TECHNICAL SUPPORT FOR
ROCKY MOUNTAIN ARSENAL

FINAL
INTEGRATED ENDANGERMENT ASSESSMENT/ RISK CHARACTERIZATION
VERSION 4.2

VOLUME III of IV

Appendix B, Sections B.5-B.8
and Appendix C

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APPENDIX B
(SECTION B.5)

HHRC SENSITIVITY ANALYSIS

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LIST OF ACRONYMS AND ABBREVIATIONS

CSS	Dust loading factor
DW	Annual frequency of exposure
HHRC	Human Health Risk Characterization
IEA/RC	Integrated Endangerment Assessment/Risk Characterization
PCC	Partial correlation coefficient
PPLV	Preliminary pollutant limit value
RAF	Relative absorption factor
RMA	Rocky Mountain Arsenal
SC	Skin soil covering
SI	Soil ingestion
SRC	Standardized regression coefficient
TE	Exposure duration

B.5 HHRC SENSITIVITY ANALYSIS

This section presents the results of a sensitivity study based on correlation analysis, which ranks the influence of several input parameters on the variability of the cumulative direct preliminary pollutant limit values (PPLVs) for aldrin, dieldrin, chlordane, dibromochloropropane (DBCP), and arsenic.

B.5.1 BACKGROUND

In a Monte Carlo simulation such as that used in the Human Health Risk Characterization (HHRC) program to compute PPLVs, parameters are represented by uncertainty distributions and sampled repeatedly to obtain the distribution for the model output. Once the model has been implemented, it is helpful to conduct a sensitivity analysis to determine which input parameters are most important in affecting model output. Such a sensitivity analysis is an essential step in the identification of those input parameters whose uncertainty drives the level of uncertainty in the model results. Identification of these drivers shows the paths for field and laboratory investigations that would be most productive in narrowing the uncertainty reflected in the output distribution for risk. This enhances the value of the risk assessment as a tool for selecting appropriate remedial action.

The question of uncertainty reduction often arises once a Monte Carlo risk model has been built and quantified using the best available information. If, for example, an investment in additional field or laboratory studies were to be made, which of the model's distributed input parameters should receive the highest priority? Which parameter would yield the greatest increase in confidence in the model's results, if it could be characterized more precisely? Sensitivity studies provide a means of answering these questions. Use of multiple regression in place of simple parametric studies provides a firmer technical basis for such recommendations. This is because the parametric methods often used for sensitivity studies are vulnerable to masking and colinearity effects that can lead to mistakes when ranking input parameters by their importance.

B.5.2 APPROACH

Iman et al. (1985) suggests that multiple regression be used to quantify the sensitivity of the model output to each of the input parameters. The regression model is defined as follows:

$$\hat{y} = b_0 + b_1x_1 + b_2x_2 + ..., \quad (1)$$

where $\hat{y}$ is the value of the output parameter that is predicted by the linear combination of the input parameters x_1, x_2,... .

B.5.2.1 Standardized Regression Coefficients

In the regression model

$$\hat{y} = b_0 + b_1x_1 + b_2x_2 + ..., \quad (2)$$

the regression coefficients b_i measure the unit change in y per unit change in x_i. The slopes b_i are therefore dependent on the units of x_i. To compare the slopes for parameters of different units, the regression coefficients must be standardized as follows:

$$B_i = b_i * \frac{SD_i}{SD_y} \quad (3)$$

where SD_i and SD_y are the standard deviations of the x_i and y samples.

The standardized regression coefficient (SRC) for each input parameter measures the importance of that parameter.

B.5.2.2 Partial Correlation Coefficients

The partial correlation coefficient (PCC) for each input parameter, which is a generalization of the coefficient of correlation (r) in ordinary linear regression, provides a second measure of that parameter's importance in the model. As noted in the standard text by Hamburg (1991), PCCs indicate the separate effect of each of the input parameters on the output parameter, after the influence of all the other input parameters has been taken into account. For example, the PCC designated $r_{y1\cdot2}$ in a model with two input parameters would show the partial correlation between y and x_1 after the effect of x_2 on y had been removed.

A variety of PCCs, each representing a different subset of parameters, can be constructed for a model with multiple input parameters. However, Iman et al. (1985) recommends the use of the full-model PCCs, which measure the contribution of each input parameter that is unique, i.e., the contribution not provided by any other parameter. For instance, the distributed input parameters in a seven-parameter model have the following full-model PCCs:

$$r_{y1\cdot234567},\ r_{y2\cdot134567},\ r_{y3\cdot124567},\ \cdots,$$

where: $r_{y1\cdot234567}$ = correlation between the output ("y") and the first input parameter, given that the remaining input parameters are in the regression model

$r_{y2\cdot134567}$ = correlation between the output ("y") and the second input parameter, given that the remaining input parameters are in the regression model

and so on.

B.5.3 CONDUCT OF THE STUDY

The biological worker's cumulative direct-pathway PPLV for aldrin and its seven distributed input parameters were the focus of this sensitivity study. Aldrin was chosen because of its strong contribution to overall risks at Rocky Mountain Arsenal (RMA), as discussed in the main body

of the Integrated Endangerment Assessment/Risk Characterization (IEA/RC) report. The distributed input parameters for the biological worker direct PPLV calculation are as follows:

DW	Annual frequency of exposure (d/y)
TE	Exposure duration (y) (carcinogens only)
RAF_{dermal}	Relative Absorption Factor for dermal absorption (unitless)
$RAF_{ingestion}$	Relative Absorption Factor for ingestion (unitless)
CSS	Dust loading factor ($\mu g/m^3$)
SC	Skin soil covering (mg/cm^3)
SI	Soil ingestion (mg/d)

For the biological worker scenario, TM is a fixed value (8 hours/day), and therefore, it is not carried through the sensitivity analysis.

The sensitivity analysis was performed for both types of health threat (carcinogen and noncarcinogen). The correlation was accomplished using analytical tools available in the S-PLUS statistical programming language. Results from a 100-sample run were used as input for the multivariate correlation.

The influence of each input parameter was quantified using both PCCs and SRCs. The PCCs and SRCs usually indicate the same order of importance among parameters; however, this is not always the case.

B.5.3.1 Evaluation of Carcinogenic PPLV

The regression analysis performed on the biological worker's direct carcinogenic PPLV for aldrin and its distributed input parameters produced the following set of SRCs and PCCs. These indicate that TE and SI are the most influential parameters, followed by $RAF_{ingestion}$(carc). The remaining variables (RAF_{dermal}[carc], SC, DW, and CSS) each contribute relatively little to variation in the carcinogenic aldrin PPLV for the biological worker.

Carcinogen (Biological Worker, Aldrin)

Parameter	PCC Value	PCC Rank	SRC Value	SRC Rank
TE (y)	+.987	1	-.804	1
SI (mg/d)	+.959	2	-.431	2
$RAF_{ingestion}$	+.740	3	-.152	3
RAF_{dermal}	+.413	4	-.075	4
SC (mg/cm^3)	+.345	5	-.066	5
DW (d/y)	+.076	6	-.026	6
CSS ($\mu g/m^3$)	+.058	7	-.022	7

B.5.3.2 Evaluation of Noncarcinogenic PPLV

The regression analysis performed on the biological worker's direct noncarcinogenic PPLV for aldrin and its distributed input parameters produced the following set of SRCs and PCCs. These indicate that SI is the most influential parameter, followed by $RAF_{ingestion}$[noncarc]. The remaining variables (RAF_{dermal}[noncarc], SC, DW, and CSS) each contribute relatively little to variation in the noncarcinogenic aldrin PPLV for the biological worker.

Noncarcinogen (Biological Worker, Aldrin)

Parameter	PCC Value	PCC Rank	SRC Value	SRC Rank
SI (mg/d)	+.964	1	-.964	1
$RAF_{ingestion}$	+.754	2	-.324	2
RAF_{dermal}	+.497	3	-.182	3
SC (mg/cm^3)	+.395	4	-.147	4
DW (d/y)	+.151	5	-.077	5
CSS ($\mu g/m^3$)	+.067	6	-.049	6
TE (y)	+.0002	7	-.004	7

B.5.4 SENSITIVITY RANKING RESULTS FOR ADDITIONAL CHEMICALS

The procedure described in Section B.5.3 for aldrin was also used to perform a sensitivity study on the cumulative-direct PPLVs for dieldrin, chlordane, DBCP, and arsenic for both the biological worker and industrial worker exposure scenarios. The dermal pathway was not evaluated for arsenic. The results of the sensitivity study are provided in Tables B.5-1 through B.5-3, which show rank, SRC, and PCC, respectively, for each input parameter and chemical/

pathway combination. The ranks in these tables are based on PCC. As shown in these tables, the results for arsenic are based on the soil ingestion and particulate inhalation routes; the dermal route was not evaluated for inorganic chemicals in HHRC.

B.5.4.1 Evaluation of Carcinogenic PPLV

B.5.4.1.1 Biological Worker

The regression analysis performed on the biological worker's direct carcinogenic PPLV for dieldrin, chlordane, DBCP, and arsenic indicates that TE is the most influential parameter. For dieldrin, chlordane, and arsenic, SI is the next most influential parameter. For DBCP, however, SC is the second most influential parameter. The remaining variables contribute relatively little to the variation in the PPLVs, as shown for each chemical in Tables B.5-1 through B.5-3.

B.5.4.1.2 Industrial Worker

The regression analysis performed on the industrial worker's direct carcinogenic PPLV for dieldrin, chlordane, DBCP, and arsenic indicate that TE is the most influential parameter. For dieldrin and arsenic, the second most influential parameter is SI. For chlordane and DBCP, the second most influential parameters are RAF_{dermal} and skin soil covering, respectively. The remaining variables contribute relatively little to the variation in the PPLVs, as shown for each chemical in Tables B.5-1 through B.5-3.

B.5.4.2 Evaluation of Non-Carcinogenic PPLV

B.5.4.2.1 Biological Worker

The regression analysis performed on the Biological Worker's direct noncarcinogenic PPLV for dieldrin, chlordane, and arsenic indicate that soil ingestion (SI) is the most influential parameter contributing to variation in the PPLV. The analysis for DBCP indicates that SC is the most influential parameter for this chemical. For dieldrin, DBCP, and chlordane, RAF_{dermal} is the second most influential parameter, while for arsenic, $RAF_{ingestion}$ is the second most influential parameter. The remaining variables for each chemical contribute relatively little to the variation in the PPLVs, as shown in Tables B.5-1 through B.5-3.

B.5.4.2.2 Industrial Worker

The regression analysis performed on the industrial worker's direct noncarcinogenic PPLV indicates that SI is the most influential parameter for dieldrin and arsenic; RAF_{dermal} is the most influential parameter for chlordane; and SC is the most influential parameter for DBCP. The second most influential parameter for dieldrin and chlordane under this scenario is SC. The second most influential parameters for arsenic and DBCP are $RAF_{ingestion}$ and RAF_{dermal}, respectively.

B.5.5 CONCLUSIONS

The results of this sensitivity study indicate that (1) variability in exposure duration is consistently the most influential contributor to the variability in the direct carcinogenic PPLVs; and (2) variability in soil ingestion, soil covering, $RAF_{ingestion}$, and RAF_{dermal} are influential contributors to variability in direct carcinogenic and noncarcinogenic PPLVs.

B.5.6 REFERENCES

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Table B.5-1 Importance of Individual Input Parameters' Contributions to PPLV Variability[1]

Input Parameter	Biological Aldrin Carc Rank	Industrial Aldrin Carc Rank	Biological Dieldrin Carc Rank	Industrial Dieldrin Carc Rank	Biological Chlordane Carc Rank	Industrial Chlordane Carc Rank	Biological DBCP Carc Rank	Industrial DBCP Carc Rank	Biological Arsenic Carc Rank	Industrial Arsenic Carc Rank
TE (y)	1	1	1	1	1	1	1	1	1	1
SI (mg/d)	2	2	2	2	2	4	5	5	2	2
$RAF_{ingestion}$	3	4	3	5	5	5	6	6	3	3
RAF_{dermal}	4	5	4	3	3	2	3	3	na	na
SC (mg/cm^3)	5	3	5	4	4	3	2	2	na	na
DW (d/y)	6	8	6	7	6	6	4	4	5	6
CSS ($\mu g/m^3$)	7	6	7	6	7	7	7	7	4	4
TM (h/d)	na	7	na	8	na	8	na	8	na	5

Input Parameter	Biological Aldrin Noncarc Rank	Industrial Aldrin Noncarc Rank	Biological Dieldrin Noncarc Rank	Industrial Dieldrin Noncarc Rank	Biological Chlordane Noncarc Rank	Industrial Chlordane Noncarc Rank	Biological DBCP Noncarc Rank	Industrial DBCP Noncarc Rank	Biological Arsenic Noncarc Rank	Industrial Arsenic Noncarc Rank
SI (mg/cm^3)	1	1	1	1	1	3	4	4	1	1
$RAF_{ingestion}$	2	5	3	4	4	4	5	7	2	2
RAF_{dermal}	3	6	2	3	2	1	2	2	na	na
SC (mg/cm^3)	4	2	4	2	3	2	1	1	na	na
DW (d/y)	5	7	5	5	5	6	3	3	3	3
CSS ($\mu g/m^3$)	6	3	6	7	6	5	6	5	4	4
TM (h/d)	na	4	na	6	na	7	na	6	na	5
TE (y)	na	na	na	na	na	na	na	na	na	na

Note:

[1] This table reflects the rankings of partial correlation coefficients (PCCs). In each regression analysis, the sets of PCCs and SRCs result in identical rankings with the exception of the following:

(1) In the SRC ranking of industrial aldrin (carc), $RAF_{ingestion}$ and RAF_{dermal} are ranked 5 and 4, respectively;

(2) In the SRC ranking of industrial dieldrin (noncarc), RAF_{dermal} and SC are ranked 2 and 3, repectively.

Table B.5-2 Standardized Regression Coefficients for Direct PPLV Input Parameters

Input Parameter	Biological Aldrin Carc SRC	Industrial Aldrin Carc SRC	Biological Dieldrin Carc SRC	Industrial Dieldrin Carc SRC	Biological Chlordane Carc SRC	Industrial Chlordane Carc SRC	Biological DBCP Carc SRC	Industrial DBCP Carc SRC	Biological Arsenic Carc SRC	Industrial Arsenic Carc SRC
TE (y)	-0.804	-0.938	-0.831	-0.923	-0.863	-0.885	-0.836	-0.939	-0.809	-0.889
SI (mg/d)	-0.431	-0.372	-0.454	-0.356	-0.330	-0.170	-0.023	-0.008	-0.436	-0.472
$RAF_{ingestion}$	-0.152	-0.129	-0.099	-0.075	-0.060	-0.036	-0.004	-0.0011	-0.141	na
RAF_{dermal}	-0.075	-0.130	-0.086	-0.187	-0.229	-0.344	-0.085	-0.089	na	na
SC (mg/cm^3)	-0.066	-0.154	-0.059	-0.172	-0.156	-0.258	-0.370	-0.387	na	-0.012
DW (d/y)	-0.026	-0.006	-0.029	-0.030	-0.048	-0.024	-0.037	-0.010	-0.040	-0.113
CSS ($\mu g/m^3$)	-0.022	-0.037	-0.0070	-0.032	-0.015	-0.016	+0.001	-0.0008	-0.114	-0.086
TM (h/d)	na	-0.026	na	-0.013	na	+0.001	na	-0.0002	na	

Input Parameter	Biological Aldrin Noncarc SRC	Industrial Aldrin Noncarc SRC	Biological Dieldrin Noncarc SRC	Industrial Dieldrin Noncarc SRC	Biological Chlordane Noncarc SRC	Industrial Chlordane Noncarc SRC	Biological DBCP Noncarc SRC	Industrial DBCP Noncarc SRC	Biological Arsenic Noncarc SRC	Industrial Arsenic Noncarc SRC
SI (mg/d)	-0.964	-0.457	-0.918	-0.781	-0.720	-0.415	-0.060	-0.021	-0.961	-0.935
$RAF_{ingestion}$	-0.324	-0.195	-0.172	-0.129	-0.150	-0.070	-0.018	-0.00192	-0.314	-0.280
RAF_{dermal}	-0.182	-0.109	-0.178	-0.377	-0.544	-0.718	-0.219	-0.215	na	na
SC (mg/cm^3)	-0.147	-0.265	-0.155	-0.376	-0.374	-0.501	-0.970	-0.952	na	na
DW (d/y)	-0.077	-0.058	-0.071	-0.054	-0.082	-0.039	-0.099	-0.024	-0.060	-0.015
CSS ($\mu g/m^3$)	-0.049	-0.232	-0.018	-0.009	-0.048	-0.065	+0.002	-0.003	-0.010	-0.011
TE (y)	na	na	na	na	na	na	na	na	na	na
TM (h/d)	na	-0.227	na	-0.045	na	-0.012	na	-0.00197	na	-0.010

Table B.5-3 Full Model Partial Correlation Coefficients for Direct PPLV Input Parameters

Input Parameter	Biological Aldrin Carc PCC	Industrial Aldrin Carc PCC	Biological Dieldrin Carc PCC	Industrial Dieldrin Carc PCC	Biological Chlordane Carc PCC	Industrial Chlordane Carc PCC	Biological DBCP Carc PCC	Industrial DBCP Carc PCC	Biological Arsenic Carc PCC	Industrial Arsenic Carc PCC
TE (y)	0.987	0.972	0.989	0.973	0.969	0.983	0.9996	0.9999	0.991	0.977
SI (mg/d)	0.959	0.837	0.967	0.838	0.831	0.682	0.695	0.544	0.969	0.924
$RAF_{ingestion}$	0.740	0.394	0.576	0.177	0.136	0.090	0.065	0.025	0.765	0.557
RAF_{dermal}	0.413	0.383	0.504	0.571	0.704	0.899	0.968	0.994	na	na
SC (mg/cm^3)	0.345	0.478	0.325	0.550	0.520	0.834	0.998	0.9996	na	na
DW (d/y)	0.076	0.001	0.108	0.033	0.093	0.042	0.852	0.652	0.193	0.008
CSS ($\mu g/m^3$)	0.058	0.049	0.006	0.039	0.010	0.020	0.004	0.012	0.683	0.405
TM (h/d)	na	0.025	na	0.007	na	0.0001	na	0.001	na	0.287

Input Parameter	Biological Aldrin Noncarc PCC	Industrial Aldrin Noncarc PCC	Biological Dieldrin Noncarc PCC	Industrial Dieldrin Noncarc PCC	Biological Chlordane Noncarc PCC	Industrial Chlordane Noncarc PCC	Biological DBCP Noncarc PCC	Industrial DBCP Noncarc PCC	Biological Arsenic Noncarc PCC	Industrial Arsenic Noncarc PCC
SI (mg/d)	0.964	0.228	0.946	0.856	0.849	0.701	0.706	0.616	0.9999	0.999
$RAF_{ingestion}$	0.754	0.051	0.388	0.141	0.199	0.064	0.177	0.013	0.9993	0.990
RAF_{dermal}	0.497	0.017	0.413	0.578	0.772	0.879	0.970	0.994	na	na
SC (mg/cm^3)	0.395	0.093	0.363	0.585	0.614	0.773	0.998	0.9997	na	na
DW (d/y)	0.151	0.005	0.107	0.029	0.072	0.021	0.870	0.683	0.982	0.223
CSS ($\mu g/m^3$)	0.067	0.074	0.007	0.0008	0.025	0.057	0.003	0.034	0.592	0.139
TM (h/d)	na	0.069	na	0.020	na	0.002	na	0.014	na	0.124
TE (y)	na	na	na	na	na	na	na	na	na	na

APPENDIX B
(SECTION B.6)

SUMMARY OF ACUTE AND SUBCHRONIC RESULTS CALCULATED FOR THE HHEA ADDENDUM

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B.6 SUMMARY OF ACUTE AND SUBCHRONIC RESULTS CALCULATED FOR THE HHEA ADDENDUM

B.6.1 INTRODUCTION

This section provides documentation supporting the acute/subchronic risk evaluation presented in Section 3.2.4. The information summarized in the following tables is based on the results and supporting methodologies included in the HHEA Addendum (EBASCO 1992), which evaluated acute and subchronic hazards for two scenarios: a most likely estimate (MLE) scenario and a reasonable maximum exposure (RME) scenario. In accordance with EPA guidance, the RME analysis was developed to represent a reasonable upperbound estimate of hazards and risks, and thus is the focus of the acute/subchronic evaluation presented in the IEA/RC. Results of the MLE evaluation are provided in the HHEA Addendum.

Acute and subchronic (deterministic) RME PPLVs developed for the cumulative direct exposure pathways are summarized in Volume I of the IEA/RC (Tables 3.2-7 and 3.2-8). The acute and subchronic RME parameters used to estimate these PPLVs are listed in Tables B.6-1 and B.6-2, respectively. The toxicity criteria used for this evaluation are listed in Table B.6-3.

As discussed in Section 3.2.4, the acute and subchronic PPLVs presented in Volume I of the IEA/RC are the same as those originally calculated for the HHEA Addendum, with the following exceptions. PPLVs for aldrin and dieldrin were recalculated for the IEA/RC to reflect revisions of the dermal RAF (all receptor scenarios), the soil covering factor (visitor populations only), and the toxicity criteria. The updated acute/subchronic RfD for aldrin and dieldrin is 1.0E-04 mg/kg/day, which was specifically developed by EPA's Office of Research and Development (1993) and supersedes the subchronic RfD used in the HHEA Addendum (5.0E-05 mg/kg/day). Figure B.6-1 presents a map of soil boring-specific hazard quotients (HQs) for aldrin/dieldrin reflecting the revised exposure parameters and toxicity criteria. HQs shown in this map correspond to the driving receptor scenario (i.e., the scenario for which PPLVs were lowest -- recreational visitor, acute exposures).

B.6.2 FACTORS TO CONSIDER WHEN EVALUATING PPLVS DERIVED FOR ACUTE/SUBCHRONIC ENDPOINTS

A comparison of acute (deterministic), subchronic (deterministic), and chronic (probabilistic) noncarcinogenic PPLVs calculated for visitor populations is provided in Table 3.2-9 (Volume I). As shown in this table, of the chemicals for which both acute and subchronic PPLVs were developed, acute PPLVs for eight COCs are lower than corresponding subchronic PPLVs, and subchronic PPLVs for eight COCs are lower than corresponding chronic PPLVs. Acute PPLVs for four chemicals (aldrin, DDT, dieldrin, and endrin) are lower than both subchronic and chronic PPLVs. Differences in exposure assumptions and the applicability of the toxicity criteria should be considered when evaluating the acute/subchronic results, and, in particular, when comparing the acute/subchronic deterministic PPLVs with corresponding chronic probabilistic PPLVs.

B.6.2.1 Differences in Exposure Assumptions

Acute/subchronic and chronic approaches differ in their use of exposure assumptions. The acute/subchronic exposures are, of course, of shorter duration than the chronic exposures. However, several other differences should be considered. For example, for recreational and regulated/casual visitor populations, both acute and subchronic hazards were calculated assuming exposure to a 10 kg child receptor (2.5 years old), whereas the chronic long-term evaluation assumed a distribution of body weights (corresponding to ages ranging from 0 to 75 years). [The assumption of a child receptor is usually implicit to the evaluation of non-worker population acute or subchronic exposures, as this results in a more conservative (health-protective) analysis.] Consequently, the soil intake (dose) to body weight ratio assumed in estimating acute/subchronic exposures is generally higher than that assumed in the chronic risk evaluation. [However, irrespective of the aforementioned factors, single-event exposures could, in some instances, be much greater than those considered to be chronic.] Also, other RME acute and subchronic parameters, such as oral RAF, dermal RAF, and skin surface area, are different from those used in the chronic risk evaluation for some COCs.

In addition to the actual exposure parameter estimates, another difference between the acute/subchronic and chronic analyses is the way in which the exposure parameters are used to

calculate the PPLVs. The acute/subchronic deterministic analysis used single point (fixed) estimates of exposure parameters to derive a single estimate of hazard; these exposure parameters represent RME parameters. The results of this evaluation thus provide a measure of the hazard to an individual exposed under those conditions. PPLVs based on these estimates are assumed to be protective for an individual exposed to this or a lesser combination of exposure factors. The probabilistic analysis, in contrast, used ranges of exposures potentially occurring within the population. It is assumed that some individuals have a high level of exposure and others have a lower level. The results of this evaluation thus provide an estimate of hazard to the population being evaluated. PPLVs based on the 95th percentile of the resulting hazard distribution are designed to protect at least 95 percent of the individuals within the population.

B.6.2.2 Applicability of Toxicity Criteria

When evaluating the acute/subchronic and chronic risk estimates, one must also consider the degree of uncertainty in the estimates contributed by the toxicity factors. In many cases, acute/subchronic toxicity values were not available for the chemicals being evaluated and a chronic toxicity value may have been substituted. If the substituted toxicity criterion was in fact based on an acute effect, then little additional uncertainty would be associated with the acute PPLV. However, if the toxicity criterion was based on a chronic effect, the acute and subchronic PPLVs are likely to contain more uncertainty than the chronic PPLV estimates.

B.6.3 REFERENCES

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EPA 1990b. Health Effects Assessment Summary Tables. (HEAST) Office of Health and Environmental Assessment.

Table B.6-1 Acute Reasonable Maximum Exposure (RME) Parameters for Cumulative Direct Soil Exposure Pathway

Parameter Name	Regulated/Casual Visitors		Recreational Visitors		Commercial Workers	Industrial Workers
Soil Ingestion	2 1/2yr	250 mg/day	2 1/2 yr	250 mg/day	100 mg/day	100 mg/day
Breathing Rate	2 1/2yr	2.016 m^3/day	2 1/2 yr	3.98 m^3/day	4.8 m^3/day	20 m^3/day
Dust Load Factor		0.042 mg/m^3		0.042 mg/m^3	0.021 mg/m^3	0.042 mg/m^3
Pulmonary Retention		0.75		0.75	0.75	0.75
Pulmonary Absorption (All compounds)		1 (100 percent)		1 (100 percent)	1 (100 percent)	1 (100 percent)
Daily Exposure Period		8 hours		8 hours	8 hours	8 hours
Annual Exposure Frequency		NA	NA	NA	NA	NA
Lifetime Exposure Duration		NA	NA	NA	NA	NA
Skin Surface Area	2 1/2yr	2,100 cm^2	2 1/2 yr	2,100 cm^2	1,120 cm^2	3,200 cm^2
Soil Covering/1/		0.51 mg/cm^2		0.51 mg/cm^2	0.11 mg/cm^2	1.5 mg/cm^2
Soil Matrix Factor		1.0		1.0	1.0	1.0
Dermal Absorption/1/		0.01 (metals) 0.10 (organics)		0.01 (metals) 0.10 (organics)	0.01 (metals) 0.10 (organics)	0.01 (metals) 0.10 (organics)
Oral Absorption		1 (100 percent)		1 (100 percent)	1 (100 percent)	1 (100 percent)
Body Weight	Child:	10 kg	Child:	10 kg	Adult: 70kg	Adult: 70kg

NA Not Applicable

/1/ RME PPLVs for aldrin and dieldrin were recalculated using an RfD recently updated by the EPA (1.0E-04 mg/kg/day; see Appendix Table B.6-3); this criterion supersedes the value used in the HHEA Addendum. These recalculated PPLVs also reflect the following: (1) dermal RAFs for aldrin and dieldrin were revised to equal 0.0052 and 0.1, respectively, consistent with the assumptions used in the IEA; and (2) concomitant with this revision of the aldrin/dieldrin dermal RAFs, the soil covering assumed for recreational and regulated/casual visitor populations was revised to equal 1.0 mg/cm^2, consistent with recent EPA dermal exposure assessment guidance.

Table B.6-2 Subchronic Reasonable Maximum Exposure (RME) Parameters for Cumulative Direct Soil Exposure Pathway

Parameter Name	Regulated/Casual Visitors		Recreational Visitors		Commercial Workers	Industrial Workers
Soil Ingestion	2 1/2yr 6 yr	250 mg/day 250 mg/day	2 1/2 yr 6 yr	250 mg/day 250 mg/day	100 mg/day	100 mg/day
Breathing Rate	2 1/2 yr 6 yr	2.016 m^3/day 6.38 m^3/day	2 1/2 yr 6 yr	3.98 m^3/day 9.74 m^3/day	4.8 m^3/day	20 m^3/day
Dust Load Factor		0.042 mg/m^3		0.042 mg/m^3	0.021 mg/m^3	0.042 mg/m^3
Pulmonary Retention		0.75		0.75	0.75	0.75
Pulmonary Absorption		1 (100 percent)		1 (100 percent)	1 (100 percent)	1 (100 percent)
Daily Exposure Period		8 hours		8 hours	8 hours	8 hours
Annual Exposure Frequency		108 days/year		108 days/year	253 days/year	253 days/year
Exposure Duration		7 years		7 years	7 years	7 years
Lifetime Exposure Duration/1/		7 years		7 years	7 years	7 years
Skin Surface Area	2 1/2yr 6 yr	2,100 cm^2 2500 cm^2	2 1/2 yr 6 yr	2,100 cm^2 2500 cm^2	1,120 cm^2	3,200 cm^2
Soil Covering/2/		0.51 mg/cm^2		0.51 mg/cm^2	0.11 mg/cm^2	1.5 mg/cm^2
Soil Matrix Factor		1.0		1.0	1.0	1.0
Dermal Absorption/2/		0.01 (metals) 0.10 (organics)		0.01 (metals) 0.10 (organics)	0.01 (metals) 0.10 (organics)	0.01 (metals) 0.10 (organics)
Oral Absorption (all compounds)		1 (100 percent)		1 (100 percent)	1 (100 percent)	1 (100 percent)
Body Weight	Child:	10th percentile (M&F)	Child:	10th percentile (M&F)	Adult: 70kg	Adult: 70kg

NA Not Applicable

/1/ Lifetime exposure duration= Q-Factor. In the SPPPLV equation, the exposure duration is divided by the lifetime exposure duration. Because exposure duration and lifetime exposure duration are the same number, this term defaults to a value of 1.0.

/2/ RME PPLVs for aldrin and dieldrin were recalculated using an RfD recently updated by the EPA (1.0E-04 mg/kg/day; see Appendix Table B.6-3); this criterion supersedes the value used in the HHEA Addendum. These recalculated PPLVs also reflect the following: (1) dermal RAFs for aldrin and dieldrin were revised to equal 0.0052 and 0.1, respectively, consistent with the assumptions used in the IEA; and (2) concomitant with this revision of the aldrin/dieldrin dermal RAFs, the soil covering assumed for recreational and regulated/casual visitor populations was revised to equal 1.0 mg/cm^2, consistent with recent EPA dermal exposure assessment guidance.

Table B.6.3 D_T(RfD) Values for Acute and Subchronic Exposure[1]

Contaminant	Acute D_TING (mg/kg/day)	D_TINH (mg/kg/day)	Subchronic D_TING (mg/kg/day)	Subchronic D_TINH (mg/kg/day)
Aldrin[2,4]	1.0E-04	1.0E-04	1.0E-04	1.0E-04
Arsenic[2]	8.0E-03	2.9E-04	1.0E-03	2.9E-04
Benzene	NA	NA	NA	NA
Cadmium[3]	4.0E-03	1.4E-01	5.0E-04	5.0E-04
Carbon Tetrachloride	4.0E-01	1.8E-01	7.0E-03	2.7E-02
Chlordane[3]	6.0E-03	6.0E-03	6.0E-05	1.4E-04
Chloroacetic Acid	NA	NA	2.0E-02	2.0E-02
Chlorobenzene[2]	2.0E-01	2.0E-01	2.0E-01	5.0E-02
Chloroform[3]	1.8e-01	4.3E-01	1.0E-02	6.8E-03
Chromium VI	1.0E-01	1.0E-01	2.0E-02	5.7E-06
Dibromochloropropane	5.0E-03	5.0E-03	NA	NA
DDE	NA	NA	NA	NA
DDT[2,3]	5.0E-04	5.0E-04	5.0E-04	5.0E-04
1,2-Dichloroethane	NA	NA	NA	NA
1,1-Dichloroethylene[3]	2.0E+00	1.0E+00	9.0E-03	2.3E-02
Dicyclopentadiene	NA	NA	3.0E-01	6.0E-04
Dieldrin[2,4]	1.0E-04	1.0E-04	1.0E-04	1.0E-04
Endrin	2.0E-03	2.0E-03	5.0E-04	5.0E-04
Hexachlorocyclopentadiene	NA	NA	7.0E-02	2.0E-04
Isodrin	NA	NA	NA	NA
Lead	NA	NA	NA	NA
Mercury(Inorganic)[3]	2.0E-01	2.0E-01	3.0E-04	8.5E-05
Methylene Chloride[3]	1.0E+00	4.9E+00	6.0E-02	8.5E-01
1,1,2,2-Tetrachloroethane	NA	NA	NA	NA
Tetrachloroethylene	2.0E-01	1.9E+00	1.0E-01	1.7E-01
Toluene[2]	2.0E+00	4.3E+00	2.0E+00	5.7E-01
Trichloroethylene	2.4E+00	4.3E-01	2.5E+00	2.5E+00

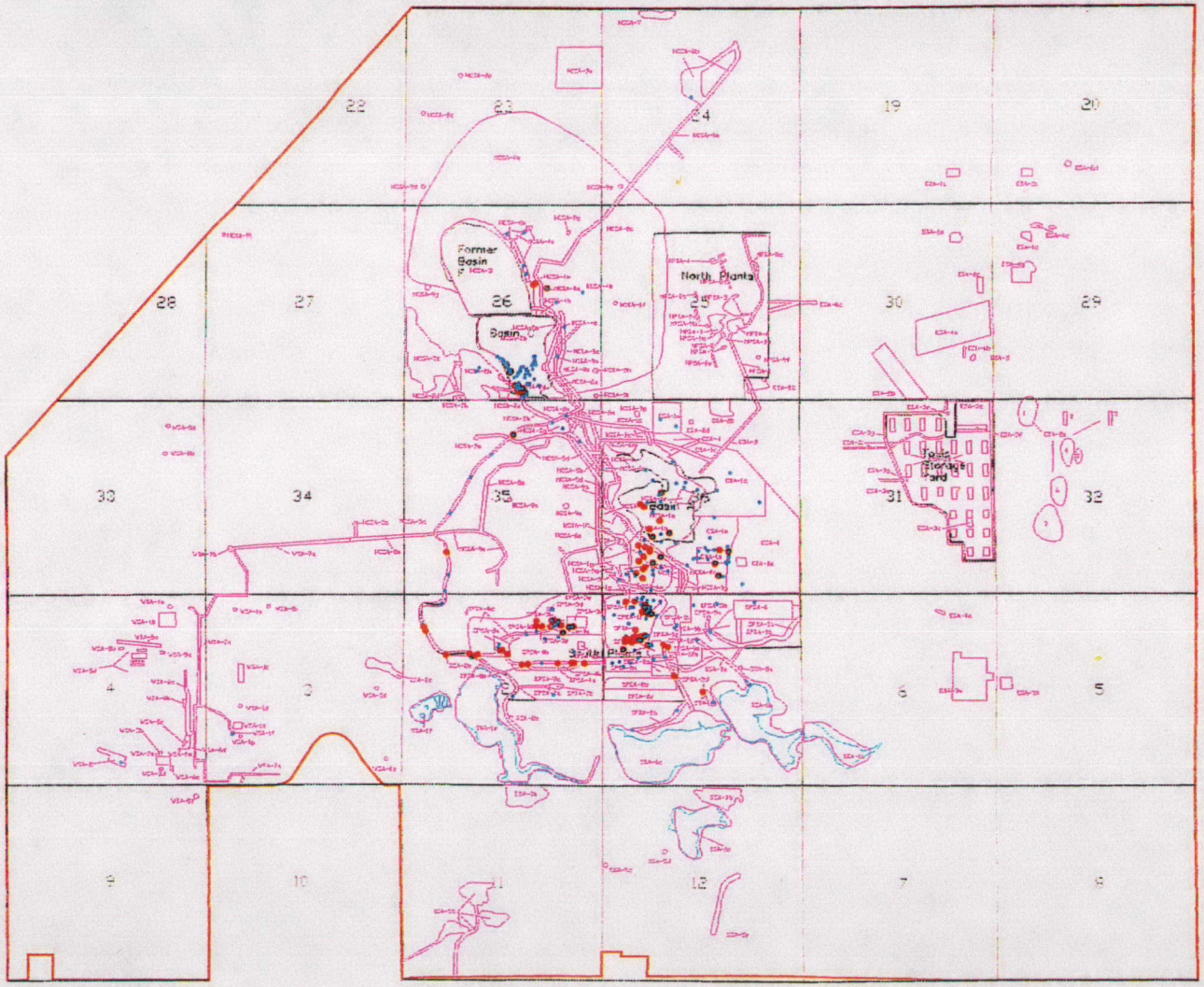
NA Dose-response data not available from EPA.
D_TING Reference dose (RfD) for oral (ingestion) pathway.
D_TINH Reference dose (RfD) for inhalation pathway.

[1] D_Ts were obtained from EPA's Integrated Risk Information System (1990a), the EPA Office of Drinking Water Health Advisories, EPA's Health Effects Assessment Summary Tables (1990b), and Agency for Toxic Substances and Disease Registry toxicology profiles. Oral RfDs and inhalation reference concentrations were given priority over other types of toxicity values.

[2] Acute DT = Subchronic DT for oral and/or inhalation pathway.

[3] Subchronic DT = Chronic DT for oral and/or inhalation pathway (see Appendix Table B.1-10 for chronic DT values).

[4] The acute and subchronic RfD for aldrin and dieldrin, 1.0E-04 mg/kg/day, reflects the December 1992 update by the EPA Office of Research and Development (EPA 1993) and supersedes the criteria used in the HHEA Addendum. This reference dose was based on a neurotoxicity study by Smith et. al. (1976) in which squirrel monkeys were exposed to dieldrin administered by a bolus dose. The no observed adverse effects level (NOAEL) in this study was 0.01 mg/kg/day; the lowest observed adverse effects level (LOAEL) was 0.10 mg/kg/day. The updated RfD was derived by applying an uncertainty factor (UF) of 100 to the NOAEL (0.01 mg/kg/day) value to account for extrapolation from animal data to humans and to protect sensitive individuals.



Legend

- Aldrin and/or Dieldrin HQ > 10
- 1 < Aldrin and/or Dieldrin HQ ≤ 10
- Lake
- 31 Section Number
- Section Line

Note:

The aldrin/dieldrin HQs shown in this map reflect the finalized acute criteria (RfDs) developed for these chemicals and correspond to an acute (deterministic) PPLV of 3.8 mg/kg and 3.7 mg/kg for aldrin and dieldrin, respectively. See Appendix B.6 text for discussion.

