past atmospheric testing of nuclear weapons. Our results should be viewed in this context.

4.3.2. Dose and risk limits

Radiological annual dose limits for various receptor populations are available from national and international organizations. For example, USEPA specifies a dose limit of 15 mrem/yr for cleanup sites (USEPA, 1997a), whereas the International Commission on Radiation Protection (ICRP) recommends a dose limit of 100 mrem/yr for the general public (ICRP, 1991). In addition, many organizations also reinforce dose limits with an objective of "as low as reasonably achievable" (ALARA) for concentrations of residual radioactive materials (NCRP, 2004). When calculated with median consumption rates and radionuclide concentrations, the annual dose estimate for the fish and wildlife consumption pathway was well below these limits (Table 6). Only with high end inputs (95th percentile) for both consumption rates and radionuclide levels did dose exceed the USEPA limit (Table 7). In addition, dose estimates in Table 7 are well below the mean natural background dose of 300 mrem/yr in the United States (USEPA, 2002a).

Dose and risk are both derived from radionuclide concentrations in environmental media. However, committed effective dose (i.e., energy absorbed in the body in a 50 year period following intake of a radionuclide, and weighted by type of radiation and tissue) is a more fundamental measure than cancer risk, since dose does not express a health risk per se. International organizations (e.g., ICRP) and the United States Nuclear Regulatory Commission (USNRC) have specified protective and regulatory limits in terms of dose, whereas USEPA has specified limits in terms of both dose and risk. Similarities and differences between dose and risk for radioactively contaminated sites have recently been examined (NCRP, 2004).

It is instructive to illustrate the correspondence between dose and risk. Table 8 lists several regulatory or protective dose limits, along with their associated risks. Lifetime cancer morbidity risks calculated for the fish and wildlife pathway in our study (Table 7) fall below or overlap the lower end of these risks in Table 8.

4.3.3. Dose and risk comparison to other studies

Comparison of dose and risk estimates in our study with estimates in the literature must recognize differences in exposure pathways and time periods evaluated, along with differences in reported statistical expressions. Our study was limited to radionuclide exposure from a single pathway, using data collected from 1995 to 2007. Fish and wildlife consumption is only one pathway within a multidimensional Native American scenario, where exposure to contaminants (including both radionuclides and non-radionuclides) may occur via ingestion, inhalation, dermal contact, and external radiation with relevant abiotic and biotic media. In addition, historical dose and risk estimates at the Hanford Site during the period of nuclear materials production

Table 8
Examples of radiological dose limits and corresponding lifetime cancer morbidity risks.

Dose description	Dose limit (mrem/yr)	Reference	Risk[a]
Drinking water standard	4	USEPA (2006)	2.1E−4
Contaminated site remediation	15	USEPA (1997a)	8.0E−4
License termination rule	25	USNRC (1997)	1.3E−3
Protection of the general public	100	ICRP (1991)	5.3E−3

[a] Assuming 7.6E−7 risk/mrem for cancer risk morbidity (NCRP, 2004) and a 70 yr lifetime.

(1944–1987) are expected to be higher than estimates for more recent time periods, due to the cessation of Hanford reactor operations, waste reduction activities on the Hanford Site, radionuclide decay, as well as the ban on atmospheric testing of nuclear weapons. Finally, reported statistical estimates (e.g., central tendency vs. high end) must be considered when comparing studies.

Several studies have estimated historical radiation dose and risk to Columbia River residents and fishermen during the operational time period at the Hanford Site (e.g., Hoffman et al., 1997; Walker and Pritchard, 1999; RAC, 2002). The Hanford Environmental Dose Reconstruction Project (HEDR) constitutes a major effort to estimate historical doses received by representative persons who lived along the Columbia River during the Hanford production years (Farris et al., 1994; ATSDR, 2006). Because our paper focuses on current conditions, however, this discussion highlights more recent dose and risk estimates.

Total lifetime radiation dose (estimated at year 2000) to individuals living a Native American subsistence lifestyle along the Hanford Reach has been predicted by Scott et al. (2005) to range from 5E−6 to 2E0 rem over a 70 year period. According to these authors, higher doses result primarily from contaminated groundwater entering the Columbia River at three locations (i.e., 100 Areas [former plutonium production reactors], Hanford townsite [discharges from 200 Area historic chemical processing and waste storage], and 300 Area [former uranium reactor fuel fabrication]). If extrapolated to lifetime dose, radiological doses calculated in our study for only a single pathway (Table 7) fall within this dose range, estimated by Scott et al. (2005).

Radiological lifetime cancer mortality risk for three Native American scenarios near the N Reactor Area at the Hanford Site (including a full array of exposure pathways) has been modeled by PNNL (1998). Median and 95th percentile risks were approximately 3E−3 and 6E−2, respectively. As expected, these complete scenario risks are greater than lifetime risks in our study, predicted for only the fish and wildlife consumption pathway (Table 7).

Radiological lifetime cancer risk, resulting from fish consumption only, was estimated for the general public and for the Columbia River Inter-tribal Fish Commission (CRITFC) member tribes at two consumption rates (USEPA, 2002a). For Native Americans, risk ranged from 3E−5 to 6E−5 (mean intake of 63 g/day) and from 2E−4 to 4E−4 (99th percentile intake of 389 g/d). Therefore, estimated lifetime cancer risks for Native Americans at high end radionuclide concentrations in our study (2.0E−4 to 1.1E−3, Table 7) were comparable to risks for Native Americans at the high intake rate in the USEPA (2002a) study, considering that our estimates also included ingestion of game birds and game mammals.

