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Risks to Ecological Receptors Posed by Contaminants of Potential Concern in the Lower Three Runs Cooling Ponds and Canals

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ABSTRACT

The upper portion of Lower Three Runs includes several ponds, reservoirs, and canals that were formerly used as a cooling system for nuclear production reactors. This area was divided into nine exposure areas (EAs) for the assessment of environmental contamination resulting from past reactor operations and other industrial processes. A tiered screening process identified several contaminants of potential concern including aluminum, cyanide, lead, manganese, mercury, DDD, DDE, and DDT. Risks posed by these contaminants to ecological receptors (river otter, belted kingfisher, raccoon, and blue heron) were assessed using contaminant exposure models that estimated contaminant intake resulting from ingestion of food, water, and sediment/ soil and compared these intakes with toxicity reference values (TRVs).

The contaminant exposure models showed that the TRVs were not exceeded in the otter model, exceeded by aluminum in EA 7 (Pond 2 and associated canals) in the raccoon model, and exceeded by mercury in EAs 2, 3 (Pond B), 6 (Par Pond), and 8 (Ponds 4 and 5 and Canal to Pond C) in both the kingfisher and blue heron models. Hazard quotients (total exposure dose divided by the TRV) were 2.8 for aluminum and 1.7- 3.6 for mercury. The primary route of exposure for aluminum was the ingestion of soil, and the primary route of exposure for mercury was the ingestion of mercury contaminated fish.

Elevated levels of mercury in fish were at least partly the result of the aerial deposition of mercury onto Lower Three Runs and its watershed. The atmospheric deposition of mercury creates pervasive contamination in fish throughout the Savannah River basin. Another possible source of mercury was the discharge of mercury contaminated Savannah River water into the Lower Three Runs cooling ponds and canals during previous years of reactor operation. This contamination originated from industries located upstream of the SRS.

The aluminum exceedance for the raccoon was likely the result of naturally high aluminum levels in SRS soils rather than SRS operations. Aluminum exceedances have previously been observed in relatively undisturbed background locations as well as areas affected by SRS operations. Aluminum exceedances are more likely with the raccoon than the other receptors because it consumes more soil as a result of its feeding habits.

Sensitivity analysis showed that model uncertainty can be reduced by adequate sampling of key variables (e.g., fish and sediments). Although sediment samples were collected from all EAs, fish samples were not collected from three EAs and some analytes (pesticides and cyanide) were not measured in fish. Water-to-fish concentration ratios were used to estimate contaminant levels in fish when direct measurements from fish were unavailable; however, such estimates are potentially less accurate than direct measurements.

INTRODUCTION

The Department of Energy (DOE) has conducted industrial operations at the Savannah River Site (SRS) near Aiken, South Carolina since the early 1950s resulting in the release of contaminants into some SRS streams. To better understand the effects of these contaminants, the 780 km^2 SRS has been partitioned into Integrator Operable Units (IOUs) that correspond to the SRS tributaries that drain into the Savannah River. IOUs are surface water bodies (e.g., streams, ponds, and lakes) and associated wetlands, including sediment/soil, surface water, and associated biota. They are "integrators" because they have the potential to receive contaminants transported by surface or subsurface flow from all the potential sources (Operable Units) within their watersheds. Animals (i.e., ecological receptors) feeding within stream-based food chains are exposed to these contaminants, and their health can be considered an integrative indicator of the severity of contamination within the stream watershed.

The SRS has six IOUs that are subdivided into IOU subunits that correspond to portions of a stream that may differ in exposure to potential sources of contamination. Lower Three Runs (LTR) is one of the six SRS IOUs. It consists of several subunits including the upper, middle, and lower portions of LTR plus ponds, reservoirs, and canals that were formerly used for reactor cooling. All have had the potential to be affected by contamination associated with past SRS operations except for the portion of LTR located upstream from the cooling ponds. These areas may also be affected by other sources of contamination including historical land uses (e.g., the application of pesticides) and the atmospheric deposition of pollutants.

The SRS has a comprehensive process for assessing the ecological effects of contaminants in the IOUs. The process involves 1) the collection of contaminant data from a variety of environmental media with an emphasis on sediment, fish, and surface water; 2) the use of contaminant exposure models that estimate potential contaminant doses to ecological receptors (EPA 1993); and 3) field bioassessments of the fish and invertebrate assemblages inhabiting SRS streams. The data generated by these studies provide a broad and integrative basis for a weight-of-evidence characterization of the extent and severity of contaminant related ecological impacts on SRS aquatic ecosystems. The LTR IOU was evaluated with this process in 2009 (Paller and Dyer 2009); however, this evaluation did not specifically assess ecological risks in each of the cooling ponds and canals formerly used for reactor cooling.

In 2015, the risks to ecological receptors posed by metals were assessed in the LTR cooling ponds and canals plus four reference areas that were largely unaffected by SRS operations (Paller and Blas 2015). This was accomplished with contaminant exposure models that estimated potential doses to the river otter *Lontra canadensis*, belted kingfisher *Ceryle alcyon*, raccoon *Procyon lotor,* and blue heron *Ardea Herodias* from metals ingested in food, water, sediment, and soil within the study area (Paller et al. 2008). The main findings of the 2015 study are listed below:

1) Metals in the LTR cooling ponds were not present at levels sufficient to harm mammals and birds with the exceptions of mercury and aluminum.

2) The primary route of mercury exposure for all receptors was the consumption of contaminated fish, which was at least partly caused by the aerial deposition of mercury from non-SRS sources. A second possible cause was the discharge of mercury contaminated Savannah River used for reactor cooling into the LTR cooling ponds during previous years of reactor operation. This contamination originated from industries located upstream of the SRS.
3) The primary route of aluminum exposure was the incidental consumption of soil, as a likely result of naturally high aluminum levels in soils rather than SRS operations.

The risks posed by contaminants in the cooling ponds and canals in the uppermost portion of LTR were reevaluated in 2017 based on a thorough screening study that identified several new constituents of potential concern (COPCs) within the study area. The 2017 study employed contaminant exposure models like those used in the previous 2015 study except that the models were updated with newer toxicity reference values (TRVs), hereafter termed LANL TRVs (LANL 2015). These TRVs were compared with the metal doses predicted by the contaminant uptake models to identify contaminants that may pose risks to the river otter, belted kingfisher, blue heron, and raccoon.

MATERIALS AND METHODS

Study Area

Lower Three Runs is a large blackwater creek that drains about 460 km^2 in the southeastern SRS. It is low gradient, generally neutral in pH, and has a sandy bottom covered with woody debris in some places (Paller and Dyer 2004). Its upper reaches were dammed to form Par Pond, a 1012 ha reservoir formerly used for cooling P and R Reactors. Several additional pre-cooler ponds were constructed in the headwaters above Par Pond. Pond B, the largest, is about 73 ha (180 acres). Smaller ponds include Pond A, Pond 2, and Ponds 4 and 5. Pond C, a pre-cooling pond for Par Pond, is separated from Par Pond by an earthen dam and is about the size of Pond B. These cooling ponds and the canal system that connects them were divided into nine exposure areas (EAs) for computation of potential ecological risks (see SRS 2017a for details) (Figure 1). The EAs are listed in Table 1 and briefly described below:

1. EA 1 – Pond A and R Discharge Canal;
2. EA 2 – Canal between Pond A and Pond B;
3. EA 3 – Pond B;
4. EA 4 – Canal between Pond B and PAR Pond;
5. EA 5 – Joyce Branch/Old Discharge Canal;
6. EA 6 – PAR Pond;
7. EA 7 – Pond 2 and associated canals;
8. EA 8 – Ponds 4 and 5 and Canal to Pond C; and
9. EA 9 – Pond C.

Environmental Samples and Contaminant Measurements

The contaminant data under analysis included inorganic and organic compounds in wetland soils and sediments, surface water, and fish measured primarily during 2009 and 2010. Sediment and sediment/soil (floodplain sediments) were evaluated as a single medium and identified as "sediment/soil." Much of the contaminant data were collected by the IOU program, supplemented by medium and high pedigree data from the IOU data base, including data collected by the SRS Environmental Monitoring Program, Savannah River National Laboratory, Savannah River Ecology Laboratory (SREL), and others. The IOU data base is described further in EGIS (2007), Paller et al. (2008), and SRS (2017a).

Contaminant concentrations in biota were usually derived from whole organisms, sediment concentrations from bulk sediment, and water concentrations from unfiltered water. Sediment and surface water were available for all EAs, although not all constituents were measured in all surface water samples, and some constituents were below detection limits (Table 2). Fish data were collected by the IOU program from EA 3 (Pond B), EA 6 (PAR Pond), EA 8 (Pond 4), and EA 9 (Pond C). Additional fish and crayfish (and limited tadpole) data were available from the Savannah River Ecology Laboratory for EA 1 (Pond A and R-Discharge Canal) and EA 7 (Pond 2).

Differences in metal speciation that could affect metal toxicity were not evaluated because supporting environmental information needed to assess metal chemistry was usually unavailable. Constituents were assumed to be present in the most toxic state likely to occur. For example, all mercury in fish was conservatively assumed to be methylmercury because methylmercury rather than inorganic mercury predominates in the bodies of fish and other aquatic organisms.

Mercury concentrations in fish are usually correlated with size, age, and trophic level (higher in predators). The fish collected from the LTR cooling system ponds and canals were typical of the larger species and size ranges occupying these waters. For example, most of the fish collected from the ponds were large specimens of largemouth bass, which likely represented the highest mercury concentrations likely to occur in LTR fish.