N

0 2000 4000

Scale in Feet

Prepared for:
U.S. Army Program Manager
for Rocky Mountain Arsenal

February 1994

Figure B.6-1
Recreational Visitor, Horizon 0 (0-1 ft): Map of Hazard Quotients (HQs) Exceeding 1.0 Reflecting Acute Exposures to Aldrin/Dieldrin
Rocky Mountain Arsenal
Prepared by: ENSERCH Environmental Corp

APPENDIX B
(SECTION B.7)

QUALITATIVE RISK ASSESSMENT DOCUMENTATION

LIST OF TABLES

Table B.7-1 Summary of Qualitative Assessment Results for Feasibility Study Action SAR Sites

Medium Group or Subgroup	SAR Site	CAR Site	Agent			Unexploded Ordnance			Number of Drums	Number Structures	Number of USTs
			Potential	Sampled	Detected	Potential	Sampled	Detected			
Munitions Testing Medium Group											
	CSA - 2c	36-NSA	Yes	Yes	No	Yes	No	No	0	0	0
	CSA - 2d	36-NSA	Yes	Yes	No	Yes	No	No	0	0	0
	ESA - 1a	19-1	Yes	Yes	No	Yes	Yes	Yes	0	0	0
	ESA - 1b	20-1	No	No	No	Yes	Yes	No	0	0	0
	ESA - 1c	29-1	No	Yes	No	Yes	Yes	No	0	0	0
	ESA - 1d	30-2	No	Yes	No	Yes	Yes	No	0	0	0
	ESA - 4a	30-1	No	Yes	Yes	Yes	Yes	Yes	0	0	0
	ESA - 4b	29-4	Yes	Yes	No	Yes	Yes	Yes	2	0	0
North Plants Subgroup											
	NPSA - 3	NP	Yes	No	No	No	No	No	0	8	0
	NPSA - 5	NP	Yes	Yes	Yes	No	Yes	No	unknown	1	0
	NPSA - 6	NP	Yes	Yes	Yes	No	Yes	No	unknown	5	0
Toxic Storage Yards Subgroup											
	ESA - 3a	5-2	Yes	Yes	No	No	No	*No	unknown	0	0
	ESA - 3b	6-6	Yes	Yes	Yes	No	No	No	unknown	6	0
	ESA - 3c	31-4	Yes	Yes	Yes	Yes	No	No	unknown	3	0
	ESA - 3d	31-6, -7	Yes	Yes	Yes	No	No	No	unknown	49	0
	ESA - 3e	31-4	Yes	Yes	Yes	No	No	No	unknown	0	0
	ESA - 3f	31-7	Yes	Yes	Yes	No	No	No	0	0	0
	ESA - 3g	31-7	Yes	Yes	Yes	No	No	No	0	0	0
	ESA - 3h	31-7	Yes	Yes	Yes	No	No	No	0	0	0
	ESA - 3i	31-7	Yes	Yes	Yes	No	No	No	0	0	0

Table B.7-1 Summary of Qualitative Assessment Results for Feasibility Study Action SAR Sites

Medium Group or Subgroup	SAR Site	CAR Site	Agent			Unexploded Ordnance			Number of Drums	Number Structures	Number of USTs
			Potential	Sampled	Detected	Potential	Sampled	Detected			
Lake Sediments Medium Group											
	NCSA - 7	24-NSA	No	Yes	No	No	Yes	No	0	1	0
	SSA - 1b	1-2	No	No	No	No	No	No	0	1	0
	SSA - 1c	1-2	No	No	No	No	No	No	0	0	0
	SSA - 1e	2-17	No	No	No	No	No	No	0	2	0
	SSA - 5b	11-2	No	Yes	No	No	No	No	0	0	0
Surficial Soils Medium Group											
	NCSA - 1g Surficial Soils Survey	36-NSA	Yes	Yes	No	Yes	No	No	0	3	0
Ditches/Drainage Medium Group											
	CSA - 2b	36-NSA	Yes	Yes	No	Yes	No	No	0	0	0
	ESA - 6c	30-1	No	Yes	No	No	No	No	0	0	0
	NCSA - 1c	36-8	Yes	Yes	Yes	No	No	No	0	0	0
	NCSA - 1d	36-11	No	Yes	No	No	No	No	0	0	0
	NCSA - 1f	36-8	Yes	Yes	Yes	No	No	No	0	0	0
	NCSA - 2d	26-7	No	Yes	No	No	Yes	No	0	0	0
	NCSA - 5d	35-NSA	No	No	No	No	No	No	0	0	0
	NCSA - 8b	24-6	No	Yes	No	No	No	No	0	5	0
	NPSA - 8c	25-NSA	No	Yes	Yes	No	Yes	No	0	1	0
	NPSA - 9f	25-NSA	No	Yes	Yes	No	Yes	No	0	0	0
	SSA - 2a	1-1	No	Yes	No	No	No	No	0	0	0
	SSA - 2c	3-2, -3	No	Yes	No	No	No	No	0	1	0
	WSA - 1f	3-NSA	No	Yes	No	No	No	No	0	0	0

Table B.7-1 Summary of Qualitative Assessment Results for Feasibility Study Action SAR Sites

Medium Group or Subgroup	SAR Site	CAR Site	Agent			Unexploded Ordnance			Number of Drums	Number Structures	Number of USTs
			Potential	Sampled	Detected	Potential	Sampled	Detected			
Basin A Medium Group											
	NCSA - 1a	36-1, -14	Yes	Yes	Yes	No	No	No	0	1	0
	NCSA - 1e	36-15	Yes	Yes	No	Yes	No	No	0	0	0
Secondary Basins Subgroup											
	NCSA - 2a	26-3	Yes	Yes	Yes	No	No	No	0	0	0
	NCSA - 2b	26-4	Yes	Yes	Yes	No	No	No	0	0	0
	NCSA - 5a	35-3	Yes	Yes	Yes	No	No	No	0	1	0
Former Basin F Subgroup	NCSA - 3	26-6	Yes	Yes	Yes	No	No	No	0	0	0
Basin F Exterior Subgroup											
	NCSA - 4a	26-1	Yes	Yes	No	No	No	No	0	3	0
	NCSA - 4b	26-NSA	No	Yes	No	No	Yes	No	0	4	0
Sanitary/Process Water Sewers Subgroup											
	NCSA - 8a	24-5, 25-2, 26-8, 34-2, 3	No	Yes	No	No	No	No	0	0	0
	SPSA - 11	1-13, 2-18	No	Yes	No	No	No	No	0	0	0
	SPSA - 12	2-UNC	No	Yes	No	No	No	No	0	0	0
	WSA - 7a	3-1, 4-1, 34-2	No	Yes	Yes	No	No	No	0	0	0
Chemical Sewers Subgroup	CSA - 3	36-20	Yes	Yes	Yes	No	No	No	0	0	0
	NCSA - 6a	25-2, 36-20	Yes	Yes	Yes	No	No	No	0	0	0

Table B.7-1 Summary of Qualitative Assessment Results for Feasibility Study Action SAR Sites

Medium Group or Subgroup	SAR Site	CAR Site	Agent			Unexploded Ordnance			Number of Drums	Number Structures	Number of USTs
			Potential	Sampled	Detected	Potential	Sampled	Detected			
	NCSA - 6b	26-9, 35-2, 36-20	Yes	Yes	Yes	No	No	No	0	0	0
	NPSA - 1	25-3	Yes	No	No	No	No	No	0	0	0
	SPSA - 10	1-13, 2-18	Yes	Yes	Yes	No	No	No	0	0	0
Complex Trenches Subgroup	CSA - 1c	36-9, -16, -17	Yes	Yes	Yes	Yes	Yes	Yes	unknown	1	0
Shell Trenches Subgroup	CSA - 1a	36-3	Yes	Yes	Yes	No	Yes	No	0	1	0
Hex Pit Subgroup	SPSA - 1f	1-13	Yes	Yes	Yes	Yes	Yes	No	unknown	1	0
Sanitary Landfills Medium Group											
	CSA - 1d	36-7	No	No	No	No	Yes	No	unknown	1	0
	ESA - 2b	30-4	No	Yes	No	No	Yes	No	1	0	0
	SSA - 4	1-12	No	Yes	No	No	Yes	No	0	0	0
	WSA - 2	4-2	Yes	Yes	No	No	Yes	No	2	0	0
	WSA - 3c	4-3	No	Yes	No	No	Yes	No	0	0	0
	WSA - 5a	4-5	Yes	Yes	No	No	Yes	No	0	0	0
	WSA - 5c	4-5	Yes	Yes	No	No	Yes	No	0	0	0
	WSA - 5d	4-5	Yes	Yes	No	No	Yes	No	0	0	0
Section 36 Lime Basins Subgroup	NCSA - 1b	36-4, -5, -10	Yes	Yes	Yes	No	No	No	150	2	0

Table B.7-1 Summary of Qualitative Assessment Results for Feasibility Study Action SAR Sites

Medium Group or Subgroup	SAR Site	CAR Site	Agent			Unexploded Ordnance			Number of Drums	Number Structures	Number of USTs
			Potential	Sampled	Detected	Potential	Sampled	Detected			
Buried M-1 Pits Subgroup	SPSA - 1e	1-13	Yes	Yes	Yes	Yes	Yes	No	unknown	1	0
South Plants Central Processing Area Subgroup	SPSA - 1a	1-13	Yes	Yes	Yes	Yes	Yes	No	unknown	230	0
South Plants Ditches Subgroup	SPSA - 1d	1-1	No	Yes	No	No	No	No	0	2	0
	SPSA - 2d	1-1	No	Yes	No	No	No	No	0	0	0
	SPSA - 3a	2-1	No	Yes	No	No	No	No	0	0	0
	SPSA - 4a	2-1	No	Yes	No	No	No	No	0	0	0
	SPSA - 5a	1-NSA	No	Yes	No	No	No	No	0	0	0
	SPSA - 8b	2-1	No	Yes	No	No	No	No	0	0	0
	SPSA - 9a	1-1	No	Yes	No	No	No	No	0	0	0
South Plants Tank Farm Subgroup	SPSA - 2a	1-10	Yes	No	No	No	Yes	No	0	11	0
	SPSA - 2b	1-9	No	No	No	No	No	No	0	2	0
South Plants Balance of Areas Subgroup	SPSA - 1b	1-3	Yes	Yes	Yes	No	No	No	0	6	0
	SPSA - 1c	1-5	No	No	No	No	No	No	0	1	0
	SPSA - 1g	1-13	Yes	Yes	Yes	Yes	Yes	No	unknown	59	0

Table B.7-1 Summary of Qualitative Assessment Results for Feasibility Study Action SAR Sites

Medium Group or Subgroup	SAR Site	CAR Site	Agent			Unexploded Ordnance			Number of Drums	Number Structures	Number of USTs
			Potential	Sampled	Detected	Potential	Sampled	Detected			
	SPSA - 2c	1-8	No	No	No	No	No	No	unknown	5	0
	SPSA - 2e	1-13	Yes	Yes	Yes	Yes	Yes	No	unknown	2	0
	SPSA - 3b	2-6	Yes	Yes	Yes	No	No	No	unknown	1	0
	SPSA - 3c	2-8	Yes	Yes	No	No	Yes	No	0	16	0
	SPSA - 3d	2-12	No	Yes	No	No	Yes	No	0	3	0
	SPSA - 3e	2-18	Yes	Yes	Yes	Yes	Yes	No	unknown	27	0
	SPSA - 4b	2-9, -18	No	Yes	No	No	No	No	0	33	0
	SPSA - 5b	1-11	No	Yes	No	No	Yes	No	0	24	0
	SPSA - 7b	2-3	No	Yes	No	No	Yes	No	0	0	0
	SPSA - 7c	2-2, -4, -13, -14	No	Yes	No	Yes	Yes	No	0	2	0
	SPSA - 8a	2-14A	No	Yes	No	No	Yes	No	44	0	0
	SPSA - 9b	1-4, -6	No	No	No	No	No	No	0	1	0
	SPSA - 12b	2-7	No	Yes	No	No	No	No	0	0	0
	SPSA - 12c	2-UNC	No	Yes	No	No	No	No	0	0	0
Buried Sediments Subgroup											
	SSA - 3a	11-1	No	Yes	No	No	No	No	0	0	0
	SSA - 3b	12-1	No	Yes	No	No	No	No	0	0	0
Sand Creek Lateral Subgroup											
	NCSA - 5b	35-4	Yes	Yes	Yes	No	No	No	0	0	0
	NCSA - 5c	35-NSA	No	No	No	No	No	No	0	0	0
	NPSA - 4	25-2	Yes	Yes	Yes	No	Yes	No	unknown	1	0
	SSA - 2b	2-1	No	Yes	No	No	No	No	0	0	0
	WSA - 6a	4-6	No	Yes	No	No	Yes	No	0	5	0

Table B.7-1 Summary of Qualitative Assessment Results for Feasibility Study Action SAR Sites

Medium Group or Subgroup	SAR Site	CAR Site	Agent			Unexploded Ordnance			Number of Drums	Number Structures	Number of USTs
			Potential	Sampled	Detected	Potential	Sampled	Detected			
Section 36 Balance of Areas Subgroup											
	CSA - 1b	36-14, -16, 17	Yes	Yes	Yes	Yes	Yes	Yes	unknown	3	0
	CSA - 2a	36-2	Yes	Yes	Yes	Yes	Yes	Yes	1	1	0
	CSA - 4	36-2, -12, -19	Yes	Yes	Yes	Yes	Yes	Yes	1	9	0
Burial Trenches Subgroup											
	ESA - 2a	32-5, -6	Yes	Yes	Yes	Yes	Yes	No	1200	0	0
	ESA - 2c	30-6	Yes	Yes	Yes	Yes	Yes	Yes	1	0	0

CAR contamination assessment report

SAR study area report

USTs underground storage tanks

Table B.7-2 Summary of Qualitative Assessment Results for Feasibility Study No Action SAR Sites

SAR Site	CAR Site (1)	Site Type	Agent			Unexploded Ordnance			Number of Drums	Number Structures	Number of USTs
			Potential	Sampled	Detected	Potential	Sampled	Detected			
ESA - 3j	31-7	storage	Yes	Yes	No	No	No	No	0	0	0
ESA - 3k	31-NSA	pit	Yes	No	No	No	No	No	0	0	0
ESA - 4c	29-2	trench	Yes	Yes	No	Yes	Yes	No	0	0	0
ESA - 5	30-5	munition disposal	No	Yes	Yes	No	Yes	No	2	2	0
ESA - 6a	6-NSA	NSA	Yes	Yes	No	Yes	Yes	No	0	0	0
ESA - 6b	30-1	NSA	No	Yes	Yes	Yes	Yes	No	0	0	0
ESA - 6d	20-2	NSA	No	No	No	Yes	Yes	No	0	0	0
NCSA - 2c	26-5	basin	Yes	Yes	Yes	No	No	No	0	0	0
NCSA - 8c	34-NSA	NSA	No	No	No	No	No	No	0	1	0
NCSA - 9a	23-NSA	NSA	No	Yes	Yes	No	Yes	No	0	0	0
NCSA - 9b	23-NSA	NSA	No	Yes	No	No	Yes	No	0	0	0
NCSA - 9c	23-NSA	NSA	No	Yes	No	No	Yes	No	0	0	0
NCSA - 9d	23-NSA	NSA	No	Yes	No	No	Yes	No	0	0	0
NCSA - 9e	24-NSA	NSA	No	Yes	No	No	Yes	No	0	0	0
NCSA - 9f	25-NSA	NSA	No	Yes	No	No	Yes	No	0	0	0
NCSA - 9g	26-NSA	NSA	No	Yes	No	No	Yes	No	0	0	0
NCSA - 9h	26-NSA	NSA	No	Yes	No	No	Yes	No	0	0	0
NCSA - 9i	26-NSA	NSA	No	Yes	No	No	Yes	No	0	0	0
NCSA - 9j	26-NSA	NSA	No	Yes	No	No	Yes	No	0	0	0
NCSA - 9k	26-NSA	NSA	No	Yes	No	No	Yes	No	0	0	0
NCSA - 9l	27-NSA	NSA	No	No	No	No	No	No	0	0	0
NCSA - 9m	35-6	firing range	No	No	No	Yes	Yes	No	0	0	0
NCSA - 9n	35-NSA	NSA	No	No	No	No	No	No	0	0	0
NCSA - 9o	35-9	NSA	No	No	No	No	No	No	0	0	0
NCSA - 9p	35-7	NSA	Yes	No	No	Yes	Yes	No	0	0	0
NCSA - 9q	36-10	NSA	No	Yes	No	No	No	No	0	0	0
NCSA - 9r	36-10	NSA	No	Yes	No	No	No	No	0	0	0
NCSA - 9s	36-NSA	NSA	Yes	Yes	No	Yes	No	No	0	0	0

Table B.7-2 Summary of Qualitative Assessment Results for Feasibility Study No Action SAR Sites

SAR Site	CAR Site (1)	Site Type	Agent			Unexploded Ordnance			Number of Drums	Number Structures	Number of USTs
			Potential	Sampled	Detected	Potential	Sampled	Detected			
NPSA - 2	25-NSA	tanks	No	Yes	No	No	Yes	No	0	6	0
NPSA - 7	25-NSA	spill	No	Yes	No	No	Yes	No	0	0	0
NPSA - 8a	25-NSA	ditch	No	Yes	No	No	Yes	No	0	0	0
NPSA - 8b	25-NSA	ditch	No	Yes	No	No	Yes	No	0	0	0
NPSA - 9a	25-NSA	NSA	No	Yes	No	No	Yes	No	0	0	0
NPSA - 9b	25-NSA	NSA	No	Yes	No	No	Yes	No	0	0	0
NPSA - 9c	25-NSA	NSA	No	Yes	No	No	Yes	No	0	0	0
NPSA - 9d	25-NSA	NSA	No	Yes	No	No	Yes	No	0	0	0
NPSA - 9e	25-NSA	NSA	No	Yes	No	No	Yes	No	0	0	0
SPSA - 6	1-7	manuf area	No	No	No	Yes	Yes	No	unknown	18	1
SPSA - 7a	2-1	ditch	No	Yes	No	No	No	No	0	0	0
SPSA - 8c	2-10, -11	manuf area	No	No	No	No	No	No	0	0	0
SSA - 1a	6-2	lake	No	Yes	No	No	No	No	0	0	0
SSA - 1d	12-2	lake	No	No	No	No	No	No	0	0	0
SSA - 1f	2-17	lake	No	No	No	No	No	No	0	0	0
SSA - 5a	1-NSA	NSA	No	Yes	No	No	No	No	0	0	0
SSA - 5c	11-NSA	NSA	No	Yes	No	No	No	No	0	0	0
SSA - 5d	11-NSA	NSA	No	Yes	No	No	No	No	0	0	0
SSA - 5e	11-NSA	ditch	No	Yes	No	No	No	No	0	0	0
WSA - 1a	3-NSA	NSA	No	Yes	No	No	No	No	0	0	0
WSA - 1b	3-NSA	spill	No	Yes	No	No	No	No	0	1	0
WSA - 1c	3-NSA	spill	No	Yes	No	No	No	No	0	0	0
WSA - 1d	3-NSA	NSA	No	Yes	No	No	No	No	0	0	0
WSA - 1e	3-4	spill	No	Yes	No	No	Yes	No	0	0	0
WSA - 1g	3-NSA	NSA	No	Yes	No	No	No	No	0	0	0
WSA - 3a	4-3	landfill	No	Yes	No	No	Yes	No	0	0	0
WSA - 3b	4-3	landfill	No	Yes	No	No	Yes	No	0	0	0
WSA - 3d	4-3	landfill	No	Yes	No	No	Yes	No	0	0	0

Table B.7-2 Summary of Qualitative Assessment Results for Feasibility Study No Action SAR Sites

SAR Site	CAR Site (1)	Site Type	Agent			Unexploded Ordnance			Number of Drums	Number Structures	Number of USTs
			Potential	Sampled	Detected	Potential	Sampled	Detected			
WSA - 4a	4-4	storage	No	Yes	No	No	Yes	No	0	0	0
WSA - 4b	4-4	storage	No	Yes	No	No	Yes	No	0	0	0
WSA - 5b	4-5	landfill	Yes	Yes	No	No	Yes	No	0	0	0
WSA - 6b	4-6	storage	No	Yes	No	No	Yes	No	0	3	0
WSA - 6c	4-6	sewer	No	Yes	No	No	Yes	No	0	1	0
WSA - 6d	4-6	ditch	No	Yes	No	No	Yes	No	0	1	0
WSA - 6e	4-6	ditch	No	Yes	No	No	Yes	No	0	0	0
WSA - 7b	34-2	sewer	No	No	No	No	No	No	0	1	0
WSA - 8a	33-NSA	NSA	No	Yes	No	No	No	No	0	0	0
WSA - 8b	33-NSA	NSA	No	Yes	No	No	No	No	0	0	0
WSA - 8c	3-NSA	NSA	No	Yes	No	No	No	No	0	0	0
WSA - 8d	4-NSA	NSA	No	Yes	No	No	No	No	0	0	0
WSA - 8e	4-NSA	NSA	No	Yes	No	No	No	No	0	0	0
WSA - 8f	9-NSA	NSA	No	Yes	No	No	No	No	0	0	0

1) Results for non-source area CARs not containing SAR sites are not included.

CAR contamination assessment report

SAR study area report

USTs underground storage tanks

APPENDIX B
(SECTION B.8)

DETERMINISTIC PARAMETERS CORRESPONDING TO 5th AND 50th PERCENTILE PPLVs

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LIST OF ACRONYMS AND ABBREVIATIONS

EPA	U.S. Environmental Protection Agency
HHRC	Human health risk characterization
IEA/RC	Integrated Endangerment Assessment/Risk Characterization
LSH	Latin Hypercube sampling
PPLV	Preliminary pollutant limit value
RAF	Relative absorption factor
SI	Soil ingestion

B.8.1 INTRODUCTION

In the Integrated Endangerment Assessment/Risk Characterization (IEA/RC), direct Preliminary Pollutant Limit Values (PPLVs) are calculated using a probabilistic Latin Hypercube analysis. Thus, an infinite number of deterministic parameter combinations could be used to calculate the 5th or 50th percentile direct PPLV values. This appendix section presents three deterministic combinations of parameters that correspond to the 5th and 50th percentile cumulative direct PPLVs for aldrin at a 10^{-6} cancer risk level. Aldrin was chosen for analysis, because it is identified as a risk-driver, based on the Human health risk characterization (HHRC) results.

In this appendix section, combinations of deterministic parameters are presented for the biological, commercial, and industrial worker receptors. Similar scenarios were not generated for the recreational and regulated/casual visitor receptors because the intermediate HHRC output files necessary to generate this information would require far more on-line data-storage capacity than is available on the HHRC code, due to the use of age-dependent probabilistic parameters for these receptor groups.

The procedures and results of this analysis are described below. Results are also summarized in Tables B.8-1, B.8-2, and B.8-3.

B.8.2 METHODS

Deterministic parameters corresponding to 5th and 50th percentile direct PPLVs were identified for biological, commercial, and industrial workers. For each of these populations, three possible combinations of deterministic parameters were identified for the corresponding 5th and 50th percentile PPLVs. These parameter values were derived from intermediate output files generated by the HHRC program.

B.8.2.1 Case A: Stratified Random Selection of Deterministic Parameters

The first set of parameter values, Case A, represents a likely combination based on the probability of each parameter being selected from its respective distribution, using stratified random sampling methods under a Latin Hypercube Sampling (LHS) routine. This case was chosen to represent a random (though stratified) selection of deterministic parameters.

To estimate the deterministic parameters under Case A, 100 direct PPLVs (one for each LHS sample) were calculated under the HHRC code. The set of parameter values corresponding to the 5th and 50th percentile direct PPLVs (i.e., the 6th and 51st lowest direct PPLVs) were then used to represent the Case A exposure parameter values.

B.8.2.2 Case B: Use of Maximum Oral and Dermal RAFs

The second set of parameter values, Case B, represents a combination which could occur if the dermal and oral relative absorption factors (RAFs) were equal to their maximum distributed values. These parameters were chosen because their variability was identified in the sensitivity analysis (Appendix Section B.5) as having a relatively large influence on the probabilistic PPLV results. Both the oral and the dermal RAFs were fixed at their maximum distributed values to represent a worst case approach for these parameters.

To estimate the deterministic parameters under Case B, the RAF dermal and RAF oral parameter values from each of the 100 LHS samples described above in Section B.8.2.1 were replaced by their maximum distribution values of 0.0052 and 0.65, respectively. All other parameter values were left unchanged. The 100 direct PPLVs were then re-calculated. The set of parameters corresponding to the value which most closely matched the 5th percentile PPLV from Case A was used to represent Case B, 5th percentile. The set of parameters corresponding to the value which most closely matched the 50th percentile PPLV from Case A was used to represent Case B, 50th percentile.

B.8.2.3 Use of 95th Percentile Soil Ingestion Values

The third set of parameter values, Case C, represents a combination which could occur if the soil ingestion (SI) parameter were equal to its 95th percentile value. This case was chosen because the variability in soil ingestion was identified in the sensitivity analysis (Appendix Section B.5) as having a large impact on the probabilistic PPLVs. The 95th percentile value was chosen because it represents an upperbound exposure value.

Because the soil ingestion parameter is population-specific, the 95th percentile values varied among the worker populations. For the biological worker, the 95th percentile soil ingestion value is 106 mg/day, and for the commercial worker, the value is 33 mg/day. For the industrial worker, the 95th percentile value is 50 mg/day, which corresponds to the reasonable maximum worker exposure parameter specified by the U.S. Environmental Protection Agency (EPA) in its Standard Default Exposure Factors (EPA, 1991).

Case C parameter sets for the 5th and 50th percentile direct PPLVs were calculated in the same way as the sets in Case B, except in instead of fixing the RAF parameters, the soil ingestion (SI) parameter values from each of the 100 LHS samples were replaced by their 95th percentile values. The RAF values and all other distributed parameters were left with their randomly generated values.

B.8.3 RESULTS AND CONCLUSIONS

The results of this evaluation are provided in Tables B.8-1, B.8-2, and B.8-3 for the biological, commercial, and industrial worker populations, respectively. These results should be interpreted with caution, as they represent three exposure possibilities out of an infinite number of exposure scenarios. It should also be kept in mind that the results presented here are exclusively for aldrin. For any worker populations, results for each chemical of concern will differ, due to the influence of the chemical-specific parameters, such as RAF and the toxicity criteria.

B.8.4 REFERENCES

RTIC 92232R03

EPA. 1991. Human Health Evaluation Manual, Supplemental Guidance: "Standard Default Exposure Factors." Office of Solid Waste and Emergency Response. Directive No. 9285.6-03. March 25, 1991.

Table B.8-1. Example Input Parameter Values That Lead to 5th and 50th Percentile Direct Carcinogenic PPLVs for Aldrin, Biological Worker[1/2/3]

	50th Percentile			5th Percentile		
	Case A: Random	Case B: Max RAFs	Case C: 95% SI	Case A: Random	Case B: Max RAFs	Case C: 95% SI
			Distributed Input Parameters [4/]			
Exposure Duration, TE (Years)	1.30E+00	9.17E+00	2.16E+00	1.16E+01	1.75E+01	1.20E+01
Percentile:	5	61	23	73	92	75
Soil Ingestion, SI (mg/day)	1.57E+02	10.7E+01	1.06E+02	9.06E+01	4.87E+01	1.06E+02
Percentile:	98	8	95	92	73	95
RAF, Dermal (unitless)	2.33E-03	5.20E-03	4.96E-03	3.78E-03	5.20E-03	1.06E-03
Percentile:	37	99	94	68	99	9
Soil Covering, SC (mg/cm^3)	2.41E-01	3.30E-01	4.94E-01	2.55E-01	2.86E-01	4.76E-01
Percentile:	18	41	74	22	30	71
RAF, Oral (unitless)	5.49E-01	6.50E-01	4.26E-01	5.90E-01	6.50E-01	5.10E-01
Percentile:	74	99	43	84	99	64
Exposure Frequency, DW (days/year)	2.23E+02	2.14E+02	2.14E+02	2.20E+02	2.20E+02	2.12E+02
Percentile	40	13	13	29	29	9
Dust Loading Factor, CSS (mg/m^3)	6.30E-02	3.44E-02	2.21E-02	2.99E-02	8.10E-02	5.50E-02
Percentile:	68	26	7	18	82	59
			Fixed Input Parameters			
Exposure Time, TM (hours/day)	8.00E+00	8.00E+00	8.00E+00	8.00E+00	8.00E+00	8.00E+00
Bodyweight, BW (kg)	67.8	67.8	67.8	67.8	67.8	67.8
Tox. Value, Ingestion (mg/kg/day)	5.90E-08	5.90E-08	5.90E-08	5.90E-08	5.90E-08	5.90E-08
Breathing Rate, BR (m^3/hour)	2.1	2.1	2.1	2.1	2.1	2.1

Table B.8-1. Example Input Parameter Values That Lead to 5th and 50th Percentile Direct Carcinogenic PPLVs for Aldrin, Biological Worker[1/2/3]

	50th Percentile			5th Percentile		
	Case A: Random	Case B: Max RAFs	Case C: 95% SI	Case A: Random	Case B: Max RAFs	Case C: 95% SI
Tox. Value, Inhalation (mg/kg/day)	5.90E-08	5.90E-08	5.90E-08	5.90E-08	5.90E-08	5.90E-08
Fraction Retained, FR (unitless)	0.85	0.85	0.85	0.85	0.85	0.85
Skin Surface Area, SSA (cm^2)	3270	3270	3270	3270	3270	3270
			Single Exposure Pathway PPLVs (mg/kg)			
SPPPLV, dermal	1.94E+02	9.41E+00	2.79E+01	1.30E+01	5.53E+00	2.46E+01
Percentile:	91	49	54	31	20	50
SPPPLV, ingestion	4.15E+00	7.61E+00	4.95E+00	7.64E-01	8.50E-01	7.53E-01
Percentile:	41	68	77	5	9	23
SPPPLV, inhalation	3.98E+02	1.07E+02	7.06E+02	9.56E+01	2.33E+01	5.18E+01
Percentile	80	49	94	43	8	24
			Cumulative Direct PPLVs (mg/kg)			
PPLV, cumulative direct	4.02E+00	4.05E+00	4.18E+00	7.16E-01	7.14E-01	7.21E-01

1/ PPLVs are soil concentrations corresponding to a 10^{-6} risk level, based on biological worker exposure to soil via the ingestion, particulate inhalation, and dermal contact pathways.

2/ Case A = Set of parameter values representing a likely combination based on parameter distributions.
Case B = Set of parameter values with RAF_{dermal} and RAF_{oral} equal to their maximum values of 0.0052 and 0.65 respectively.
Case C = Set of parameters with soil ingestion equal to its 95th percentile value of 106 mg/day.

3/ Exposure and toxicity parameters listed here are discussed in detail in Appendix Sections B.1 and B.3

4/ Distributed input parameter percentiles were approximated by ranking the 100 LHS sample values for each parameter, respectively. The input parameters were given percentile values corresponding to their ranking order. For example, a percentile value of 10 would indicate that the corresponding parameter was the 11th smallest out of 100 LHS sample values. Because the LHS sampling routine uses stratified random sampling, duplicates in LHS sample values for a given parameter can occur when the original parameter distributions are relatively steep and narrow. When a parameter value appeared more than once in the LHS sample, the lowest percentile value was used. See Appendix Section E.8.2 for more information on the LHS sampling process.

Table B.8-2. Example Input Parameter Values That Lead to 5th and 50th Percentile Direct Carcinogenic PPLVs for Aldrin, Commercial Worker [1/2/3/]

	50th Percentile			5th Percentile		
	Case A: Random	Case B: Max RAFs	Case C: 95% SI	Case A: Random	Case B: Max RAFs	Case C: 95% SI
			Distributed Input Parameters [4/]			
Exposure Duration, TE (Years)	2.54E+00	1.31E+00	9.59E-01	8.97E+00	6.96E+00	7.32E+00
Percentile:	53	30	21	88	83	84
Soil Ingestion, SI (mg/day)	6.14E+00	1.19E+01	3.30E+01	2.50E+01	1.84E+01	3.30E+01
Percentile:	25	59	95	89	80	95
RAF, Dermal (unitless)	4.21E-03	5.20E-03	2.46E-03	2.12E-03	5.20E-03	4.06E-03
Percentile:	78	99	40	32	99	75
Soil Covering, SC (mg/cm^3)	5.16E-02	6.34E-02	9.24E-02	6.04E-02	6.92E-02	4.77E-02
Percentile:	37	70	99	62	82	27
RAF, Oral (unitless)	6.34E-01	6.50E-01	3.41E-01	4.02E-01	6.50E-01	3.91E-01
Percentile:	96	99	22	37	99	35
Exposure Frequency, DW (days/year)	2.37E+02	2.36E+02	2.41E+02	2.39E+02	2.38E+02	2.37E+02
Percentile	52	38	93	78	66	52
Dust Loading Factor, CSS (mg/m^3)	5.08E-03	9.20E-03	3.23E-03	2.45E-03	3.81E-03	4.03E-03
Percentile:	56	88	27	14	37	41
Exposure Time, TM (hours/day)	3.39E+00	5.43E+00	6.66E+00	4.84E+00	6.09E+00	9.01E+00
Percentile:	14	27	40	21	34	68
			Fixed Input Parameters			
Bodyweight, BW (kg)	68.7	68.7	68.7	68.7	68.7	68.7
Tox. Value, Ingestion (mg/kg/day)	5.90E-08	5.90E-08	5.90E-08	5.90E-08	5.90E-08	5.90E-08

Table B.8-2. Example Input Parameter Values That Lead to 5th and 50th Percentile Direct Carcinogenic PPLVs for Aldrin, Commercial Worker [1/2/3/]

	50th Percentile			5th Percentile		
	Case A: Random	Case B: Max RAFs	Case C: 95% SI	Case A: Random	Case B: Max RAFs	Case C: 95% SI
Breathing Rate, BR (m^3/hour)	0.83	0.83	0.83	0.83	0.83	0.83
Tox. Value, Inhalation (mg/kg/day)	5.90E-08	5.90E-08	5.90E-08	5.90E-08	5.90E-08	5.90E-08
Fraction Retained, FR (unitless)	0.85	0.85	0.85	0.85	0.85	0.85
Skin Surface Area, SSA (cm^2)	1550	1550	1550	1550	1550	1550
			Single Exposure Pathway PPLVs (mg/kg)			
SPPPLV, dermal	5.12E+02	6.54E+02	1.27E+03	2.43E+02	1.12E+02	1.99E+02
Percentile:	32	68	68	12	13	8
SPPPLV, ingestion	4.42E+01	4.33E+01	3.99E+01	4.81E+00	5.23E+00	4.64E+00
Percentile:	51	63	85	5	13	20
SPPPLV, inhalation	1.22E+04	9.48E+03	2.95E+04	5.76E+03	3.82E+03	2.33E+03
Percentile	56	47	79	34	26	20
			Cumulative Direct PPLVs (mg/kg)			
PPLV, cumulative direct	4.06E+01	4.04E+01	3.86E+01	4.71E+00	4.99E+00	4.53E+00

1/ PPLVs are soil concentrations corresponding to a 10^{-6} risk level, based on commercial worker exposure to soil via the ingestion, particulate inhalation, and dermal contact pathways.

2/ Case A = Set of parameter values representing a likely combination based on parameter distributions.
Case B = Set of parameter values with RAF_{dermal} and RAF_{oral} equal to their maximum values of 0.0052 and 0.65, respectively.
Case C = Set of parameters with soil ingestion equal to its 95th percentile value of 33 mg/day.

3/ Exposure and toxicity parameters listed here are discussed in detail in Appendix Sections B.1 and B.3

4/ Distributed input parameter percentiles were approximated by ranking the 100 LHS sample values for each parameter, respectively. The input parameters were given percentile values corresponding to their ranking order. For example, a percentile value of 10 would indicate that the corresponding parameter was the 11th smallest out of 100 LHS sample values. Because the LHS sampling routine uses stratified random sampling, duplicates in LHS sample values for a given parameter can occur when the original parameter distributions are relatively steep and narrow. When a parameter value appeared more than once in the LHS sample, the lowest percentile value was used. See Appendix Section E.8.2 for more information on the LHS sampling process.

Table B.8-3. Example Input Parameter Values That Lead to 5th and 50th Percentile Direct Carcinogenic PPLVs for Aldrin, Industrial Worker [1/2/3/]

	50th Percentile			5th Percentile		
	Case A: Random	Case B: Max RAFs	Case C: 95% SI	Case A: Random	Case B: Max RAFs	Case C: 95% SI
			Distributed Input Parameters [4/]			
Exposure Duration, TE (Years)	1.51E+00	1.83E+00	1.32E+00	1.36E+01	7.84E+00	4.94E+00
Percentile:	35	41	30	94	86	74
Soil Ingestion, SI (mg/day)	2.23E+01	1.51E+01	5.00E+01	1.22E+01	1.26E+01	5.00E+01
Percentile:	74	57	95	46	48	95
RAF, Dermal (unitless)	4.49E-03	5.20E-03	3.03E-03	3.27E-03	5.20E-03	9.24E-04
Percentile:	84	99	52	57	99	6
Soil Covering, SC (mg/cm^3)	4.74E-01	3.42E-01	4.11E-01	6.79E-01	5.93E-01	8.80E-01
Percentile:	51	26	39	79	70	91
RAF, Oral (unitless)	5.36E-01	6.50E-01	3.65E-01	2.68E-01	6.50E-01	5.77E-01
Percentile:	71	99	28	4	99	81
Exposure Frequency, DW (days/year)	2.38E+02	2.37E+02	2.34E+02	2.34E+02	2.37E+02	2.36E+02
Percentile	66	53	15	15	53	38
Dust Loading Factor, CSS (mg/m^3)	2.09E-02	1.52E-02	2.40E-02	1.33E-02	1.20E-02	3.39E-02
Percentile:	51	28	61	20	15	82
Exposure Time, TM (hours/day)	3.18E+00	2.17E+00	4.67E+00	1.07E+01	1.04E+01	6.06E+00
Percentile:	9	5	19	83	81	33
			Fixed Input Parameters			
Bodyweight, BW (kg)	68.7	68.7	68.7	68.7	68.7	68.7
Tox. Value, Ingestion (mg/kg/day)	5.90E-08	5.90E-08	5.90E-08	5.90E-08	5.90E-08	5.90E-08

Table B.8-3. Example Input Parameter Values That Lead to 5th and 50th Percentile Direct Carcinogenic PPLVs for Aldrin, Industrial Worker [1/2/3/]

	50th Percentile			5th Percentile		
	Case A: Random	Case B: Max RAFs	Case C: 95% SI	Case A: Random	Case B: Max RAFs	Case C: 95% SI
Breathing Rate, BR (m^3/hour)	2.1	2.1	2.1	2.1	2.1	2.1
Tox. Value, Inhalation (mg/kg/day)	5.90E-08	5.90E-08	5.90E-08	5.90E-08	5.90E-08	5.90E-08
Fraction Retained, FR (unitless)	0.85	0.85	0.85	0.85	0.85	0.85
Skin Surface Area, SSA (cm^2)	3270	3270	3270	3270	3270	3270
			Single Exposure Pathway PPLVs (mg/kg)			
SPPPLV, dermal	4.15E+01	4.12E+01	8.22E+01	4.50E+00	5.52E+00	3.34E+01
Percentile:	46	67	61	3	11	38
SPPPLV, ingestion	2.42E+01	2.44E+01	1.83E+01	9.96E+00	6.82E+00	3.08E+00
Percentile:	45	56	73	17	19	16
SPPPLV, inhalation	2.44E+03	4.07E+03	1.67E+03	1.29E+02	2.50E+02	2.42E+02
Percentile	83	90	70	9	24	23
			Cumulative Direct PPLVs (mg/kg)			
PPLV, cumulative direct	1.52E+01	1.53E+01	1.48E+01	3.02E+00	3.01E+00	2.79E+00

1/ PPLVs are soil concentrations corresponding to a 10^{-6} risk level, based on industrial worker exposure to soil via the ingestion, particulate inhalation, and dermal contact pathways.