4.4. Uncertainty of dose and risk estimates

4.4.1. Analytical approach

Dose and risk can be estimated with either deterministic or stochastic methods. These methods are typically applied to exposure data, and both uniquely provide information on uncertainty of results. Although each has its own set of advantages and disadvantages (USEPA, 2001), a deterministic approach was selected, based on the following rationale.

Both USEPA (1997c) and NCRP (1996) have stated that a formal uncertainty analysis (e.g., Monte Carlo) may be unnecessary when conservatively biased screening calculations indicate that the exposure or risk is clearly below regulatory levels of concern. We believe this is the case in our study, considering that consumption rates are biased high (relative to the general population), the estimated median annual dose of 0.36 mrem/yr (Table 7) is well below the regulatory limit (15 mrem/yr), and (importantly) no difference was observed between onsite (Hanford) vs. offsite (background) radionuclide concentrations in biota.

Furthermore, a deterministic analysis is supported by results of a simple sensitivity analysis. Median inputs (i.e., radionuclide concentration in biota and consumption rates) were increased by 50%, resulting in a dose of 0.81 mrem/yr. Because this dose remains well below the 15 mrem/yr regulatory limit, a stochastic analysis would add little value to our results.

Finally, it is clear that total uncertainty in our dose and risk estimates cannot be captured with either deterministic or stochastic analysis. As described in the following section, several other sources contribute uncertainty beyond those factors addressed by the analytical approach (e.g., data gaps in radionuclide and biota data). For these reasons, the simpler deterministic approach was employed to estimate dose and risk.

4.4.2. Sources of uncertainty

Sources of uncertainty in radiological dose and risk estimates include factors related to exposure (e.g., consumption rates, biota radionuclide concentrations). Consumption rates are largely self-reported estimates, based on surveys and interviews (Table 1). Limitations associated with these data collection methods include dietary recall, timing and location of survey, and tribal member selection (CRITFC, 1994). More recently, problems associated with conventional survey methods for collecting fish consumption data from Native Americans have been examined by Donatuto and Harper (2008) and Harper and Harris (2008).

Several assumptions relate to consumption rates in our study. In particular, consumption rates of wildlife required assumptions for cases where game animals were not specifically identified. Ingestion of non-muscle tissues was assumed to represent 10% of total animal intake (Harris and Harper, 1997). Lastly, in terms of estimating lifetime risk, adult consumption rates were assumed over the entire lifetime.

In general, exposure assessment was also constrained by assumptions and limitations relating to biota groups included, radionuclides evaluated, and time period examined (i.e., 1995–2007), along with analytical aspects of radionuclide measurement. In particular, shellfish and all plant materials were excluded from our analysis of fish and wildlife. Radionuclide data gaps (e.g., insufficient data for Tc-99 in fish and wildlife muscle tissue) also contributed uncertainty to exposure estimates.

Biota radionuclide concentrations comprised a source of uncertainty for exposure in terms of completeness and representativeness of fish and wildlife samples in the HEIS database (HEIS, 1989). For example, in terms of completeness, although salmon are a primary food fish, salmon radionuclide data were absent in the HEIS database for the period examined. With respect to representativeness, sample designs for collection of biota data in HEIS are not intended to be entirely statistically based. Also, biological processes and attributes may contribute uncertainty to exposure estimates. For example, differences in toxicokinetic factors (i.e., absorption, distribution, biotransformation, and elimination) and life history traits (e.g., migratory behavior) in fish and wildlife species influence radionuclide tissue concentrations.

In addition to exposure-related factors, sources of uncertainty in dose and risk estimates include factors related to toxicity (e.g., dose conversion factors, risk coefficients, and $f1$ values). Uncertainties in toxicity factors have been reviewed by USEPA (1999). For example, dose conversion factors and risk coefficients are derived from models representing the biological behavior of radionuclides in the human body, doses to radiosensitive tissues, and lifetime cancer risk per unit dose to these tissues. These biokinetic, dosimetric, and radiation risk models have substantial uncertainties associated with their predictions which are propagated in the derivation of dose conversion factors and risk coefficients. Similar to many cancer risk assessments for non-radionuclides, another major source of uncertainty in radiogenic cancer risk estimation is the use of a linear no-threshold model to calculate risks for low acute doses or low dose rates.

5. Conclusions

Published tribal consumption rates, along with site-specific radionuclide concentrations in fish, game birds, and game mammals compiled over the period 1995–2007, were used to estimate current radiological dose and risk for a fish and wildlife consumption pathway within an adult Native American exposure scenario on the Hanford Site. No significant differences were observed for radionuclide concentrations in biota onsite vs. offsite (background). By extension, dose and risk for this pathway were not significantly different from background.

Furthermore, using median consumption rates and radionuclide tissue concentrations, estimated dose was well below USEPA and ICRP dose limit recommendations. Although present in Hanford waste streams, the primary radionuclides contributing to dose and risk are also ubiquitous in the environment, due to past atmospheric testing of nuclear weapons. Recognizing uncertainties in exposure and toxicity assessments, our results may facilitate informed decision making and optimize resource allocation within a risk assessment framework at the Hanford Site.

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