Contaminant Exposure Models

Multiple constituents in sediments/soils, water, and fish were screened in each EA using a tiered process described in the LTR Remedial Investigation/Baseline Risk Assessment (RI/BRA) report (SRNS 2017a) to identify contaminants present at concentrations high enough to be of concern. The initial ecological screening process conducted to support the RI/BRA was documented in the scoping summary for the LTR IOU (SRNS, 2017b). The screening consisted of comparing maximum concentrations in sediment/soil and surface water to No Adverse Effect Level (NOAEL) ecological screening values (ESVs) followed by a refined screen against Lowest Observed Adverse Effect Levels. Sediment/soil or surface water constituents that failed the initial and refined screening were evaluated in an uncertainty evaluation that considered mean values, background levels, frequency of detections, and age/quality of the data to determine refined constituent of potential concern (RCOPCs). Ultimately, the screening process identified

nine constituents in seven of the nine EAs as RCOPCs for further evaluation using the contaminant exposure models (Table 1). These constituents are the focus of this report.

The contaminant exposure models calculated exposure doses resulting from the ingestion of contaminants in fish, crayfish, water, and sediment/soil. Exposure point concentrations (EPCs) represented the doses of contaminants in each of these media for each contaminant in each EA (Table 2). The EPCs were calculated from the environmental data by a process described in detail in the LTR RI/BRA report (SRNS 2017a). EPCs were represented by the lesser of the maximum detected concentration and the 95% upper confidence limit (UCL) of the mean concentration. Some EPCs were calculated using concentration ratios, as described later.

The EPCs for each of the four media served as inputs to contaminant exposure models developed for the river otter, belted kingfisher, blue heron, and raccoon. The first three species are representative of the aquatic environments under study, locally common, vulnerable to contaminants because they feed largely on aquatic organisms, and are near the apex of the aquatic food chain. The raccoon is an omnivore that commonly forages in wetland and floodplain habitats. The duration of exposure for all receptors was assumed to be long-term, and the receptors were assumed to spend all of their time in the evaluation areas. Ingestion of food, surface water, and soil were assumed to be the primary exposure pathways. Dermal and inhalation pathways are generally insignificant compared with ingestion pathways and insufficiently understood to properly evaluate.

The diet of the river otter consists largely of fish but includes invertebrates (assumed to be crayfish), amphibians and reptiles (collectively termed herptiles), birds, and mammals. Estimated dietary composition was 65% fish, 15% crayfish, 10% herptiles, 5% birds, and 5% mammals. The belted kingfisher has a more restricted diet consisting of 70% fish, 15% amphibians and reptiles, and 15% crayfish. The blue heron diet was estimated to be 95% fish, 1% crayfish, 3% amphibian, and 1% birds and mammals. The diverse diet of the raccoon includes animal and vegetable matter derived from both aquatic and terrestrial environments. Estimated composition was 43% fruit, 20% grains and nuts, 18% insects, 8% crayfish, 4% herptiles, 3% rodents, 2% molluscs, 1% fish, and 1% birds and mammals. Dietary estimates are derived from EPA (1993).

No assumptions were made regarding the species and size of fish consumed; however, as previously described, the fish were generally among the larger specimens characteristic of the EAs. Metal concentrations in herptiles were unmeasured but assumed to be the same as in fish because both types of organisms are ectothermic and feed mainly on aquatic/riparian invertebrates and small vertebrates. Birds and mammals consumed by the otter, raccoon, and blue heron were assumed to be mainly waterfowl and rodents, which may have different contaminant body burdens than fish because they are homeothermic and may feed outside of aquatic food chains.

The contaminant exposure models required information on the ingestion rates for different food sources, water, and sediment/soil for each receptor. Ingestion of sediment/soil was included because soil or sediment is often ingested inadvertently while

feeding or intentionally to meet trace mineral needs (EPA 1993). Ingestion rates were computed on the basis of dietary composition (previously described), gross energy content, assimilation efficiency for each food, and the metabolic rates of the receptors. These computations and associated parameters are shown in detail in Appendices 1-4. The metabolic rates of the receptors were computed on the basis of body weight using allometric models (EPA 1993). Allometric models (EPA 1993) were also used to compute water ingestion rates for each receptor (Appendices 1-4). The soil ingestion rate for the otter was assumed to be 2.8%, which is the soil consumption rate of the red fox (EPA 1993), another mammalian carnivore of approximately comparable size. The soil consumption rates of the kingfisher and blue heron, species for which soil consumption data for comparable species were lacking, was assumed to be 2%. The soil ingestion rate for the raccoon was 9.4% (EPA 1993). Ingestion rates for all pathways were summed as follows:

$ED_{total} = \sum_{i=1}^{n} ED_{food\ i} + ED_{water} + ED_{soil}$, where:

ED_{total} = total exposure dose from all sources (mg/kg/d)
$ED_{food\ i}$ = exposure dose from ingestion of food source i
ED_{water} = exposure dose from ingestion of water
ED_{soil} = exposure dose from ingestion of soil.

The exposure dose resulting from each pathway was represented as a daily intake normalized to body weight (mg/kg/d) (Appendices 1-4). Total daily exposure (ED_{total}) was subsequently compared with the LANL TRVs (LANL 2015) to identify potentially hazardous constituents (Table 3). A hazard quotient (HQ) was calculated by dividing ED_{total} by the TRV. The percent contribution of each pathway (each food source, water, and soil) to ED_{total} was also computed.

A TRV for the ingestion of iron by mammals and birds was unavailable. Therefore this constituent, which exceeded screening levels in EA 7 (Table 1), was not evaluated with the contaminant exposure models.

Concentration ratios

Concentration data needed to calculate EPCs for use in the contaminant exposure models were unavailable for some environmental media. For example, concentrations of DDE, DDD, DDT, aluminum and iron were not measured in fish because these constituents were not associated with reactor operations. Also, constituents were sometimes below detection levels making it impossible to compute EPCs for some media in some EAs. Last, data to calculate EPCs were unavailable for receptor foods that were difficult or impossible to collect (small mammals, birds, fruits, insects). In the absence of EA specific EPCs, concentration ratios (CRs) were used to estimate contaminant levels needed as input for the contaminant exposure models. CRs express the relationship between the concentration of a contaminant in one environmental medium (e.g., sediment/soil) and the concentration of the same contaminant in another medium (e.g., animal tissue). These relationships made it possible to estimate the contaminant concentration in a medium that was unmeasured (e.g., small mammal tissue) from the contaminant concentration in another medium that was measured (e.g., sediment). Six

types of CRs were used: soil/sediment-to- mammal, soil/sediment-to- bird, fish-to-crayfish, water-to-fish, soil/sediment-to-plant reproductive tissue, and soil/sediment-to-invertebrate (Table 4).

Soil-to-mammal CRs were used to compute contaminant concentrations in the tissues (whole body) of small mammals potentially consumed by the ecological receptors. CRs for metals were computed from metal concentrations in cotton rats *Sigmodon hispidus* and soil collected from five sample sites on the Par Pond lake bed after it had been exposed during an extended drawdown (Paller and Wike 1996). The CRs were computed as follows:
$CR = C_{animal}/C_{soil\ or\ sediment}$ where:
CR = the tissue to soil CR for a particular metal
C_{animal} = average metal concentration in animal whole body (wet weight)
$C_{soil\ or\ sediment}$ = average metal concentration in sediment (dry weight).
Average CRs for the metals, computed from the five sites in Par Pond, were used in the contaminant exposure models (Table 4).

Soil-to-tissue CRs computed from SRS data, as described above, are termed SRS-specific CRs. SRS-specific data were unavailable to compute soil/sediment-to-mammal CRs for cyanide, DDD and DDT. CRs for these contaminants were obtained from "Table 1. Biouptake Factors" shown in the ACP Regulatory Document Handbook (ERD 1999).

With the exception of mercury, the soil/sediment-to- mammal CRs for metals described above were also used for birds. Mercury data collected by SREL from Par Pond waterfowl were divided by the average mercury levels in Par Pond sediments (sediment data from the IOU data bases) to compute a bird-specific CR for mercury (Table 4). Soil-to-bird CRs for cyanide and pesticides were taken from ERD (1999) and shown in Table 4).

Water-to-fish CRs were used to estimate contaminant concentrations in fish for EAs that lacked fish data for some or all constituents (Table 4). Water- to-fish CRs for aluminum, cyanide, DDD, DDT, and lead were taken from ERD (1999). Water-to-fish CRs for DDE were computed by averaging the CRs for DDE reported in the "Health Effects Support Document for DDE" (EPA 2008). An SRS-specific water-to-fish CR was calculated for mercury from environmental data in the SGCP data bases.

Fish-to-crayfish CRs were used to compute aluminum, mercury, manganese, and lead levels in crayfish. These CRs were estimated from fish and crayfish data collected in five SRS IOU subunits including lower Fourmile Branch, lower Lower Three Runs, middle Lower Three Runs, lower Pen Branch, and lower Steel Creek (see Paller et al. 2008 for a description of locations). The fish-to-crayfish CRs were calculated as follows:
$CR = C_{crayfish}/C_{fish}$ where:
CR = the tissue to tissue concentration ratio for a particular metal
$C_{crayfish}$ = average metal concentration in crayfish whole body (wet weight)
C_{fish} = average metal concentration in fish (whole body wet weight).

CRs were averaged across the five subunits to produce a final CR for each metal (Table 4). Crayfish-to-water and crayfish-to-soil/sediment CRs were also examined but not used because they were more variable than fish-to-crayfish CRs. Fish-to-crayfish CRs were unavailable for organic contaminants necessitating the use of water-to-fish CRs to compute DDD, DDE, and DDT levels in crayfish.