2/ Case A = Set of parameter values representing a likely combination based on parameter distributions.
Case B = Set of parameter values with RAF_{dermal} and RAF_{oral} equal to their maximum values of 0.0052 and 0.65 respectively.
Case C = Set of parameter with soil ingestion equal to its 95th percentile value of 106 mg/day.

3/ Exposure and toxicity parameters listed here are discussed in detail in Appendix Sections B.1 and B.3

4/ Distributed input parameter percentiles were approximated by ranking the 100 LHS sample values for each parameter, respectively. The input parameters were given percentile values corresponding to their ranking order. For example, a percentile value of 10 would indicate that the corresponding parameter was the 11th smallest out of 100 LHS sample values. Because the LHS sampling routine uses stratified random sampling, duplicates in LHS sample values for a given parameter can occur when the original parameter distributions are relatively steep and narrow. When a parameter value appeared more than once in the LHS sample, the lowest percentile value was used. See Appendix Section E.8.2 for more information on the LHS sampling process.

APPENDIX C
(SECTION C.1)

COMPUTATIONAL METHODOLOGY

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LIST OF ACRONYMS AND ABBREVIATIONS

ARAR	applicable or relevant and appropriate requirement
Army	U.S. Department of the Army
AWQC	ambient water quality criteria
BAF	bioaccumulation factor
BCRL	below certified reporting limit
BMFs	biomagnification factors
CMP	comprehensive monitoring program
COCs	contaminant of concerns
C_{ow}	average exposure area water concentration of contaminant
CPMS	chlorophenylmethyl sulfide
$CPMSO_2$	chlorophenylmethyl sulfone
C_{rep}	representative site concentration
CRL	certified reporting limit
C_{sed}	average exposure area sediment concentration of contaminant
C_w	contaminant concentration in surface water
DBCP	dibromochloropropane
DCPD	dicyclopentadiene
DDE	dichlorodiphenyldichloroethene
DDT	dichlorodiphenyltrichloroethane
EPA	U.S. Environmental Protection Agency
ERC	Ecological Risk Characterization
ESC	average exposure area soil concentration of contaminant
<ESC>	estimated average exposure area soil concentration of contaminant
FR	dietary fraction
FS	Feasibility Study
ft	foot/feet
ft^2	feet square
ha	hectares
HHRC	Human Health Risk Characterization
HI	hazard index
HQ	Hazard Quotient
IEA/RC	Integrated Endangerment Assessment/Risk Characterization
kg/kg-bw/day	kilograms per kilogram of body weight per day
LT	less than
MATC	maximum allowable tissue concentration (micrograms contaminant in tissue per gram body weight)
NOAELs	no observed adverse effects levels
NPL	National Priority List
QA/QC	quality assurance/quality control
ppm	parts per million
R	feed rate (kilogram food per kilogram body weight per day)

LIST OF ACRONYMS AND ABBREVIATIONS
(continued)

RI	Remedial Investigation
RI/FS	Remedial Investigation/Feasibility Study
RMA	Rocky Mountain Arsenal
SAR	Study Area Report
Shell	Shell Oil Company
TC	tissue concentration
TPC	tissue partitioning coefficient
TRV	toxicity reference value (microgram ingested per gram body weight per day)
UFs	uncertainty factor(s)
WASP	Water Quality Analysis Simulation Program

C.1 COMPUTATIONAL METHODOLOGY

C.1.1 INTRODUCTION

To characterize risk to the diverse plants and animals at Rocky Mountain Arsenal (RMA) from the contaminants present as a result of human activities on the site, it was first necessary to select the contaminants of concern (COCs) for biota from among those present and to identify the receptors potentially exposed to those contaminants. The COCs were selected using a set of criteria that evaluated their toxicity and the likelihood of receptors being exposed to them. The receptors were identified as representatives of important food webs that interrelate the biota on RMA in a matrix of predator and prey relationships.

Potential risk to the identified receptors was characterized by estimating the concentrations to which they were exposed and their resulting tissue concentrations and then comparing these measured or predicted site-specific tissue concentrations to toxicological threshold values. This general strategy was implemented as shown in Figure C.1-1, using two basic approaches, a tissue-based approach and a dose-based approach. To calculate potential risk for a particular trophic box and chemical combination, measurements of its tissue concentration were compared to its maximum allowable tissue concentration (MATC); measurements of the tissue concentration in its daily food (and soil/sediment and water) intake were compared to its toxicity reference value (TRV). The estimation of tissue concentrations was necessary in three situations: (1) for terrestrial prey species because soil concentrations are variable across RMA and tissue concentrations were not measured at all locations; (2) for top-level predator trophic boxes (i.e., bald eagle, great horned owl, American kestrel, and great blue heron) because tissue concentrations were not measured anywhere on site for these species; and (3) for aquatic trophic boxes where tissue concentration data for specific trophic box/chemical combinations were missing.

In the first situation, the potential exposure among individuals within a given trophic box varies because the concentrations of COCs in soil vary markedly across RMA. The potential exposure among different trophic boxes varies still more because the size of the exposure area for the trophic boxes varies. The measured tissue concentrations of prey species collected from RMA

are the best available representation of the results of exposing an organism to the variety of soil concentrations at RMA. The mean ratio of these tissues to the "exposure area soil concentration" (ESC) is the biomagnification factor (BMF). Measured tissue concentrations and their associated estimated exposure area soil concentrations (<ESC>s) were used to define a sample BMF distribution for each trophic box/chemical combination. The mean value from this distribution was then used to estimate tissue concentrations associated with trophic box-specific <ESC> values calculated for locations 100 feet apart all across RMA. Three different approaches for defining the BMF distribution are used in this document (Army's, Shell's, and EPA's).

In the second situation, where tissue concentrations for predators are lacking, doses to predator from the estimated tissue concentrations in their food (and soil/sediment and water) can be calculated; this is the dose-based approach. This is done using the prey tissue concentrations estimated above and weighted as to the proportion of the daily diet they comprise; however, to make this dose specific to exposure incurred across RMA, it must be predicted from BMF and <ESC> values at specified locations and adjusted for daily food intake (R). Risk is then based on a comparison of dose to the dose-based toxicity threshold (i.e., TRV). Alternatively, the tissue concentration in a predator (rather than in its food) can be calculated from <ESC> and the predator's BMF calculated by multiplying its bioaccumulation factor (BAF) by its dose from food. The BAF is needed to convert prey BMF to predator BMF because BAF is defined as the ratio of tissue concentrations between a predator and its food. Risk in this case is based on a comparison of tissue concentration to the tissue-based toxicity threshold (i.e., MATC). Again, these calculations can be made for locations 100 feet apart all across RMA.

In the third situation, which applies to aquatic trophic boxes, exposure is to relatively homogeneous concentrations in water. Therefore, the variability of exposure across RMA, other than by lake, is not an issue. Further, because risk could be evaluated only for highly mobile aquatic receptors (e.g., birds) for reasons explained below, even the variability among lakes is not an issue, since a given bird could readily feed at all RMA lakes and was therefore exposed to an average of their contaminants.

The remainder of Appendix Section C.1 is devoted to providing methodological detail on the selection of COCs (Appendix Section C.1.2), identification of target biota receptors (Appendix Section C.1.3), calculation of exposure concentrations (Appendix Section C.1.4), development of BMF (Appendix Section C.1.5), and on the calculation of potential risk from this information (Appendix Section C.1.6). The final discussion (Appendix Section C.1.7) summarizes the quantitative uncertainty analysis that was performed on this overall approach.

C.1.2 SELECTION AND EVALUATION OF ECOLOGICAL RISK CHARACTERIZATION CHEMICALS

The COCs for which ecological risk was characterized were selected in two stages. Initially, seven COCs were selected that had also been identified in both the Biota Remedial Investigation (RI) (ESE 1989) and Biota Comprehensive Monitoring Program (CMP) (RLSA 1992) reports as chemicals that were the most widespread, bioaccumulative, persistent, and most toxic among the chemicals known to be present at RMA. These COCs were the target analytes for biota tissue samples analyzed under these programs, as well as under the ERC, and thus tissue concentrations are available for them. These initial seven COCs are aldrin, dieldrin, endrin, dichlorodiphenyltrichloroethane (DDT), dichlorodiphenyldichloroethene (DDE), arsenic, and mercury. For the purposes of modeling and risk characterization, aldrin was combined with dieldrin, and DDT with DDE because the first (parent) compound in each pair is readily metabolized to the second. With the exception of arsenic, each of the seven COCs bioaccumulates substantively.

During the ERC, Phase II RI data became available and were used to reevaluate the areal extent of chemicals given less importance in the Biota RI because they had been detected in fewer than 5 acres on RMA. As a result of this reevaluation, seven additional contaminants were added to the list of COCs: cadmium, chlordane, chlorophenylmethyl sulfide (CPMS), chlorophenylmethyl sulfone ($CPMSO_2$), copper, dibromochloropropane (DBCP), and dicyclopentadiene (DCPD). Except for chlordane, these contaminants are generally considered nonbioaccumulative. Chlordane is a chlorinated hydrocarbon insecticide that can potentially accumulate in lipid-containing tissue following prolonged exposure to low concentrations in contaminated media.

Cadmium and copper may accumulate in selected tissues under unusual circumstances of high exposure. The four metal COCs (mercury, arsenic, cadmium, and copper) are all naturally occurring in soils at RMA, in addition to being present as a result of anthropogenic activity. Potential risks to biota from these 14 chemicals were evaluated in the ERC.

C.1.3 IDENTIFICATION OF TARGET BIOTA RECEPTORS

Exposure of biota to contaminants can follow pathways of ingestion, imbibition, inhalation, and dermal contact. The primary pathway through which biota are exposed to contaminants on RMA is ingestion, especially for the bioaccumulative COCs (aldrin/dieldrin, DDE/DDT, endrin, and mercury). By definition, bioaccumulative chemicals concentrate to higher levels in each successive level in a food web. To select receptors of concern as a result of exposure from this pathway, a food web was developed to describe the relationships of predator and prey from the level of autotrophs (i.e., self-feeders, or plants) to the top predators in the ecosystems on RMA (e.g., bald eagles, great horned owls).

The food web includes representative site-specific species in food chains originating in aquatic or terrestrial ecosystems that lead to site-specific taxa at the top of the food web. Food webs culminating in four species and a species group were selected for the RMA ecological risk evaluations: the bald eagle (Figure C.1-2), the American kestrel (Figure C.1-3), the great horned owl (Figure C.1-4), the great blue heron (Figure C.1-5), and the shorebird group, represented on RMA by the killdeer (Figure C.1-6). The bald eagle, an endangered species, represents the highest avian trophic level potentially affected by the biomagnification of contaminants at RMA. The American kestrel, also an avian top predator, is abundant at RMA. Food webs for each of the three additional top food-web species (great horned owl, great blue heron, and shorebird) addressed different considerations. While the great horned owl food web generally incorporated terrestrial food chains from the kestrel and bald eagle food webs, this species feeds at night and is a year-round resident at RMA. The great blue heron food web evaluated risk to birds at RMA that consume primarily fish. Since shorebirds probe for food in sediments, the shorebird food web incorporated a pathway of direct exposure to sediments, which is not covered in the other food webs.

To construct each of these food webs, data were collected from literature sources on ecosystems similar to those at RMA, information from regional experts was obtained, and on-site observations were performed to determine major plant and animal species and habitats that occur within site boundaries. From this information trophic boxes, groups of species with similar feeding requirements at the same feeding level, were developed for each food web.

To select the target biota receptors representative of each trophic box, the IEA/RC employed the same approach presented in the Biota RI (ESE 1989) and Biota CMP (RLSA 1992). The information evaluated for the selection of receptors included species abundance, home range, and distribution, as well as whether the species is threatened or endangered, economically or socially important, or an important component of regional food webs. Data on the food habits of each of the representative species at RMA (i.e., the target biota receptors) collected from literature and field observations at RMA were also considered. In addition to the top trophic box representatives (bald eagle, great horned owl, great blue heron, American kestrel, and shorebirds), the other target biota receptors are: black-tailed prairie dog and desert cottontail (medium mammal trophic box); deer mouse and thirteen-lined ground squirrel (small mammal trophic box); bullsnake and toad (reptile and terrestrial amphibian trophic box); mourning dove, vesper sparrow, and western meadowlark (small bird trophic box); mallard, blue-winged teal, and American coot (water bird trophic box); earthworm (earthworm trophic box); grasshoppers and ground beetles (insect trophic box); cheatgrass, kochia, lactuca, morning glory, and sunflower (terrestrial plant trophic box); northern pike and largemouth bass (large fish trophic box); bluegill, bullheads, and channel catfish (small fish trophic box); salamanders (amphibian trophic box); aquatic invertebrates (aquatic invertebrate trophic box); American pondweed, sago pondweed, coontail, and various aquatic macrophytes (aquatic plant trophic box); and plankton (plankton trophic box). Each of the representative target biota receptors below the top trophic boxes was sampled for analysis of its tissues. Samples of great horned owl eggs and American kestrel eggs and juveniles were also collected. These representative target biota receptors were exposed either directly or indirectly (i.e. through eating) to contaminant concentrations in soil, sediment, and/or water. A combined total of 1,897 biota tissue samples were collected under the Biota RI (ESE 1989), the Biota CMP (RLSA 1992), and the ERC. Once off-post control samples, fortuitous

(i.e., found sick or dead) samples, and QA/QC rejections were removed, 1,328 samples remained to provide tissue concentrations representative of the terrestrial and aquatic trophic boxes. Further information on the collection of these biota samples is in Appendix A and Appendix Section C.4.

C.1.4 CALCULATION OF EXPOSURE CONCENTRATIONS

Due to the large area and variable contamination patterns across the RMA, the spatial distribution of risk from terrestrial food-web exposure was modeled based on exposure to contaminant concentrations in the soil. The development of estimated exposure area soil concentrations is described in Appendix Section C.1.4.1.

In contrast, aquatic species were assumed to integrate their exposure over an entire lake, and therefore tissue concentration samples from a given lake adequately described the mean tissue concentration and risk for that lake (as long as sample size and spatial representation were adequate). Birds with aquatic food webs were assumed to be exposed to either a lake-specific mean tissue concentration in aquatic prey items (and aquatic media ingested directly) or, if individuals were likely to divide their feeding time over several lakes, to a mean tissue/media concentration contributed by a group of lakes. Therefore, aquatic risk for these species was estimated based on direct estimates of observed tissue concentrations in the predator or in prey items. The estimation of aquatic prey tissue concentrations is described in C.1.4.2.

C.1.4.1 Characterization of Exposure Concentration for Terrestrial Food Webs

It is necessary to begin the discussion of exposure concentrations by defining terms, particularly to distinguish exposure soil concentration, exposure area soil concentration (ESC), and estimated exposure area soil concentration (<ESC>). These definitions, given below, indicate assumptions and uncertainties in using <ESC> to estimate risk.

Exposure concentrations are the contaminant concentrations in source media (i.e., soil, sediment, and water) that are bioavailable and accessible to the receptor (i.e., the contaminant concentrations to which organisms are exposed). Thus, an individual organism's exposure soil concentration is the bioavailable and accessible contaminant concentration in the actual soil to

which the individual is exposed over a specified interval of time. The exposure area soil concentration (ESC) differs from the exposure soil concentration in that, rather than describing exposure to an organism, it describes the average soil concentration in an area, i.e., an "exposure area". Thus, the ESC is the average contaminant concentration in a specified soil depth profile over a circular area with radius determined on a species-by-species or trophic-box-specific basis. The estimated exposure area soils concentration (<ESC>) is an estimate of the exposure area soil concentration, derived from the RMA soil concentration database using the statistical estimation techniques described in this section. The RMA Soil data base that was used resulted from a March 5, 1993 data pull from D.P. and Associates that was screened and subjected to quality assurance checks as described in Appendix Section D.1.4.1.

The estimated ESCs were used to calculate site-specific RMA biomagnification factors (BMF_{obs}) and potential risk estimates expressed as hazard quotients (HQs) and hazards indices (HIs) for terrestrial food chain trophic boxes. The applications are explained, respectively, in Appendix Sections C.1.5 and C.1.6.

<ESC>s were calculated based on an area-wide average (i.e., an arithmetic mean) concentration, an area being defined as an organism's estimated foraging or exposure area. The trophic-box-specific values for the individual exposure areas, as well as the methods used to develop the values, are described in Appendix Section C.2.4. The area-averaged concentration was computed from spatially interpolated soil concentrations in the 0- to 1-foot (ft) depth interval (except for the prairie dog's exposure area, which incorporates a vertical average for the 0- to 20-ft depth interval). The interpolated soil concentrations were calculated on grid with 100-ft spacing using surrounding actual soil sample concentration data and the inverse distance-squared algorithm. Before the soil data were interpolated, values that were below certified reporting limits (BCRL) were replaced with estimated values when the surrounding data were sufficient. More specifically, exposure area soil concentrations were estimated in three steps: spatial interpolation of BCRL data, interpolation of soil concentrations onto an RMA-wide grid, and averaging of interpolated data within an exposure area to compute <ESC>. These steps are described in further detail below and are illustrated in Appendix Section C.1.4.1.4 using a detailed example

taken from the data. Appendix Section C.1.4.1.5 discusses the rationale for selecting the inverse distance-squared interpolation method.

C.1.4.1.1 Spatial Interpolation of BCRL Data

A replacement value for each BCRL was interpolated based on nearby detections and other estimated BCRLs using the inverse distance-squared algorithm described in Appendix Section C.1.4.1.2. The BCRL data replacement procedure is described in this section.

The spatial interpolation of BCRL data proceeded iteratively. An initial estimate for each grid point was calculated using the detections found within a specified search radius. BCRL data within the search radius were not factored into the estimation during this first iteration, but were used in latter iterations when they had received an estimate. After estimates were calculated for all BCRL samples, each estimate was compared to the associated CRL and adjusted as follows. If the calculated estimate was less than CRL, the BCRL data point was assigned the calculated estimate. If the calculated estimate was greater than or equal to the CRL, the BCRL data point was assigned the CRL value. If no detections were found within the search radius of a given BCRL, then no replacement was made for that BCRL data point. Successive iterations calculated a new estimate for each BCRL data point, each time including both detections and other estimated BCRL values (from the previous iteration) in the calculation. The second iteration estimated values for previously unestimated BCRL data points if some first iteration estimates fell within the search radius, thereby allowing an estimate. The third and fourth iterations were only used to calculate new estimates for BCRL data points that had been previously estimated.

A maximum of six points was used to calculate any given estimate (the six closest points if the search radius included more than six points). The search radii used were 1,200 ft outside of designated sites (including Basin F Exterior and the wind dispersion area) and 400 ft for all other designated sites. A vertical search radius of 5 ft was used with each of these search radii, regardless of the depth of the BCRL sample being estimated. This allowed the inclusion of nearby data from a different depth in the BCRL interpolation for samples in the 0- to 1-ft depth interval as well as for samples from the 1- to 20-ft deptn interval that were used for prairie dog

risk calculations. (The relative scaling of vertical and horizontal distances is discussed in C.1.4.1.2.) Estimates for borings outside of designated sites made use only of data from outside of designated sites to retain differences in soil concentration characteristics within and outside of designated sites.

Depending on the COC, 6.5 to 85.1 percent of the BCRL data points did not receive an estimate during the BCRL interpolations. Further details and a discussion of associated uncertainty are given in Appendix Section E.12.4.2.2.

This treatment of BCRLs differed from that used in the Human Health Risk Characterization (HHRC) because the estimation of the HHRC site-wide representative concentrations (C_{rep}) did not require spatial interpolation. Therefore, established statistical methods for handling BCRL data (e.g., robust method) were applied for the nonspatial HHRC estimation procedure. However, since established BCRL methods do not consider the spatial distribution of samples, they could not be used in the spatial estimation procedure for the ERC.

C.1.4.1.2 Interpolation onto the RMA-Wide Grid

The spatial distribution of RMA-wide soil concentrations was interpolated from soil samples by using the inverse distance-squared algorithm again. Interpolations were created for the 0- to 1-ft depth interval, for use in characterizing exposure for most species, and for the 0- to 20-ft depth interval for use in characterizing exposure for the prairie dog. The general methods used for both methods are described first, followed by specifics pertaining to the 0- to 20-ft interpolation for prairie dog.

For both interpolations, a grid was developed that divided RMA into 100- by 100- by 1-ft blocks. Each point on the grid represented the center of a particular grid block. Concentrations in each block were estimated for each contaminant using the inverse distance-squared algorithm. This algorithm has also been used in the interpolation of contaminant concentrations for the feasibility study at RMA. The inverse distance squared algorithm calculates an estimate for grid point k based on a weighted average of samples that fall within a search radius around k, using the

equation given below. The weights are inversely proportional to the distance between the sample and the grid point to be estimated. A maximum of six samples are used to calculate an estimate for a given grid point; the six closest samples to this point are used if there are more than six samples within the search radius.

$$Estimate_k = \sum_{i \in SR(k)} X_i W_i \qquad (1)$$

where:

k	=	grid point being estimated
SR(k)	=	search radius of grid point k (only 6 closest points)
X_i	=	chemical concentration of sample at location i
W_i	=	$\frac{1}{D(i,k)^2} \div \sum_{i \in SR(k)} \frac{1}{D(i,k)^2}$
D(i,k)	=	distance between i and k

The search radius in equation (1) is specified horizontally for interpolations for both depth intervals, and vertically for the interpolation over 0 to 20 feet. Horizontal search radii were assigned by grouping designated sites according to similarities in contamination characteristics or location as follows: outside designated sites, 1,200 ft; sites in South Plants, 800 ft; sites in the central and north central areas, 750 ft; sites in the eastern, western, southern, and North Plants areas, 400 ft; Shell trenches and complex disposal trenches, 200 ft. A maximum of six points were used in all cases to estimate soil concentration for each model block. Estimates for grid blocks within a given site were based only on samples located within sites of the same site group. In all cases if there was no information to model a given grid block, that block was labeled as "not estimated" (NE).

The above algorithms and specifications were applied to the interpolations for both depth intervals, 0- to 1-ft (general) and 0- to 20-ft (prairie dogs). Interpolated soil concentrations for the 0- to 1-ft depth interval, which were used for all species including prairie dog, were calculated based on samples only from the top 1 foot. The rationale for excluding lower soil

depths from the interpolation was that exposure of most target receptors is primarily to the top 1 ft and contaminant transport processes on RMA imply that surface contamination tends to be much higher and possibly unrelated to contamination at lower depths. Additional specifications used in the interpolation for lower depths are described below.

Prairie dogs are known to burrow to a depth of 20 ft. They may ingest soils throughout this depth range when digging or grooming, although they feed primarily on insects and terrestrial plants, which are exposed primarily to soils near the surface. Therefore, the exposure area soil concentration for prairie dogs was calculated as a weighted average of soil concentrations within the circular prairie dog exposure area, but extending to a depth of 20 ft. The weights, based on the prairie dog dietary fractions for terrestrial plants exposed to the 0- to 1-ft depth interval and soil ingested uniformly from the 0- to 20-ft interval were as follows: 0- to 1-ft depth interval weight, 0.9981; 1- to 20-ft depth interval weight, 0.0019.

Accordingly, exposure area soil concentration estimations for prairie dogs included both a 0- to 1-ft depth interpolation layer and a second interpolation layer for the 1- to 20-ft depth interval within designated sites. Within designated sites, a three-dimensional grid of soil concentrations was constructed by interpolating among the depth-specific data of the sample borings. The concentration for each grid block was estimated from the samples that fell within an elliptical sphere surrounding that grid block. The sphere was defined by a vertical axis that was much shorter than the horizontal axis because concentrations were expected to change faster with depth than laterally. For the three dimensional interpolation, the distance parameter in equation (1) was defined as the elliptical distance, i.e., the true distance normalized to the equation of an ellipsoid. For example, all points on the edge of the ellipsoid have an elliptical distance of 1.0. Interpolated soil concentrations within the 0- to 1-ft layer were calculated based only on samples within the 0- to 1-ft depth interval. Interpolated concentrations within the remaining 1- to 20-ft depth interval were calculated based on a search radius of 10 feet, where samples from the top 0- to 1-ft depth interval were excluded. The 0- to 1-ft samples were excluded from estimations at lower depths and lower samples were excluded from estimation at the 0- to 1-ft depth because, as discussed above, contaminant transport processes on RMA imply that surface contamination

tends to be much higher and possibly unrelated to contamination at lower depths. It was judged that the estimations would be more representative of contaminant concentrations if the 0- to 1-ft and 1- to 20-ft data sets were separated during the interpolation. Outside designated sites, nearly all of the reliable data are from the 0- to 1-ft depth interval.

After all grid blocks were modeled, the three dimensional grid was transformed to a two dimensional grid where each point represented the weighted average concentration for the depth profile associated with this point. Specifically, the concentrations estimated for the 0- to 1-ft and 1- to 20-ft depth intervals for a given column on the three dimensional grid were averaged according to the dietary weights developed above for prairie dog and then assigned to the corresponding point on the two dimensional grid. Exposure area soil concentrations (ESC) for the prairie dog were then calculated (as described in C.1.4.1.3) by averaging concentrations on the two dimensional grid for prairie dog.

C.1.4.1.3 Averaging Within the Exposure Area

Estimated exposure area soil concentration (<ESC>) is the average of the interpolated soil concentrations at all grid points within a given species- or trophic-box-specific exposure area. For the purpose of estimating BMF_{obs}, <ESC> is calculated for an exposure area centered on a specific biota sample collection location. For the characterization of risk, <ESC> is calculated for exposure areas centered at each point of the 100- by 100-ft RMA-wide grid. The <ESC> for a given grid point was used to calculate potential risk at that grid point. If the exposure range for a trophic box was small enough to encompass at most one grid point at a time, the <ESC> value for that grid point is the interpolated value, since there are no other values within the exposure area to be averaged.

If a small fraction of the cells within a given exposure area did not contain soil concentration information, then the ESC for that exposure area was estimated with these cells excluded from the average. If the fraction of cells without concentration estimates was large (e.g. greater than about 1/2) then <ESC> and risk were assigned a value of NE ("not estimated"). The treatment of areas of inadequate soil data and the lake and RMA boundaries, is described in more detail

in Appendix Section D.1.3.4. Nearly all of the areas lacking concentration estimates were either sampled and recorded as BCRL or were not sampled because there was no historic reason for a given COC to be present. Therefore, on the risk maps, the areas lacking risk estimates are combined with areas of no risk, and identified as representing HQ < 1. Special treatment of shorebird, great blue heron, and bald eagle trophic boxes are also described in Appendix Section D.1.3.4. The selection of the exposure area radii is explained in Appendix Section C.2.4.

C.1.4.1.4 Soil Concentration Estimation Example

This section provides a detailed example (involving aldrin concentrations in soil associated with a specific cottontail sample) of how individual soil samples were used to calculate <ESC> from measured surficial soil and boring data. A trophic box-specific <ESC> value was calculated for each model block.

Figure C.1-7 shows a cottontail tissue sample location with its corresponding exposure area circle (radius of 346 feet). In this example, the exposure area overlaps two different designated sites: site CSA-1a (Shell Trenches) and CSA-1b (Complex Disposal Area South).

Soil borings used to estimate aldrin concentrations for grid cells within this exposure area are shown in Figure C.1-8. In general, soil borings both inside and outside of the given exposure area may be used to estimate grid point concentrations within the exposure area. For this example, 45 soil borings were utilized to calculate <ESC> for the exposure area centered around the single cottontail tissue sample. The measured aldrin concentration at each sample location is shown. Concentrations with an LT (less than) designation reflect BCRL data, i.e. data reported as below the certified reporting limit (CRL) value shown.

Spatial interpolation calculations were performed, as detailed in Appendix Section C.1.4.1.1, for all BCRL data points prior to modeling onto the grid. The estimates are based on the spatial interpolation algorithm described in Appendix Section C.1.4.1.1. Table C.1-1 presents results of the first two iterations of the spatial interpolation for the BCRL point indicated by a "★" on Figure C.1-8. For the first iteration, only the six closest points indicated as detections were used

for the estimation, resulting in a value of 0.4796 parts per million (ppm). As this interpolated value was lower than the CRL for this point, the CRL was replaced with the interpolated value. The remaining BCRL soil borings were replaced similarly. For the second and successive iterations, all detections and replaced BCRL points were used for estimation. A new value of 0.3568 ppm was estimated in the second iteration. Again, this value was used to replace the first-iteration value since it was lower than the original CRL. Two additional iterations were completed using the same method to produce a final result of 0.3558 ppm for the interpolated aldrin concentration at the point indicated by a "★" on Figure C.1-9.

Following the BCRL replacement process, the soil data were used to estimate a concentration for each of the grid blocks described above in Appendix Section C.1.4.1.2. Figure C.1-10 shows the grid blocks located within the cottontail exposure area and the concentrations of aldrin modeled for each block. Concentrations for each grid block were calculated individually from nearby borings using the inverse distance squared algorithm described in Appendix Section C.1.4.1.2. Grid blocks were considered to be 1 ft thick and 100 ft^2 in area.

The estimation of grid-block concentrations also used the inverse distance-squared interpolation process. However, the search radius for each grid block was varied to reflect the physical or geographical characteristics of the designated site in which it was found. In this example, concentrations for grid blocks within the Complex Disposal Area South were estimated using a 750-ft search radius, while concentrations for blocks within the Shell Trenches were estimated using a 200-ft search radius. Figure C.1-9 includes search radii circles used for estimating aldrin concentrations for the two sample grid blocks indicated with +. The vertical search radius was 1 ft in all cases. A maximum of six samples was used for any single estimation regardless of the number of samples available within the search radius. The six samples nearest to each grid block were used when more than six samples were located within the search radius.

The grid-block modeling algorithm further constrained the borings used for each block by allowing only borings within the same designated site as the grid block to be used in the estimation for that block unless an adjacent site had similar characteristics. This prevented the

use of data from one site type in estimations for a different site type, and prevented data from designated sites that are contaminated from being used in estimation of soil concentrations in areas where contamination is unlikely and vice versa. For this example, concentrations for grid blocks located within the Shell Trenches were estimated using only those samples located within the trench area, as the surrounding disposal area was determined to have different characteristics. These sample locations are shown in blue in Figure C.1-9. Similarly, estimation for grid blocks located in the Complex Disposal Area did not use data from borings located within the Shell Trenches. The effect can be seen by comparing Figures C.1-9 and C.1-10. Note that grid blocks within the southeast portion of the Shell Trenches area did not use the much higher concentration data (190 ppm) found just south of the site boundary. Although this point is within the search radius, the differences in site characteristics precluded its use. Grid block concentrations inside the Shell Trenches boundary reflect only the data samples found within the Shell Trenches area.

Table C.1-2 illustrates the hand calculation of the inverse distance-squared algorithm for one grid block. The example block is numbered in red on Figure C.1-10. Note that the hand-calculated concentration (0.841 ppm) does not match exactly the computer-calculated value (0.909 ppm) for this block in Figure C.1-10. Since the model considers the center of each grid block (1 ft by 100 ft by 100 ft) to be at a depth of 0.5 ft, distances measured by hand from Figure C.1-9 did not include the vertical component contributing to the true distance between the block and sample location used by the computer. In addition, inaccuracies in measuring distance from the Figure C.1-9, as well as rounding differences between the computer and hand calculations, contributed to the error. This illustrated calculation was applied to each block within the cottontail exposure area.

The <ESC> for aldrin that is associated with the cottontail tissue sample is indicated in Figure C.1-10. This value was calculated as the average of all aldrin concentrations for the blocks within the associated exposure area.

C.1.4.1.5 Selection of Interpolation Method

The inverse distance-squared method was chosen for the soil and sediment spatial interpolations because of its strong advantages in handling BCRL data and in accommodating the separate interpolation of soil data by site, or groups of similar sites, to avoid spatial discontinuities across sites. The three interpolation methods considered for the RMA data are described below followed by further discussion of the rationale for selecting the inverse distance squared method for the RMA data set.

Methods for Interpolation

Three common methods of spatial interpolation were considered for application to the RMA soil and sediment data: inverse distance-squared (the method selected), kriging, and Dirichlet polygons. These methods are all based on the premise of spatial continuity: two data points that are close to each other are more likely to have similar values than two data points that are far apart. Spatial continuity exists in most earth science data sets (Isaaks and Srivastava 1989). The least sophisticated interpolation method considered for application at RMA was Dirichlet tessellation. This method makes the simple assumption that the concentrations at points between measured samples are equal to the concentration of the nearest sample. This rule implies the construction of a network of polygons, each around a single sample, within which concentrations are assumed to equal the value of the sample. The inverse distance-squared and kriging methods assume that the concentrations of points between samples vary smoothly, resulting in gradients between sample locations. For these methods, the estimation incorporates information from not only the nearest neighboring sample, as is done in Dirichlet tessellation, but also other nearby samples. The estimate for a given point is based on a weighted linear combination of nearby sample concentrations.

In the inverse distance-squared method, the weights assigned to each sample are proportional to one divided by the square of the physical distance between the sample and the point to be estimated. Smaller distances imply that the sample has an increased informational value to the estimation point, which in turn implies a higher weight assigned to the particular sample concentration. In kriging the weights are dependent on the "statistical distance" between the

sample and estimation point. The statistical distance incorporates physical distance but is a more sophisticated description of the value of information because it also incorporates the spatial correlations between individual samples contributing to the estimate. For example, if a particular sample occurs near many other samples, the group of samples are spatially correlated and will, therefore, give somewhat redundant information regarding the estimation point. Assuming all else is equal, the kriging weights of each of these correlated samples will be lower than the inverse distance-squared weights. As discussed below, the incorporation of statistical distance rather than physical distance is especially important when the sampling locations are clustered. The three main advantages of kriging are the following: (1) it provides rigorous treatment of spatial correlations due to clustered data, (2) it allows uncertainty to be estimated, and (3) it allows the relationship between inter-point distance and value of information to be fine tuned for a specific data set rather than assumed a priori as in the case of the inverse distance-squared rule.

Selection of Method for RMA

Three characteristics of the RMA data set are especially pertinent to the selection of an interpolation method: (1) abundance of BCRL data, (2) spatial discontinuities caused by site phenomena such as trenches, basins, and other containments, and (3) clustered sampling. The first two characteristics are better dealt with by the inverse distance-squared method than the others.

Abundance of BCRL Data

The BCRL interpolation method discussed in Appendix Section C.1.4.1.1 was developed to make use of the spatial structure of the soil concentration data in estimating likely values for BCRL data. The incorporation of censored data into a kriging framework would be relatively complex and has not yet been published in the statistical literature. In contrast, under the basic premise of the Dirichlet tessellation approach, an estimate for a BCRL data point would based only on the value of its nearest neighbor, an overly simplistic assumption. A Thiessen polygon approach also could have been implemented with BCRLs replaced by the conditional expected values from distributions of soil concentrations in the same area. The inverse distance- squared method and

software was modified to incorporate "less than" information and provide estimates of BCRL data points.

Spatial Discontinuities

RMA soil concentrations are dependent on site-specific activities which are not necessarily spatially continuous. For example, trenches and basins create containments that hinder contaminant transport or diffusion. Because of potential discontinuities, concentrations within a given site were thought to be best estimated from only the samples occurring within this site, or in some cases within adjacent sites that had similar characteristics. Therefore, the soil concentrations were estimated by separate spatial interpolations within each site, and one overall interpolation for all areas outside the sites. The division of the data into designated sites, groups of similar and adjacent sites, and areas outside designated sites, left many data sets with relatively low sample sizes. The sophistication offered by the kriging method (e.g., fine tuning of correlation function, handling of clustered data) did not seem warranted for these small data sets. The specification of kriging parameters (e.g. correlation function) in such cases could be highly subjective. A further consideration is that kriging would require a large increase in analytical complexity and time compared to the inverse distance-squared method.

Clustered Data

RMA soil sampling was highly concentrated in the designated sites, where contamination tended to be highest, and was much less concentrated outside the designated sites. Therefore, taken as a whole, the RMA data are highly clustered, with areas of high sample density corresponding to areas of high concentrations. If data within and outside of sites were interpolated as a single data set, the following interpolation bias, referred to as a "halo effect", would arise: areas surrounding a contaminated site would receive upwardly biased estimates of concentrations from the undue influence of the more dense site data. Dirichlet polygons constructed in highly clustered data are often assigned anomalous shapes and may produce arbitrary or biased estimates. Of the three methods, Kriging minimizes this interpolation bias because it rigorously handles spatial correlation due to clustered data. (However even kriging does not rigorously handle the correlation structure formed by heterogeneity in the underlying variance of soil concentrations

likely to occur at RMA and with hot spot contamination in general.) As described above, the RMA data were not interpolated as a single data set but divided into groups of similar and adjacent sites or areas outside designated sites. Within these groupings, the sample pattern ranged from random to slightly clustered, and therefore interpolation bias was believed to be minimal for all three methods considered. In summary, the inverse distance-squared method was chosen based on the need to incorporate BCRL data points and to interpolate sites separately, and because interpolation bias from clustering was not expected to be substantial when sites were interpolated separately.