Soil/sediment-to-plant reproductive tissue (i.e., fruits and grains) CRs and soil/sediment-to-invertebrate CRs were used only in the contaminant exposure models for the raccoon to estimate contaminant concentrations in the fruit, grains and nuts, and insects consumed by this organism (Table 4). These CRs were obtained from ERD (1999). Concentrations in the crayfish, herptiles, mammals, and birds consumed by raccoons were estimated as previously described. Concentrations in molluscs consumed by raccoons were assumed to be the same as in fish.

Contaminant Exposure Model Uncertainty

Like similar models, the contaminant exposure models presented herein, are underpinned by various assumptions that contribute to uncertainty in model output. Some can only be evaluated qualitatively, but other sources of uncertainty can be analyzed rigorously using a quantitative sensitivity analysis. Sensitivity analyses based on Monte Carlo simulations were used to evaluate uncertainty in exposure model results resulting from variability in metal concentrations and CRs.

These analyses, which were initially reported in Paller et al. (2008), made use of data from Lower Three Runs and Fourmile Branch (Paller et al. 2008). Both streams have been impacted by reactor operations and industrial processes on the SRS and are representative of locations with relatively high contaminant concentrations that resulted in TRV exceedances (Paller et al. 2008). Representative models were used in the sensitivity analysis to identify environmental media that had the greatest influence on total exposure and examine the impact of the concentration ratios (CRs) on model results.

The sensitivity analysis employed a Monte Carlo simulation approach in which contaminant concentrations and CRs were represented as probability distributions rather than point estimates. The probability distributions for each contaminant described the range of values that the contaminants could take and the likelihood of occurrence of each value within the range. The probability distribution for each contaminant in each medium was based on the distribution of the contaminant data in the IOU subunit under analysis with goodness of fit determined by a chi-square test. In cases where data distributions did not correspond with commonly used probability models (e.g., log-normal, normal, exponential, etc.), the probability model used in the simulation was modeled after a histogram of the concentration data. Contaminant probability distributions were truncated at the maximum observed concentration to avoid unrealistically high exposure scenarios. Model realism was also maintained by measuring correlations among contaminant concentrations in different media (soil/sediment, water, fish, and crayfish) and constraining the simulation so that values for each variable were selected based on the correlations among variables rather than independently. This precluded unrealistic situations in which, for example, a very high

value for one medium (e.g., aluminum levels in crayfish) would be selected in conjunction with a very low value for an associated determinative medium (e.g., aluminum levels in sediment). Model results were expressed as histograms showing the probability of different levels of exposure. Variables that had the greatest influence on model output were identified by calculating regression coefficients (R^2s) describing the strength of the relationships between input variable values and model output values. All Monte Carlo simulations and associated analyses were conducted with @RISK software (Palisade Corp. 2004).

RESULTS AND DISCUSSION

Contaminant Exposure Models

No exposure doses for the potentially problematic contaminants identified by the screening process exceeded the TRVs for the otter (Table 5). These results differed from those of the 2015 study on the LTR cooling ponds, which showed exceedances for mercury in several EAs (Paller and Blas 2015). The reason for this difference is that the mammalian TRV for methlmercury used in the 2015 study (0.025 mg/kg/day) was substantially lower than the updated TRV for mammals now being used (0.16 mg/kg/day).

The only TRV exceedance for the raccoon was aluminum in EA7 (Table 6). Most of the aluminum intake was associated with incidental soil consumption followed by insect consumption. Aluminum exceedances have been observed before in a number of IOU subunits, including reference subunits (Paller et al. 2008). These exceedances are likely related to naturally high aluminum levels in SRS soils rather than to SRS operations.

TRV exceedances for mercury occurred in EAs 2, 3, 6, and 8 for both the kingfisher and the blue heron (Tables 7 and 8). Hazard quotients for the kingfisher ranged from 2.9-3.6, and HQs for the blue heron ranged from 1.7-2.1. Hazard quotients were somewhat lower for the blue heron than for the kingfisher because it consumed fish at a lower rate (14% of body weight per day compared with 22% for the kingfisher). Birds exhibited exceedances for mercury and mammals did not because the TRV for birds (0.064 mg/kg/day) was lower than the TRV for mammals (0.16 mg/kg/day).

The principal route of mercury exposure in all EAs was the consumption of fish (Tables 5, 6, 7, and 8). Mercury typically reaches higher levels in fish tissues than in sediment or water as a result of bioaccumulation (Bahnick et al. 1994). Relatively high levels of mercury in fish have been observed in many water bodies on the SRS including reference areas not directly affected by SRS operations. A contributing factor is the atmospheric deposition of mercury from non-SRS sources, which has resulted in relatively high levels of mercury in fish throughout the Savannah River basin (EPA 2000). However, Savannah River water contaminated with substantial amounts of mercury from industries located upstream of the SRS was formerly pumped through the LTR cooling ponds (Paller and Littrell 2007). SRS water bodies that received reactor cooling water from the Savannah River typically have elevated levels of mercury in biota compared with those that were not used for reactor cooling (Newman and Messier 1994). Thus, elevated mercury levels

in the LTR cooling ponds are probably unrelated to ongoing SRS industrial processes but may be associated with the former use of contaminated Savannah River water by the SRS.

It is well known that mercury levels in fish are directly correlated with trophic level and size/age (Bahnick et al. 1994). With the exception of EA7, most of the fish collected from the LTR EAs were relatively large (up to about 30 cm total length) specimens of largemouth bass *Micropterus salmoides*. These fish were probably among the most contaminated fish in LTR, and mercury levels in them were likely higher than in the smaller and lower trophic level fish that are usually eaten by the blue heron and kingfisher. Therefore, it is possible that the contaminant exposure models overestimated mercury intake for both receptors.

Contaminant Exposure Model Uncertainty

Monte Carlo simulations showed that the distribution of total exposures for mercury ingestion by the otter was positively skewed, with less than half of the total exposure estimates exceeding the TRV for methylmercury (Figure 2). Concentrations of mercury in fish had a much greater influence on the estimates of total exposure (R^2=0.84) than concentrations of mercury in the other environmental media (R^2=<0.01 to 0.18, Table 9) because of relatively high and variable mercury concentrations in fish combined with a high proportion of fish in the diet of the otter. In contrast, concentrations of mercury in water, which were very low, had almost no influence on the total exposure estimates. The CRs used to determine mercury levels in mammals and bird consumed by otters had comparatively small effects on the total exposure estimates (R^2s of 0.06 and 0.18, respectively), with the CR for birds being somewhat more influential because of its greater magnitude (Table 9). These results suggest that moderate errors in CR estimates and in sediment, water, and crayfish methylmercury concentrations resulting from analytical problems, unrepresentative sampling or other factors were unlikely to strongly affect the results of the mercury exposure model. Of greater potential importance were errors in estimating fish mercury concentrations.

Mercury is a constituent for which fish ingestion was a dominant exposure pathway. In contrast, sediment/soil ingestion was an important exposure pathway for aluminum. The distribution of total exposures for aluminum ingestion in the FMB-lower subunit was positively skewed with 60% of the total exposure estimates exceeding the TRV (Figure 2). Concentrations of aluminum in sediment had the greatest influence (R^2=0.69) on the exposure model results because of the relatively high concentrations of aluminum that occurred in the sediment/soil (Table 9), but concentrations of aluminum in crayfish (R^2=0.54) and fish (R^2=0.44) were also important. Concentrations in water and the CR for aluminum uptake by mammals had almost no influence on the model output.

CRs were calculated from data collected on the SRS where possible; however, such data were unavailable for some contaminants necessitating the use of a default CR derived from literature sources. Sensitivity analysis was used to evaluate the use of default CRs by comparing aluminum model results based on the default value of one (ERD 1999) with model results based on a calculated SRS-specific CR of 0.0016. Sixty percent of the

exposure estimates for aluminum in the FMB-lower subunit exceeded the LOAEL when a calculated CR was used (Figure 2) compared with 80% when the default CR of one was used in the same model. The default CR also resulted in highly skewed model results almost completely determined by aluminum concentrations in sediment (R^2>0.99, Table 9) as a result of both high concentrations of aluminum in ingested sediment and high calculated uptake of aluminum in mammal and bird prey items resulting from the use of the default CR. These results show that the use of default CRs can strongly influence model output for contaminants in which the default CR differs substantially from the site-specific CR and suggest greater uncertainty concerning exceedances associated with exposure models that employed default CRs.

These sensitivity analyses showed that input variables with high and varying values were usually the most important determinants of exposure model output. Thus, errors in fish mercury levels strongly affected mercury exposure model output because mercury levels were higher in fish than in other media. In contrast, the accuracy of the aluminum exposure model was strongly dependent on representative measurements of aluminum concentrations in sediment because aluminum levels were much higher in sediment than in other media.

The preceding results indicate that adequate sampling of key variables (e.g., fish and sediments) is necessary for model results with high certainty. Sediment sampling was adequate; however, fish samples were not collected from some EAs (Table 2) creating a problem for constituents like mercury that can reach high levels in fish and strongly influence model output. Water-to-fish CRs were used to estimate contaminant levels in fish when fish measurements were unavailable, thereby contributing uncertainty related to the accuracy of the CRs. Water-to-fish CRs could not be used to estimate concentrations in fish when water concentrations were below detection limits. This resulted in an absence of fish data that produced underestimates of total exposure, as was the case for DDD and DDE in EA 8 (Table 2). However, it is unlikely that DDD and DDE were an appreciable concern in EA8 considering that these constituents were below detection limits in surface waters.

A number of additional factors can contribute to uncertainty in the contaminant exposure model results. These are listed and discussed qualitatively below:

1) *Temporal changes in contaminant concentrations*. The use of data collected in 2009 and 2010 ensures that the exposure model results reported for the LTR cooling ponds are indicative of recent conditions.
2) *Uncertainty in contaminant ingestion rates*. Data are unavailable to fully evaluate the uncertainty associated with this factor. However, information regarding diet, physiology, and behavior used to compute ingestion rates for the otter and belted kingfisher were taken from the Wildlife Exposure Factors Handbook (EPA 1993), which provided specific dietary and physiological information for each species. Where possible, data for southeastern populations were used to estimate dietary intake to produce more realistic estimates of exposure. The greatest source of this type of uncertainty likely stemmed from the estimates of soil ingestion, which were unavailable for all receptors except the raccoon and, therefore, extrapolated from

other species. Uncertainty in this parameter could have affected the aluminum exposure estimates which were strongly influenced by the soil ingestion pathway, but it is unlikely that uncertainty in contaminant ingestion rates strongly affected the results for the other contaminants.

3) *Individual variation in receptor exposure*. Contaminant ingestion rates were computed for average individuals and may not encompass the full range of variation in the diets of the receptors under study. However, the primary goal of ecological risk assessments is protection of the population rather than the individual, which is also the goal of this exposure dose assessment for the SRS IOUs. In addition, the conservatism associated with the use of upper 95% confidence limits (or maxima where data were too sparse to compute upper 95% confidence limits) serves to protect individuals that experience atypical exposure situations or are unusually susceptible.
4) *TRV accuracy*. The accuracy of TRVs can be affected by a number of factors including extrapolating between species and exposure scenarios (acute to chronic) and from laboratory to field conditions. In general, TRVs are computed using appropriate safety factors with the objective of evaluating the potential for long-term effects including impaired reproduction. However, the TRVs were extrapolated from related species because toxicity data were unavailable for the receptors, thus introducing an element of uncertainty that cannot be directly evaluated.
5) *No investigation of dermal and inhalation pathways*. Dermal contact and inhalation pathways were not investigated because the information needed to evaluate these pathways are largely unavailable for wildlife. However, it is unlikely that dermal contact and inhalation contributed substantially to total exposure in most IOU subunits when compared with ingestion.
6) *Assumptions concerning contaminant concentrations in herptiles*. Lack of data describing contaminant concentrations in herptile tissues necessitated the assumption that herptile contaminant burdens were the same as fish contaminant burdens. This assumption is reasonable because fish and herptiles are both ectothermic and because the amphibious herptiles of interest probably consume much the same food as fish.
7) *Use of concentration ratios to estimate contaminant concentrations*. Lack of data describing contaminant levels in some media necessitated the use of CRs to estimate these values. There were three types of uncertainty affecting the CRs: 1) uncertainty associated with the SRS-specific CRs, 2) uncertainty associated with the extrapolation of the mammalian CRs to metal uptake by birds, and 3) uncertainty resulting from the use of default CRs (equaling one) when SRS-specific CRs were unavailable. The uncertainty associated with the extrapolation of mammalian CRs to birds cannot be evaluated with existing data. Uncertainty associated with the other two factors was addressed by the previously described sensitivity analysis.
8) *Interactions among contaminants*. Exposure to multiple contaminants involves the risk of synergistic or antagonistic interactions that can change toxicity. Generally, data are unavailable to permit quantitative adjustments for interactions among chemicals, and the addition of effects is not advocated unless there is strong evidence for a similar mode of action on the same tissues.
9) *Environmental factors affecting toxicity*. Environmental conditions can alter the physical and chemical state of a contaminant with resulting effects on toxicity and bioavailability. For example, mercury occurs in several forms in the environment

including methylmercury, which is significantly more toxic than inorganic mercury. All mercury consumed by the receptors was assumed to be methylmercury, and the mercury total exposure dose was compared with a TRV for methylmercury rather than a higher TRV for inorganic mercury. Total exposure doses for other contaminants were likewise compared with TRVs for the more toxic chemical species when appropriate.

10) *Failure to include all relevant contaminants in the assessment*. Most contaminants likely to be associated with SRS operations or other anthropogenic sources were measured in the SRS IOUs, although TRVs were unavailable for some (e.g., iron) resulting in an inability to evaluate their toxicological significance. An absence of toxicity data for a particular metal generally implies that this metal is not particularly hazardous. Therefore, omitting it is unlikely to constitute a major source of error when calculating ecological risk.
11) *Representativeness of the receptors*. The otter, belted kingfisher, and blue heron are relatively common on the SRS and are integral components of aquatic habitats and food chains in the IOU subunits. They rely heavily on aquatic food sources throughout their lives and feed near the top of the food chain making them vulnerable to contaminants that may collect and bioaccumulate within aquatic habitats. The raccoon is a more versatile feeder that often makes use of food sources derived from wetland and stream environments. These species are appropriate ecological receptors for the purposes of this study although it is possible that other ecological receptors on the SRS could be more sensitive to some types of contamination. This source of uncertainty is difficult to evaluate but is unlikely to be a major source of error.
12) *Representativeness of contaminant measurements*. A failure to accurately characterize contaminant concentrations as a result of unrepresentative or insufficient sampling can produce inaccurate exposure model results. Such failures will have especially large effects if they involve variables that strongly influence the exposure model results as discussed more fully in the sensitivity analysis.

CONCLUSIONS

1. Preliminary screening identified several contaminants of concern in the LTR cooling ponds and canals including aluminum, cyanide, DDD, DDE, DDT, iron, lead, manganese, and mercury. Further evaluation of these constituents by contaminant exposure models showed that only mercury and aluminum were present at levels sufficient to be potentially harmful to birds and mammals.
2. The primary route of mercury exposure for all receptors was the consumption of contaminated fish, which was at least partly caused by the aerial deposition of mercury from non-SRS sources. A second possible cause was the discharge of mercury contaminated Savannah River used for reactor cooling into the LTR cooling ponds during previous years of reactor operation. This contamination originated from industries located upstream of the SRS.
3. The primary route of aluminum exposure was the incidental consumption of soil, as a likely result of naturally high aluminum levels in soils rather than SRS operations.
4. Sensitivity analysis showed that model uncertainty is reduced by adequate sampling of key variables (e.g., fish and sediments). Sediment sampling was adequate. Fish

samples were not collected from three EAs, and some analytes (pesticides and cyanide) were not measured in fish. Water-to-fish concentration ratios were used to estimate contaminant levels in fish when direct measurements were unavailable; however, such estimates are potentially less accurate than direct measurements.

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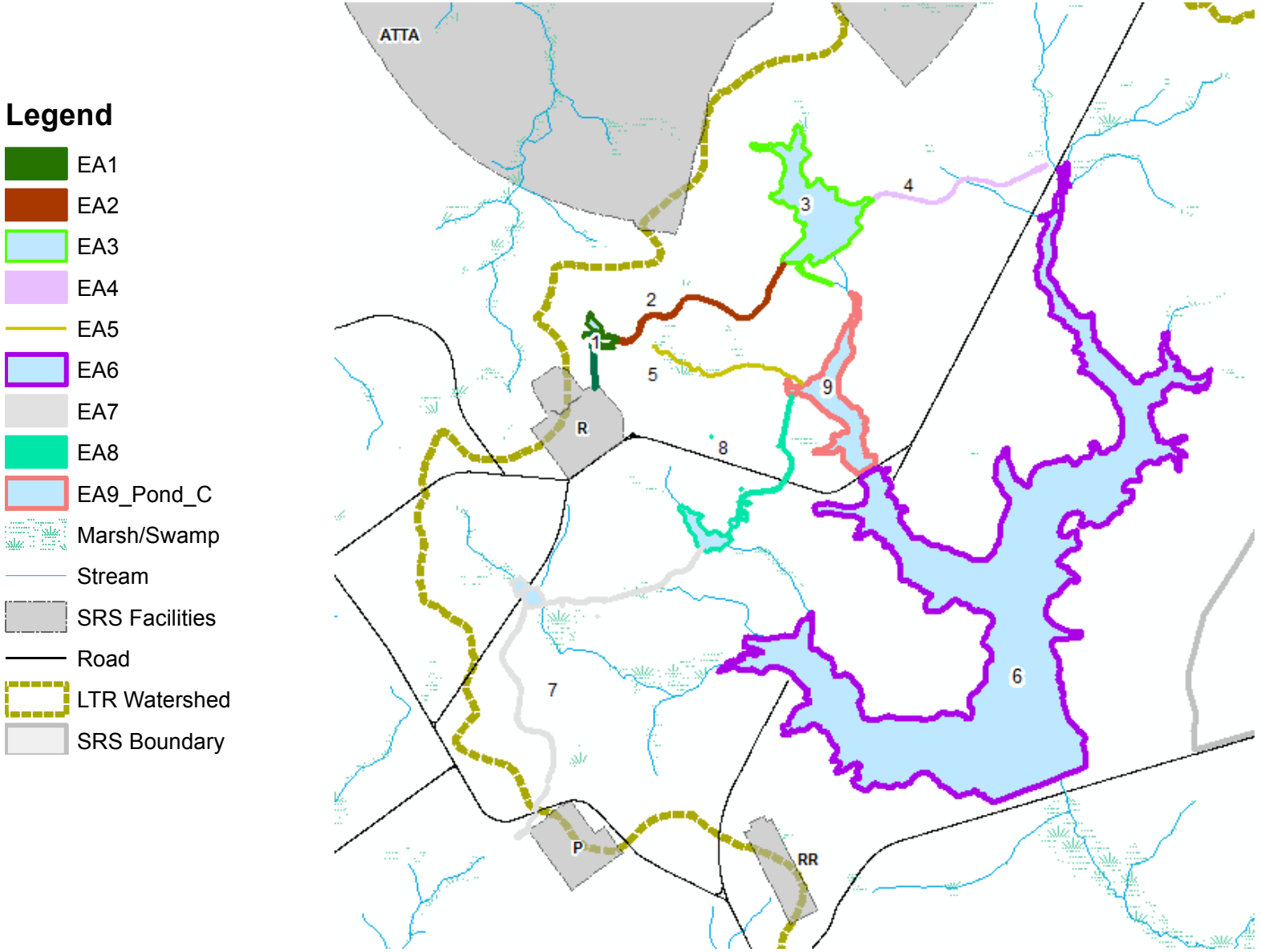

Figure 1. Nine exposure areas in the former reactor cooling ponds and canals in the upper portion of Lower Three Runs.