C.1.4.2 Characterization of Exposure Concentration for Aquatic Food Webs

The primary contaminant source medium for exposure of biota in the aquatic ecosystem is water. Nearly all of the RMA water sample concentrations were found to be BCRL and therefore exposure water concentrations could not be estimated directly from these samples. An attempt was made to estimate exposure water concentrations from sediment concentrations and then to derive media-based estimates of risk as follows. A single exposure area sediment concentration was estimated for each of the RMA lakes considered in the risk evaluations. These estimates were calculated by replacing BCRL sediment samples with estimates based on the expected value robust (EVR) method, applying the inverse distance-squared algorithm to interpolate sediment concentrations onto a grid (similar to the interpolation of soil data described in Appendix Section C.1.4.1), and then averaging concentrations from all grid blocks within a lake. The estimated average exposure area sediment concentrations for a given lake could then be converted to pore water concentrations, using the equilibrium partitioning concept, and from pore water concentrations to overlying water concentration, using EPA's Water Quality Analysis Simulation Program, version 4.31 (WASP4; Ambrose et al. 1988). These conversions were rejected because of the substantial uncertainty in applying the equilibrium partitioning concept for systems where the percentage of organic carbon in sediments is as high as that observed for RMA lakes (10 to 20 percent). This is the reason exposure area sediment concentrations were not used to estimate risk for trophic boxes with aquatic food chains. Instead, since exposure concentrations in aquatic ecosystems are relatively homogeneous and well represented by the tissue concentrations, aquatic

risk was calculated directly from the ratio of tissue concentration or dose to MATC or TRV, respectively (Appendix Section C.1.6).

For aquatic food chains, measured exposure tissue concentrations were occasionally missing for a particular lake, COC or trophic box. Plankton data were unavailable for East Upper Derby and Upper Derby Lakes and Rod & Gun Club Pond. Aquatic plant data were unavailable for East Upper Derby and Upper Derby Lakes. Aquatic invertebrate data were available only for Lower Derby Lake and Lake Ladora. Amphibian data were available only for Lower Derby Lake. Field data on small fish were available for Upper and Lower Derby Lakes, and Lakes Ladora and Mary. East Upper Derby Lake and Rod & Gun Club Pond were assumed not to contain small fish. Large fish data were available for Lower Derby Lake, Lake Ladora, and Lake Mary; East Upper Derby, Upper Derby, and Rod & Gun Club Pond were assumed not to contain large fish. The individuals of these aquatic trophic boxes were confined to the specific lakes where the trophic box was assumed to occur. For each lake where a trophic box was assumed to occur but lacked data, average tissue concentrations were estimated as the weighted average of the sample averages from the lakes with field data. Sample sizes reported in Appendix Section D.1, Figure D.1-10 were used as weighting factors. Using this method, data were sufficient to estimate lake-specific tissue concentrations for all trophic box/chemical combinations except DDT/DDE concentrations in aquatic invertebrates and amphibians. DDT/DDE was not detected in these trophic boxes in any of the lakes.

For DDT/DDE in aquatic invertebrates and amphibians, average tissue concentrations were calculated using the ratios of measured organochlorine pesticides (OCPs; i.e., DDE/DDT relative to aldrin/dieldrin and to endrin) in tissue found in other aquatic trophic boxes. Aquatic invertebrate (aq.invert.) DDT/DDE tissue concentrations were calculated using ratios of DDT/DDE to aldrin/dieldrin (ald/dld) and DDT/DDE to endrin tissue concentrations in plankton and aquatic plants (aq.plants) as follows:

$$[DDT/DDE]_{aq.invert} = \frac{1}{4} * \left\{ \left(\frac{[DDT/DDE]}{[ald/dld]} \right)_{plankton} + \left(\frac{[DDT/DDE]}{[ald/dld]} \right)_{aq.plants} \right\} * [ald/dld]_{aq.invert}$$

$$+ \frac{1}{4} * \left\{ \left(\frac{[DDT/DDE]}{[endrin]} \right)_{plankton} + \left(\frac{[DDT/DDE]}{[endrin]} \right)_{aq.plants} \right\} * [endrin]_{aq.invert}$$

where the OCP ratios represent the averages over all lakes having tissue concentration data (Lower Derby, Ladora, and Mary for plankton; Lower Derby, Ladora, Rod & Gun Club Pond, and Mary for aquatic plants). Amphibian DDT/DDE tissue concentrations were calculated using ratios of DDT/DDE to aldrin/dieldrin and DDT/DDE to endrin tissue concentrations in small and large fish:

$$[DDT/DDE]_{amphib} = \frac{1}{4} * \left\{ \left(\frac{[DDT/DDE]}{[ald/dld]} \right)_{sm.fish} + \left(\frac{[DDT/DDE]}{[ald/dld]} \right)_{lrg.fish} \right\} * [ald/dld]_{amphib}$$

$$+ \frac{1}{4} * \left\{ \left(\frac{[DDT/DDE]}{[endrin]} \right)_{sm.fish} + \left(\frac{[DDT/DDE]}{[endrin]} \right)_{lrg.fish} \right\} * [endrin]_{amphib}$$

The OCP ratios are averages over all lakes having tissue concentration data (Upper Derby, Lower Derby, Ladora, and Mary for small fish; Lower Derby, Ladora, and Mary for large fish). The great blue heron trophic box is more heavily impacted than the bald eagle by the missing DDE/DDT data replacement algorithms described above, so a sensitivity analysis was performed to evaluate the impact of estimation uncertainty on the great blue heron tissue concentration predictions. The results, reported in Table C.1-3, indicate that heron tissue concentration predictions, and therefore risk estimates, are insensitive to the replacement values used for missing DDE/DDT data.

The bird species with aquatic food chains (shorebird, water bird, bald eagle, and great blue heron) were assumed to be exposed to all the lakes evaluated, since they could readily fly from one to

another. Measured tissue concentrations were available from some lakes for shorebird and water bird. How these data were used to represent all the lakes is described below, as is the estimation of water concentrations to be used as dietary components for water bird and great blue heron. Tissue concentrations were not measured in the bald eagle or great blue heron, so their tissue concentrations from aquatic as well as terrestrial food chains were modeled from tissue concentrations in their prey as described in Appendix Section C.1.5.2.

Water bird data were available for all except East Upper Derby Lake, so the water bird risk characterization was based on the samples from these five lakes. Weighted averages of lake-specific water bird sample averages were calculated, averaging over all lakes except East Upper Derby Lake, and the resulting RMA-wide estimates of average tissue concentration were used to characterize risk to this trophic box. The weighting factors, given in Appendix Section D.1, Figure D.1-15, were based on the assumption that the level of predation at a lake is proportional to the assumed size of the trophic box exposure area at that lake. Exposure areas for water bird were assumed to equal the lake's water surface area.

Shorebird data were available from the vicinity of Lower Derby Lake and Lake Mary. Because this trophic box has both terrestrial and aquatic food chains, these shorebird sample averages were partitioned to attribute a portion of the observed tissue concentration to exposure through the aquatic food web, and the balance to exposure through the terrestrial food web. Partitioning coefficients, derived as detailed in Appendix Section C.1.5.1.4, are given in Appendix Section D.1, Figure D.1-8. The average shorebird tissue concentration for the other four lakes was estimated by the average of the partitioned sample averages for Lower Derby Lake and Lake Mary. For the shorebird, the feeding area was assumed to be a band around the perimeter of the lake, extending inward three feet from the shoreline (Appendix Section D.1, Figure D.1-15). The resulting RMA-wide mean tissue concentrations for water bird and shorebird are given in Appendix Section D.1, Figure D.1-16.

As an estimate of COC concentrations in the water (C_w) that was assumed ingested by the great blue heron, the certified reporting limits (CRLs) were used for aldrin/dieldrin, DLE/DDT, and

endrin, because less than 25 percent of the lake water samples had concentrations above the CRL for the contaminants evaluated. Mercury was the only bioaccumulative contaminant of concern for which measured data could be used because it was detected consistently above its CRL in water samples.

C.1.5 DEVELOPMENT OF BIOMAGNIFICATION FACTORS

As noted in Appendix Section C.1.1, BMF is defined in the IEA/RC as an empirical quantity relating an organism's tissue concentration to its ESC. Terrestrial risk was calculated using spatially distributed population mean tissue concentrations that were estimated based on three sets of site-specific BMFs derived by approaches put forth by the U.S. Army (Army), EPA, and Shell. All three approaches use the same initial data set of measured tissue concentrations and the <ESC> values paired with them. The EPA approach further screened these data. The outputs of direct calculations using these field data are referred to as BMF_{obs} (observed BMFs), which was calculated for prey. BMFs can also be estimated using a food-web model, as was done for top predators. This section describes the development of final BMFs for both prey (Appendix Section C.1.5.1) and top predator (Appendix Section C.1.5.2) trophic boxes.

C.1.5.1 Development of Final BMFs for Prey

The final BMF values used by EPA and Shell for prey were the BMF_{obs} values from their respective approaches. BMF_{obs} from the Army approach was used to derive "calibrated" Army BMFs for prey. Calibrated BMF_{obs} values were available only for prey trophic boxes because tissue concentrations were measured only for prey.

C.1.5.1.1 Implications of the Sampling Design

The sampling design for collecting tissue and soil concentration data affected whether or not they were representative of the site and determined the appropriateness of the way in which they were combined to estimate BMF_{obs}. Under ideal conditions, a sampling design for the collection of data to quantify BMF_{obs} would provide tissue samples representative of a population's exposure to the full range of soil concentrations, and soil samples collected sufficiently near the tissue sampling locations to provide a precise measure of the chemical concentrations in the area where

the organisms were exposed. Under ideal sampling conditions, variability in the tissue concentration/ESC relationship would be due primarily to processes which cannot be accounted for in the risk model (e.g., individual physiology, behavior, age, and bioavailability of contaminants in RMA soils). Under real field circumstances, however, ideal sampling goals are unlikely to be met, adding to the variability in the observed relationship between tissue concentration and ESC. Only a limited number of organisms can be collected without impacting populations. Further, an individual's true exposure area may be different from the assumed exposure area used to estimate its ESC. Accurately pairing tissue concentration and the appropriate ESC is difficult because organisms are mobile and not necessarily collected at the center of their exposure area as is assumed in estimating ESC. Further detail regarding these uncertainties is provided in Appendix Section E.12.

Because the locational association of tissue concentration and ESC added the greatest uncertainty to the calculation of BMF_{obs}, there was considerable discussion as to the appropriate way in which to use these data. Divergent opinions existed as to whether all the pairs of tissue concentration and ESC data should be used (or only a portion of them), whether the data should be used to define tissue concentration and ESC distributions (or calculate individual BMF_{obs} values), and, if tissue concentration and ESC distributions were used, how best to combine them. Three approaches to calculating BMF_{obs} emerged from these discussions: the Army's collocated-distribution approach, Shell's collocated-distribution approach, and EPA's modified paired approach. All three methods are described below. In addition, EPA has prepared a document that describes analyses it has performed and steps it would like to have followed in estimating BMF_{obs} by the EPA approach (see Appendix C.6.2). Appendix C.6.1 is a jointly prepared description of differences between the Army and EPA regarding the process and purpose of BMF estimation.

C.1.5.1.2 Approaches to Calculate BMF_{obs}

All three of the approaches used to calculate BMF_{obs} start with the same initial databases of ESC and tissue concentration. The Army and Shell collocated distribution approaches used the entire tissue concentration and ESC databases. The modified paired approach screened out tissue

concentration/ESC pairs associated with the highest concentrations of aldrin/dieldrin and endrin, as well as the onsite control samples before calculating individual BMFs.

As discussed in Appendix Section C.1.4, ESC was estimated by first interpolating the original soil boring and surficial soil sample data onto a grid. Values for sample borings that were BCRL were replaced where possible with values based on information from surrounding sample borings. At each tissue sample location, <ESC> was calculated as the average of grid point concentrations that fell within the sample's species-specific exposure area. This process is described in Appendix Sections C.1.4.1 and E.12.4.2.2. Total measured values were used for all chemicals, including those that occur naturally (mercury, arsenic, cadmium, and copper). For all trophic boxes, soil data from the 0- to 1-ft depth interval were used to calculate <ESC>; for prairie dogs, data from the 1- to 20-ft depth interval were also used to calculate <ESC>.

Tissue concentration data used in BMF_{obs} were from dressed carcass, whole body, and composite samples. For most trophic boxes, the tissue concentration data were grouped by trophic box to determine BCRL replacement. BCRLs were replaced using the robust method (Gilliom and Helsel 1986) when the number and percentage of detections for a given trophic box were sufficient; in other cases the one-half CRL method was used. In cases where paired data were required (e.g., for the modified paired approach and for summing of concentrations for aldrin and dieldrin or DDE and DDT), the expected value robust method was used in place of the robust method. Methodological details for replacing BCRL tissue concentration data and a table of sample sizes and BCRL replacement methods used for each case are provided in Appendix Section E.12.4.1.1.

For three trophic boxes (small bird, small mammal, and medium mammal), exposure areas for component species differed sufficiently to warrant initial calculation of species-specific BMF_{obs} values when data for a particular chemical were sufficient for this to be done (e.g., aldrin/dieldrin). In these cases, BCRLs were replaced and BMF distributions estimated for each species. A weighted average of these species-specific BMF_{obs} distributions was then used to represent the trophic box BMF distribution in the Army and Shell approaches.

The Army, Shell, and EPA approaches are described below, preceded by the definition of the statistical terminology used in describing them.

Statistical Terminology

Three types of statistical distributions are discussed below. These are the lognormal distribution of a given quantity X, the normal distribution corresponding to the natural logarithm of X, and the approximately normal distribution describing the estimated mean of X. The true descriptors for these distributions (e.g., mean and standard deviation) are unknown. Estimates for these descriptors are calculated from the data based on standard statistical formulas. The terminology used for these descriptors is as follows.

μ_X – the true mean of the probability distribution for X.

σ_X – the true standard deviation of the probability distribution for X.

$\hat{\mu}_X$ – an estimate of μ_x. In this section, $\hat{\mu}_X$ refers to either the arithmetic sample mean or the mean calculated using an estimator specific to the lognormal distribution.

$\hat{\sigma}_x$ – an estimate of σ_X. In this section, $\hat{\sigma}_X$ refers to either the arithmetic sample standard deviation or the standard deviation calculated using an estimator specific to the lognormal distribution.

$\mu_{ln(X)}, \sigma_{ln(X)}$ – the true mean and standard deviation of the probability distribution for the natural logarithm of X.

$\hat{\mu}_{ln(X)}$, $\hat{\sigma}_{ln(X)}$ – estimates of $\mu_{ln(X)}$ and $\sigma_{ln(X)}$. In this section, these terms refer to either the sample mean and standard deviation calculated on the log transformed values, or the estimates back calculated from the arithmetic estimators using formulas specific to the lognormal distribution. It is important to note that $\hat{\mu}_{ln(x)} \neq \ln(\hat{\mu}_x)$.

$\overline{BMF}$ – an estimation of the mean BMF. This quantity depends on the correlation between tissue concentration and <ESC> and thus the correlation between ln(TC) and ln(<ESC>).

$\hat{\mu}_{\overline{BMF}}$, $\hat{\sigma}_{\overline{BMF}}$ – the mean and standard deviation of different estimates of $\overline{BMF}$ that arise from different values of correlation or different bootstrap samples of the data.

r – the correlation between ln(TC) and ln(<ESC>).

$r_{ln(<ESC>),\ ln(BMF)}$ – the correlation between ln(<ESC>) and ln(BMF).

BMF_{obs} Calculation Methods

Army Collocated Distribution Approach

The Army collocated distributions approach implements the following steps for a particular trophic box/chemical combination:

1. Calculate an <ESC> value for each measured tissue concentration as the average concentration of all grid points in the exposure area centered at the location of the tissue sample.

2. Disassociate the pairs of tissue concentration and <ESC> data.

3. Calculate arithmetic sample average and sample standard deviation for the tissue concentration data ($\hat{\mu}_{TC}$, $\hat{\sigma}_{TC}$). Use these estimates as the mean and standard deviation of the lognormal tissue concentration distribution.

4. Calculate arithmetic sample average and sample standard deviation for the <ESC> data ($\hat{\mu}_{<ESC>}$, $\hat{\sigma}_{<ESC>}$). Use these estimates as the mean and standard deviation of the lognormal <ESC> distribution.

5. Derive the values for the parameters of ln(TC) and ln(<ESC>), which are found by rearranging Gilbert's equations 13.7 and 13.8 (1987, p. 167):

$$\ln(TC) \sim \text{Normal}(\hat{\mu}_{\ln(TC)}, \hat{\sigma}_{\ln(TC)}) \quad (2)$$

$$\ln(ESC) \sim \text{Normal}(\hat{\mu}_{\ln(<ESC>)}, \hat{\sigma}_{\ln(<ESC>)}) \quad (3)$$

$$\hat{\mu}_{\ln(TC)} = \ln(\hat{\mu}_{TC}) - \frac{\hat{\sigma}^2_{\ln(TC)}}{2} \quad (4)$$

$$\hat{\sigma}^2_{\ln(TC)} = \ln\left(\left(\frac{\hat{\sigma}^2_{TC}}{\hat{\mu}^2_{TC}}\right) + 1\right) \quad (5)$$

$$\hat{\mu}_{\ln(<ESC>)} = \ln(\hat{\mu}<ESC>) - \frac{\hat{\sigma}^2_{\ln(<ESC>)}}{2} \quad (6)$$

$$\hat{\sigma}^2_{\ln(<ESC>)} = \ln\left(\left(\frac{\hat{\sigma}_{<ESC>^2}}{\hat{\mu}_{<ESC>^2}}\right) + 1\right) \quad (7)$$

6. Calculate the mean ($\hat{\mu}_{\ln(BMF)}$) and variance ($\sigma^2_{\ln(BMF)}$) of the distribution of ln(BMF) as a function of r, the assumed correlation between ln(TC) and ln(ESC), using the definitions of expectation and variance of the differences between two normal random variables (ln(BMF) = ln(TC) - ln(<ESC>)):

$$\hat{\mu}_{\ln(BMF)} = \hat{\mu}_{\ln(TC)} - \hat{\mu}_{\ln(<ESC>)} \quad (8)$$

$$\hat{\sigma}^2_{\ln(BMF)} = \hat{\sigma}^2_{\ln(TC)} + \hat{\sigma}^2_{\ln(<ESC>)} - 2r\hat{\sigma}_{\ln(TC)}\hat{\sigma}_{\ln(<ESC>)} \quad (9)$$

where:

r = assumed (ln(TC), ln(<ESC>)) correlation.

Note that the assumed correlation between ln(TC) and ln(<ESC>) results in a correlation between ln(<ESC>) and the estimates of ln(BMF). That is, while BMF is treated as a constant when estimating risk, estimates of BMF based on observed data are dependent on <ESC> in cases where the correlation between the TC and <ESC> distributions is not high.

7. Compute $\overline{BMF}_{obs}$ as a function of r:

$$\overline{BMF}(r) = \exp\left(\hat{\mu}_{\ln(BMF)} + \frac{\hat{\sigma}^2_{\ln(BMF)}}{2}\right) \quad (10)$$

where:

r enters through $\hat{\sigma}^2_{\ln(BMF)}$. Values for r are drawn from a triangular distribution with limits of 0.3 and 0.7 and a most likely value of 0.5. A total of 100 Latin hypercube samples were drawn from the triangular distribution of r using the Excel/@RISK software program. Each r sample was used in equations (9) and (10) to estimate a value for $\overline{BMF}_{obs}$, resulting in 100 estimates of $\overline{BMF}_{obs}$.

8. Calculate the sample average and sample standard deviation for the $\overline{BMF}_{obs}$ sample:

$$\hat{\mu}_{\overline{BMF}_{obs}} = \frac{\sum_{i=1}^{100} \left(\overline{BMF}_{obs}\right)_i}{100} \quad (11)$$

$$\hat{\sigma}_{\overline{BMF}_{obs}} = \sqrt{\frac{\sum_{i=1}^{100} \left[\left(\overline{BMF}_{obs}\right)_i - \hat{\mu}_{\overline{BMF}_{obs}}\right]^2}{99}} \quad (12)$$

9. Estimate the distribution of $\overline{BMF}_{obs}$ as:

$$\overline{BMF}_{obs} \sim N\left(\hat{\mu}_{\overline{BMF}_{obs}}, \hat{\sigma}_{\overline{BMF}_{obs}}\right) \quad (13)$$

These computations were performed in a batch mode for all chemicals and trophic boxes. Equations were coded in S-Plus, rather than Excel/@RISK, to facilitate batch calculation.

For cases in which species-specific BMFs were calculated, a weighted average of the species-specific distributions was used to define the trophic box (group) distribution, as in equations (14) and (15):

$$\hat{\mu}_{BMF,MDMML} = \hat{\mu}_{BMF,cylu} w_1 + \hat{\mu}_{BMF,syau} w_2 \quad (14)$$

$$\hat{\sigma}^2_{BMF,MDMML} = \hat{\sigma}^2_{BMF,cylu} w_1^2 + \hat{\sigma}^2_{BMF,syau} w_2^2 \quad (15)$$

The weights for each species were as follows: small mammal (deer mouse [PEMA], 0.90; thirteen-lined ground squirrel [SPTR], 0.10), medium mammal (prairie dog [CYLU], 0.72; desert cottontail [SYAU], 0.28), and small bird (vesper sparrow [AOGR], 0.10; western meadowlark [STNE], 0.20; mourning dove [ZEMA], 0.70).

The expected value of the BMF_{obs} distribution for each trophic box/chemical combination calculated by the Army collocated distributions approach is used as the initial value in the Army BMF calibration process, described below in Appendix Section C.1.5.1.6.

Shell Collocated Distribution Approach

Shell's approach is similar to the Army's approach except that different estimators are used in Steps 1 through 4 and different steps are applied subsequent to Step 4.

1. Same as the Army's approach.
2. Same as the Army's approach.
3. Calculate the arithmetic sample mean and standard deviation of the log-transformed tissue concentration data.
4. Calculate the arithmetic sample mean and standard deviation of the log-transformed <ESC> data.
5. Estimate the distribution parameters for $\ln(BMF_{obs})$ using the following equations:

$$\hat{\mu}_{\ln(BMF_{obs})} = \hat{\mu}_{\ln(TC)} - \hat{\mu}_{\ln(<ESC>)} \quad (16)$$

$$\hat{\sigma}^2_{\ln(BMF_{obs})} = MAX((\hat{\sigma}^2_{\ln(TC)} - \hat{\sigma}^2_{\ln(<ESC>)}), 0) \quad (17)$$

Equation (16) is equal to equation (8) of the Army's approach. Equation (17) reflects the assumption that the correlation between <ESC> and BMF equals zero. (Under this assumption, equation (17) is consistent with equation (19).)

6. Estimate the mean BMF_{obs} based on the following property of lognormal distributions:

$$\overline{BMF_{obs}} = EXP(\hat{\mu}_{\ln(BMF_{obs})} + 0.5 * \hat{\sigma}^2_{\ln(BMF_{obs})}) \quad (18)$$

Note that the basic assumption of equation (17) is that the correlation between $\ln(BMF_{obs})$ and ln(<ESC>) is zero, which also implies that the correlation between ln(TC) and ln(<ESC>) is positive and is given by the following equation:

$$\text{Correlation } (\ln(TC), \ln(<ESC>)) = \frac{\hat{\sigma}_{\ln(<ESC>)}}{\hat{\sigma}_{\ln(TC)}} \quad (19)$$

7. Estimate the distribution of the arithmetic mean BMF_{obs} based on bootstrap resampling of the tissue concentration values and estimated ESCs. (Bootstrap resampling is described in Noreen 1989.) This procedure includes five steps (a through e):

   a. Draw N pairs {TC, <ESC>} of data randomly from their individual data sets with replacement, where N is the sample size for the trophic box or species.

   b. Disassociate these pairs {TC, <ESC>}, then calculate the mean and standard deviation for the TC and <ESC> distributions.

$$\hat{\mu}_{\ln(TC)} = \frac{\Sigma \ln(TC)}{N} \quad (20)$$

$$\hat{\sigma}_{\ln(TC)} = \text{Std. Dev. } (\ln(TC)) \quad (21)$$

$$\hat{\mu}_{\ln(ESC)} = \frac{\Sigma \ln(<ESC>)}{N} \quad (22)$$

$$\hat{\sigma}_{\ln(ESC)} = \text{Std. Dev. } (\ln(<ESC>)) \quad (23)$$

c. Calculate $\overline{BMF_{obs}}$ using equations (16), (17) and (18).

d. Repeat Steps a through c 1,000 times to obtain 1,000 estimates of $\overline{BMF_{obs}}$.

e. Calculate the final distribution for μ_{BMFobs} as follows:

$$\hat{\mu}_{\overline{BMF_{obs}}} = \frac{\sum_{i=1}^{1000} (\overline{BMF_{obs}})_i}{1000} \quad (24)$$

$$\hat{\sigma}_{\overline{BMF_{obs}}} = \text{Std. Dev. } (\overline{BMF_{obs}}) \quad (25)$$

where:

$\overline{BMF_{(obs)_i}}$ = i th bootstrap estimate.

When BMF_{obs} was calculated for individual species rather than a trophic box, a weighted average of species-specific values was used for the trophic box as was done in the Army's approach. The expected value of the BMF_{obs} distribution for each trophic box/chemical combination calculated by the Shell collocated distributions approach is used as the final Shell BMF value for that combination.

EPA Modified Paired Approach

The EPA approach uses a subset of the same paired tissue concentration and <ESC> data used in the Army and Shell approaches. Appendix Section C.6.1 contains additional information on the EPA approach, including differences between the EPA and Army approaches. During the process to screen data from this subset, soil data were used to identify areas of RMA from which tissue concentration and <ESC> data pairs would not be used. The purpose of screening data in the modified paired approach was to remove the pairs of tissue concentration and <ESC> data that were associated with particularly high and particularly low soil concentrations. This exclusion of data from the tails of the soil distribution was intended to facilitate a more precise estimate of the mean BMF_{obs} in the range of contaminant levels that are most uncertain with regard to the need for cleanup. The following steps were performed during the screening procedure:

1. Remove from the data set biota samples that fell within a boundary line located 50 ft outside the 10 part per million (ppm) contours for total "drin" (i.e., aldrin plus dieldrin plus endrin) concentrations in soil (i.e., remove tissue concentration data from areas where total "drin" in soil exceeded 10 ppm and from a buffer zone of 50 ft outside the 10-ppm contours). Do this in the five following steps ((a) through (e)):

    a. Develop the boundary and 50-ft contours based on interpolated soil data at each grid point (not actual borings and not <ESC> values) and the sum of concentrations of "drins" (aldrin, dieldrin, and endrin) at each grid point.

b. Display the boundaries and contours on a map, with each contour identified individually (with a letter or number).

c. Add symbols at the collection locations for samples of the following species: earthworm, deer mouse, ground squirrel, prairie dog, cottontail, vesper sparrow, meadowlark, mourning dove, and shorebird. Note that four species of plants and two insect groups occurred at the same locations as the deer mice.

d. Identify (and list for exclusion) all tissue concentration samples that are within the boundary around each designated contour polygon.

e. In cases where very small contour polygons (<5 hectare [ha]) contain tissue concentrations to be excluded, do not exclude them if their associated exposure area is larger than the polygon. (This rule resulted in keeping some mourning dove samples in the data set because small polygons of high concentration (> 10-ppm) made up only a small portion of the individuals exposure area.

2. Remove data collected in control areas at RMA from the tissue concentration database.

3. After the screening process, calculate sample-specific BMF_{obs} values directly as the ratio of tissue concentration/<ESC> for each remaining data pair. Calculations were done for the same trophic box/chemical (or species/chemical) combinations as in the Army and Shell approaches.

   Trophic-box-specific (or species-specific) arithmetic mean BMF_{obs} distributions were defined using the sample arithmetic mean and the sample standard error.

$$\hat{\mu}_{BMF} = \frac{\sum_{i=1}^{N} BMF_i}{N} \tag{26}$$

$$\hat{\sigma}_{\overline{BMF}} = \frac{\text{Std. Dev. (BMF)}}{N^{1/2}} \tag{27}$$

where:

BMF_i = BMF for individual i.

Screening resulted in a reduction of the number of data points by 0 to 54 percent, depending on the trophic box/chemical combination. The smallest data sets (n=6) were not reduced during the screening process. Further discussion of the results of screening is provided in E.12.5.

The expected value of the BMF_{obs} distribution for each trophic box/chemical combination calculated by the EPA modified paired data approach is used as the final EPA BMF value for that combination.

C.1.5.1.3 Rationale for the Three BMF_{obs} Calculation Approaches

The three approaches for calculating BMF_{obs} were developed in response to estimation problems implied by a lack of correlation between the tissue concentrations and associated ESC estimates. Error associated with random sampling reduces the correlation in paired data and imparts an upward bias to estimation of the mean BMF.

Random error affects the estimation of BMF as follows. Sample correlations and graphical analysis of the tissue concentration and <ESC> data indicated a general lack of correlation or, in some cases, a negative correlation, which indicates that <ESC> is not fully representative of the exposure resulting in the tissue concentrations and/or that other factors may have affected tissue concentrations (e.g., off-site exposure for migrant or dispersing individuals). Under ideal collocation of samples, the error in interpolating soil concentrations and in associating the appropriate exposure area with each tissue sample would be minimal. However, even under such ideal collocation, the correlation between tissue concentration and ESC may be relatively low because ESC does not account for individual variability in true exposure due to such factors as physiology, behavior, age, and bioavailability processes in different soils. Variability in the tissue concentration/ESC relationship from these factors appropriately influences both the true mean BMF and the sample mean and so does not impart a bias. In contrast, random error in measuring tissue concentration and ESC, and in assigning a specific exposure area to a specific individual, reduces the sample correlation, increasing the sample mean of the individual BMFs but not influencing the true mean. Therefore, this random error imparts a positive bias to the BMF estimation if the paired approach is used. Random error is always present in sampling and does

not impart a bias to the estimation of the mean for the sampled populations, i.e., the tissue concentration and <ESC> distributions in this case. The bias arises in the estimation of a constructed variable (not sampled directly) such as the ratio of tissue concentration to <ESC> for each pair.

The Army and Shell approaches attempt to avoid this type of bias in BMF estimation by working with tissue concentration and <ESC> distributions, while the EPA approach attempts to minimize the degree of bias by screening the data before working with the constructed values, the individual BMFs. The rationales for these methods are based on considerations of data set screening, collocation assumptions, correlation assumptions, estimation of uncertainty, and application of arithmetic vs logarithmic estimators. Major differences in rationale are discussed below.

Data Set Screening

EPA's approach attempted to reduce variability and nonlinearity in the tissue concentration/<ESC> relationship by discarding parts of the data set associated with extreme soil concentrations. A successful reduction in variability and nonlinearity would tend to increase the correlation and thus reduce the upward bias in estimating the BMF from the paired data. The Army and Shell approaches involved correlation assumptions (discussed below) that reduced the dependency of the BMF estimates on these sources of bias. The Army and Shell collocated distributions approaches imply the underlying assumption that the advantages of screening are outweighed by the disadvantages (lower sample sizes) and therefore screening is not performed for these methods.

Collocation Assumptions

The EPA approach is based on the assumption that each tissue sample and associated ESC estimate are accurately collocated and therefore together provide an independent random sample of BMF. The Army and Shell approaches are based on the assumption that tissue concentration and <ESC> are random samples from collocated distributions; that is, the group of ESC estimates are assumed to be representative of the true exposure area concentrations which gave rise to the

group of tissue concentrations. This assumption is reasonable whether or not the individual samples are accurately paired (individually collocated). In any case where individual samples are collocated, the collocated distribution assumption is met as well. However, the converse is not true. Collocated distributions can be achieved from sampling schemes that result in noncollocated individual samples. For example, if ESC and tissue concentration estimates were drawn randomly from the same general vicinity (no collocated pairing), the distribution of tissue concentrations would be expected to be dependent on the distribution of <ESC> samples. In such a case, the mean BMF would depend on the unknown correlation of the collocated distributions. Because the sampling design underlying the data used in the IEA/RC was not random and attempted to collocate individual tissue concentrations and <ESC>s, the lack of correlation in the data implies uncertainty in both of the assumptions of collocation (pairwise and distributional) used in the three approaches.

If the data are appropriately paired, then an accurate estimate of the mean BMF is provided by utilizing this pairing (i.e., the EPA approach). However, if the data are not appropriately paired, the EPA approach will impart an upward bias to the estimation of the mean BMF, while the Army and Shell distribution approaches may impart either a negative or positive bias, depending on the extent to which their assumptions are realistic for the data set at hand.

Correlation Assumptions

EPA's approach assumes that the data pairing an observed tissue concentration (TC_{obs}) with a "predicted <ESC>" (possibly containing "location error") provides appropriate information on the relationship between TC_{obs} and "estimated actual ESC" (<ESC> without location error). Location error is the error associated with the assumption that tissue samples were taken at the center of the sampled organism's home range. "Predicted <ESC>" is the <ESC> estimate centered at the location where an organism is sampled. "Estimated actual ESC" is the <ESC> concentric with the organism's (unknown) home range.

The Army and Shell approaches assume that the predicted <ESC> and TC_{obs} data are inaccurately paired in the sense that the predicted <ESC> paired with a TC_{obs} contains location error, and

therefore the relationship between the TC_{obs} and actual <ESC> distributions cannot be estimated based on the paired sample data which typically have correlations near zero. If this is the case, then the estimation of the mean BMF will be biased upward. To avoid this suspected bias, the Army and Shell approaches make assumptions regarding the correlation between the variables TC_{obs}, <ESC>, and BMF.

As shown below, any assumption regarding the correlation of TC_{obs} and <ESC> has implications regarding the correlation of <ESC> and BMF, and visa versa. The Army approach restricts the correlation between $\ln(TC_{obs})$ and ln(actual <ESC>) to what the Army assumed to be a plausible range of values. The assumption used in the Army approach implies that BMF and <ESC> are correlated. The rationale for assuming non-zero correlation between BMF and <ESC> is that the IEA/RC estimates risk under the constraint that the true mean BMF and TC_{pred} have zero correlation. (The correlation of BMF and <ESC> implies non-zero correlation between BMF and TC_{obs}. Non-zero correlation between BMF and TC_{obs} is needed to obtain zero correlation between BMF and TC_{pred}, because of errors in the measured values TC_{obs} as estimates of the spatially distributed population mean tissue concentration.) The assumption used in the Shell approach assumes that BMF and <ESC> are uncorrelated and that the estimates of BMF obtained from the available <ESC> data are appropriate for estimating population mean tissue concentrations at RMA. The rationale for assuming zero correlation between BMF and <ESC> is that the IEA/RC estimate risk under the assumption that the true mean BMF and <ESC> have a zero correlation.

The Army and Shell approaches for calculating the variance of $\ln(BMF_{obs})$, and from this the mean BMF_{obs}, both make assumptions about the correlations between TC_{obs}, <ESC>, and BMF_{obs}. They are derived using standard statistical theory from two forms of the same general equation, the first relating TC_{obs}, <ESC>, and BMF_{obs}:

$$BMF_{obs} = \frac{TC_{obs}}{\langle ESC \rangle}$$

$$\ln(BMF_{obs}) = \ln(TC_{obs}) - \ln(\langle ESC \rangle)$$

$$\sigma^2_{\ln(BMF_{obs})} = \sigma^2_{\ln(TC_{obs})} + \sigma^2_{\ln(\langle ESC \rangle)} - 2 \cdot \rho_{(\ln(TC_{obs}),\ln(\langle ESC \rangle))} \cdot \sigma_{\ln(TC_{obs})} \cdot \sigma_{\ln(\langle ESC \rangle)} \qquad (28)$$

and the second relating BMF_{obs}, TC_{pred}, and <ESC>:

$$
\begin{aligned}
TC_{pred} &= BMF_{obs} \cdot \langle ESC \rangle \\
\ln(TC_{pred}) &= \ln(BMF_{obs}) + \ln(\langle ESC \rangle) \\
\sigma^2_{\ln(TC_{pred})} &= \sigma^2_{\ln(BMF_{obs})} + \sigma^2_{\ln(\langle ESC \rangle)} + 2 \cdot \rho_{\ln(BMF_{obs}),\ \ln(\langle ESC \rangle)} \cdot \sigma_{\ln(BMF_{obs})} \cdot \sigma_{\ln(\langle ESC \rangle)} \\
\sigma^2_{\ln(BMF_{obs})} &= \sigma^2_{\ln(TC_{pred})} - \sigma^2_{\ln(\langle ESC \rangle)} \ 2 \cdot \rho_{\ln(BMF_{obs}),\ \ln(\langle ESC \rangle)} \cdot \sigma_{\ln(BMF_{obs})} \cdot \sigma_{\ln(\langle ESC \rangle)}
\end{aligned}
\tag{29}
$$

If one assumes that:

$$\sigma^2_{\ln(TC_{obs})} = \sigma^2_{\ln(TC_{pred})} = \sigma^2_{\ln(TC)}$$

and

$$\rho_{\ln(TC_{obs}),\ \ln(\langle ESC \rangle)} = \rho_{\ln(TC_{pred}),\ \ln(\langle ESC \rangle)} = \rho_{\ln(TC),\ \ln(\langle ESC \rangle)}$$

then equations (28) and (29) are equivalent and therefore indicate that any assumption regarding the values of the correlation between BMF_{obs} and <ESC> implies a formula for the correlation between <ESC> and TC_{obs}, and visa versa. In particular, the Shell method assumes:

$$\text{Relationship(i)} \quad \rho_{\ln(BMF_{obs}),\ \ln(\langle ESC \rangle)} = 0$$

which implies:

$$\text{Relationship(ii)} \quad \rho_{\ln(TC),\ \ln(\langle ESC \rangle)} = \frac{\sigma_{\ln(\langle ESC \rangle)}}{\sigma_{\ln(TC)}}$$

The equivalency of the implications of these relationships can be seen by substituting them into the respective forms of the equation for the variance of $\ln(BMF_{obs})$, that is, relationship (i) is applied using the form given in equation (29) and the relationship (ii) is applied using the form given in equation (28). With these substitutions made, both equations simplify to:

$$\sigma^2_{\ln(BMF_{obs})} = \sigma^2_{\ln(TC)} - \sigma^2_{\ln(\langle ESC \rangle)} \tag{30}$$

subject to the constraint that

$$\sigma^2_{\ln(BMF_{obs})} \geq 0 \qquad (31)$$

Equations (30) and (31) result in equation (27) used in the Shell approach. The variance of $\ln(BMF_{obs})$ can be calculated from either formula by plugging in the assumed variance of zero for $\rho_{\ln(BMFobs),\ \ln(<ESC>)}$ into equation (29) or by estimating $\rho_{\ln(TC),\ \ln(<ESC>)}$ using relationship (ii) and applied using the form given in equation (28). The motivation behind Shell's approach is that if BMF_{obs} and <ESC> are assumed to be independent when risk and TC predictions are made, then BMF_{obs} and <ESC> should be treated as independent when the mean BMF_{obs} is estimated.