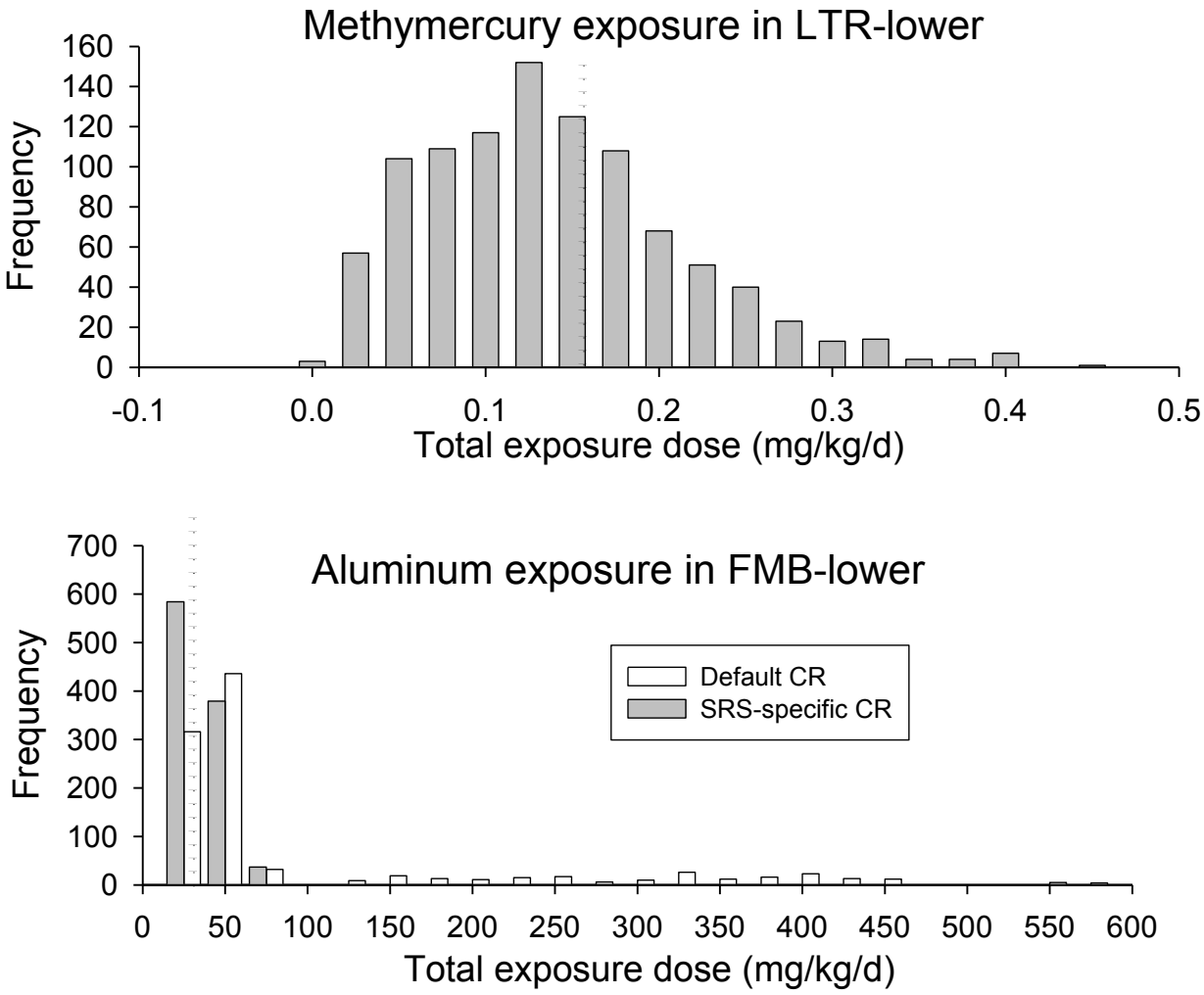

Figure 2. Results of Monte Carlo simulations for the river otter contaminant exposure models. Dotted lines represent TRVs.

Table 1. Contaminants exceeding screening level exposure criteria for each exposure area (EA).

		Contaminants								
EA	Description	Aluminum	Cyanide	DDD	DDE	DDT	Iron	Lead	Manganese	Mercury
1	Pond A and R Discharge Canal					X				
2	Canal between Pond A and Pond B									X
3	Pond B									X
4	Canal between Pond B and PAR Pond									
5	Joyce Branch/Old Discharge Canal					X				
6	PAR Pond									X
7	Pond 2 and associated canals	X				X	X		X	X
8	Ponds 4 and 5 and Canal to Pond C		X	X	X	X		X		X
9	Pond C									

Table 2. Exposure point concentrations (EPCs) for the four types of environmental media that served as inputs to the contaminant exposure models. EPCs for fish and crayfish that are followed by a "CR" were calculated from concentration ratios (see text for explanation).

Location	Analyte	Fish (mg/kg)	n	Crayfish (mg/kg)	n	Sediment (mg/kg)	n	Water (mg/l)	n
EA1	DDT	4.658 CR	0	4.658 CR	0	0.003	12	0.00014	6
EA2	Mercury	0.804 CR	0	0.225 CR	0	0.021	16	0.00045	12
EA3	Mercury	0.734	81	0.206 CR	0	0.061	7	0.00045	3
EA5	DDT	2.244 CR	0	2.244 CR	0	0.011	4	0.00007	7
EA6	Mercury	0.689	227	0.193 CR	0	0.081	371	BDL[b]	0
EA7	Aluminum	23.640 CR	0	168.918 CR	0	8605.000	36	2.36400	24
EA7	DDT	0.707 CR	0	0.707 CR	0	0.001	34	0.00002	24
EA7	Iron	NMC[a]	0	NMC	0	10908.000	36	5.08100	24
EA7	Manganese	156.000	71a	1114.841 CR	0	109.000	36	1.28000	24
EA7	Mercury	0.228	71a	0.064 CR	0	0.297	76	0.00007	24
EA8	Cyanide	0 CR	0	0 CR	0	2.331	4	0.00185	4
EA8	DDD	NMC	0	NMC	0	0.370	18	BDL	0
EA8	DDE	NMC	0	NMC	0	0.262	18	BDL	0
EA8	DDT	0.483 CR	0	0.483 CR	0	0.224	18	0.00001	9
EA8	Lead	0.522 CR	0	1.821 CR	0	10.530	65	0.00174	9
EA8	Mercury	0.854	1b	0.240 CR	0	0.367	65	BDL	0

[a]NMC – not measured or calculated from a concentration ratio.
[b]BDL – below detection limits or not measured.

Table 3. Toxicity reference values (TRVs taken from LANL 2015) used for mammals and birds in the contaminant exposure models.

Analyte	TRV mammals	TRV birds	TRV units
Aluminum	19.3	1100	mg/kg/day
Cyanide	687	0.4	mg/kg/day
DDD	11.7	0.083	mg/kg/day
DDE	22.7	2.4	mg/kg/day
DDT	0.694	5.96	mg/kg/day
Mercury	0.16	0.064	mg/kg/day
Lead	8.9	3.26	mg/kg/day
Manganese	515	1790	mg/kg/day
Nickel	3.4	67.1	mg/kg/day

Table 4. Concentration ratios and source references (Refs) used to estimate exposure point concentrations for receptor foods when direct measurements were unavailable. For example, the concentration of aluminum in crayfish was estimated by multiplying the concentration of aluminum in fish by 7.145.

	Water-to-fish		Soil/sediment-to- mammal		Soil/sediment-to-bird		Fish-to-crayfish		Soil/sediment-to-plant reproductive tissue		Soil/sediment-to-invertebrate	
Constituent	CR	Ref	CR	Ref	CR	Ref	CR	Ref	CR	Ref	CR	Ref
Aluminum	10	a	0.0016	d	0.0016	d	7.145	c	0.0001	a	0.0750	a
Cyanide	0	a	1	a	1	a			1.0000	a	0.0000	a
DDD	0	a	1	a	1	a			0.0013	a	3.3000	a
DDE	161764	b	1	a	1	a			0.0020	a	1.7000	a
DDT	34000	a	1	a	1	a			0.0008	a	0.5700	a
Lead	300	a	0.2535	d	0.2535	d	3.489	c	0.0018	a	0.3270	a
Manganese			0.0144	d	0.0144	d	7.146	c	0.0100	a	0.0200	a
Mercury	1790	c	0.1243	d	0.3630	d	0.281	c	0.0400	a	0.3400	a

a ERD-AG-003
b EPA (2008)
c Computed from unpublished SRS data (IOU data base)
d Paller and Wike (1996)

Table 5. Contaminant exposure model results for the otter. Abbreviations are as follows: TRV= toxicity reference value (mg/kg/day), EXP =total exposure dose (mg/kg/day), HQ= hazard quotient (EXP/TRV), FIR=fish ingestion rate, IIR=crayfish ingestion rate, HIR=herptile ingestion rate, MIR=mammal ingestion rate, BIR=bird ingestion rate, SIR=soil ingestion rate, and WIR=water ingestion rate. Ingestion rates are expressed as a percentage of EXP.