The Army method assumes values for the correlation between $\ln(TC_{obs})$ and $\ln(<ESC>)$ and therefore simultaneously implies a relationship for the correlation between $\ln(<ESC>)$ and $\ln(BMF_{obs})$:

$$\rho_{\ln(BMF_{obs}),\ \ln(<ESC>)} = \frac{\rho_{\ln(TC_{obs}),\ \ln(<ESC>)}\sigma_{\ln(TC_{obs})} - \sigma_{\ln(<ESC>)}}{\sigma_{\ln(BMF_{obs})}}$$

The motivation behind the Army's approach is that if BMF and TC_{pred} are defined to be independent, then the correlation between BMF_{obs} and <ESC> should be non-zero. This is because BMF_{obs} is calculated from <ESC> and TC_{obs}, and if BMF_{obs} and TC_{obs} have non-zero correlation, then BMF_{obs} and <ESC> also must have non-zero correlation in order for BMF_{obs} and TC_{pred} to be independent.

Estimation of Uncertainty

The uncertainty in estimating mean BMF from the field data from two interrelated sources: uncertainty of the degree of representativeness of tissue concentration and <ESC> distributions, and uncertainty in the appropriate correlation between these distributions. The Army's approach accounts for these interrelated uncertainties by allowing the correlation between tissue concentration and <ESC> to vary according to a distribution of plausible values, and through its

calibration process. The variation in the correlation results in variation in the estimated mean BMFs. The adjustment of the expected values of the BMF distributions implicitly modifies the underlying tissue concentration, <ESC>, and {tissue concentration, <ESC>} correlation assumptions. Shell's approach accounts from the uncertainty in representativeness and correlation by focusing on the sampling variability in tissue concentration and <ESC> data. It uses bootstrap re-sampling to incorporate sampling variability, which in turn produces variation in the tissue concentration/<ESC> correlation and mean BMF. EPA's approach accounts for the uncertainties of representativeness and correlation by incorporating the sampling variability in the BMF "samples", i.e., the individual tissue concentrations divided by their associated <ESC>s.

Application of Arithmetic vs. Logarithmic Sample Estimators.

The three BMF approaches used different statistical formulas to estimate means and standard deviations of the data distributions. Arithmetic estimators were applied by the Army and EPA approaches, while logarithmic estimators were applied by the Shell method. These different estimators had a large impact on the resulting BMFs, as described in Appendix Section E.12.5. Although both estimators are unbiased for lognormally distributed data, one or both of the estimators were apparently biased for the RMA data due to nonlognormality and the high proportions of BCRL data points. The potential bias is indicated by the fact that the arithmetic estimators tended to produce lower means and variances for the RMA data than the logarithmic estimators. The advantages and disadvantages of these different estimators are discussed in Appendix Section E.12.5. The rationale for selecting the estimators for each method is described below.

The Army's approach uses arithmetic sample estimators, i.e., the mean and variance of the untransformed tissue concentration and <ESC> data, to estimate the parameters of the lognormal distribution. Arithmetic estimators were used in the Army's approach because of its reliance on the robust method for handling BCRL data points. This method estimates replacement values for the BCRLs based on the assumption of lognormality. The main advantage of the robust method for handling BCRLs is that these replacement values can be transformed back to linear space and arithmetic estimators applied, reducing the reliance on the assumption that the entire

data set is lognormal (Gilliom and Helsel 1986). Therefore, the use of arithmetic estimators was considered part of the robust estimation methodology. This same rationale for using arithmetic estimators, even when the data sets are skewed, was applied in the human health risk characterization (HHRC) to estimate C_{rep} because of the high frequency of BCRLs.

Shell's approach used logarithmic sample estimators, i.e., the mean and variance of the log-transformed tissue concentration and <ESC> data. These estimators are the maximum likelihood estimations (MLEs) of the log transformed tissue concentration and <ESC> distributions. These MLEs were used because the parameters of the BMF distributions are functions of the log transformed parameters.

The EPA used arithmetic estimators applied to the BMF estimate associated with each tissue concentration; there was no strong rationale for the use of arithmetic rather than logarithmic estimators. Both sets of estimators have little or no bias for both the mean and standard deviation if the data set is a random sample from a lognormal distribution.

C.1.5.1.4 Special Cases

The shorebird trophic box presented a special case in estimating BMF because it had additive aquatic and terrestrial food-web components, and also tissue data that could be used to calibrate the model. Therefore, measured tissue concentrations in shorebirds were partitioned into aquatic and terrestrial components before being used to estimate the terrestrial BMF_{obs} and to directly calculate aquatic HQs. Estimation of BMF_{obs} involved the following steps: (1) development of tissue partitioning constants; (2) calculation of partitioned tissue concentrations, $TC_{obs,\,AQ}$ and $TC_{obs,TR}$, for each of the 10 tissue samples; and (3) calculation of BMF_{obs} based on $TC_{obs,TR}$ and the <ESC>s associated with the 10 shorebird samples. The first two steps are described in more detail below. The final step was conducted in accordance with each of the three BMF_{obs} approaches as described above. (The partitioned $TC_{obs.AQ}$ values from step 2 were used to directly calculate aquatic HQs.)

The tissue partitioning constants were developed based on the following rationale. The true partitioning fractions (i.e. the true proportions of dose due to aquatic and terrestrial components) vary for individuals according to the aquatic and terrestrial doses they receive. Doses in turn vary according to the individuals location and exposure media concentrations. However, the accuracy of predicting dose for a given sample individual was not sufficient to warrant the calculation of individual partitioning constants. Therefore, a specific set of partitions was calculated for each of the two areas from which tissue samples were taken: area 1, between Lake Ladora and Lake Mary and area 2, between Upper and Lower Derby Lakes.

The tissue partitioning constants (TPC) for shorebird area k were calculated as follows:

$$TPC_{TR,k} = \frac{T_k}{(T_k + A_k)} \tag{32}$$

$$TPC_{AQ,k} = \frac{A_k}{(T_k + A_k)} \tag{33}$$

where:

T_k	=	$BMF_{trplt,k}*C_{soil,k}*FR_{trplt} + BMF_{insct,k}*C_{soil,k}*FR_{insct}$
A_k	=	$TC_{aqinv,k}*FR_{aqinv} + C_{sed,k}*FR_{sed}$
aqinv	=	aquatic invertebrates
sed	=	sediment
trplt	=	terrestrial plants
insct	=	terrestrial insects
$C_{sed,k}$	=	average sediment concentration in area k
$C_{soil,k}$	=	average soil concentration in area k

Note that it is not necessary that A_k and T_k include feeding rate (R_k) since the term cancels in the TPC ratio. For the prey BMFs, BMF_{trplt} and BMF_{insct}, the Army calibrated model BMFs (BMF_{Army}) were used. (For terrestrial plants and insects, the EPA and Shell BMFs are larger than the Army BMFs and, therefore, would have implied a larger terrestrial tissue concentration and

TPC_{TR}, and a lower aquatic tissue concentration and TPC_{AQ}.) FR values were as derived in Appendix Section C.2.2. $C_{sed,k}$ and $C_{soil,k}$ were calculated as the area-weighted average sediment and soil concentrations, respectively, within a given shorebird area. The shorebird areas were defined so that they equaled the number of shorebird tissue samples multiplied by the size of the shorebird exposure area.

Once the tissue partitioning constants were derived, each of the shorebird tissue concentrations was multiplied by the associated tissue partitioning constants to partition it into terrestrial and aquatic components:

$$TC_{obs,TR} = TC_{obs} * TPC_{TR} \quad (34)$$

$$TC_{obs,AQ} = TC_{obs} * TPC_{AQ} \quad (35)$$

The terrestrial component was then used in the calculation of BMF_{obs} for shorebird.

C.1.5.1.5 Development of $BMF_{lit/model}$

This section and Appendix Section C.1.5.1.6 describe the final two steps in developing the Army's calibrated BMF (BMF_{Army}). These steps were designed to integrate field and literature data to obtain a "most informed" estimate of BMF. While BMF_{obs} represents a BMF estimate based entirely on field data, it was not, in general, possible to calculate a literature-dependent BMF for a given trophic box based entirely on literature data for that box. Any literature-dependent estimate of BMF is influenced by uncertainties or errors in the BMFs of prey items. Therefore it is reasonable to reduce uncertainty in the prey BMFs, to the extent possible, before applying the literature BAF to estimate BMF. For this reason, $BMF_{lit/model}$ was defined as that value implied by the literature BAFs and the final calibrated BMFs (BMF_{Army}) for prey trophic boxes, i.e.:

$$\overline{BMF}_{lit/model(k)} = \overline{BAF}_{lit(k)} * \sum_j FR_{k,j} * \overline{BMF}_{Army(j)} \quad (36)$$

where:

$\overline{BMF}_{lit/model(k)}$ = Mean biomagnification factor distribution for predator k predicted by the terrestrial food-web model

$\overline{BMF}_{Army(j)}$ = Weighted average of $\overline{BMF}_{lit/model}$ and $\overline{BMF}_{obs}$ for prey j

$BAF_{lit(k)}$ = Mean bioaccumulation factor distribution for predator k derived from literature values

$FR_{(k, j)}$ = Fraction of predator diet contributed by trophic box j

k = Predator trophic box index variable

j = Prey trophic box index variable

BMF_{Army}, which represents the most informed combination of $BMF_{lit/model}$ and BMF_{obs}, is developed during calibration.

C.1.5.1.6 Development of Calibrated BMF

BMF_{Army} was the final BMF used for prey trophic boxes in the Army's approach. It was developed through a calibration procedure that evaluated two estimates of BMF, BMF_{obs} and $BMF_{lit/model}$, as well as intermediate values, and from these selected a single BMF considered to provide the best prediction of measured tissue concentrations. The values of BMF_{obs} and $BMF_{lit/model}$ represented both field and literature data and therefore were considered to provide the range of likely values for BMF. Specifically, BMF_{Army} was defined as the weighted average of BMF_{obs} and $BMF_{lit/model}$:

$$\overline{BMF}_{Army} = w * \overline{BMF}_{obs} + (1-w) * \overline{BMF}_{lit/model} \quad (37)$$

$$w = \{0, 0.1, 0.2, ..., 0.9, 1.0\} \quad (38)$$

For those trophic boxes having field data appropriate for calculating BMF_{obs} (terrestrial plant, worm, insect, small bird, small mammal, medium mammal, herptile, and shorebird), relative weights for BMF_{obs} and $BMF_{lit/model}$ were assigned individually.

The goal in calibrating BMF was to select a value that was able to reproduce site phenomena; however, the ambiguity in interpreting the relationship between RMA soil and tissue concentration data precluded the use of a single numerical criterion to evaluate alternative BMFs. For example, the BMF that best predicts the measured tissue concentrations from the associated ESC estimates differs considerably for a regression slope approach (described below), and the paired data and collocated distributions approaches. Since the paired data and collocated distribution approaches were used to estimate BMF_{obs} they would provide a circular (unfair) criteria for calibrating BMF; each would always select the BMF_{obs} developed under that particular approach. The paired data and collocation distribution approaches also do not incorporate information contained in the RMA database regarding spatial structure. For these reasons, the calibration protocol evaluated RMA data using two very different types of analyses: the evaluation of tissue concentration versus <ESC> scatter plots, and the spatial comparison of predicted population mean tissue concentrations and observed individual tissue concentrations (using GIS maps). Both analyses relied on professional judgement rather than a numerical criterion for reasons discussed below.

Tissue Concentration vs <ESC> Scatter Plots

Tissue concentrations were compared to <ESC> using scatter plots and professional judgment. The conventional criterion pertaining to a regression analysis of the scatter plots (constrained through the origin) could not be used for reasons discussed below.

The tissue concentration vs <ESC> scatterplots (where tissue concentration and <ESC> were plotted on the Y and X axis, respectively) were used to qualitatively assess the frequency and magnitude of over and underpredictions of measured tissue concentrations. The tissue concentration vs <ESC> scatter plots were visually inspected for each trophic box/chemical combination and predictions of tissue concentration were evaluated relative to the prescribed relationship between tissue concentration and <ESC>:

$$TC = \overline{BMF} * \langle ESC \rangle \qquad (39)$$

where:

TC = biota tissue concentration
<ESC> = estimated exposure area soil concentration

Equation (39) is a formal statement of the assumption that the relationship between tissue concentration and <ESC> is linear, and that when <ESC> equals 0, tissue concentration equals 0. The slope of a straight line through the origin of an {<ESC>, TC} scatterplot is, by equation (39), the BMF. $BMF_{lit/model}$ and BMF_{obs} appear on the {TC, <ESC>} scatterplot as the slopes of two lines through the origin. Equations (37) and (38) define the slope of any line through the origin of the {<ESC>, TC} scatterplot over the range bounded by BMF_{obs} and $BMF_{lit/model}$. Only slopes within this range were considered as admissible values for BMF_{Army}.

If the model equation (39) described the true relationship between the variables ESC and tissue concentration and also the relationship between the paired sample data, a BMF selected by a constrained least squares regression criterion would provide the best estimates of tissue concentrations and risks. In fact, the scatterplots strongly indicated that the paired {<ESC>, TC} data did not follow this prescribed relationship and often had values distributed along the two axes (i.e., high tissue concentrations paired with very low <ESC>s and very low tissue concentrations paired with high <ESC>s). In such cases, the standard least squares regression fit through the origin ignores the values distributed along the y axis near x equals 0 and results in relatively low BMFs. Therefore, this criterion, although considered, was not often used to select a BMF. Instead, a set of eleven candidate BMFs were considered using the tissue

concentration map evaluation protocol described below. The set of BMFs were bounded by $BMF_{lit/model}$ and BMF_{obs} and given by equations (37) and (38).

Tissue Concentration Map Evaluations

A qualitative protocol was developed to evaluate tissue concentration maps. The qualitative protocol was used because the tissue concentration map evaluation depended on nuances of spatial pattern of contamination that could not be adequately characterized by a single numerical criterion.

For a given trophic box/chemical combination, one or more BMFs (each representing a different weighting of BMF_{obs} and $BMF_{lit/model}$) were evaluated using maps of predicted and observed tissue concentration. Maps were produced that displayed the surface of predicted tissue concentrations implied by a given BMF, overlain by symbols indicating the location of measured tissue samples. The concentration class (e.g., 0.1 - 0.5 ppm) was indicated by the color of the predicted tissue concentration (TC_{pred}) surface or the measured tissue concentration (TC_{obs}) symbol so that substantial differences in concentration could be readily identified. Based on these maps, measured tissue concentrations were compared to tissue concentration predictions in the vicinity of each sample location. This information about spatial structure in the data is extremely important in the assessment of tissue concentration predictions because biomagnification in mobile organisms is inherently a spatial process (e.g., the size, shape, habitat characteristics, location, and other spatial properties of hot spots- not just the concentration- determines exposure).

The tissue concentration map analysis follows from the premise that the model used to estimate risk (whether based on literature or site data) should ideally be able to reproduce phenomena that are observed on the site. The strong advantage of evaluating mapped tissue and estimated exposure area soil concentrations is that they reflect minimal assumptions and are therefore less disputable than the (presumably) paired tissue concentration and <ESC> data used in the scatter plots. (Assumptions regarding BCRLs, exposure area size, and spatial interpolation are involved in processing the maps; however, these assumptions cannot be avoided and permeate any use of

the field data.) The map comparison displays the relatively accurate tissue concentration measurements without any assumptions of association. The question is left open as to which <ESC>s represent a possible exposure for a given sample individual. For example, if an individual with a high measured tissue concentration was collected from a point where tissue concentration is predicted to be low, several interpretations are possible. For example, BMF may be underestimated or BMF may be accurate but the sampled individual received a higher exposure from a nearby hot spot than was estimated at its exact collection point. Or, the high measured tissue concentration could be interpreted in conjunction with other nearby low tissue concentrations as an indication of high tissue concentration variability on the edge of a hot spot, a phenomena that may also be predicted by the model. Because proximity and spatial structure are displayed, the tissue concentration maps represent the site data in the most complete manner and with the fewest assumptions possible.

Due to the large number of maps generated, a tabular summary was devised to aid in describing the map comparisons. The map summaries were developed in two steps. First, the quality of individual predictions (TC_{obs} compared to surrounding TC_{pred} values) was evaluated by one of five categories and the fraction of data which fell within each category was estimated. Second, an overall rating was assigned to each map based on the fractions determined in step one, the magnitude of over and under estimates, considerations regarding sample size adequacy, and an assessment of whether an increase or decrease in BMF would enhance the predictions. The categories and ratings are described below.

Individual Prediction Categories

Good: TC_{obs} was within the same class as TC_{pred} value at the exact location where the TC_{obs} individual was collected.

OK: Correct values for TC_{pred} did not occur at the precise sampling location but did occur within a specified "allowable distance" from a given TC_{obs}. Because of the many sources of error in estimating the true exposure area for each TC_{obs}, this case (close proximity of correct tissue concentration estimates) did not represent a clear contradiction of the model estimates. In general, the potential error in correctly identifying the center of the exposure area for a given sample individual is at least as large as the

exposure area radius, since the individual could have been caught at the edge of its exposure area. The allowable distance used to evaluate the "OK" category was approximately one-half the exposure area radius with a minimum of 250 ft imposed to reflect the possibility of a sample location error this large even for very small exposure areas.

Over: TC_{pred} values within the allowable distance from a given TC_{obs} were too high.

Under(hit): TC_{pred} values within the allowable distance of a given TC_{obs} were too low, and TC_{obs} was a detected concentration.

Under(BCRL): TC_{pred} values within the allowable distance of a given TC_{obs} were too low and TC_{obs} was BCRL. Since the true tissue concentration of the sample may be much lower than its estimated BCRL replacement value on the map, this case was not considered to be a definite underestimation.

Overall Ratings

Good: The model closely predicted a substantial portion of the TC_{obs} data. Significant increases or decreases in the BMF would reduce the fit. Small changes in BMF (< 1/4th order of magnitude) may or may not enhance the fit.

OK: The model resulted in a substantial portion of both over and under estimates. While the fit was not good, improvement in the fit was not likely to result from either an increase or decrease in BMF.

Good-Over: The model closely predicted a substantial portion of the data. Improvement was more likely to result from a decrease, rather than increase in BMF.

Good-Under: The model closely predicted a substantial portion of the data. Improvement was more likely to result from an increase, rather than decrease in BMF.

NC: The model was not contradicted by the data; however, the power to discriminate between different BMFs was extremely low.

Poor: The model over/under predicted a substantial portion of the data so that a smaller/larger BMF was indicated.

Map summaries for the final prey BMFs selected using this calibration procedure (BMF_{Army}) are reported in Table C.1-4.

C.1.5.2 Development of Final BMF for Predators

Final BMF values for top predators (bald eagle, great horned owl, American kestrel and great blue heron) were developed differently from those for their prey because appropriate tissue sample data were not available for these predators. In the absence of field data, final BMFs for top predators were developed from a food-web model. This is equivalent to assigning BMF_{obs} a relative weight of zero (w = 0) and $BMF_{lit/model}$ a relative weight of 1. Thus, the final BMF for predators was based on a revision of equation (34):

$$\overline{BMF}_{lit/model(k)} = \overline{BAF}_{lit(k)} * \sum_j FR_{k,j} * \overline{BMF}_{Army(j) \text{ or } Shell(j) \text{ or } EPA(j)} \quad (40)$$

where:

terms in this equation are as defined for equation (36).

More information on this food-web model and the basis for equation (36) is provided below.

In general, a food-web model is a representation of energy (food) flow from lower trophic-(feeding) level biota to upper trophic-level organisms. As applied to contaminant transport, the model is used to simulate the movement of a chemical from the exposure medium (soil, sediment, or water) into successive trophic levels, with the eventual result being its biomagnification in the top predator through the same pathways by which food is transported to the top species (Cohen 1978). Various food-web models were considered for RMA (i.e., Schnoor 1981; Thomann 1981; Spacie and Hamelink 1982; Mackay and Paterson 1981; Barber et al. 1988); however, none of these were directly applicable to multiple food chains in either aquatic or terrestrial ecosystems, so elements from several models were used to develop an RMA-specific approach. The ERC food-web model is an expanded version of the model first applied at RMA

by Fordham and Reagan (1991). For species that spend only a part of their life cycle at RMA, the food web model approach is conservative because it assumes year-round exposure.

The fully developed version of the food-web model addresses both terrestrial and aquatic ecosystems and has parameters developed to estimate BMF for predators and prey in both terrestrial and aquatic food chains from data in the literature. However, as discussed above, site-specific field data were used as the basis for final BMF values whenever possible because the BMF serves as a proportionality constant between site-specific TC and <ESC> data and between toxicological thresholds and TC estimates that reflect site-specific <ESC> data. To be an effective proportionality constant, BMF must also be site-specific. To maximize the site-specificity of BMFs developed for predators, their BMFs were modeled directly from the BMF_{obs} values for their prey, using the portion of the food-web model for terrestrial food chains as follows. Further explanation of the food chain model is provided in Appendix D, and the quantification of the parameters that were used in the IEA/RC is documented in Appendix Section C.2.

In terrestrial food-chain equations, the BMF was computed as a function of the bioaccumulation factor (BAF) and the dietary fraction (i.e., the fraction of any prey species in a predator's diet, represented by "FR" in the model). Aquatic food chains for birds were modeled using the BAF and FR parameters like the terrestrial food chains. The rationale behind the model's calculation of BMF is given in an example below.

For a simple, straight-line terrestrial food chain of three trophic levels, with the first level equal to the source medium and the third level equal to the top predator, the BMF for level 3 can be expressed as follows:

$$BMF_3 = BAF_3 * BMF_2 \qquad (41)$$

where:

BMF_3 = Biomagnification factor for level 3
BAF_3 = Bioaccumulation factor for level 3
BMF_2 = Biomagnification factor (equals the bioaccumulation factor) for level 2

Bioaccumulation is the process by which a chemical is accumulated into an organism by direct exposure as well as ingestion of food and soil (or other media) and the bioaccumulation factor (BAF) is the ratio of contaminant concentration between a predator and its prey.

If the second trophic level contains two trophic boxes, the equation can be expanded to account for contributions from two prey trophic boxes to the top predator's diet as follows:

$$BMF_3 = BAF_3 * (FR_{3,2A} * BMF_{2A} + FR_{3,2B} * BMF_{2B}) \quad (42)$$

where:

2A and 2B = Trophic boxes in the second level
$FR_{3,2A}$ and $FR_{3,2B}$ = Dietary fractions of trophic boxes 2A and 2B in the diet of the predator trophic box at level 3

This simple equation can be further expanded to accommodate the much greater size and complexity of the RMA food webs.

The top predator BMFs were computed probabilistically because the BAF and prey BMFs used to calculate them are represented by distributions. These distributions were represented by a mean and standard deviation (or other descriptors appropriate to the distribution type) in the model spreadsheets. The probabilistic, rather than deterministic, approach quantified a portion of the uncertainty in the input data and propagated it through the computation of the predator BMFs. The input parameters used in the food-web model are discussed and quantified in Appendix Section C.2.

Some measured tissue concentration data were available for the top predators (American kestrel, bald eagle, and great horned owl). Some of these field data were used in Appendix Section E.12.7.6 to provide a reality check on the tissue concentrations implied by the BMFs estimated using the food-web model. The available kestrel samples consisted of eggs and dressed carcasses of juveniles that had not fledged and therefore received food from adults that foraged largely or exclusively within RMA. Although the juvenile dressed carcass samples could have been used to calculate BMF_{obs}, their concentrations were correlated to the concentrations in egg samples taken from the same nest (R=.67) and the concentrations in juveniles and eggs collected from the same nest were similar. These two observations indicate that the juvenile tissue concentrations were likely to depend in large part on the prenesting exposure of their parents, which are not restricted to RMA. To avoid potential bias, the kestrel data set was not used to estimate BMF_{obs}, but was used for comparison with tissue concentrations implied by the modeled kestrel BMF. The only tissue samples available for bald eagles (blood) and great horned owls (eggs) were not considered to be appropriate for comparison with values that reflected whole body concentrations.

C.1.6 CALCULATION OF POTENTIAL RISK

As stated in the introduction to Appendix Section C.1, calculations of potential risk were based on a comparison of predicted or measured site-specific tissue concentrations to toxicological threshold values using either a tissue-based approach or a dose-based approach. For both approaches, the ways in which risk was calculated for various trophic box/chemical combinations, except where absent data prevented a specific calculation, varied in response to (1) the type of food chains leading to a trophic box and (2) the type of COC being evaluated as follows.

Calculations differed depending on whether a trophic box had only terrestrial food chains, only aquatic food chains, or both leading to it in a food web:

- For trophic boxes with terrestrial food chains, BMFs, however calculated, were multiplied by <ESC> values to predict tissue concentrations for each block in the RMA-wide grid; the comparison of each of these tissue concentrations to a toxicological threshold resulted in a calculation of potential risk.

- For trophic boxes with aquatic food chains, measured tissue concentrations were compared directly to a toxicological threshold to calculate potential risk.

- For trophic boxes with mixed food chains, potential risk was calculated from both terrestrial and aquatic sources.

Regardless of the type of food chains leading to a trophic box, calculations of potential risk to the trophic box also differed depending on whether or not the COC being evaluated was bioaccumulative.

- For the bioaccumulative COCs (aldrin/dieldrin, DDT/DDE, endrin, and mercury), both tissue-based and dose-based calculations were done and the more certain of these values was used.

- For the remaining COCs, only dose-based calculations were done and this application of the dose-based approach considered only contaminant uptake from abiotic media, not from food.

A few special cases that were outside these situations are identified below. Each of these situations, except risk calculations for aquatic food chains and for nonbioaccumulative COCs, uses a BMF. When a BMF was used, calculations were done three times, using final values from each of the BMF approaches.

The remainder of this section provides more information on the ERC calculation of potential risk, first defining risk (Appendix Section C.1.6.1), identifying the data used to quantify risk (Appendix Section C.1.6.2), and then detailing the ways in which it was calculated in each of the situations above (Appendix Section C.1.6.2).

C.1.6.1 Definition of Potential Ecological Risk

The term risk implies that there is some probability of an adverse effect occurring in an exposed or potentially exposed population, e.g., the probability of excess cancer cased in a given human population. For the ERC, risks were quantified on the basis of HQs and His that do not represent probabilities but rather estimates of the magnitude of difference between a measure of exposure (tissue concentration or dose) and a toxicity threshold value considered to be protective (MATC or TRV). It is assumed that the magnitude of the potential adverse effect (risk) will be proportional to the magnitude of the HQ or HI. Based on best professional judgment, an HQ or

HI of 1.0 represents the highest level of chronic exposure that is unlikely to result in adverse effects on populations exposed chronically in the field. For values of HQ or HI greater than 1.0, the potential for adverse effects increases as the HQ or HI value increases. The range of uncertainty in these statements regarding HQs or HIs spans at least one order of magnitude. This uncertainty exists in both directions; hence, some risk may occur at values of HQ or HI as low as 0.1, and no risk may occur at values of HQ or HI as high as 10.

C.1.6.2 Data Used to Quantify Risk

A combined total of 1897 biota tissue samples were collected under three programs: the Biota RI (ESE 1989), the Biota CMP (RLSA 1992), and the ERC. Once off-post control samples, fortuitous (i.e., found dead) samples, and QA/QC rejections were removed, 1328 samples remained to provide tissue concentrations representative of the terrestrial and aquatic trophic boxes. Further information on the collection of these biota samples is in Appendix A and Appendix Section C.4.

Soil concentration data used for risk calculations were described in Appendix Section C.1.4. The soil boring and surficial soil data used in risk calculations (and in BMF calculations) were taken from the RMA environmental database and modified as described in that section through screening, quality assurance checks, spatial interpolation of BCRL samples, interpolation of soil concentrations into an RMA-wide grid, and spatial weighting of interpolated data using trophic-box-specific or species-specific exposure areas to get <ESC>.

Sediment and water concentrations of the bioaccumulative COCs entered the risk calculations indirectly through the tissue concentration data collected from aquatic biota. Water concentrations of the bioaccumulative COCs also entered the risk calculations explicitly in that ingestion of water accounted for 7.1 percent by mass of the assumed diet for the great blue heron. Virtually all of the water concentration measurements for the nonmetal COCs were BCRL. In calculating exposure through direct ingestion of water, these COC concentrations were set equal to their respective CRLs. It was noted in Appendix Section C.1.4.2 that the risk calculation was found to be insensitive to the water concentrations selected (over the range of

zero to the CRL). Measured water concentrations for mercury did exceed CRLs, so the average observed mercury concentration was used in the risk calculations.

C.1.6.3 Calculation of Potential Risk

Potential risk is expressed as a hazard index (HI) for all COCs collectively or a hazard quotient (HQ) for a single chemical. Thus, the total cumulative risk for all contaminants and exposure pathways can be expressed as:

$$Risk_{total} = Hazard\ Index \tag{43}$$

where:

HI is defined as:

$$HI = \sum_i \sum_j \frac{Estimated\ Exposure_{ij}}{Toxicity\ Threshold_{ij}} \tag{44}$$

The i and j refer to contaminant "i" and exposure pathway "j", respectively.

The ratio of the estimated exposure to the toxicity threshold for a single contaminant is defined as the HQ and expressed as:

$$HQ_i = \sum_j \frac{Estimated\ Exposure_{ij}}{Toxicity\ Threshold_{ij}} \tag{45}$$

The estimated exposure can be represented as the contaminant concentration in biota tissue, or the estimated contaminant intake rate (dose) at the point of exposure. The corresponding toxicity thresholds can be represented by the maximum allowable concentration in biota tissue (MATC)

or the maximum allowable intake rate or dose (TRV). Therefore, the HQ for contaminant *i* can be expressed as:

$$HQ_i = \frac{Tissue\,Concentration_i}{MATC_i} \tag{46}$$

or

$$HQ_i = \frac{Dose_i}{TRV_i} \tag{47}$$

for the tissue-based and dose-based approaches, respectively.

C.1.6.2.1 Variations in Risk Calculation for Different Types of Food Chains

Risk Calculations for Terrestrial Food Chains

When the tissue-based approach was used for terrestrial food chains, the tissue concentrations used to calculate potential risk were estimated using BMF and <ESC> so that potential risk could be calculated at each grid point on RMA. Thus, equation (46) can be expanded for terrestrial food chains by replacing tissue concentration with the product of the BMF and <ESC>:

$$(HQ_i)_{terrestrial} = \frac{BMF * <ESC>}{MATC} \tag{48}$$

When the dose-based approach was used for terrestrial food chains, BMF and <ESC> were also used to estimate tissue concentrations at individual grid points; to convert the tissue concentration of the trophic box being evaluated into a dose, R (feed rate) was added to express tissue concentration on a daily basis and BAF was added to convert from the tissue concentration in the trophic box to that in its prey. Thus, equation (47) can be expanded for terrestrial food chains as follows:

$$(HQ_i)_{terrestrial} = \frac{R * BMF * <ESC>}{BAF * TRV} \quad (49)$$

Note that the sum of prey tissue concentrations, represented by their respective BMF and <ESC> values then weighted by the dietary fraction, can be substituted in equation (49) to avoid the use of BAF. Thus:

$$(HQ_i)_{terrestrial} = \frac{R * \sum BMF_{prey} * FR * <ESC>}{TRV} \quad (50)$$

The parameters used above are discussed further in Appendix Section C.2.

Risk Calculations for Aquatic Food Chains

For most of the trophic box/chemical combinations in aquatic food chains, measured tissue data that were considered representative of the relatively homogeneous aquatic environment were available. Therefore, measured tissue concentrations could be used directly in both the tissue-based and dose-based approach equations (equations (46) and (47) above) to calculate risk to aquatic trophic boxes, and estimations of tissue concentrations using BMF were not needed. This is how risk to the water bird was calculated as explained further in Appendix Section C.1.4.2. However, toxicity threshold values (MATC and TRV) were lacking in the literature for trophic boxes representing strictly aquatic organisms such as fish, aquatic plants, and aquatic invertebrates. Therefore, while risk to these trophic boxes could not be calculated, their tissue concentrations were used to estimate the tissue concentrations of and dose to top predators that had aquatic food chains (bald eagle and great blue heron). These calculations are shown in Appendix Section C.1.4.2.

The calculation of potential risk to strictly aquatic organisms by use of the food-web model and by comparison to EPA's ambient water quality criteria (AWQCs) was also investigated. However, both of these approaches required information on COC concentrations in surface water.

Because the COCs (except arsenic and mercury) had few or no detections in analyzed surface water samples, surface water COC concentrations would have had to be estimated. Efforts at estimation using equilibrium partitioning and the WASP model proved too uncertain.

The final approach to evaluating risk to strictly aquatic organisms was to identify the sources of COCs that might be contributing to potential risk and the likely magnitude of that risk. This approach revealed that the ultimate source of contamination found in RMA lake sediments and water is soils from the shoreline and surrounding upland areas. This is qualitatively supported by the documentation of highly elevated aldrin, dieldrin, and endrin sediment concentrations near the north inlet of Upper Derby Lake that are orders of magnitude above sediment concentrations elsewhere in Upper Derby Lake, many of which are BCRL. The inlet sediment concentrations are similar to nearby soil concentrations, which are on the order of 0.1 to 10 ppm. The magnitude of the risk to strictly aquatic trophic boxes from contaminants with their origin in surrounding soils can be assumed lower than the magnitude of risk from aquatic food chains to the bird species evaluated because the strictly aquatic trophic boxes are lower in the food web. For the water bird (assumed to be strictly aquatic but having toxicity threshold values), bald eagle, and shorebird the potential risk is of relatively low magnitude ($2.0 > HI > 1.0$). The great blue heron (HI=13), which consumes primarily predatory fish, exhibited the greatest risk from aquatic food chains.

Risk Calculations for Trophic Boxes with Both Terrestrial and Aquatic Food Chains

The shorebird, eagle, and heron food webs have both terrestrial and aquatic food web components. Therefore, potential risk to these trophic boxes results from both terrestrial and aquatic contributions to their tissue concentrations or doses and their total potential risk can be calculated as the sum of the partial risks from these sources. Partial tissue- and dose-based risks are defined for a single chemical as follows:

$$HQ_{TR} = \frac{TC_{TR}}{MATC} \quad (51)$$

$$HQ_{AQ} = \frac{TC_{AQ}}{MATC} \quad (52)$$

$$HQ_{TR} = \frac{dose_{TR}}{TRV} \quad (53)$$

$$HQ_{AQ} = \frac{dose_{AQ}}{TRV} \quad (54)$$

Total risk is equal to the sum of partial tissue-based or dose-based risks. Thus, for example:

$$HQ_{TOTAL} = \frac{\text{tissue concentration}_{TOTAL}}{MATC} \quad (55)$$

$$= \frac{\text{tissue concentration}_{TR} + \text{tissue concentration}_{AQ}}{MATC} \quad (56)$$

$$= HQ_{TR} + HQ_{AQ} \quad (57)$$

The calculation of total risk based on dose is comparable.

Potential risk to top predator trophic boxes is presented separately in the ERC for terrestrial and aquatic food chain sources. The additive nature of these risks should be kept in mind when evaluating current risk and when considering future scenarios where the balance between terrestrial and aquatic contributions may be varied.

C.1.6.2.2 Variations in Risk Calculation for Different Types of COCs

Of the 14 COCs evaluated by the ERC, six are defined as bioaccumulative: aldrin/dieldrin, DDT/DDE, endrin, and mercury. Chlordane, although it bioaccumulates, was treated with the rest of the COCs as nonbioaccumulative for reasons discussed elsewhere. These two groups of COCs differ in the approaches that were available to evaluate potential risk and in the way dose was defined.

Evaluation of Potential Risk for Bioaccumulative COCs

For the bioaccumulative COCs, potential risk could be evaluated using either the tissue-based or dose-based approach. A slightly different rationale was used to select the final approach for terrestrial and aquatic food chains.

Approach Selection for Bioaccumulative COCs in Terrestrial and Mixed Food Chains

Protocol—The selection of the final approach to be employed in the estimation of potential risk from terrestrial and mixed food chains was based primarily on the uncertainty in the toxicological threshold values (MATC and TRV) for each trophic box/bioaccumulative chemical combination. The aquatic component of mixed food chains was generally treated like the terrestrial food chains because there were no appropriate measured tissue samples for the trophic boxes having mixed food chains. Uncertainty factors (UFs; described in greater detail in Appendix Sections C.2.3 and C.2.5) for the toxicological threshold values were the primary basis for the decision between the tissue-based and dose-based approaches. For each trophic box/chemical combination considered, the numerical values of the UFs were compared, and the threshold value with the lower UF was typically selected. When the UF values were very similar for MATCs and TRVs, other considerations entered into the selection.

The other considerations used in selecting the type of risk calculation approach to be used for bioaccumulative COCs were uncertainty about the BAF, source of information, and the possibility of future sampling. The tissue-based approach incorporates one more level of computation than the dose-based approach. This is because the dose-based approach uses prey tissue concentrations as the dose to the predator, while the tissue-based approach must transform the predator dose into a predator tissue concentration by using the predator's BAF. Thus, when UF values were numerically close, the relative values of the predator BAFs, mean and standard deviation were also evaluated. Further, when the two toxicological endpoints were developed from different papers, the relative strength of the papers and the pertinence of the test organisms were considered. When the two criteria were developed from the same paper, the total UF reflected any differences in extrapolations needed to derive the pre-UF values because of the way data were presented in the paper so that the quality of the paper and pertinence of the test organisms were not an issue. Finally, the dose-based approach was given more weight for the bald eagle trophic box when uncertainty was similar for the two approaches since no post-remediation tissue samples can be taken to test for effectiveness of remediation.