Location	Analyte	TRV	EXP	>TRV	HQ	FIR	IIR	HIR	MIR	BIR	SIR	WIR
EA1	DDT	0.694	0.68	No	0.99	72.2	16.7	11.1	0.0	0.0	0.0	0.0
EA2	Mercury	0.16	0.104	No	0.65	82.0	5.3	12.6	0.0	0.1	0.0	0.0
EA3	Mercury	0.16	0.10	No	0.59	81.9	5.3	12.6	0.1	0.2	0.0	0.0
EA5	DDT	0.694	0.33	No	0.48	72.2	16.7	11.1	0.0	0.0	0.0	0.0
EA6	Mercury	0.16	0.09	No	0.56	81.8	5.3	12.6	0.1	0.3	0.0	0.0
EA7	Aluminum	19.3	17.87	No	0.93	14.0	23.2	2.2	0.6	0.6	58.3	1.1
EA7	DDT	0.694	0.10	No	0.15	72.2	16.7	11.1	0.0	0.0	0.0	0.0
EA7	Manganese	515	46.66	No	0.09	35.5	58.5	5.4	0.0	0.0	0.3	0.2
EA7	Mercury	0.16	0.03	No	0.19	78.9	5.1	12.1	1.0	2.9	0.1	0.0
EA8	Cyanide	687	0.04	No	0.00	0.0	0.0	0.0	46.4	46.4	6.9	0.4
EA8	DDD	11.7	0.001	No	0.00	0.0	0.0	0.0	46.5	46.5	6.9	0.0
EA8	DDE	22.7	0.01	No	0.00	0.0	0.0	0.0	46.5	46.5	6.9	0.0
EA8	DDT	0.694	0.08	No	0.11	68.7	15.9	10.5	2.4	2.4	0.0	0.0
EA8	Lead	8.9	0.17	No	0.02	33.6	27.1	5.2	13.2	13.2	7.7	0.1
EA8	Mercury	0.16	0.11	No	0.70	81.0	5.2	12.4	0.3	1.0	0.0	0.0

Table 6. Contaminant exposure model results for the raccoon. Abbreviations are as follows: TRV=toxicity reference value (mg/kg/day), EXP =total exposure dose (mg/kg/day), HQ= hazard quotient (EXP/TRV), CIR=crayfish ingestion rate, FRIR=fruit ingestion rate, FSIR=fish ingestion rate, GIR=grain ingestion rate, HIR=herptile ingestion rate, NIR=insect ingestion rate, LIR=molluSC ingestion rate, MIR=mammal ingestion rate, BIR=bird ingestion rate, SIR=soil ingestion rate, and WIR=water ingestion rate. Ingestion rates are expressed as a percentage of EXP.

Location	Analyte	TRV	EXP	>TRV	HQ	CIR	FRIR	FSIR	GIR	HIR	NIR	LIR	MIR	BIR	SIR	WIR
EA1	DDT	0.694	0.361	No	0.5	92.6	0.0	1.8	0.0	1.8	0.0	3.8	0.0	0.0	0.0	0.0
EA2	Mercury	0.16	0.005	No	0.0	0.0	0.9	22.1	0.4	22.1	3.0	46.9	1.4	0.7	1.8	0.7
EA3	Mercury	0.16	0.029	No	0.2	81.4	0.4	3.5	0.2	3.5	1.5	7.4	0.7	0.4	0.9	0.1
EA5	DDT	0.694	0.275	No	0.4	95.3	0.0	1.1	0.0	1.1	0.0	2.4	0.0	0.0	0.0	0.0
EA6	Mercury	0.16	0.034	No	0.2	83.6	0.5	2.8	0.2	2.8	1.7	6.0	0.8	0.4	1.1	0.0
EA7	Aluminum	19.3	53.149	Yes	2.8	3.2	0.1	0.1	0.0	0.1	25.6	0.1	0.1	0.0	70.3	0.4
EA7	DDT	0.694	0.010	No	0.0	60.5	0.0	9.5	0.0	9.5	0.1	20.3	0.0	0.0	0.0	0.0
EA7	Manganese	515	12.787	No	0.0	87.5	0.4	1.7	0.2	1.7	0.4	3.6	0.0	0.0	3.7	0.8
EA7	Mercury	0.16	0.008	No	0.0	25.4	7.6	4.0	3.5	4.0	27.1	8.5	2.3	1.2	16.4	0.1
EA8	Cyanide	687	0.209	No	0.0	0.0	56.3	0.0	25.8	0.0	0.0	0.0	8.5	4.5	4.8	0.1
EA8	DDD	11.7	0.028	No	0.0	0.0	0.1	0.0	0.0	0.0	91.7	0.0	1.6	0.8	5.7	0.0
EA8	DDE	22.7	0.011	No	0.0	0.0	0.2	0.0	0.1	0.0	86.1	0.0	2.1	1.1	10.4	0.0
EA8	DDT	0.694	0.008	No	0.0	16.9	0.1	8.3	0.0	8.3	33.5	17.6	2.0	1.1	12.1	0.0
EA8	Lead	8.9	0.151	No	0.0	12.1	0.6	0.5	0.3	0.5	45.7	1.0	5.8	3.1	30.3	0.1
EA8	Mercury	0.16	0.139	No	0.9	91.5	0.5	0.8	0.2	0.8	1.9	1.8	0.8	0.4	1.1	0.0

Table 7. Contaminant exposure model results for the kingfisher. Abbreviations are as follows: TRV= toxicity reference value (mg/kg/day), EXP =total exposure dose (mg/kg/day), HQ= hazard quotient (EXP/TRV), FIR=fish ingestion rate, IIR=crayfish ingestion rate, HIR=herptile ingestion rate, SIR=soil ingestion rate, and WIR=water ingestion rate. Ingestion rates are expressed as a percentage of EXP.

Location	Analyte	TRV	EXP	>TRV	HQ	FIR	IIR	HIR	SIR	WIR
EA1	DDT	5.96	1.439	No	0.2	70.0	15.0	15.0	0.0	0.0
EA2	Mercury	0.064	0.218	Yes	3.4	79.9	3.0	17.1	0.0	0.0
EA3	Mercury	0.064	0.198	Yes	3.1	80.1	2.7	17.1	0.0	0.0
EA5	DDT	5.96	0.693	No	0.1	70.0	15.0	15.0	0.0	0.0
EA6	Mercury	0.064	0.186	Yes	2.9	80.3	2.6	17.2	0.0	0.0
EA7	Aluminum	1100	26.051	No	0.0	19.6	30.0	4.2	44.9	1.2
EA7	DDT	5.96	0.218	No	0.0	70.0	15.0	15.0	0.0	0.0
EA7	Manganese	1790	92.857	No	0.1	36.3	55.5	7.8	0.2	0.2
EA7	Mercury	0.064	0.060	No	0.9	81.6	0.9	17.5	0.0	0.0
EA8	Cyanide	0.4	0.003	No	0.0	0.0	0.0	0.0	92.6	7.4
EA8	DDD	0.083	0.001	No	0.0	0.0	0.0	0.0	100.0	0.0
EA8	DDE	2.4	0.000	No	0.0	0.0	0.0	0.0	100.0	0.0
EA8	DDT	5.96	0.149	No	0.0	70.0	15.0	15.0	0.0	0.0
EA8	Lead	3.26	0.236	No	0.1	47.9	35.7	10.2	6.1	0.1
EA8	Mercury	0.064	0.232	Yes	3.6	79.8	3.2	17.1	0.0	0.0

Table 8. Contaminant exposure model results for the blue heron. Abbreviations are as follows: TRV= toxicity reference value (mg/kg/day), EXP =total exposure dose (mg/kg/day), HQ= hazard quotient (EXP/TRV), FIR=fish ingestion rate, IIR=crayfish ingestion rate, HIR=herptile ingestion rate, MIR=mammal ingestion rate, BIR=bird ingestion rate, SIR=soil ingestion rate, and WIR=water ingestion rate. Ingestion rates are expressed as a percentage of EXP.