Results—Table C.1-5 summarizes the risk calculation approach selected for each trophic box/ bioaccumulative chemical combination. The UF values associated with MATC and TRV values for each combination are listed in Table C.1-6. Also shown are the relative uncertainty associated with the BAF and other information considered to decide which type of risk calculation approach was more appropriate for each trophic box/chemical combination.

For eagle, heron, owl, and kestrel, the selection of risk calculation approach was based solely on a comparison of the MATC and TRV UFs in all cases except three: aldrin/dieldrin for eagle; aldrin/dieldrin for owl; and aldrin/dieldrin for kestrel. In these three cases, the MATC and TRV UFs were the same, so the dose-based approach was selected on the basis of the assumption that there is less uncertainty in estimating the predator's dose from the available estimate of the average prey tissue concentration than there is in estimating the predator's tissue concentration from the average prey tissue concentration estimate. It is implicit in this assumption that there

is more uncertainty in the BAF needed to predict predator tissue concentration from prey tissue concentration than in the R value needed to adjust prey tissue concentration to a predator dose.

For shorebird, the tissue-based approach was selected for DDT/DDE and endrin and the dose-based approach was selected for aldrin/dieldrin and mercury (for terrestrial food chains only) on the basis of the relative magnitude of MATC and TRV UFs. Use of the tissue-based approach to calculate risk from mercury to this trophic box from aquatic food chains allowed direct use of the partitioned measured shorebird tissue data. Even though the TRV UF was lower, the absence of a BAF for this trophic box/chemical combination precluded the use of shorebird tissue data to calculate its dose.

For both small and medium mammal, the dose-based approach was the only approach used; it had the lower UF for aldrin/dieldrin and DDT/DDE and was the only available approach for endrin and mercury. For small bird, the tissue-based approach was selected for endrin and the dose-based approach was selected for aldrin/dieldrin, DDT/DDE, and mercury.

Approach Selection for Bioaccumulative COCs in Aquatic Food Chains

Protocol—The calculation of risk from aquatic food chains emphasized the use of measured tissue concentrations, which were assumed representative of the lakes from which they were collected. Therefore, the tissue-based approach was generally used whenever measured tissue data were available (without consideration of UFs for MATC and TRV) because it allowed the calculation of risk from unmodified measured data and avoided the additional uncertainty that would be introduced by converting the tissue concentration data into a dose estimate.

Results—For water bird, the tissue-based approach was selected for aldrin/dieldrin, DDT/DDE, and mercury. The water bird MATC UF was twice the TRV UF for aldrin/dieldrin, one and two-thirds the TRV UF for DDT/DDE, and one and one-half times the TRV UF for mercury, but the tissue-based approach was selected for all these cases because measured tissue data could be used. The dose-based approach was selected for endrin, where the MATC UF was three-fourths

the TRV UF; only two of the 67 tissue samples measured contained detectable concentrations of endrin.

Evaluation of Potential Risk for Nonbioaccumulative COCs

For the nonbioaccumulative COCs, a modified version of the dose-based approach was used to evaluate risk because tissue concentrations of these chemicals were not measured at RMA. Therefore, there are no direct measurements of contaminant concentrations in food items, which are a much less important contaminant source for COCs that do not bioaccumulate. Thus, when the dose-based approach is applied to nonbioaccumulative COCs, the dose used is based only on the contaminant contributions from soil/sediment and, in some cases, water.

C.1.6.2.3 Special Cases in Risk Calculation

There are two special cases that require further discussion of their risk calculation: where species-specific rather than trophic-box specific BMFs were calculated and when COCs were naturally occurring as well as present as a result of human activities. Each of these cases is discussed below.

Risk Calculations Using Species-Specific BMFs

The species that were grouped together in a trophic box determined the exposure area appropriate to the trophic box as described in Appendix Section C.2.4. As was noted in Appendix Section C.1.5.1.2, there were a few instances in which exposure ranges were somewhat variable within a trophic box and data were sufficient to calculate species-specific exposure areas (i.e., small bird, small mammal, and medium mammal trophic boxes in combination with aldrin/dieldrin). In these cases, the species-specific BMF_{obs} values were then combined in a weighted average. To calculate risk for these three trophic box/chemical combinations, the <ESC>s calculated using the smallest of the exposure areas available for the trophic box were paired with the weighted average BMF to estimate tissue concentrations for comparison with the appropriate toxicological threshold at each grid point. This is a conservative assumption in the sense that remediation protective of smaller home ranges will always be protective of a larger home range composed of the smaller areas. The use of <ESC>s associated with the smallest exposure range helped

ensure that risk to prairie dog's (which have a smaller exposure area than cottontails) would not be underestimated. This was particularly important because this species is an important prey item for wintering raptors, including the bald eagle. This approach was also used to try to ensure that risk to the diverse species of small birds on RMA would not be underestimated by using the large and atypical exposure area defined for the mourning dove.

Risk Calculations for Naturally Occurring COCs

Four of the COCs are naturally occurring metals: arsenic, cadmium, copper, mercury. For these metals an indicator level range was developed for and used in the RI/FS at RMA (ESE 1986). The indicator range was based on an evaluation of the natural ranges of potential contaminants in soils of the western U.S., the results from chemical analysis of a bulk soil sample collected just off the northeast corner of RMA, and the soil quality data from the uncontaminated portions of 24 of the 27 land sections at RMA. Background levels of metals, defined in the IEA/RC as the upper end of the indicator range, are: arsenic, 10 ppm; mercury, 0.1 ppm; cadmium, 2 ppm; and copper, 35 ppm. ESE (1986) recognized the inherent variability in trace metal concentrations at RMA due to the variety of soil series present, but concluded: "...comparison of these values with action levels selected by the U.S. EPA and various states at NPL sites and with recommended concentrations for land treatment of hazardous waste indicates these levels are very conservative" (ESE 1986, pg 4-18). The types of data on which the indicator range was based are shown in Table C.1-7.

C.1.6.3 Evaluation of Risk

Potential ecological risk at RMA was quantified using total HIs, HIs for chemical groupings, and HQs. Total HI was calculated for all trophic boxes that had available MATCs or TRVs (bald eagle, great horned owl, American kestrel, great blue heron, shorebird, water bird, small bird, medium mammal, and small mammal). For these trophic boxes, HIs were also calculated for three groupings of chemicals. These three chemical groupings were aldrin/dieldrin, DDT/DDE, and endrin; mercury, arsenic, cadmium, and copper; and chlordane, CPMS, $CPMSO_2$, DBCP, and DCPD. Thus, the contributions to total risk of the bioaccumulative COCs (minus mercury), the metals, and the other COCs were evaluated separately. In addition, for the metals grouping,

potential risk was also calculated using <ESC> values from which background concentrations at RMA had been subtracted; this was done to allow consideration of the relative contribution to risk of concentrations from human activities versus naturally occurrence. HQs were calculated for all trophic box/chemical combinations, since their sum is equal to the HI for that combination. The calculations of total HI, HIs for chemical groupings, and HQs just described were repeated three times, once using the Army's BMF, once using Shell's BMF, and once using EPA's BMF.

To evaluate risk from terrestrial food chains on RMA, maps were used because the calculations of risk from concentrations in soil were done for each grid point and vary across the site. Maps were prepared that show the number of trophic boxes with HIs that exceed 1.0, show soil concentrations of individual COCs, show HIs for particularly important trophic boxes, and that show the effect of exposure range size and contaminant concentration magnitude on the size of areas of risk. These maps were based on total HI and on chemical grouping HIs. Some of the maps showed the results of the Army, Shell, and EPA approaches on the same map; others presented these approaches separately. HQs were mapped for the most important (i.e., most widespread, most bioaccumulative, and most toxic) chemicals, and for any other individual chemicals when the HI for their grouping exhibited substantive exceedances. HIs based on concentrations above background were also mapped. Section 4.2 and Appendix Section C.3 provide the results of the ERC.

C.1.7 QUANTITATIVE UNCERTAINTY ANALYSIS

Uncertainties about the variability in the parameters and structures of the models for characterizing risks to representative biota at RMA were identified and analyzed to provide more realistic and informative risk characterizations. As necessitated by the complexity of RMA ecosystems and ambiguities in relevant databases, a wide range of methods were employed to investigate the impacts of process and parameter uncertainties on RMA risks. The analyses are presented in Appendix E.

A number of benefits were derived from the explicit consideration of uncertainty in RMA risk characterizations. For example, analysis of uncertainty about exposure soil concentrations helped

explain the quantitative and qualitative differences between BMFs reported in the scientific literature and those computed for RMA and facilitated the choice of appropriate BMFs for the ERC. As a second example, analysis of the spatial distribution of biota tissue concentration predictions helped explain the lack of correlation between tissue and home-range soil concentration databases. These findings and other implications of uncertainty analyses are discussed in detail in Section 5 and Appendix E.

C.1.8 CALCULATION OF BIOTA SOIL CRITERIA

C.1.8.1 Bioaccumulative COCs

Biota soil criteria for the bioaccumulative COCs, derived for the three BMF calculation approaches, are reported in Section 4, Table 4.6-1. The values were derived using the terrestrial HQ equations (Section 4, equations 8 and 9), which are reproduced as equations 59 and 60:

$$(HQ_i)_{terrestrial} = \frac{BMF \cdot ESC}{MATC} \quad (59)$$

$$(HQ_i)_{terrestrial} = \frac{R \cdot BMF \cdot ESC}{BAF \cdot TRV} \quad (60)$$

The biota soil criterion is the value of ESC for which $(HQ_i)_{terrestrial} = 1$. Thus, criteria are calculated by setting $(HQ_i)_{terrestrial} = 1$ and rearranging equations (8) and (9) to solve for ESC. The resulting average soil concentrations are the biota soil criteria:

$$soil\ criterion = \frac{MATC}{BMF} \quad (61)$$

when risk is calculated by the tissue based approach, or

$$soil\ criterion = \frac{BAF \cdot TRV}{R \cdot BMF} \quad (62)$$

when risk is calculated by the dose based approach. When risk is calculated by the tissue based approach, the term BMF in the biota soil criterion equation (eq. 61) represents the BMF for the trophic box in question, calculated by the Army, EPA, and Shell methods. When risk is calculated by the dose based approach, the term BMF in the biota soil criterion equation (eq. 62) is calculated using the BMFs for the prey trophic boxes and the ERC food web model:

$$BMF = BAF \cdot \sum_{\substack{prey \\ trophic \\ boxes}} FR \cdot BMF_{prey} \quad (63)$$

where FR is the prey fraction and BMF_{prey} is the prey BMF calculated by the Army, EPA, or Shell method. The procedure for selecting the tissue or dose based approach for each trophic box/bioaccumulative COC is described in Appendix Section C.1.6.2.2, beginning on page C.1-60. Selections are summarized in Table C.1-5.

C.1.8.2 Non-Bioaccumulative COCs

Biota soil criteria for the non-bioaccumulative COCs are reported in Table 4.6-2. The dose based approach was used to calculate risk for all "non-bioaccumulative" COCs, so these biota soil criteria were calculated using a modified form of equation (62). For the non-bioaccumulative COCs, it is assumed that exposure to soil contaminants occurs only through direct ingestion of soil (because the contaminant is assumed not to bioaccumulate). This implies that for the direct soil ingestion pathway, BAF = BMF = 1, and for all other exposure pathways within the food web model, BAF = 0. Consequently, equation 62 becomes:

$$soil criterion = \frac{TRV}{FR_{soil} \cdot R} \quad (64)$$

for the non-bioaccumulative COCs, where FR_{soil} is the assumed quantity of ingested soil in the trophic box's diet as a mass fraction of total consumption, and the denominator is a "soil ingestion rate coefficient."

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Table C.1-1 Example Calculations for Spatial Interpolation of a BCRL Data Point

First Iteration

x_i	distance (ft)	d_i	$1/d_i^2$	W_i	x_iW_i
0.0076	44	0.11	82.64	0.815	0.0062
7.40	184	0.46	4.73	0.047	0.3478
0.32	194	0.485	4.25	0.042	0.0134
0.0279	199	0.498	4.03	0.040	0.0011
3.0	206	0.515	3.77	0.037	0.1110
0.0039	285	0.713	1.97	0.019	0.0001
			sum 101.39	1.0	sum 0.4796

Second Iteration

x_i	distance (ft)	d_i	$1/d_i^2$	k_i	x_ik_i
0.0076	44	0.11	82.64	0.723	0.0055
0.18	112	0.28	12.78	0.112	0.0202
0.18	165	0.413	5.86	0.051	0.0092
7.4	184	0.46	4.72	0.041	0.3034
0.32	191	0.478	4.38	0.038	0.0122
0.18	199	0.498	4.03	0.035	0.0063
			sum 114.41	1.0	sum 0.3568

x_i Aldrin concentration of soil sample (ppm)
d_i Normalized distance between BCRL point and soil sample
W_i $1/d_i^2$ divided by sum $1/d_i^2$ to provide the inverse distance squared based weight

Table C.1-2 Example Calculations for Modeling of a Grid Block

x_i	distance (ft)	d_i	$1/d_i^2$	k_i	$x_i k_i$
0.18	198	0.263	14.42	0.163	0.029
0.435	145	0.193	26.75	0.302	0.131
0.134	235	0.313	10.19	0.115	0.015
0.94	220	0.293	11.62	0.131	0.123
3.6	215	0.287	12.17	0.137	0.495
0.31	205	0.273	13.39	0.151	0.047
			sum 88.54	1.0	sum 0.841

x_i Aldrin concentration of soil sample (ppm)
d_i Normalized distance between grid block center and soil sample
k_i $1/d_i^2$ divided by sum $1/d_i^2$ to provide the inverse distance-squared based weight

Table C.1-3 Heron DDE/DDT Tissue Concentration Prediction Sensitivity to Assumed DDE/DDT Concentrations in Aquatic Invertebrates and Amphibians

Lake ¥	Original heron TCpred	Modified heron TCpred*	Percent increase in heron TCpred*
Lower Derby	2.00E + 01	2.31E + 01	16%
Ladora	1.14E + 01	1.36E + 01	19%
Mary	1.34E + 01	1.60E + 01	20%

* when aquatic invertebrate and amphibian tissue concentration predictions are set = 10xTCpred

Lake ¥	Original heron TC predictions	Modified heron TCpred †	Percent decrease in heron TCpred †
Lower Derby	2.00E + 01	1.97E + 01	2%
Ladora	1.14E + 01	1.12E + 01	2%
Mary	1.34E + 01	1.31E + 01	2%

† when aquatic invertebrate and amphibian tissue concentration predictions are set = 0

¥ Sensitivity analysis was limited to Lower Derby, Ladora, and Mary Lakes because these are the only lakes in which all trophic boxes in the heron's food chain (especially aquatic invertebrates, amphibians, and small and large fish) were present when sampling was performed.

Table C.1-4 Quality of TC Predictions for Army Calibrated BMFs

Species/Chemical	N	AD (feet)[1/]	Fraction of Points in Different Prediction Classes					
			Good	OK	Over	Under (HIT)	Under (BCRL)	Rating[2/]
American Kestrel		1,000						
Aldrin/Dieldrin	19		0.37	0.26	0.32	0.05	0	Good
DDE/DDT	19		0.37	0.53	0	0	0.10	Good
Endrin	19		0.05	0.21	0	0	0.74	NC
Mercury	19		0.84	0.11	0	0	0.05	NC
Prairie Dog		250						
Aldrin/Dieldrin	126		0.54	0.15	0.23	0.07	0.01	Good-Over
DDE/DDT	95		0.17	0.31	0	0.01	0.51	NC
Endrin	128		0.09	0.10	0	0.01	0.80	NC
Mercury	110		0.40	0.41	0.19	0	0	Good-Over
Cottontail		250						
Aldrin/Dieldrin	28		0.08	0.33	0.38	0.08	0.13	Good-Over
DDE/DDT	14		0.14	0.29	0	0	0.57	NC
Endrin	24		0.05	0.38	0.33	0	0.24	Good-Over
Mercury	24		0.54	0.42	0	0.04	0	Good
Deer Mouse		250						
Aldrin/Dieldrin	87		0.58	0.19	0.01	0.13	0.09	Good-Under
DDE/DDT	90		0.13	0.33	0.06	0.01	0.47	Good-Over
Endrin	90		0.22	0.35	0.04	0	0.39	Good-Over
Mercury	90		0.03	0.01	0.93	0.03	0	Over
Ground Squirrel		250						
Aldrin/Dieldrin	3		0.33	0.67	0	0	0	Good
DDE/DDT	3		0	0.67	0	0	0.33	NC
Endrin	2		0	1.0	0	0	0	NC
Mercury	2		0	0	1.0	0	0	Over
Mourning Dove		1,000						
Aldrin/Dieldrin	68		0.46	0.21	0.29	0.03	0.01	Good-Over
DDE/DDT	68		0.14	0.52	0	0.02	0.32	Good

Species/Chemical	N	AD (feet)[1/]	Fraction of Points in Different Prediction Classes					
			Good	OK	Over	Under (HIT)	Under (BCRL)	Rating[2/]
Endrin	68		0.42	0.46	0.07	0.05	0	Good
Mercury	68		0.31	0.12	0.07	0	0.50	NC
Meadow Lark		500						
Aldrin/Dieldrin	10		0.3	0.3	0.10	0.3	0	Good
DDE/DDT	10		0	0	0	0	1.0	NC
Endrin	10		0.4	0.6	0	0	0	NC
Mercury	10		0	0	0	0	1.0	NC
Sparrow		500						
Aldrin/Dieldrin	5		0.20	0.60	0	0	0.20	Good
DDE/DDT	5		0	0.80	0	0	0.20	NC
Endrin	5		0	0	0	0	1.0	NC
Mercury	5		0	0.40	0	0	0.60	NC
Shorebird[3/]		500						
Aldrin/Dieldrin	10		0.70	0	0.20	0.10	0	Good
DDE/DDT	10		0.6	0.10	0.20	0.10	0	OK
Endrin	10		0.50	0	0.10	0.40	0	Good-Under
Mercury	10		1.0	0	0	0	0	Good
Bullsnake		250						
Aldrin/Dieldrin	3		0	0.334	0.333	0.333	0	NC
DDE/DDT	3		0.334	0	0	0.333	0.333	NC
Endrin	3		0.334	0.333	0	0	0.333	NC
Mercury	3		0.33	0	0	0	0.67	NC

1/ AD = Allowable distance from TCobs within which TCpred values are considered for class "ok".
2/ Includes qualitative visual assessment of map.
3/ TCobs and TCpred reflect contribution from terrestrial food web only.
NC = Model is not contradicted; however power to discriminate different BMFs is low.
OK = A substantial percentage of over and under estimates occurred.

Table C.1-5 Summary of Selected Risk Calculation Approach

Trophic Box	Chemical			
	Aldrin/Dieldrin	Endrin	DDT/DDE	Mercury
Bald Eagle	dose-based	tissue-based	tissue-based	dose-based
Great Blue Heron	tissue-based	tissue-based	tissue-based	dose-based
Shorebird	dose-based	tissue-based	tissue-based	dose-based*
Great Horned Owl	dose-based	tissue-based	tissue-based	dose-based
American kestrel	dose-based	tissue-based	tissue-based	dose-based
Medium Mammal	dose-based	dose-based	dose-based	dose-based
Small Mammal	dose-based	dose-based	dose-based	dose-based
Small Bird	dose-based	tissue-based	dose-based	dose-based
Water Bird	tissue-based	dose-based	tissue-based	tissue-based

*the tissue-based approach was used for calculation of risk from mercury to shorebird from aquatic food chains; all other trophic boxes having mixed food chains (bald eagle and great blue heron) used the same approach for both aquatic and terrestrial food chains.

Table C.1-6 Information Used to Decide Between the Tissue-Based and Dose-Based Approaches to Risk Evaluation*

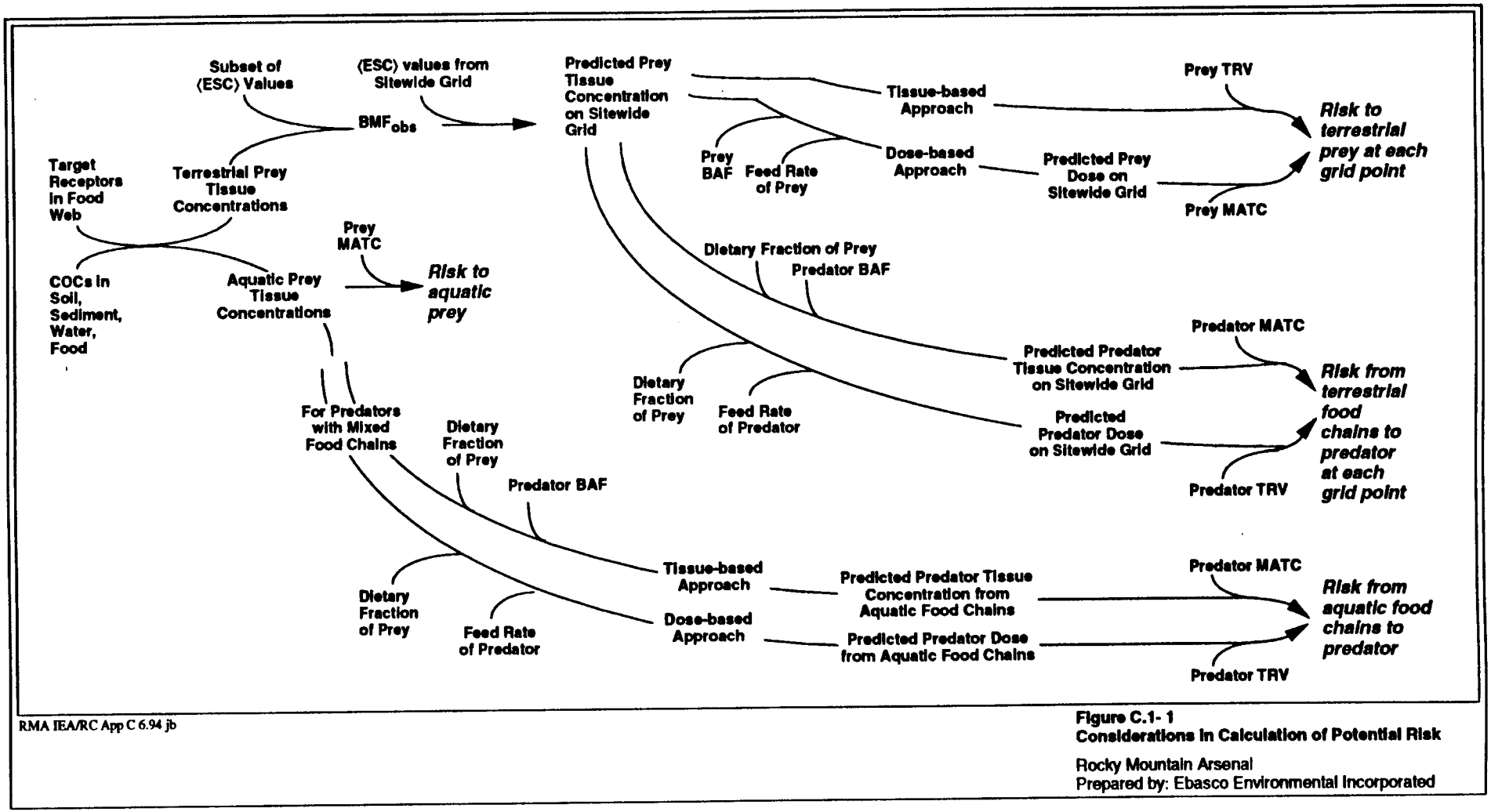
	CHEMICAL							
	Aldrin/Dieldrin				DDT/DDE			
Trophic Box	MATC UF	TRV UF	BAF Mean, SD	OTHER	MATC UF	TRV UF	BAF Mean, SD	OTHER
Eagle	30	30	15.9, 3.9	T&E species	3	80	27.1, 2.4	eagle vs. kestrel
Heron	1.5	15	16, 5.1	heron vs. mallard	2	15	93.5, 20	heron vs. bl. duck
Shorebird	20	10	13.3, 4.2	kestrel vs. quail/pigeon	8	50	NA	tern vs. kestrel
Owl	16	16	21.1, 3.4	same paper	12	40	43.7, 2.4	same paper
Kestrel	4	4	10.5, 1.2	same paper	3	10	NA	same paper
Med. Mammal	24	16	NA		60	12	NA	
Sm. Mammal	24	16	NA		60	12	NA	
Small Bird	20	10	6.6, 1.8	kestrel vs. pigeon	375	250	NA	same paper
Water Bird	30	15	16, 5.1	same paper	25	15	96, 26.2	same paper

Page TABLE C.1-6 PAGE 2 OF 2 is missing from the original.

Table C.1-7 Data Used in Development of Indicator Ranges for Metals at RMA

Metal	Number of Detections/Number of Samples	Average Concentration in Western Soils in ppm	Indicator Range in ppm	Comment
arsenic	80/798	5.5	4.7-10	24 of 80 detections in Section 36; more than 90 % of detections <10 ppm
mercury	27/798	0.46	0.05-0.1	18 of 27 detections in Section 36; more than 60 % of detections <0.1 ppm
cadmium	12/798	NA	1-2	generally BCRL in the bulk soil sample
copper	580/798	21	20-35	detections in uncontaminated area samples ranged from 7-55 ppm and were skewed toward the lower end of the range; 60 % of detections <10 ppm

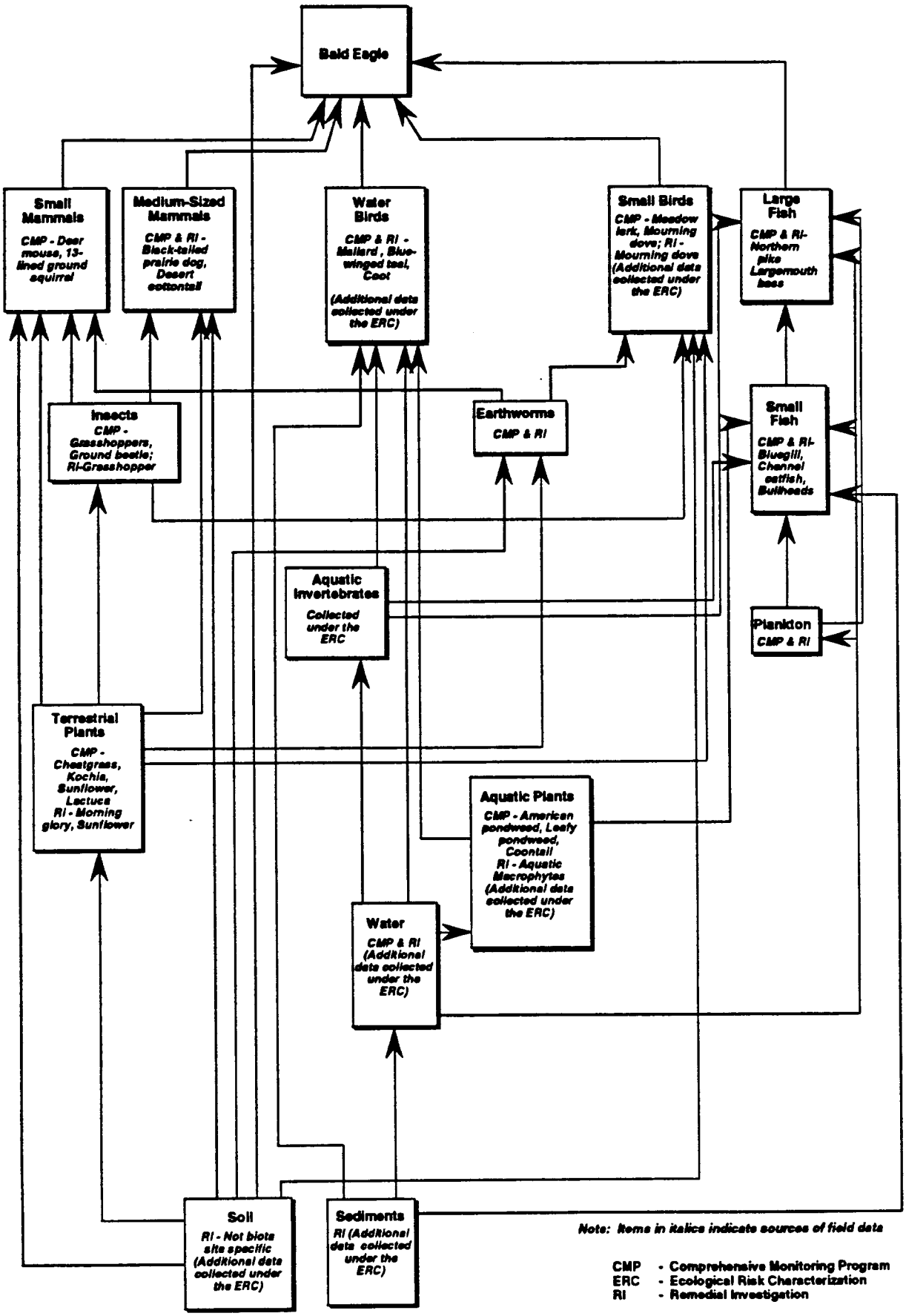
ppm = parts per million
BCRL = below certified reporting limit



RMA IEA/RC App C 6.94 jb

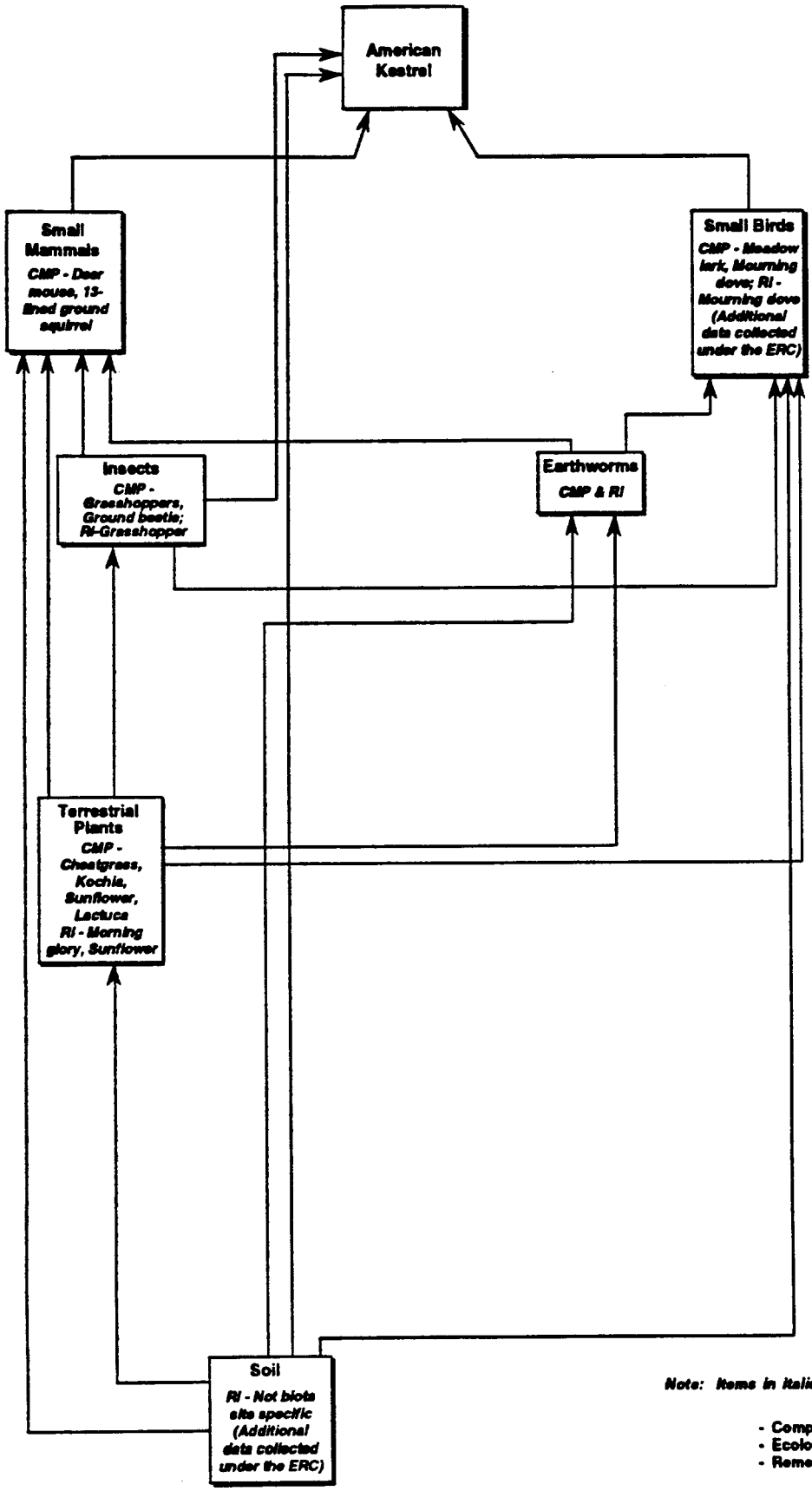
Figure C.1- 1
Considerations in Calculation of Potential Risk

Rocky Mountain Arsenal
Prepared by: Ebasco Environmental Incorporated



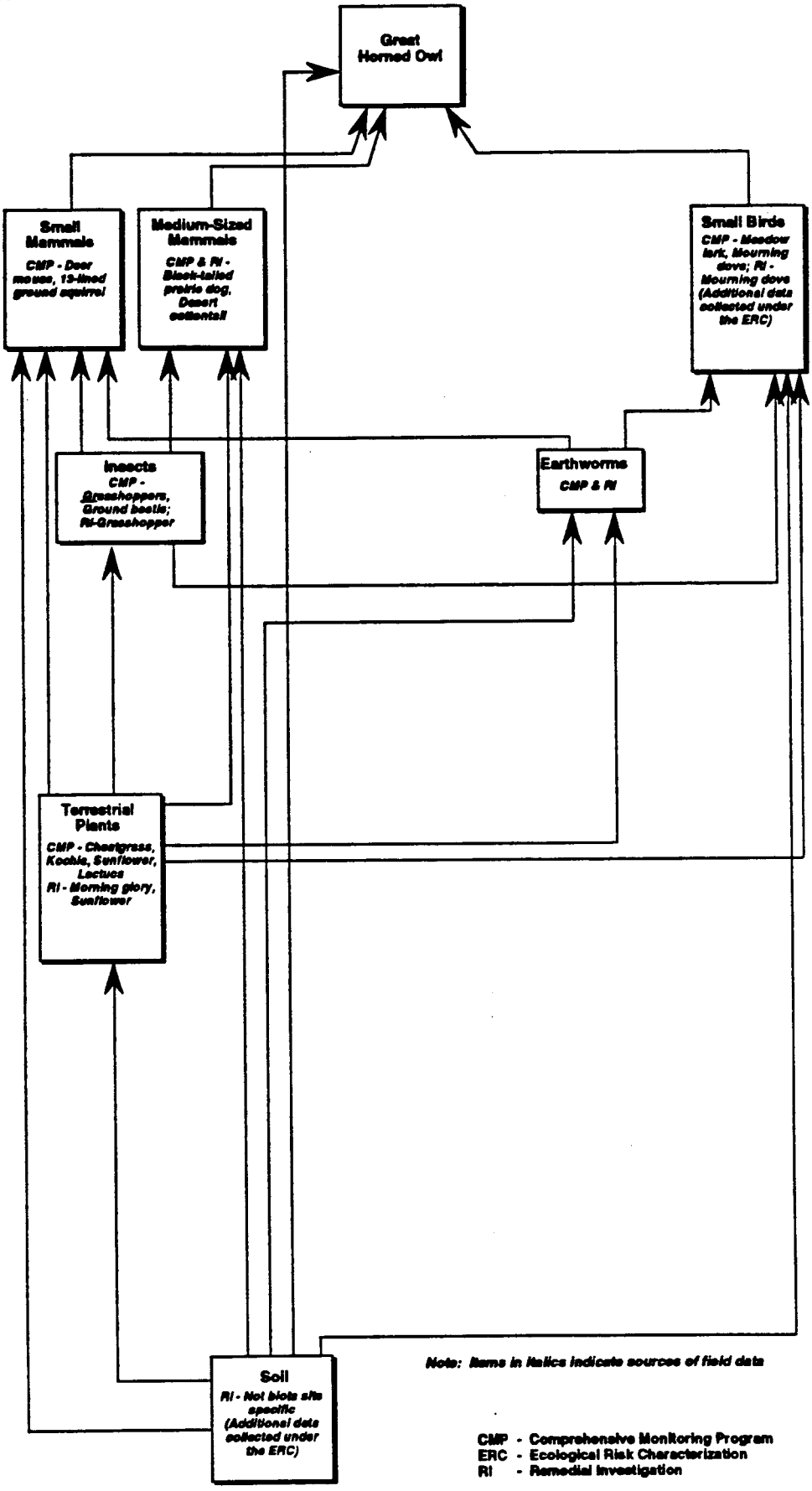
RMA IEA/RC App. C 2.94.jb

Figure C.1-2 • Bald Eagle Food Web
Rocky Mountain Arsenal
Prepared by: Ebasco Services Incorporated



RMA IEA/RC App. C 2.94.jb

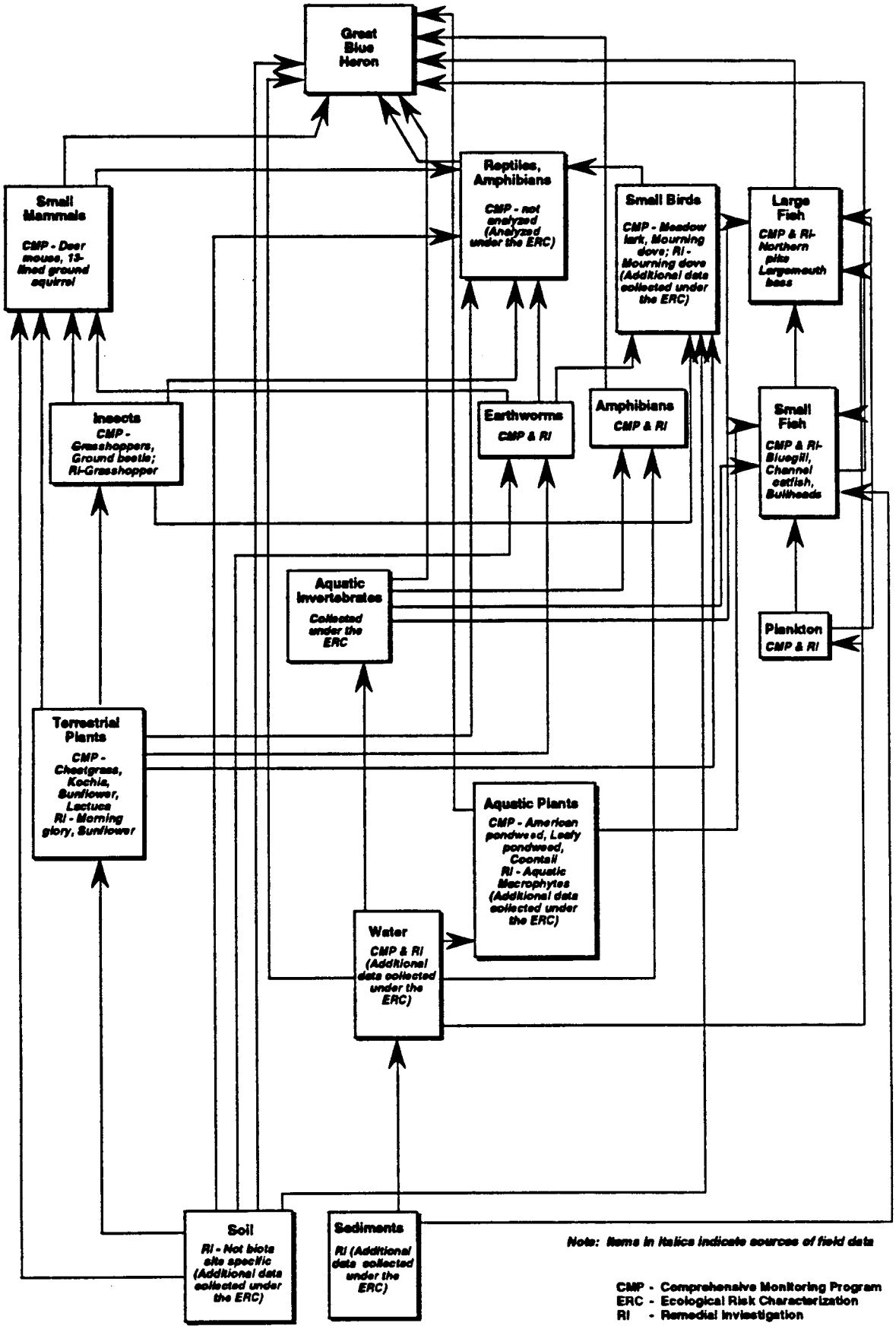
Figure C.1-3 • American Kestrel Food Web
Rocky Mountain Arsenal
Prepared by: Ebasco Services Incorporated



RMA IEA/RC App. C 2.94.jb

Figure C.1-4 • Great Horned Owl Food Web

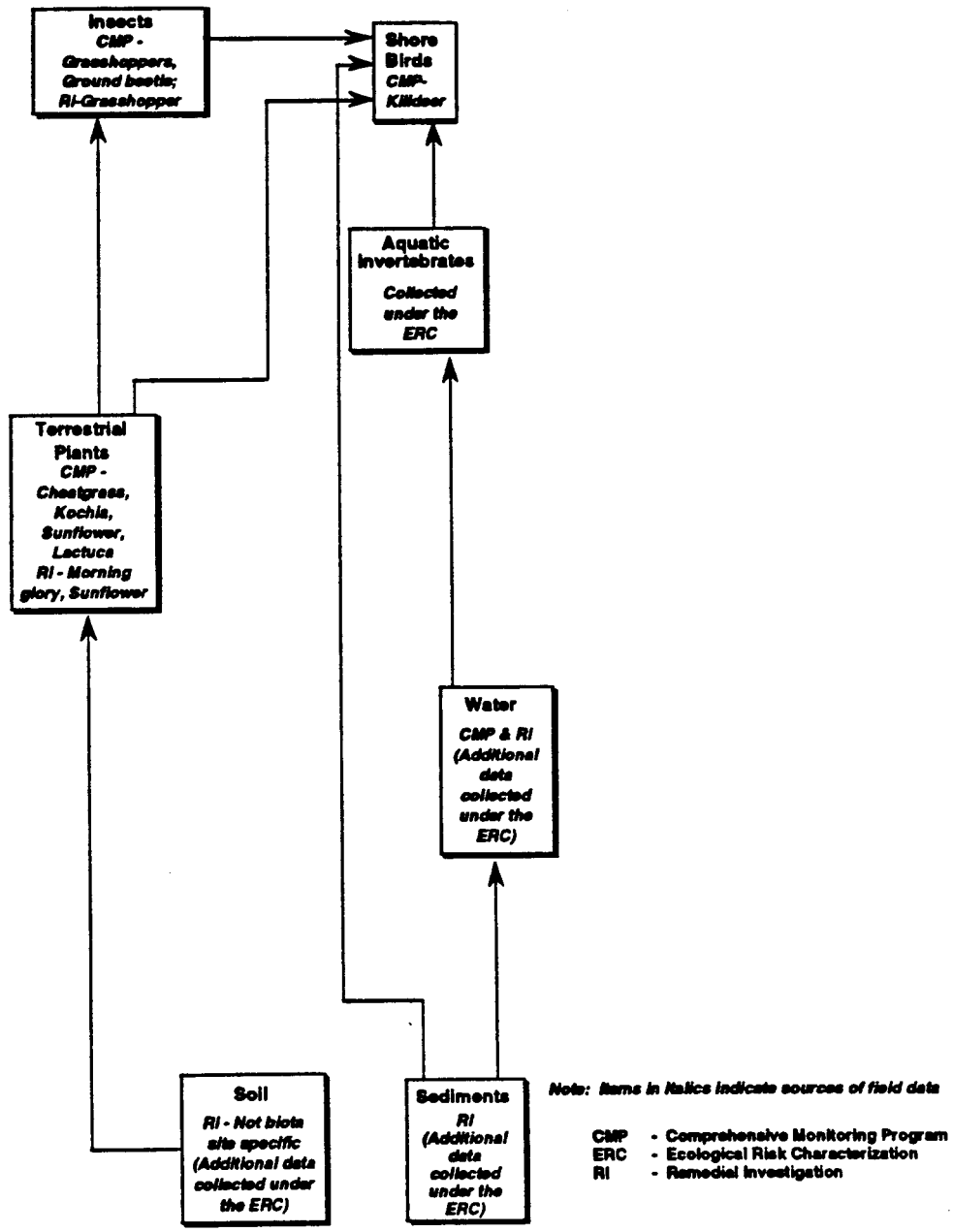
Rocky Mountain Arsenal
Prepared by: Ebasco Services Incorporated



RMA IEA/RC App. C 2.94.jb

Figure C.1-5 • Great Blue Heron Food Web

Rocky Mountain Arsenal
Prepared by: Ebasco Services Incorporated
Revised August 5, 1993



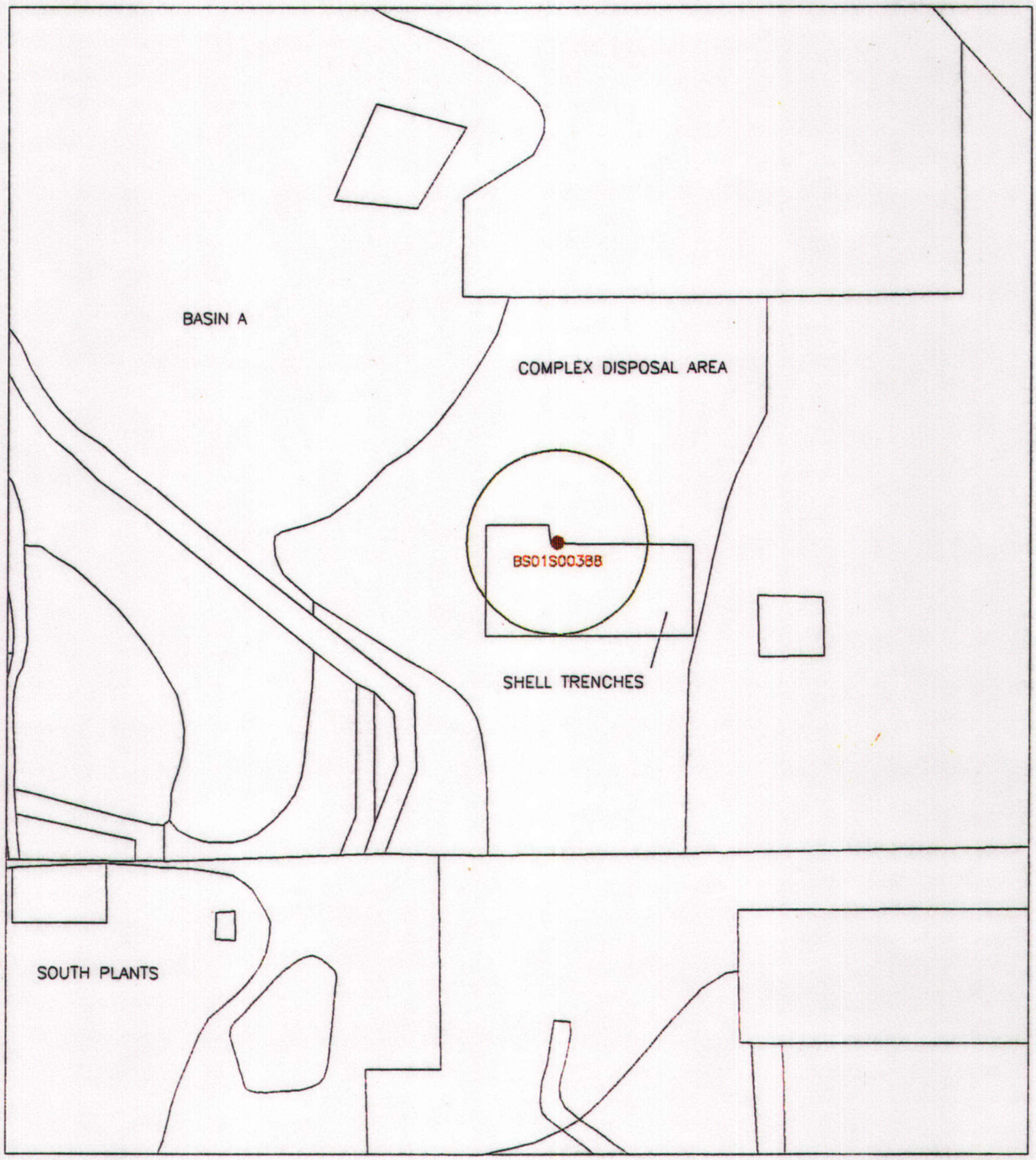
RMA IEA/RC App. C 2.94.jb

Figure C.1-6 • Shorebird Food Web

Rocky Mountain Arsenal
Prepared by: Ebasco Services Incorporated
Revised August 5, 1993

APPENDIX C
(SECTION C.1)

COMPUTATIONAL METHODOLOGY



Legend

SARsite Boundary

Tissue Sample Location

Exposure Area Circle

300 0 300 600 Feet

-N-

Prepared for: U.S. Army Program Manager for Rocky Mountain Arsenal

Figure C.1–7

Cottontail Tissue Sample Location and Corresponding Exposure Area

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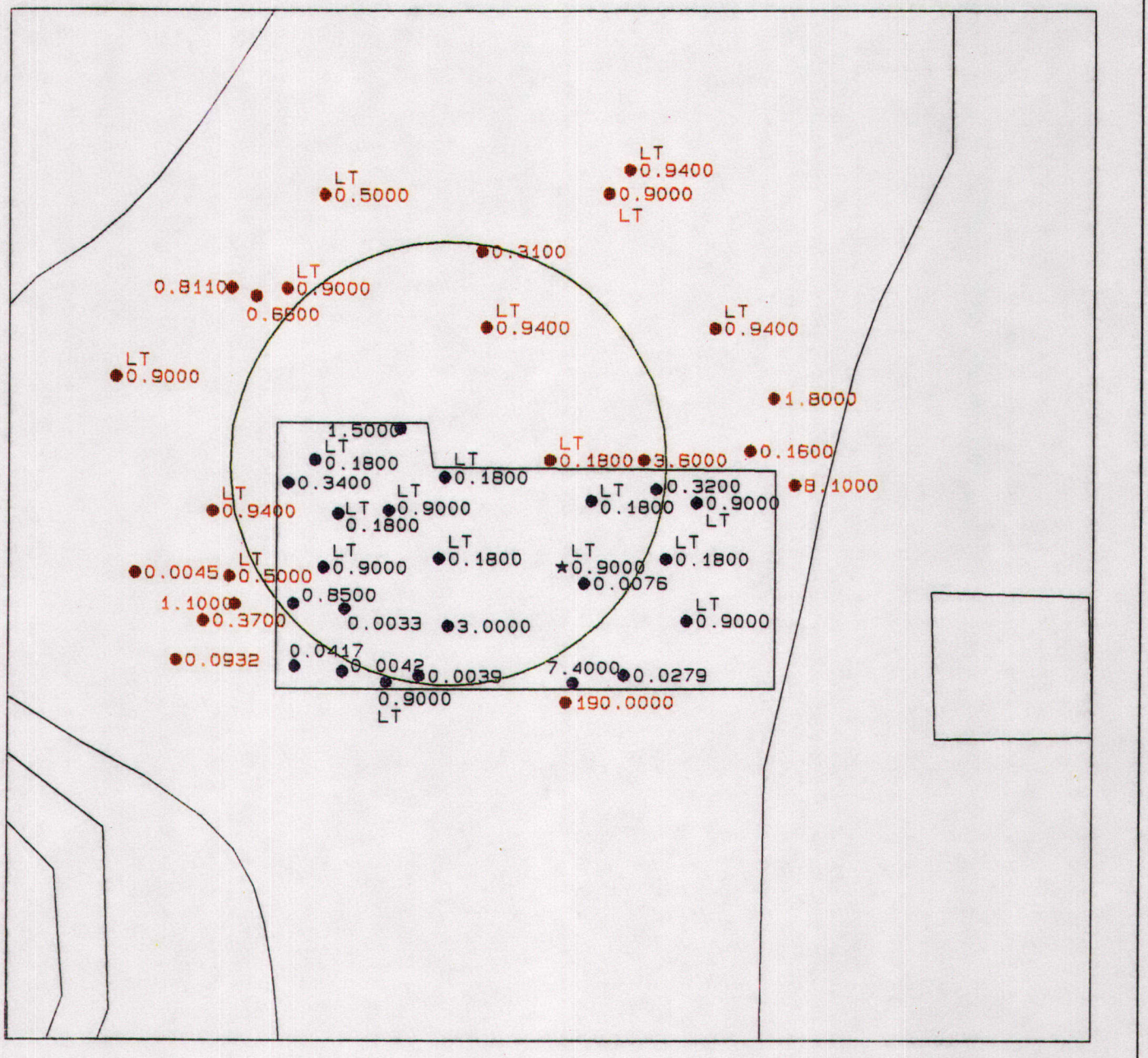
February 1994



Legend

SARsite Boundary

Soil Sample Locations

Exposure Area Circle

-N-

125 0 125 250 Feet

Prepared for: U.S. Army Program Manager for Rocky Mountain Arsenal

Figure C.1-8

Soil Boring Locations and Aldrin Soil Concentrations (ppm) (prior to BCRL spatial interpolation)

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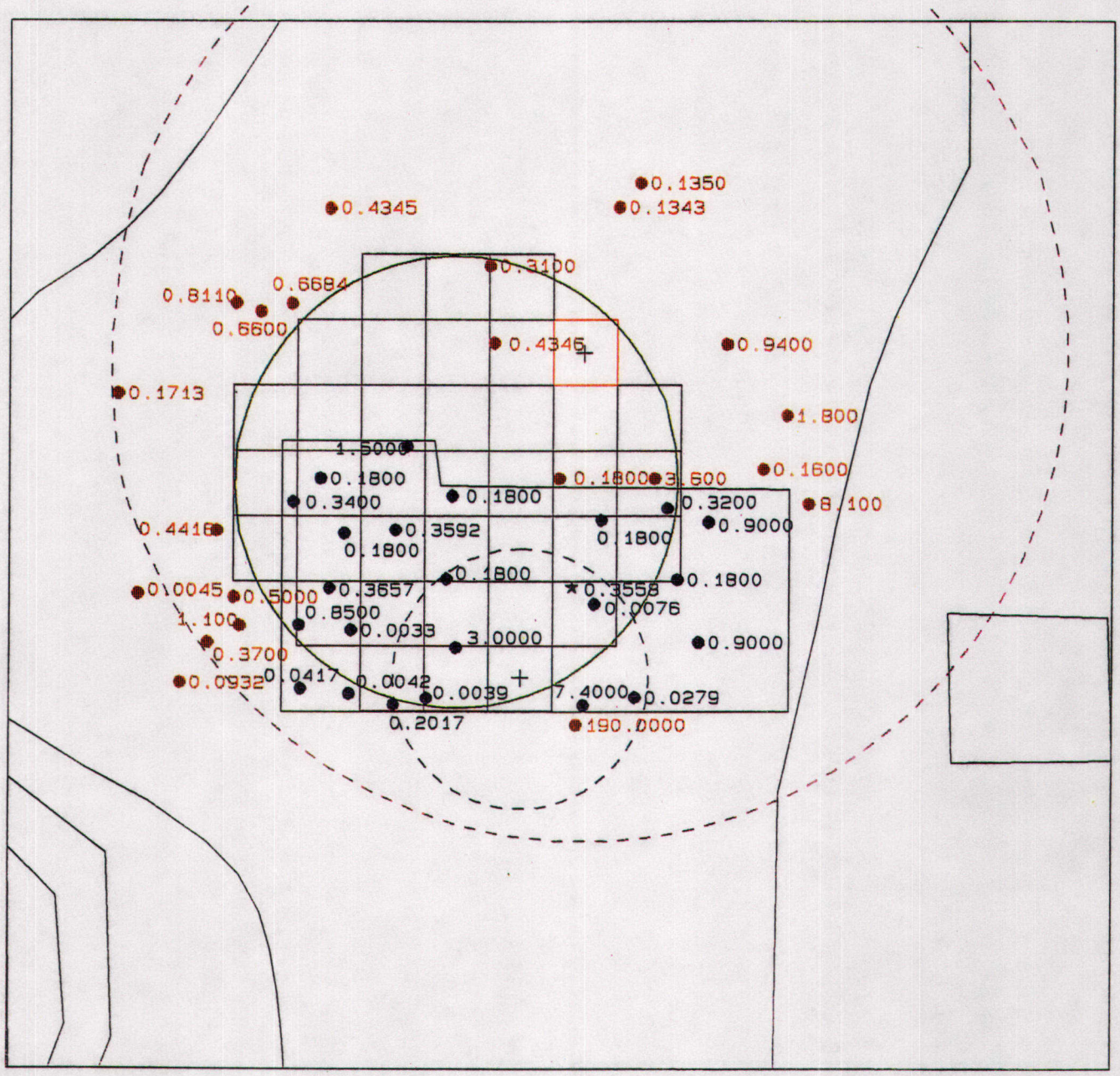
February 1994



Legend

- SARsite Boundary
- Search Radius
- Soil Sample Locations
- Exposure Area Circle

-N-

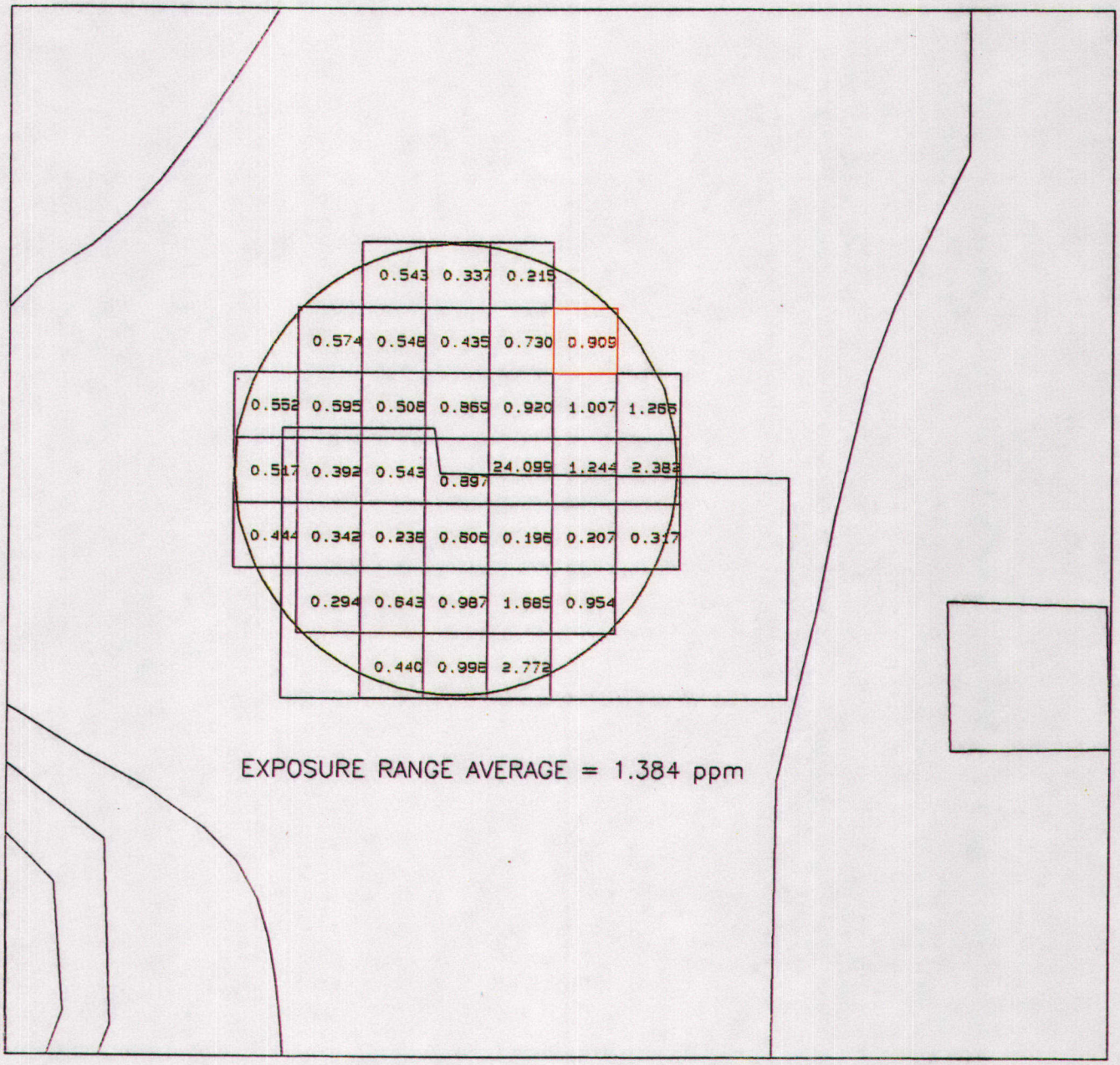
125 0 125 250 Feet

Prepared for: U.S. Army Program Manager for Rocky Mountain Arsenal

Figure C.1-9

Soil Boring Locations and Aldrin Soil Concentrations (ppm) (following BCRL spatial interpolation)

Ebasco Services Incorporated
February 1994



Legend

SARsite Boundary

0.909 Estimated Soil Concentration

Exposure Area Circle

125 0 125 250 Feet

-N-

Prepared for: U.S. Army Program Manager for Rocky Mountain Arsenal

Figure C.1–10

Modeled Grid Soil Concentrations for Aldrin within the Cottontail Exposure Area (ppm)

Ebasco Services Incorporated
February 1994

APPENDIX C

(SECTION C.2)

ECOLOGICAL RISK CHARACTERIZATION PARAMETERS

TABLE OF CONTENTS

LIST OF TABLES

LIST OF ACRONYMS AND ABBREVIATIONS

% bw/d	percent body weight per day
µg/l	micrograms per liter
AWQC	Ambient Water Quality Criteria
BAF	bioaccumulation factor
BCRL	below certified reporting limit
BMF	Biomagnification factor
$C_{(0)}$	initial tissue concentration
$C_{(t)}$	tissue concentration at time t
COC	contaminant of concern
CPMS	chlorophenylmethyl sulfide
$CPMSO_2$	chlorophenylmethyl sulfone
CRL	certified reporting limit
DBC	Dose-based criteria
DBCP	dibromochloropropane
DCPD	dicyclopentadiene
DDE	dichlorodiphenyldichloroethene
DDT	dichlorodiphenyltrichloroethane
e	base of natural logarithm system (2.7168)
EA	Endangerment Assessment
EPA	U.S. Environmental Protection Agency
ERC	Ecological Risk Characterization
<ESC>	Estimated exposure area soil concentration
ESE	Environmental Science and Engineering
FEL	Frank effect level, the concentration causing obvious adverse effects
FR	dietary fraction
HHRC	Human Health Risk Characterization
HQ	hazard quotient
HSDB	Hazardous Substances Data Bank
IEA/RC	integrated endangerment assessment/risk characterization
IRIS	Integrated Risk Information System
kg/kg-bw/d	kilograms per kilogram of body weight per day
LD_{50}	lethal dose to 50 percent of population (micrograms contaminant per gram body weight)
LOAEL	Lowest Observed Adverse Effect Level (micrograms contaminant per gram tissue or body weight)
m	meters
m^3/hr	cubic meters per hour
MATC	maximum allowable tissue concentration (micrograms contaminant per gram tissue)
mg/kg/d	milligram per kilogram body weight per day
mg/kg	milligram per kilogram
N	number

NOAEL	No Observed Adverse Effect Level (micrograms contaminant per gram tissue or body weight
OAS	Organizations and State
PBC	tissue-based criteria
R	feed rate (grams food per grams body weight per day)
RfD	reference dose
RI	Remedial Investigation
RMA	Rocky Mountain Arsenal
RRCs	regulatory risk criteria
SEP	Standard Evaluation Procedure
STSC, Inc.	STATGRAPHICS
TRV	toxicity reference value (milligrams contaminant per kilogram body weight per day)
UF	uncertainty factor
USFWS	U.S. Fish and Wildlife Service

C.2 FOOD-WEB MODEL INPUT PARAMETER DEVELOPMENT SUMMARIES

The tissue- and dose-based approaches both use the biomagnification factor (BMF) to predict biomagnification in biota at Rocky Mountain Arsenal (RMA). For species having measured tissue data from RMA, the BMF was calculated as the ratio of measured tissue concentration to <ESC> using three different approaches (Army, Shell, and EPA approaches) as described in Appendix C.1. For the top predators (bald eagle, great horned owl, American kestrel, and great blue heron), appropriate measured tissue concentrations were not available, and the food web model described in Appendix C.1 was used to calculate BMF. When applied to the tissue-based approach, the food web model used prey BMF values, bioaccumulation factor (BAF), and dietary fraction (FR), to calculate predator BMFs. This approach then used the predator BMF and <ESC> values to calculate its tissue concentration for comparison with its literature based MATC value. When applied to the dose-based approach, the food web model used prey BMF values, dietary fraction, feed rate (R), and <ESC> to calculate dose for comparison with its literature based TRV values. The various parameters used in the food web model for these two approaches, as well as the exposure range value used to calculate <ESC> were quantified with data from the literature or, in some cases, from RMA-specific data.

This appendix documents the way in which each of the parameters used in the food web model (BAF, R, and FR) and in the calculation of exposure (<ESC>) and of risk (MATC and TRV) were quantified. In the sections that follow, each of these parameters is also defined and characterized; a characterization of the database available for each parameter is included, as well.

The parameters used in the tissue-based approach were quantified for aldrin/dieldrin, DDT/DDE, endrin and mercury. The parameters used in the dose-based approach were quantified for these same COCs as well as for arsenic, cadmium, chlordane, copper, chlorophenylmethyl sulfide (CPMS), chlorophenylmethyl sulfone ($CPMSO_2$), dibromochloropropane (DBCP), and dicyclopentadiene (DCPD).

It should be noted that the approach to calculating potential risk has gone through a substantive evolution during the preparation of this document. Initially, an updated, probabilistic, and

calibrated version of the approach used in the Biota RI was used for all trophic boxes in the IEA/RC. This meant that the food web model was used to predict BMF values from parameters quantified by the literature for trophic boxes having strictly terrestrial, strictly aquatic, and mixed food chain input. These literature-based BMFs were then calibrated with BMFs calculated from site-specific measured tissue data. The calibrated BMFs were used to calculate criteria for soil, sediment, and water to serve as benchmark values against which concentrations in these media could be compared to calculate potential risk. However, this approach did not adequately account for the averaging of contaminant concentrations that biota do as they move and feed throughout the range of their exposure. With the introduction of spatial averaging considerations to better represent the variability of exposure, the concept of a media criterion became less useful. This is because a criterion represents the average concentration within an exposure range and concentrations at individual locations may be higher or lower than the criterion, so long as collectively they do not result in potential risk. Further, use of the food web model and criterion development for strictly aquatic trophic boxes was forced to use conversions between concentrations in sediment and those in water because CRLs for water were higher than calculated criteria and most concentrations in water were BCRL. This meant that calculation of potential risk via a criterion that was based on BMFs for strictly aquatic trophic boxes was very uncertain. Finally, some parameter literature data were lacking for many strictly aquatic trophic box/chemical combinations. Changes in the initial approach occurred as a result of all of these considerations. The outcome of these changes is the process presented in the IEA/RC, which maximizes the use of site-specific data and minimizes the uncertainty relative to the various types of implementation considered. Appendix E documents the still considerable uncertainty that remains.

The remainder of this section explains the literature search that was done for each of the parameters used in the tissue-based and dose-based approaches. Also presented is the basis for the initial decision to make some of the parameters deterministic (i.e., represented by a single fixed value) and some probabilistic.

Literature Search

Initially, an effort was made to limit the toxicological literature search to those species found at RMA (see the Biota Remedial Investigation [RI] report [ESE 1989] for the complete species list). However, because toxicological references or review articles did not always identify species in key words, the final search was guided by using specific COCs and input parameters as key words (Tables C.2-1 and C.2-2). Articles on obviously inappropriate taxa (e.g., marine species and large, domesticated mammals) were neither reviewed nor incorporated into this report.

Potential literature references were identified and compiled using references from the Biota RI report; Environmental Science and Engineering's (ESE's) in-house BIOTA.BIB database; the RMA Resource Technical Information Center; the U.S. Fish and Wildlife Service (USFWS) Patuxent Wildlife Research Center; DIALOG, a computerized information system; secondary references, and recommendations from the Organizations and State (OAS). References from the Biota RI report and BIOTA.BIB contained the results of the ESE literature search for the Biota RI, including the results of a search of Biosis Preview database information (1969 to 1988) within the DIALOG system. The RMA-specific Resource Technical Information Center, located at RMA, provided articles that were identified from a printout of its card catalog. The USFWS Patuxent Wildlife Research Center has a library of toxicological literature from which requested references on toxicity of the COCs were obtained. Studies evaluated for the TRVs derived for the Off-Post Endangerment Assessment (EA) (HLA 1993) were also considered. In addition, a literature search from 1989 to 1993 was conducted for TRV information on cadmium and copper because these elements were not evaluated in the Off-Post EA. Toxicological studies on cadmium and copper prior to 1989 were reviewed during development of the toxicity assessments for the Biota RI report. The Integrated Risk Information System (IRIS) (EPA 1993) and the Hazardous Substances Data Bank (HSDB) (NLM HAZ 1993), both U.S. Environmental Protection Agency (EPA)-supported electronic databases, were searched for additional information.

DIALOG is a computerized information system containing approximately 400 databases, of which approximately 24 pertain to medicine and bioscience subject categories. Although DIALOG references go back as far as 1908, the system was only searched for the period 1985–90.

DIALOG search strings were developed in consultation with a librarian experienced with the DIALOG search protocols. Only articles written in English were considered.

The search strategy varied between toxicological parameters (BAF, BCF, α, K2, MATC, and TRV) and ecological parameters. For the biota/chemical parameters, the following items were searched for in the Biosis Preview database:

- MATC or TISSUE CONCENTRATION or TISSUE TOXIC or ASSIMILATION or UPTAKE or DEPURATION or LOSS or BIOCONCENTRATION or BIOACCUMULATION or BIOMAGNIFICATION or SOIL INGEST or SOIL FEED or SOIL FOOD or DIET or SOIL EATING.

These items were searched in combination with all nonhuman citations.

For ecological parameters (R, FR, and ER), the following items were searched for in the Zoological Record database:

- FOOD or PREY AVAILABILITY or HABITS or PREFERENCE or SELECTION or PREY ITEM or PREY RESOURCE or PREY UTILIZATION or DIET or FEEDING or FEED or FORAGE or FORAGING.

These items were searched in combination with each of the species sampled to represent the trophic boxes but with the food habits string limited to title or descriptor field and the species limited to title or taxonomic field.

The review of the literature sources listed above resulted in the identification of approximately 2,800 references. These are listed in the dBASE bibliographic file (ECOREFS.EXE) included on the diskette in Appendix Section C.7. The diskette file represents the literature considered and the results of all the literature searches performed including the literature for parameters that are no longer used in the final implementation of the tissue- and dose-based approaches. The literature pertinent to α, K2, and BCF is still included in ECOREFS.EXE, but these values were not used because potential risk was calculated directly from the more certain measured tissue concentration data.

Each literature reference reviewed was given an unique identification number and a single-letter prefix that identified its source. Hardcopies of articles were obtained from various libraries, photocopied, and marked with the unique identification number. Following the initial literature search, additional articles were added as they were identified from various sources. In addition to general articles on exposure range, food webs, toxicity, and feed rate, to name a few, there are many chemical-specific citations. For example, approximately 358 sources provided information on aldrin/dieldrin, approximately 355 sources (several of which provided information on more than one chemical) were selected for information on DDT/DDE, approximately 176 sources were reviewed for information on endrin, and another 422 sources were reviewed for mercury. The 2,800 references are chronologically grouped as follows:

1900–30 =	8
1931–40 =	10
1941–50 =	24
1951–60 =	107
1961–70 =	265
1971–80 =	894
1981–90 =	1,422
1991–93 =	71

All the articles, (or in the case of DIALOG abstracts), compiled in the dBASE citation files were considered, and those that appeared pertinent were tagged for hardcopy retrieval. Each article was selected for review based on its pertinence to populations of biota or habitats at RMA, to the parameters or other measures of toxicity listed in Table C.2-1, and for the toxicological parameters, its reference to the Ecological Risk Characterization (ERC) COCs.

Information recorded from each article on toxicological parameters included: species on which data provided, value and units given, chemical and parameter addressed, trophic box to which applicable, literature citation information, comments on derivations or calculations needed, and where appropriate, associated toxicological endpoints. Articles providing information on toxicological parameters were distributed to individual reviewers who were responsible for summarizing data for individual COCs. In this way, one reviewer would review all articles for one COC, thus ensuring consistency and accuracy in data interpretation as well as a thorough

understanding of the literature pertaining to a particular chemical. Reviewers met often to discuss parameter derivation to ensure consistency across a chemical group as well as for an individual COC. Information recorded from each article on ecological parameters included the applicable model parameter and model trophic box or species, the value and units of the parameter, location of the study, season(s) covered by the study, life stage of the species studied, the food item(s), sample size, and comments. To achieve final agreement on the values for BAF, FR, R, MATC, TRV, and ER, an extensive consensus-building process among the participating scientists representing the OAS was used. The final values that were used in the IEA/RC are provided in Table C.2-3.

A perusal of Table C.2-3 and Sections C.2.1 to C.2.5 reveals that some of the input parameters are expressed as fixed values and some are expressed as distributions. The initial decisions as to whether a parameter should be fixed or stochastic were based on the results of an importance analysis. The importance analysis was performed on a set of deterministic values representing the literature input parameter estimates considered to be the best for a deterministic version of the terrestrial and aquatic model equations. This importance analysis was designed to provide information regarding the relative influence of an individual input parameter's value to the model output value. Importance was determined by the magnitude of change in the final result of the deterministic model equations attributable to changing the value of one input parameter and leaving the others constant. The rationale for this approach is as follows.

If:

$$A = X * Y * Z \quad (1)$$

$$A' = X * 2Y * Z \quad (2)$$

then:

$$\% \text{ change caused by doubling } Y = \frac{A - A'}{A} * 100 \quad (3)$$

The importance analysis compared the results from the systematic adjustment of all model input parameters. In addition, the importance analysis provided information on the parameters that were high contributors to the overall uncertainty of model output.

A ranking system was developed to rank the importance of each input parameter for each top-level trophic box and COC in the terrestrial and aquatic models. Four importance categories were assigned: 1 = important parameter for 90 to 100 percent of COCs, 2 = important parameter for 40 to 90 percent of COCs, 3 = important parameter for 10 percent of COCs, and 4 = not important for any COC. The aggregate rankings of a parameter and its variability among the four top-level trophic boxes evaluated were used to decide which input parameters to fix as deterministic values (i.e., those that did not have much impact on model output or uncertainty) and which input parameters to treat stochastically (i.e., those that did have impact on model output and/or uncertainty) in the probabilistic terrestrial and aquatic models.

The final importance categories that resulted from analysis of the four food webs (i.e., bald eagle, great blue heron, great horned owl, and American kestrel) are shown in Tables C.2-4 and C.2-5 for those parameters that are used in the final ERC protocols. Based on these results, all three of the input parameters for terrestrial food chains (FR, BAF, and MATC) were to be stochastic. As a result of decisions that are explained in the appropriate sections of this appendix, some parameters originally intended to be stochastic later became fixed parameters (MATC, FR); TRV was also used as a fixed parameter.

C.2.1 BIOACCUMULATION FACTOR

C.2.1.1 Parameter Definition and Characteristics

Bioaccumulation of environmental contaminants in food chains, with subsequent biomagnification, has been recognized and documented in the literature for more than three

decades. Bioaccumulation is affected by many of the same factors as bioconcentration and each COC behaves differently within food chains. For example, metal contaminants exhibit different BMFs and BAFs in ecosystems depending on their concentration. Biomagnification is defined as the amplification of a contaminant's concentration from the initial source (e.g., water, soil, or sediment) to a specified target species or trophic level. The BMF reflects uptake and transfer between trophic levels, resulting from both bioconcentration and bioaccumulation, in conjunction with other parameters. BMF can be calculated for each specified organism level or trophic box above the first trophic level.

The BAF is defined as the ratio of the concentration of a contaminant in an organism to the concentration in its diet. Contaminant concentrations were provided for whole-body homogenate samples where data are available and otherwise for specific tissues (e.g., fat, liver, brain) and labeled as such. For terrestrial organisms, BAFs were taken directly from the literature, or when no direct literature values were available, they were calculated as the ratio of tissue concentrations as follows:

$$BAF_j = \sum_{i=1}^{n} \left(\frac{C_j}{C_i} * FR_i \right) \quad (4)$$

where: j = predator's trophic level

i = trophic level for n prey items

For birds with aquatic food chains, the BAF was defined as for terrestrial organisms.