Location	Analyte	TRV	EXP	>TRV	HQ	FIR	IIR	HIR	MIR	BIR	SIR	WIR
EA1	DDT	5.96	0.698	No	0.1	96.0	1.0	3.0	0.0	0.0	0.0	0.0
EA2	Mercury	0.064	0.128	Yes	2.0	90.3	6.8	2.9	0.0	0.0	0.0	0.0
EA3	Mercury	0.064	0.117	Yes	1.8	90.3	6.8	2.9	0.0	0.0	0.0	0.0
EA5	DDT	5.96	0.336	No	0.1	96.0	1.0	3.0	0.0	0.0	0.0	0.0
EA6	Mercury	0.064	0.110	Yes	1.7	90.3	6.8	2.9	0.0	0.0	0.1	0.0
EA7	Aluminum	1100	10.414	No	0.0	32.6	2.5	1.0	0.1	0.1	62.7	1.0
EA7	DDT	5.96	0.106	No	0.0	96.0	1.0	3.0	0.0	0.0	0.0	0.0
EA7	Manganese	1790	24.958	No	0.0	89.8	6.8	2.8	0.0	0.0	0.3	0.2
EA7	Mercury	0.064	0.036	No	0.6	89.8	6.8	2.8	0.0	0.0	0.6	0.0
EA8	Cyanide	0.4	0.010	No	0.0	0.0	0.0	0.0	40.8	40.8	17.6	0.8
EA8	DDD	0.083	0.000	No	0.0	0.0	0.0	0.0	21.2	21.2	57.5	0.0
EA8	DDE	2.4	0.000	No	0.0	0.0	0.0	0.0	17.2	17.2	65.7	0.0
EA8	DDT	5.96	0.073	No	0.0	95.6	1.0	3.0	0.1	0.1	0.2	0.0
EA8	Lead	3.26	0.092	No	0.0	81.3	3.0	2.6	2.2	2.2	8.7	0.1
EA8	Mercury	0.064	0.136	Yes	2.1	90.2	6.8	2.8	0.0	0.0	0.2	0.0

Table 9. Sensitivity analysis of selected river otter exposure models for data collected from the lower Lower Three Runs (LTR) and lower Fourmile Branch (FMB) Integrator Operable Unit (IOU) subunits. Values in the table are regression coefficients (R^2s) that represent the strength of the relationship between input distributions representing different sources of contaminant ingestion and model output.

Input distribution	Mercury exposure in LTR Lower IOU subunit (SRS-specific CR)[a]	Aluminum exposure in FMB Lower IOU subunit (SRS-specific CR)[a]	Aluminum exposure in FMB Lower IOU subunit (default CR of 1)
Fish	0.84	0.44	0.05
Crayfish	0.02	0.54	0.06
Sediment	0.18	0.69	>0.99
Water	<0.01	<0.01	<0.01
CR mammal	0.06	<0.01	<0.01
CR bird	0.18	NA[b]	NA[b]

[a] SRS-specific soil to animal concentration ratios (CRs) were calculated from data collected on the SRS (Paller and Wike 1996). A default CR of 1 (ERD 1999) was used for metals that lacked data to calculate SRS-specific CRs.

[b] Data were available to compute a CR for birds only in the case of mercury. Mammal CRs were used to represent birds for Al and other metals (see text for more explanation).

Appendix 1. River otter contaminant exposure model.

A. Computations for food ingestion rates (modified from USEPA 1993)

Food source*	Proportion (P) in diet (wet weight)*	Gross energy content (GE) (kcal/g wet)*	Assimilation efficiency (AE) (unitless)*	Metabolizable energy (ME=GExAE) (kcal/g wet)	ME weighted by P (PxME) kcal/g wet)	Ingestion rate (IR=TIRxP/1000) (kg wet/day)	Ingestion rate symbol
Fish	0.65	1.2	0.91	1.09	0.71	0.788	FIR
Invertebrates	0.15	1.0	0.87	0.87	0.13	0.182	CIR
Herptiles	0.10	1.3	0.84	1.09	0.11	0.121	HIR
Birds	0.05	2.0	0.84	1.68	0.08	0.061	BIR
Mammals	0.05	1.7	0.84	1.43	0.07	0.061	MIR

TIR (g/day) = total ingestion rate = BW** x NFMR*** / WAME) = 1212.11

WAME = weighted average metabolizable energy = (Sum PxME for all prey / Sum P) = 1.105

* Otter food source P, GE, and AE values taken from USEPA 1993

** BW (g) = average adult body (BW) for otters in GA and AL (USEPA 1993) = 7430

***NFMR (kcal/g/day) = normalized field metabolic rate = 0.6167 BW^0.862 (g) / BW (USEPA 1993, equation 3-47) = 0.1803

B. Computations for sediment/soil ingestion rate (estimated as percentage of dry weight food intake following USEPA 1993)

Food source*	Ingestion rate (IR=TIRxP/1000) (kg wet/day)	Percent moisture*	IR kg/d-dry
Fish	0.788	75.0	0.197
Invertebrates	0.182	75.5	0.045
Herptiles	0.121	71.3	0.035
Birds	0.061	68.0	0.019
Mammals	0.061	68.0	0.019

Total dry weight food ingestion rate (kg/d) = 0.315

% soil in diet (dry weight assuming red fox) (USEPA 1993) = 2.8

SIR = soil ingestion rate (kg dry/d) = 0.0088

C. Computations for water ingestion rate

WIR (L/d) = water ingestion rate = 0.099 BW^0.90 (kg) (USEPA 1993, equation 3-17) = 0.60

D. Contaminant exposure computations

Exposure source	Contaminant exposure (mg/kg/d)
Fish (FE)	FE = fish concentration (mg/kg) x FIR (kg/day) / BW (kg)
Invertebrates (IE)	IE = crayfish concentration (mg/kg) x IIR (kg/day) / BW (kg)*
Herptiles (HE)	HE = fish concentration (mg/kg) x HIR (kg/day) / BW (kg)**
Birds (BE)	BE = sediment concentration (mg/kg) x TF (soil to tissue) x BIR (kg/day) / BW (kg)***
Mammals (ME)	ME = sediment concentration (mg/kg) x TF (soil to tissue) x MIR (kg/day) / BW (kg)***
Sediment/soil (SE)	SE = sediment concentration (mg/kg) x SIR (kg/day) / BW (kg)****
Water (WE)	WE = water concentration (mg/kg) x WIR (kg/day) / BW (kg)*****
Total Exposure (TE)	TE = FE + IE + HE + BE + ME + SE + WE

* All invertebrates assumed to be crayfish

** Herptile concentration assumed same as fish concentrations because frogs, snakes, baby alligators, turtles, and salamanders potentially consumed by otters resemble fish in being ectothermic and feeding in the aquatic food chain. Actual herptile concentrations were unavailable.

*** Birds and mammals consumed by otters are assumed to be primarily water fowl and rodents. These organisms may be largely herbivorous, are homeothermic, and may not feed entirely within the aquatic food chain. Therefore, unlike herptiles, their body burdens may differ from fish. Levels in these food sources are computed from sediment levels (either average or maximum) using a tissue to soil concentration ratio computed from the data in Paller and Wike (1996) or from default TF values given in ERD-AG---3

**** For MeHG computations, sediment MeHg concentrations are assumed to be 5% of the total Hg concentration (USEPA 1997, Vol. III, Table 3-10).

***** For MeHg computations, water MeHg concentrations are assumed to be 7.6% of the total Hg concentration (SRS Mercury report: Bowers et al. 2003)

Appendix 2. Belted kingfisher contaminant exposure model.

A. Computations for food ingestion rates (modified from USEPA 1993)

Food source*	Proportion (P) in diet (wet weight)*	Gross energy content (GE) (kcal/g wet)*	Assimilation efficiency (AE) (unitless)*	Metabolizable energy (ME=GExAE) (kcal/g wet)	ME weighted by P (PxME) (kcal/g wet)	Ingestion rate (IR=TIRxP/1000) (kg wet/day)	Ingestion rate symbol
Fish	0.7	1.2	0.91	1.09	0.76	0.0318	FIR
Invertebrates	0.15	1.0	0.87	0.87	0.13	0.0068	CIR
Herptiles	0.15	1.3	0.84	1.09	0.16	0.0068	HIR

WAME = weighted average metabolizable energy = (Sum PxME for all prey / Sum P) = 1.059
TIR (g/day) = total ingestion rate = BW** x NFMR*** / WAME) = 45.47

* Kingfisher food sources, P, GE, and AE values taken from USEPA (1993)
** BW (g) = average adult body (BW) (USEPA 1993) = 147
*** NFMR (kcal/g/day) = kingfisher normalized field metabolic rate 1.146 BW^0.749 (g) / BW (USEPA 1993, equation 3-37) = 0.3275

B. Computations for sediment/soil ingestion rate (estimated as percentage of dry weight food intake following USEPA 1993)

Food source*	Ingestion rate (IR=TIRxP/1000) (kg wet/day)	Percent moisture*	IR kg/d-dry
Fish	0.032	75.0	0.008
Invertebrates	0.007	75.5	0.002
Herptiles	0.007	71.3	0.002

Total dry weight food ingestion rate (kg/d) = 0.012
Estimated % soil in diet = 2% (no similar spp in USEPA 1993) = 2
SIR = soil ingestion rate (kg dry/d) = 0.0002

C. Computations for water ingestion rate

WIR (L/d) = water ingestion rate = 0.059 BW^0.67 (kg) (USEPA 1993, equation 3-15) = 0.02

D. Contaminant exposure computations

Exposure source	Contaminant exposure (mg/kg/d)
Fish (FE)	FE = fish concentration (mg/kg) x FIR (kg/day) / BW (kg)
Invertebrates (IE)	IE = crayfish concentration (mg/kg) x IIR (kg/day) / BW (kg)*
Herptiles (HE)	HE = fish concentration (mg/kg) x HIR (kg/day) / BW (kg)**
Sediment/soil (SE)	SE = sediment concentration (mg/kg) x SIR (kg/day) / BW (kg)***
Water (WE)	WE = water concentration (mg/kg) x WIR (kg/day) / BW (kg)****
Total Exposure (TE)	TE = FE + IE + HE + SE + WE

* All invertebrates assumed to be crayfish
** Herptile concentration assumed same as fish concentrations because frogs, snakes, baby alligators, and salamanders potentially consumed by kingfishers resemble fish in being ectothermic and feeding mainly on aquatic animals. Actual herptile concentrations were unavailable.
*** For MeHG computations, sediment MeHg concentrations are assumed to be 5% of the total Hg concentration (USEPA 1997, Vol. III, Table 3-10).
**** For MeHg computations, water MeHg concentrations are assumed to be 7.6% of the total Hg concentration (SRS Mercury report: Bowers et al. 2003)

Appendix 3. Raccoon contaminant exposure model.

A. Computations for food ingestion rates (modified from USEPA 1993)

Food source*	Proportion (P) in diet (wet weight)*	Gross energy content (GE) (kcal/g wet)*	Assimilation efficiency (AE) (unitless)*	Metabolizable energy (ME=GExAE) (kcal/g wet)	ME weighted by P (PxME) kcal/g wet)	Ingestion rate (IR=TIRxP/1000) (kg wet/day)	Ingestion rate symbol
Crayfish	0.08	1.0	0.87	0.87	0.07	0.058	CIR
Fruit	0.43	1.1	0.76	0.84	0.36	0.292	RIR
Fish	0.01	1.2	0.91	1.09	0.01	0.008	FIR
Grain/nut	0.20	4.6	0.85	3.91	0.76	0.134	GIR
Herptile	0.04	1.3	0.84	1.09	0.04	0.027	HIR
Insects	0.18	1.6	0.87	1.39	0.25	0.122	NIR
Molluscs	0.02	0.8	0.87	0.70	0.02	0.017	LIR
Mammals	0.03	1.7	0.84	1.43	0.04	0.019	MIR
Birds	0.01	1.9	0.84	1.60	0.02	0.010	BIR

TIR (g/day) = total ingestion rate = BW** x NFMR*** / WAME) = 685.37

WAME = weighted average metabolizable energy = (Sum PxME for all prey / Sum P) = 1.574

* Raccon food source P, GE, and AE values taken from USEPA 1993

** BW (g) = average adult body (BW) for raccoons in IL, MS, and AL (USEPA 1993) = 5782

***NFMR (kcal/g/day) = normalized field metabolic rate = 0.6167 BW^0.862 (g) / BW (USEPA 1993, equation 3-47) = 0.1866

B. Computations for sediment/soil ingestion rate (estimated as percentage of dry weight food intake following USEPA 1993)

Food source*	Ingestion rate (IR=TIRxP/1000) (kg wet/day)	Percent moisture*	IR kg/d-dry
Crayfish	0.058	74.0	0.015
Fruit	0.292	77.0	0.067
Fish	0.008	75.0	0.002
Grain/nut	0.134	9.3	0.121
Herptile	0.027	75.5	0.007
Insects	0.122	65.0	0.043
Molluscs	0.017	82.0	0.003
Mammals	0.019	68.0	0.006
Birds	0.010	68.0	0.003

Total dry weight food ingestion rate (kg/d) =	0.267
% soil in diet (USEPA 1993) =	9.4
SIR = soil ingestion rate (kg dry/d) =	0.0251

C. Computations for water ingestion rate

WIR (L/d) = water ingestion rate = 0.099 BW^0.90 (kg) (USEPA 1993, equation 3-17) = 0.48

D. Contaminant exposure computations

Exposure source	Contaminant exposure (mg/kg/d)	
Crayfish (CE)	CE = crayfish concentration (mg/kg) x CIR (kg/day) / BW (kg)	CIR
Fruit (RE)	RE = sediment concentration (mg/kg) x TF (soil to tissue) x RIR (kg/day) / BW (kg)[1]	FRIR
Fish (FE)	FE = fish concentration (mg/kg) x FIR (kg/day) / BW (kg)	FSIR
Grain/nut (GE)	GE = sediment concentration (mg/kg) x TF (soil to tissue) x GIR (kg/day) / BW (kg)[1]	GIR
Herptiles (HE)	HE = fish concentration (mg/kg) x HIR (kg/day) / BW (kg)[2]	HIR
Insects (NE)	NE = sediment concentration (mg/kg) x TF (soil to tissue) x NIR (kg/day) / BW (kg)[3]	NIR
Molluscs (LE)	LE = fish concentration (mg/kg) x LIR (kg/day) x BW (kg)[4]	LIR
Mammals (ME)	ME = sediment concentration (mg/kg) x TF (soil to tissue) x MIR (kg/day) / BW (kg)[5]	MIR
Birds (BE)	BE = sediment concentration (mg/kg) x TF (soil to tissue) x BIR (kg/day) / BW (kg)[5]	BIR
Sediment/soil (SE)	SE = sediment concentration (mg/kg) x SIR (kg/day) / BW (kg)[6]	SIR
Water (WE)	WE = water concentration (mg/kg) x WIR (kg/day) / BW (kg)[7]	WIR
Total Exposure (TE)	TE = CE + RE + FE + GE + HE + NE + LE + ME + BE + SE + WE	EXP

1 Soil to plant reproductive tissue transfer factors (TF) from ERD (1999).

2 Herptile concentration assumed same as fish concentrations because frogs, snakes, salamanders, etc. potentially consumed by raccoons resemble fish in being ectothermic and feeding (usually) in the aquatic food chain. Actual herptile concentrations were unavailable.

3 Soil to invertebrate transfer factors (TF) from ERD (1999).

4 Concentrations in molluscs assumed to be the same as in fish

5 Concentrations in birds and mammals (primarily rodents) were computed from sediment levels (either average or maximum) using a tissue to soil concentration ratio computed from the data in Paller and Wike (1996) or from default TF values given in ERD-AG---3

6 For MeHG computations, sediment MeHg concentrations are assumed to be 5% of the total Hg concentration (USEPA 1997, Vol. III, Table 3-10).

7 For MeHg computations, water MeHg concentrations are assumed to be 7.6% of the total Hg concentration (SRS Mercury report: Bowers et al. 2003)

Appendix 4. Blue heron contaminant exposure model.

A. Computations for food ingestion rates (modified from USEPA 1993)

Food source*	Proportion (P) in diet (wet weight)*	Gross energy content (GE) (kcal/g wet)*	Assimilation efficiency (AE) (unitless)*	Metabolizable energy (ME=GExAE) (kcal/g wet)	ME weighted by P (PxME) (kcal/g wet)	Ingestion rate (IR=TIRxP/1000) (kg wet/day)	Ingestion rate symbol
Fish	0.950	1.2	0.91	1.09	1.04	0.320	FIR
Crayfish	0.010	1	0.87	0.87	0.01	0.003	CIR
Herptiles	0.030	1.3	0.84	1.09	0.03	0.010	HIR
Birds	0.005	1.7	0.84	1.43	0.01	0.002	BIR
Mammals	0.005	1.9	0.84	1.60	0.01	0.002	MIR

WAME = weighted average metabolizable energy = (Sum PxME for all prey / Sum P) = 1.094
TIR (g/day) = total ingestion rate = BW** x NFMR*** / WAME) = 337.22

* Blue heron food sources, P, GE, and AE values taken from USEPA (1993)
** BW (g) = average adult body (BW) (USEPA 1993) = 2229
*** NFMR (kcal/g/day) = Blue heron normalized field metabolic rate = 1.146 BW^0.749 (g) / BW (USEPA 1993, equation 3-37) = 0.1655

B. Computations for sediment/soil ingestion rate (estimated as percentage of dry weight food intake following USEPA 1993)

Food source*	Ingestion rate (IR=TIRxP/1000) (kg wet/day)	Percent moisture*	IR kg/d-dry
Fish	0.3204	75.0	0.080
Crayfish	0.0034	74.0	0.001
Herptile	0.0101	75.5	0.002
Mammals	0.0017	68.0	0.001
Birds	0.0017	68.0	0.001

Total dry weight food ingestion rate (kg/d) = 0.085
Estimated % soil in diet = 2% (no similar spp in USEPA 1993) = 2
SIR = soil ingestion rate (kg dry/d) = 0.0017

C. Computations for water ingestion rate

WIR (L/d) = water ingestion rate = 0.059 BW^0.67 (kg) (USEPA 1993, equation 3-15) = 0.10

D. Contaminant exposure computations

Exposure source	Contaminant exposure (mg/kg/d)
Fish (FE)	FE = fish concentration (mg/kg) x FIR (kg/day) / BW (kg)
Crayfish (CE)	CE = crayfish concentration (mg/kg) x IIR (kg/day) / BW (kg)
Herptiles (HE)	HE = fish concentration (mg/kg) x HIR (kg/day) / BW (kg)[1]
Birds (BE)	BE = sediment concentration (mg/kg) x TF (soil to tissue) x BIR (kg/day) / BW (kg)[2]
Mammals (ME)	ME = sediment concentration (mg/kg) x TF (soil to tissue) x MIR (kg/day) / BW (kg)[2]
Sediment/soil (SE)	SE = sediment concentration (mg/kg) x SIR (kg/day) / BW (kg)[3]
Water (WE)	WE = water concentration (mg/kg) x WIR (kg/day) / BW (kg)[4]
Total Exposure (TE)	TE = FE + IE + HE + SE + WE

1 Herptile concentration assumed same as fish concentrations because frogs, snakes, salamander, etc potentially consumed by kingfishers resemble fish in being ectothermic and feeding mainly on aquatic animals. Actual herptile concentrations were unavailable.
2 Concentrations in birds and mammals (primarily rodents) were computed from sediment levels (either average or maximum) using a tissue to soil concentration ratio computed from the data in Paller and Wike (1996) or from default TF values given in ERD-AG---3
3 For MeHG computations, sediment MeHg concentrations are assumed to be 5% of the total Hg concentration (USEPA 1997, Vol. III, Table 3-10).
4 For MeHg computations, water MeHg concentrations are assumed to be 7.6% of the total Hg concentration (SRS Mercury report: Bowers et al. 2003)