C.2.1.2 Database Characterization

C.2.1.2.1 Literature Description

In the introduction to this section, literature search procedures for BAF were provided that were further detailed in Tables C.2-1 and C.2-2. The final BAF values are presented in Table C.2-6. Characteristics of the specific values and how the BAF values were used are summarized below.

For the bioaccumulative chemicals such as the organochlorine pesticides and mercury, articles were sought that quantify the amount of bioaccumulation for different trophic components. The articles used to derive the BAF values cover various species of terrestrial plants, invertebrates, birds, reptiles, insects, and mammals. The BAF is a direct model input parameter for terrestrial food chains. The BAF values included in Table C.2-6 were taken directly from literature sources.

There are several limitations that affect the ability of the literature data to precisely quantify BAF. For example, the paucity of steady-state data for the contaminants that equilibrate slowly is one factor; many studies were terminated at a given time even if equilibrium was not achieved. Only equilibrium values were used whenever possible. Site-specific food habits data and site-specific environmental factors (e.g., pH, water temperature) are also factors that may affect BAF values. Variation in these factors across studies could result in values seemingly aberrant from others (i.e., outliers).

In addition, many authors failed to mention whether their experiment was performed under steady-state conditions, which is necessary to obtain accurate BAF values. Often papers were not selected because authors failed to report sample size, use of controls, raw data, or statistical results. Furthermore, some BAF values were taken from studies that began several years after contaminant application ("aged" contaminants), while others involved direct dosing of the media or prey item ("fresh" contaminants). Both types of values were used as there were too few studies that examined only the impacts of "aged" chemicals. Since "fresh" contaminants may be more bioavailable than "aged" contaminants (MRI 1991), the BAFs may slightly overestimate risk at RMA. However, this potential is expected to be quite small. Finally, several of the values needed to be converted to the corresponding wet-weight basis. Data were assumed to be on a wet-weight basis and to have reached approximate equilibrium conditions when the authors failed to specify otherwise and data appeared so reasonable as to make this assumption, based on comparison with other values known to be wet-weight and at equilibrium and from other papers by the same author where this information was given, a valid one. As always, the use of an assumption increases the uncertainty in a parameter distribution. However, only values deemed

acceptable on the basis of professional judgment calibrated to field data were used, which reduces uncertainty.

C.2.1.2.2 Data Variability

BAFs generally spanned less than one order of magnitude within a trophic box. Exceptions to this trend include the earthworm trophic box for endrin that spanned 2.5 orders of magnitude and the great blue heron trophic box for mercury that spanned 1.4 orders of magnitude. The range of variability in literature BAF values is considered small in light of species variability and other variable factors. For example, reported values for each parameter are affected by the time at which the measurement is taken (i.e., whether the organism has reached equilibrium with its immediate environment), although only equilibrium values were used whenever possible. Other factors contributing to BAF are mentioned below.

C.2.1.2.3 Final Value Selection and Assessment

All final values represent the best values available in the literature and were accepted based on a consensus of the participating scientists. Long-term studies or studies that indicated a steady-state condition was achieved were used whenever possible. However, studies of short duration were accepted for BAF values when the exposure duration was appropriate for the species under consideration (short-lived organism) or when studies of chronic duration were lacking.

Tissue-specific and carcass values were converted to whole-body measurements when possible. A factor of 1.3 was used to convert from carcass to whole body, and a conversion factor of 0.6 was used to convert from egg to carcass on the basis of data from Wiemeyer et al. (1986) at the U.S. Fish and Wildlife Service Patuxent Wildlife Research Center. The fat to whole-body conversion factor was 0.2. Values given on a dry-weight basis were converted to wet-weight contaminant measurements when possible. Conversion factors were frequently based on data presented in Wiemeyer et al. (1986), although a conversion factor from Gish (1970) was used when paper-specific values were unavailable. Conversion factors from other sources are noted in Table C.2-6.

Acceptable mercury BAF values could not be located in the scientific literature for the insect, earthworm, and reptile trophic boxes. RMA field data were used to derive a BAF for mercury; accordingly, there is no calibration of the model for these trophic boxes. Likewise, no acceptable literature data could be found for endrin in plants or reptiles. Because the toxicity of endrin is relatively similar to that of aldrin/dieldrin, it was deemed appropriate to use aldrin/dieldrin BAFs to represent these trophic levels for endrin.

The data points presented in Table C.2-6 were used to develop data distributions for BAFs for use in the ERC model. Data for distribution development were not grouped ("lumped") unless deemed appropriate by the availability of data (e.g., all bird values for endrin were combined for each bird trophic box), and as determined by the results of the importance analysis of the model input parameters, distributions for BAF values were developed. The methods used to develop distributions for these parameters are summarized below.

C.2.1.3 Distribution Development

The Army used statistical information from the consensus papers, when available, to develop the BAF distribution for each trophic box. Statistical information available in the papers ranged from a standard error about the mean tissue concentration (e.g., Rudolph et al. 1983) to more detailed information such as means and standard deviations about the mean for both male and female tissue concentrations as well as the variance in the dose (e.g., Mendenhall et al. 1983). Statistical information was available for BAFs in the consensus papers for all trophic boxes except insect, small and medium mammal, and reptile for aldrin/dieldrin and reptile for DDE.

C.2.1.3.1 Criteria Used

The combination of statistical information from the literature (whenever available), STATGRAPHICS (when $n \geq 4$), and comparison with other chemicals and trophic boxes was used to determine the standard deviation and distribution type for each trophic box and each chemical. The amount of variability assigned to the distribution was determined based on comparison with like chemicals (e.g., dieldrin for endrin) and closely related trophic boxes (e.g., small fish for large fish).

Generally, when the sample size was two and no statistical information was presented in the papers, a uniform distribution was selected and the consensus values were used as the endpoints. However, if descriptive data from the paper or comparison with other chemicals indicated a high likelihood of substantial variability, then the range was expanded above and below the consensus values. The mammal and reptile trophic boxes had only two values and so were often assigned a uniform distribution (13 out of 72 possible trophic box/chemical combinations).

When the sample size was three and there was no statistical information presented in the consensus papers, a uniform, triangular, or lognormal distribution was assigned, depending on the spread of the values and comparison with other chemicals. Only four trophic box/chemical combinations fell into this category. When sample size was four or greater and statistical information was not available in the papers, STATGRAPHICS was used to assign the distribution type. Nineteen trophic box/chemical combinations fell into this category. For aldrin/dieldrin, the earthworm, terrestrial plant, and plankton trophic boxes contained more than four consensus values (n = 30, 7, and 6, respectively). Only two trophic boxes, insect and earthworm, contained more than four consensus values for endrin (n = 14 and n = 5, respectively). There were 4 trophic boxes containing 4 to 18 consensus values for DDE (bald eagle, insect, earthworm, and terrestrial plants). For each of these trophic box/chemical combinations, an estimated distribution was assessed using the Kolmogorov-Smirnov goodness-of-fit test. When the test indicated that the best fit was a normal distribution, but the data ranged close to zero, the next best fit was chosen, usually lognormal or uniform, to avoid sampling negative values.

When the standard deviation about the mean dose was known from the literature, the following formula was used to calculate the variance in the BAF when one study was recommended for use for a particular trophic box:

$$(S_{BAF,pred})^2 = X_{BAF,pred} * \left[\left(\frac{S_{pred}}{X_{pred}}\right)^2 + \left(\frac{S_{prey}}{X_{prey}}\right)^2\right] \quad (5)$$

where: $S_{BAF,pred}$ = Standard deviation about the mean BAF for the predator

s_{pred} = Standard deviation about the mean tissue concentration in the predator
s_{prey} = Standard deviation about the mean tissue concentration in the prey
$x_{BAF,pred}$ = Mean BAF for the predator
x_{pred} = Mean tissue concentration in the predator
x_{prey} = Mean tissue concentration in the prey

When the variance in the mean tissue concentration in the prey was not given, the variance about the BAF was assumed equal to the variance about the mean tissue concentration in the predator. When more than one reference was used to develop the mean BAF for a particular trophic box (i.e., mean of means), the following formula was used to compute the total standard deviation (S_T) about the mean of means:

$$S_T = \left[\frac{\left[(n_x-1) * S_x^2\right] + \left[(n_y-1) * S_y^2\right] + \left[(n_z-1) * S_z^2\right] + \ldots}{(n_x-1) + (n_y-1) + (n_z-1) + \ldots} \right]^{1/2} \quad (6)$$

where each subscript (x, y, z) denotes the study from which the data came, S^2 denotes the variance about the mean from the specified study, and n stands for the number of individuals involved in the study. This formula was also used for values derived from individual studies if the mean tissue concentrations of males and females were given separately.

C.2.1.3.2 Distributions Developed

The following two examples illustrate calculation of the standard deviation about the mean BAF for the DDE and aldrin/dieldrin based on statistical information presented in the relevant consensus papers in which statistical information was provided. The first column of Table C.2-6 provides the same information (but in less detail) for each chemical/trophic box combination.

DDE

American kestrel: Consensus values of 7.7 and 29 were selected from Rudolph et al. (1983) and Wiemeyer et al. (1986), respectively. Rudolph et al. provided the following information: mean dose = 5.9 ppm, n = 6, mean carcass concentration = 35.3, and se = 1.9 (s = 4.7). Wiemeyer et

al. provided the following information in the paper: mean dose = 2.8 ppm, n = 3, mean carcass concentration = 40.3, and s = 34.7. The information from Wiemeyer et al. was based on data for birds that died as a result of exposure to pesticides. However, the BAF from this study was computed from data for sacrificed birds for which no statistical values were provided. The standard deviation for the mean BAF developed based on the information given above, using equation (5), was greater than the mean value from both papers. Thus, if a normal distribution had been chosen it would have included negative values (mean = 18.4, sd = 19). Therefore, a uniform distribution was chosen as most representative of the true distribution because there was no indication that lower values were any more likely to occur than higher values, as would have been inferred by a lognormal distribution, for this trophic box.

Great horned owl: The consensus value of 43.7 was selected from Mendenhall et al. (1983). Data presented in Mendenhall et al. included the following: mean dose = 2.83 ppm, se = 0.1, n = 10; female mean carcass concentration = 78, se = 0.34, and n = 6; mean male carcass concentration = 112, se = 0.056, and n = 9. To calculate the standard deviation about the mean BAF, several steps were required. First, standard deviations were computed from the associated standard errors and sample sizes ($s = se\sqrt{n}$). Then, the total standard deviation (S_T) associated with the mean of male and female carcass concentrations was computed using equation (6). Finally, the standard deviation associated with the mean BAF was calculated using equation (5). The Mendenhall et al. (1983) study had a mean BAF of 43.7 and a standard deviation of 0.55. It is highly unlikely that this very small standard deviation is a realistic measure of the variability in BAF for all great horned owls at RMA because of differences in species, size, and variability in field conditions. Hence, the four consensus values from the bald eagle trophic box were used to compute a pooled coefficient of variation (CV_p), which was multiplied by the mean owl BAF (x_{gho}) to generate a standard deviation about the mean for the owl trophic box ($CV_p * x_{gho} = s_{gho}$). The data are as follows: CV_p (bald eagle trophic box) = 0.6, x_{owl} = 43.7, and therefore, s_{gho} = 26.2.

Aldrin/dieldrin

Bald eagle: The consensus values of 10.5, 16, and 21.1 were selected from Wiemeyer et al. Braune and Norstrom (1989), and Mendenhall et al. (1983), respectively. Information provided in Wiemeyer et al. included the following: mean dose = 0.28 ppm, n = 6, mean carcass concentration = 1.64 ppm, and s = 1.2. This information was for birds that died; however, the BAF was computed from sacrificed birds for which no statistical information was available. Braune and Norstrom provided the following information: mean = 16, sd = 5.1, and n = 10. Mendenhall et al. provided the following information: mean dose = 0.58 ppm, se = 0.028, n = 10; mean female carcass concentration = 9.2 ppm, se = 0.12, and n = 7; mean male carcass concentration = 9.6 ppm, se = 0.17, and n = 12. Since only the carcass values had associated variability presented in Wiemeyer et al. the standard deviation about the BAF from this paper was assumed to be the same as that associated with the mean carcass concentration (1.2). Moreover, since the standard deviation about the mean BAF was reported by Braune and Norstrom, no further calculations were necessary to use the values from this paper. To calculate the standard deviation about the mean BAF from Mendenhall et al. the steps outlined above for the great horned owl under DDE were followed. The BAF and its associated variability derived from this approach is 21.2 ± 3.4. The three BAF values with associated uncertainty were then used to calculate the variability about the mean BAF for the trophic box, which is ± 3.9 about 15.9. Since the individual BAFs were approximately evenly spaced, the standard deviation was much less than the mean BAF and, since there was no information to suggest otherwise, a normal distribution was chosen for this trophic box/chemical combination.

C.2.2 FEED RATE AND DIETARY FRACTION

C.2.2.1 Parameter Definition and Characteristics

C.2.2.1.1 Feed Rate

The R is defined as the quantity of food ingested by an organism relative to its body weight per unit time kg/kg-bw/d. The feed rate is species-specific and varies with location, season, and the age, size, appetite, reproductive stage, and condition of the organism. Feed/rate is used in the dose-based approach because TRVs are expressed on a per-day basis.

C.2.2.1.2 Dietary Fraction

The FR represents the fraction of the biomass of total food ingested that is contributed by a given food item in the diet of a consumer species. As with feed rate, dietary fraction is species-specific and can vary with location, season, age, reproductive stage, and other individual characteristics of the organism. In addition, dietary fraction can also vary with habitat type since more adaptable species are opportunistic feeders and may use different food sources as they become available in various habitats. The dietary fraction parameter is used in both aquatic and terrestrial food chain equations.

C.2.2.2 Database Characterization

C.2.2.2.1 Literature Description

Literature search procedures for feed rate and dietary fraction were provided in the introduction to Section C.2, along with Tables C.2-1 and C.2-2, which further detail the search. The parameter-specific literature review for feed rate and dietary fraction is summarized below.

The articles reviewed during the literature search provided feed rate and dietary fraction values for raptors, including great horned owl, burrowing owl, barn owl, bald eagle, ferruginous hawk, and American kestrel, fish, waterfowl, small and medium birds, mammals, reptiles (here defined as reptiles and terrestrial amphibians), and insects. No information was found on feed rate or dietary fraction for earthworms. In the literature, feed rate data were generally expressed as, or readily converted to, kg/kg-bw/d. Values for feed rate were reported in terms of both wet and dry weights; however, only values based on wet weight were used in the model. The measurement units for dietary fraction in the literature included percent by volume, percent by occurrence, and percent biomass.

C.2.2.2.2 Parameter Quantification

Values from the literature were converted to the appropriate units. Feed rate values reported in the literature as total daily food intake were converted to a corresponding rate per unit body weight by dividing the reported value by the average body weight (in kilograms [kg]) for that species. Values such as percent body weight per day (% bw/d) were multiplied by the literature-derived average body weight of the species and divided by 100 to convert them to kg/kg-bw/d.

Wet-weight feed rate values were converted to kg/kg-bw/d. Dry-weight measurements were used only when they could be converted to wet-weight equivalents.

All of these measurement units were considered; however, percent biomass was used and converted from the most prevalent form, percent occurrence, in the literature because it is more representative of contaminant loads that are on a mass basis. To base dietary fraction on biomass rather than percent occurrence, a consistent conversion protocol was used. Dietary fraction based on biomass contributed by dietary items was calculated as the product of percent occurrence and representative weight divided by total average biomass (Table C.2-9). To develop average weights for a trophic box, probable RMA prey species in that trophic box were determined from site-specific food-item studies, the literature, or local experts. Next, where weight information was available, the average weights of prey species were calculated from the hundreds of individual weights for species collected on RMA under the Biota CMP. In the instance that body weight information was lacking from RMA, values were selected based on information from local experts, best professional judgment, and field guides, or estimated based on a portion of the predator's body weight. Since the sum of the food items in the diet should equal the total amount of food ingested, the dietary (prey) fractions were summed to 1.0 before the percent biomass of soils/sediment and water ingested were added. The total fractions were then adjusted to sum to 1.0.

The feed rate and dietary fraction values and citations for various trophic boxes are listed in Tables C.2-7 and C.2-8. The number of values for dietary fraction reflects the total number of dietary items consumed by a given trophic box in the model food web (e.g., bald eagle dietary items include water birds, medium mammals, small mammals, and small birds). In some cases, adjustments supported by best professional judgment were based on available knowledge of feeding rates or food habits of relevant or similar species at RMA, or similar study areas in eastern Colorado or the Great Plains.

C.2.2.2.3 Data Variability

Both feed rate and dietary fraction values spanned about two orders of magnitude across all trophic boxes (Tables C.2-7 and C.2-8). Within individual trophic boxes, values for both parameters spanned one order of magnitude and usually much less. For a parameter such as dietary fraction, with absolute boundaries of 0.0 and 1.0, variability is limited by definition. For any particular trophic box, the range of dietary fraction is usually described by a factor of 2 to 10. However, for each dietary item in a particular trophic box, the range of literature values seldom spans more than a factor of 2. Bald eagle and great blue heron have the highest values for feed rate.

Some of the variability is attributable to geographic and seasonal differences among studies (e.g., bald eagle studies in Alaska and coastal areas vs. those from inland areas). To minimize this variability, data were preferentially selected that represented habitat types and geographic areas similar to those at RMA. Further, site-specific data were used whenever available. Variation in reported values for feed rate and dietary fraction is affected by an organism's natural response to changes in its environment, such as seasonal changes. In addition, an individual's sex, life stage, and reproductive state affect these parameters.

C.2.2.2.4 Final Value Selection and Assessment

The merits of the literature data were evaluated in round table discussions. Preference was given to dietary information from geographic and habitat types similar to those at RMA. For those trophic boxes for which there was food-item data from RMA, the RMA-specific dietary fractions were used instead of literature values (Section C.4.2). For those trophic boxes for which no data was available for either feed rate or dietary fraction, best professional judgment based on information from taxonomically related organisms was used to select a value.

C.2.2.3 Distribution Development

C.2.2.3.1 Criteria Used

Based on a review of the literature, the species-specific parameters, feed rate and dietary fraction, are believed to be normally distributed. A normal distribution is typical of many biological variables such as morphometric characteristics and behavioral frequency measurements.

Initially, dietary fraction was to be treated as a probabilistic parameter. Closer inspection revealed that dietary fraction has very limited variability, usually within a factor of 2 to 10, for any trophic box. Finally, the use of site-specific data was facilitated by establishing dietary fraction as a fixed value.

Distributions for feed rate were fit using STATGRAPHICS. Normal distributions developed for all trophic boxes were assessed using the Kolmogorov-Smirnov goodness-of-fit test. A normal distribution fitted to feed rate data was not significantly different from the theoretical normal distribution. Other available model distributions did not produce as good a fit for the majority of feed rate data. Even when an alternative distribution produced a better fit, the developed normal distribution goodness-of-fit significance always was greater than 0.853. The taxon-specific nature of feed rate dictated development of separate distributions for each trophic box. Literature values were considered adequate if three or more data points were available.

C.2.2.3.2 Distributions Developed

Literature values for feedrate (Table C.2-8) were adequate for all trophic boxes: bald eagle, n = 7; great blue heron, n = 7; great horned owl, n = 7; and American kestrel, n = 7.

Values for dietary fraction were comprised from literature values, field studies (Section C.4.2), and professional judgment when either literature values or field values were not available.

C.2.3 MAXIMUM ALLOWABLE TISSUE CONCENTRATION

C.2.3.1 Parameter Definition and Characteristics

The tissue-based method used to estimate ecological risk from the bioaccumulative COCs involves the maximum concentration of contaminant in tissue that is unlikely to be associated with harmful effects. This concentration is referred to as the maximum allowable tissue concentration (MATC). Therefore, MATC, as used in this report, is defined as the whole-body tissue concentration that is unlikely to be harmful to the average individual of a population over prolonged exposure under field conditions. MATC values, expressed as the mass of contaminant per unit of body weight, were derived from data in the literature on tissue concentrations associated with the presence or absence of observed effects in organisms. Literature-based MATC tissue values were divided by uncertainty factors (UFs) to attempt to ensure adequate protection of the biota at RMA.

The MATC is not applicable to contaminants that are readily metabolized and/or rapidly excreted. The MATC applies only to bioaccumulative contaminants that accumulate over extended periods of time to toxic levels in the organism, exerting toxic effects as a function of residue concentration. When the MATC is divided by the site-specific issue concentration of the same organism, the result is a measure of potential risk from the contaminant being evaluated at the site.

C.2.3.2 Database Characterization

C.2.3.2.1 Literature Description

Literature search procedures for MATC data were provided in the introduction to Appendix C.2, along with Tables C.2-1 and C.2-2, which further detail the search. Only literature that indicates a correlation between tissue residues and toxic effects should be relied upon to provide a MATC for assessing toxicity levels.

In the literature, tissue concentration data associated with endpoints for organochlorine pesticides were often available for avian species and sometimes available for mammals. Less information was available for endrin than for the other organochlorine pesticides. Data for potential use in

derivation of MATC for mercury were also available for birds and laboratory mammals. Information that could be used to develop MATCs for fish and other aquatic life is generally lacking in the literature.

Tissue-based values related to the toxicity of all the COCs for the reptile trophic box were unavailable in the literature, and the values related to endrin and mercury for the small and medium mammal trophic boxes were either nonexistent or ambiguous. To date, reptiles have not been a common taxonomic group in toxicity testing. Most of the endrin and mercury toxicity studies on mammals that were reviewed investigated the effects of various dosing regimens without tissue residue measurements. The data compiled from the literature were screened for quality and appropriateness.

C.2.3.2.2 Parameter Quantification

Because most site-specific tissue data and BMFs are reported on a wet-weight basis, all MATC values were converted to a wet-weight concentration before being considered for use in the food-web model.

Tissue concentration values associated with specific endpoints in the literature were often provided for a specific organ or other component of the whole animal. These organ or carcass concentrations were converted to an estimation of the comparable whole-body concentration. Professional judgment and consensus discussions were used to select the most appropriate of these values from the available literature studies. These MATCs, expressed on a whole-body basis, were then divided by a UF to produce final MATC values.

Since many of the tissue residue data were associated with different toxicity endpoints, final MATC values were not summed and averaged to arrive at a final recommended value. Likewise, it was not appropriate to attempt to develop actual or hypothetical distributions. Instead, MATC was treated as a deterministic parameter represented by fixed values.

C.2.3.2.3 Data Variability

The MATC is preferably based on no observed toxic effect levels or concentrations associated with sublethal or minimal toxic effects. However, lethal or seriously adverse tissue concentrations are sometimes the only data reported in the literature. In these cases, one of the UFs divided into the tissue concentration reduced to a no adverse effects level or minimal adverse effects level.

MATC or residue data are often reported for a target organ or carcass, but are not always reported for nontarget organs or the whole body. The BMF for the food-web model is calculated on a whole-body basis (from the BAF and dietary fraction), and represents the amplification of a contaminant through an organism's food chain up to itself from the source media, due to its own bioaccumulative capability and that of others in the food chain. Site-specific tissue data are also generally reported on a whole-body basis. Therefore, all MATC values were converted to a whole-body concentration before use in the food-web model.

Avian toxicity data are relatively common for organochlorine pesticides, especially for lethal residue concentrations in brain tissue. For avian species, organochlorine pesticides stored in adipose tissue may be mobilized during times of stress, such as breeding or migration, and reach toxic levels in the brain. This, in turn, may cause the bird to cease feeding and may result in the death of the bird. Avian brain-to-carcass ratios can be developed for lipid-soluble contaminants such as organochlorine pesticides, but only if these ratios are derived from data in healthy birds in stable body condition. Because brain organochlorine concentrations are regulated by lipid levels, there is a definable relationship between contaminant concentration in the brain and that in carcass lipid levels. Thus, the brain-to-carcass ratio depends on the fraction of lipid in the carcass, and will fluctuate as this fraction changes. The ratio becomes meaningless in the circumstance of lethal or near-lethal poisoning because body lipids are then depleted and the contaminant is mobilized to the brain. It is this situation, and its sometimes questionable and difficult clarification, that has discouraged the use of brain-to-carcass ratios to estimate MATC values.

C.2.3.2.4 Final Value Selection and Assessment

Many of the toxicity data points related to tissue residues were discounted for reasons such as inappropriate toxicity endpoint (e.g., excessive mortality in the group of experimental animals or a histopathological change that could not be associated with a toxic effect on the organism), inappropriate species (e.g., livestock or other species only distantly related to those known to occur at RMA), or poor experimental protocol (e.g., the lack of a control group of experimental animals, small group of experimental animals, or an inaccurate measure of tissue residues). The consensus pre-UF MATCs (Table C.2-10) were selected after intense scrutiny of the scientific literature.

C.2.3.2.5 Uncertainty Factor Development

The UFs applied to the final literature-based, pre-UF MATCs attempt to ensure adequate protection of biota populations. The UFs were developed for the MATC and the TRV (Section C.2.6) approaches in parallel, i.e., it was decided to apply the same rationale and values for each derivation process.

Four uncertainty factor categories were selected to account for 1) the intertaxon variability in toxicological responses to contaminants when extrapolating from the species used in an experimental study to a target species at RMA; 2) the extrapolation from the duration of an experimental study to the chronic exposure being assessed at RMA; 3) the extrapolation from a toxicity endpoint in an experimental study to the desired no adverse effects endpoint for the ecological risk assessment at RMA; and 4) a modifying factor to account for additional sources of uncertainty. The final UF, the product of the results of these four categories, is divided into the pre-UF MATC critical value to determine a final MATC value (Table C.2-11). The same procedure is followed for the derivation of the TRVs.

Intertaxon Variability Extrapolation

For studies with the same species of interest, the minimum amount of uncertainty for the intertaxon variability extrapolation category was 1. The maximum amount of uncertainty was

applied when the study animals were in the same class but a different order as those animals at RMA. Critical values were not extrapolated beyond different orders.

Study Duration Extrapolation

The concentrations that are reported in the literature may reflect biological responses from acute, subchronic, or chronic exposures. Because the food-web model assumes that a contaminant is at equilibrium between an organism and the environment, values from studies employing chronic exposure, where equilibrium has been achieved, are most appropriate to provide data points for the MATC parameter.

Study Endpoints Extrapolation

Generally, endpoints for MATC values are presented as a biological response that varies from death to sublethal effects (e.g., physical deformations, disease, behavioral abnormalities, physiological abnormalities) to no observed effects. For a particular endpoint, the tissue concentration may be presented as a general toxicity level measured when the endpoint was observed, or it may be a more precisely defined lowest observable adverse effect level (LOAEL) or no observed adverse effect level (NOAEL). These more precise endpoints are determined by experiments that precisely measure incremental concentrations to pinpoint the specific concentration at which or before which the endpoint is first observed. A frank effects level (FEL) is that contaminant concentration in the tissue or organism that is highly certain to cause an adverse toxic effect. A decision must be made as to what biological response is an appropriate endpoint for each trophic box/chemical combination and what should be done if the appropriate endpoint is not available.

Modifying Factor

Eight additional sources of uncertainty were summed to comprise the modifying factor. These are adjustments for threatened and endangered species status, endpoint relevance, laboratory to field extrapolation, contaminant presence, unclear endpoint, species sensitivity, tissue to whole-body conversion, and intraspecies variability.

The UF intended to represent a safety factor for threatened and endangered species was used only for the bald eagle. The bald eagle is protected as an endangered species by the Endangered Species Conservation Act (16 U.S.C. 668aa-668cc-6) and the Bald and Golden Eagle Protection Act (16 U.S.C. 668-0668d). The MATC (or the TRV) developed using this UF, therefore, was assumed to be protective of sensitive individuals of this particular species. The remaining trophic boxes represent animal groups for which a MATC (or TRV), protective of populations and representing a concentration that potentially results in minimal adverse effects to some individuals but is protective of the average individual, is appropriate.

The uncertainty associated with relevance of an endpoint resulted when behaviors or pathology could not be directly linked to adverse health effects. Laboratory to field extrapolation uncertainty was applied to studies conducted with laboratory-bred animals such as rats or chickens. Co-contaminant uncertainty resulted from studies in which the results for the contaminant of interest may have been impacted by the presence of other chemicals, such as the administration of DDT or DDE with dieldrin, or the use in an experiment of an organic form of mercury rather than an inorganic form. Endpoint uncertainty resulted when the validity of an endpoint was questionable, such as when the number of deaths in test birds did not exceed the number of deaths observed in control birds. The modification of uncertainty for sensitive species was used when an unusually sensitive species, such as the brown pelican (which is impacted by relatively low concentrations of DDT), was used in a toxicological study. For the derivation of MATCs only, a modifying factor was used when data from one organ were extrapolated to a whole-body basis because of uncertainty about the applicability of the extrapolation ratio to individual animals at RMA. Uncertainty also resulted from intraspecific variability associated with small sample sizes of individuals in experimental groups, toxicity responses related to gender differences, and other variability among individuals as well as from the extrapolation of those differences to different individuals of the same species at RMA.

When the total modifying factor summed from the individual modifying factors had a negative sign, it denoted a reduction in the uncertainty from the eight various issues. The negative modifying factor was replaced with a value of 0.5 and then multiplied with the other categories

of UFs. When the total modifying factor was zero, which denoted a null impact on the overall uncertainty, it was replaced with a value of 1 so that it would not affect the product of the other three UF categories. When the total UF exceeded 400, it was replaced with the value 400 prior to division into the pre-UF MATC. An upper bound was established because at some point uncertainty becomes difficult to define and takes the MATC critical value to a point where only the doubt around the data point is expressed. The upper-bound UF value of 400 is supported by Barnthouse et al. (1990). Barnthouse et al. showed that the maximum uncertainty demonstrated was 417 for acute exposure studies based on the results of extrapolations of various types of toxicity data to obtain lifetime concentrations of triflurallin in water protective of Gulf menhaden and Chesapeake striped bass.

C.2.3.2.6 Uncertainty Factor Summary

Ecological risk assessments are forced to make many assumptions, especially when extrapolating toxicity data across taxa, because quantitative ecological risk assessment is in the formative stages and because there is a lack of sufficient field toxicity studies on many of the receptors of concern. Because many of these assumptions are extremely uncertain, UFs or safety factors have been used for the protection of species of interest.

For example, the Natural Resource Damage Assessment Model for Coastal and Marine Environments estimates potential hazards for aquatic biota using acute toxicity (lethal concentration to 50 percent of a population, or LC_{50}, and lethal dose to 50 percent of a population, or EC_{50}) data. For this model, a hazard value for each aquatic species is derived by dividing the acute toxicity value by 100 to estimate a no-effects level for the species of interest (EPA 1988).

Further, a Standard Evaluation Procedure (SEP) developed by the EPA Office of Pesticide Programs uses safety factors for conducting ecological risk assessments relative to pesticide product registration (1986, 1988). The SEP approach is a modified quotient method, which is similar to the hazard quotient method used in the Integrated Endangerment Assessment/Risk Characterization in which environmental concentrations are compared to environmental toxicity

endpoint values (i.e., regulatory risk criteria, or RRC). Both aquatic and terrestrial receptors are evaluated by this method. The assessments are directed at the population level, with members of threatened and endangered species addressed further by the application of more stringent RRC. For acute toxicity, the RRC are equal to the LC_{50} or lethal dose to 50 percent of population (LD_{50}) divided by a safety factor of either 5, 10, or 20. The general principle of the SEP approach and associated safety factors support the application of smaller UFs to derive MATCs and TRVs, because the total uncertainty associated with the SEP approach is intended to derive acceptable concentrations for acute exposure rather than chronic exposure (HLA 1993).

UFs were initially applied in the Biota RI (ESE 1989) to derive toxicity benchmark values. The Off-Post EA for RMA expanded the uncertainty development presented in the Biota RI (HLA 1993). For the Off-Post EA, a step-wise protocol was used to derive a TRV from a critical study dosage with a UF being applied at each step (UFs were not applied to the MATC values used in the Off-Post EA). The UFs used in the IEA/RC vary from those used in the Biota RI and the Off-Post EA because the derivation of MATCs and TRVs is an evolving process, and new information has become available. The UF values listed in Table C.2-11 were selected on the basis of consensus after thorough discussion of the available literature and application of informed best professional judgment. The total uncertainty values used result in MATCs and TRVs that are reasonably conservative relative to existing toxicological data.

C.2.4 EXPOSURE AREA

C.2.4.1 Parameter Definition and Characteristics

The exposure area parameter defines the area within which an organism is potentially exposed to contaminants. Soil contaminant concentrations averaged within the exposure area provide the <ESC> value used in the calculation of the BMF_{obs} (Section C.1.6) and in the estimation of risk (Section C.1.8). The concept of exposure area can be applied to sediment or water concentrations. Exposure area values for each trophic box, and in some cases for individual species within trophic boxes, were derived from the literature on home range, foraging area, and other areas of use.

Home range is defined as the area traversed by an organism in its normal activities of food gathering, loafing, mating, and caring for its young. Home range is often defined for a species during its breeding season. At this time, individuals use a defined area because their mobility is restricted by the presence of young. Foraging area is that portion of the home range which an individual uses for food gathering. Depending on the species, foraging area may cover the entire home range, or only a relatively small portion of the home range. For example, wide-ranging species, such as carnivores and birds, may have dens or nests in one area, resting areas in another, and feeding areas in a select, small area of high prey density.

For most species, exposure to soil concentrations is expected to result from feeding activities. However, some species may also be exposed to soil concentrations in portions of the home range used for other activities such as burrowing and dusting. Therefore, the selection of home range, foraging area, or some other area of use to represent exposure area varied by species.

C.2.4.2 Database Characterization for Exposure Area

C.2.4.2.1 Literature Description

General literature search procedures were described in the introduction to Section C.2. However, an independent literature search was conducted for information pertinent to the definition of exposure area. Most articles were located in three information databases (Wildlife Review, Wildlife Information Services, and CARL Uncover) or were provided by the technical representatives of the Organizations and State.

Studies on home range are available for most bird and mammal species found at RMA. Most literature studies are directed toward quantifying the areas used by species with relatively predictable use patterns, such as breeding birds or small mammals. Studies for wide-ranging species tend to focus on how far from their nest or den they might feed, not on the area actually used for feeding. No home-range information was available in the literature for shorebirds, insects, earthworms, and terrestrial plants.

C.2.4.2.2 Parameter Quantification

Literature studies on home range were evaluated and screened on the basis of characteristics such as geographic location, habitat, season, and methodology, which determined their usefulness in defining exposure area on RMA. For example, a study of home-range size in a grassland community in Nebraska would be valuable. On the other hand, methodologies such as mark-recapture studies or radio-transmitter studies that determined maximum distances traveled for food were not useful in quantifying RMA home ranges. The screened studies provided ranges of values for comparison and discussion.

Most of the literature studies provided the size of species-specific areas of use. The shape of the areas of use was often irregular and was defined by such factors as adjoining areas of unsuitable habitat, neighboring individuals of varying aggressiveness, and the energy needs of the organism using the area. Even within the area of use, patchiness of habitat often precluded homogeneous use of all portions of the area. However, the exposure area values based on home range were expressed in acres and later assumed to be to circles with their radii expressed in feet. Since this approach assumes that the exposure area has a circular shape and homogeneous use, the exposure area is only an approximation of the true area of use and exposure.

For wide-ranging species, best professional judgment was necessary to define the exposure areas since the area of potential use was often much larger than the area of likely exposure. For example, a bald eagle or great blue heron may range more than 10 miles in search of prey, yet feed consistently in a small area with high prey density.

C.2.4.2.3 Data Variability

There was considerable variability among species in the size of their areas of use. As would be expected, use areas for birds were larger than those for comparably sized mammals, and use areas for large birds and mammals were larger than those for small birds and mammals. Behavioral patterns also influenced the size of areas of use. For example, prairie dogs, which inhabit a burrow system, range less widely than cottontails. Even within a species, the size of the use areas varied given such factors as the quality of the habitat, the availability of the food, and the

aggressiveness of neighboring individuals. For example, home range for kestrels in an industrial complex was larger than in more natural surroundings. Areas of use were variously based on such characteristics as area defended, area of foraging, or area of greatest movement. Best professional judgment was used to determine the area, defined on the basis of activities, that would best correspond with exposure to contaminants. Final values were used deterministically.

C.2.4.2.4 Final Value Selection and Assessment

Best professional judgment and consensus discussions were used to select the home range (or other area of use) values that were most appropriate to estimate exposure area. During the selection process, many values were discounted due to inappropriate methodology or inappropriate values for determining exposure area (i.e., discarding general home range values when more refined values, such as foraging range or defended coterie territories, were available). Conversely, some values were deliberately chosen because they were measured by a researcher known to be an expert with regard to a certain species.

When values were not available for certain species or when all available literature values were discarded, local and regional experts were contacted. Many unpublished values or methodologies giving insight to exposure area were discovered using this method. For example, radioisotope tracer studies on root distributions for terrestrial plants and small-scale insect movement studies were used to estimate an appropriate exposure area for these trophic boxes.

Consensus on final values (Table C.2-13) was achieved in two stages. First, consensus was achieved on lower trophic box species such as terrestrial plants, insects, amphibians and reptiles, small birds, and mammals. Such species were either represented by consistent values in the literature or by exposure areas that could be reasonably approximated for reasons such as low mobility. Within this group, only the mourning dove was problematic, because of its habit of traveling considerable distances away from its nesting territory to obtain water. The distance traveled to water was ultimately ignored for this species and the exposure area was defined as the foraging area around its nest site where most of its exposure was believed to occur. Because organisms with aquatic food chains are exposed to water that is continually mixed within a lake,

even if they do not travel throughout the lake themselves, the entire lake was considered the exposure area.

Consensus on values for the wide-ranging high trophic level species (bald eagle, great blue heron, great horned owl, American kestrel and shorebird) and water birds was more difficult to achieve. Particular care was given the selection of exposure area for these species because they are at or near the top of their food webs and, therefore, experience the greatest biomagnification.

Within this group of six, bald eagle and great blue heron exposure area values were the most difficult to ascertain because both species range widely yet may feed selectively in small areas. The concern was that overestimating exposure area would underestimate risk. Based on consideration of whether the bald eagle exposure area should extend beyond RMA boundaries, or whether it should be restricted to a portion of RMA, the selection of a single exposure area for the bald eagle at RMA was made, which consists of the Bald Eagle Management Area as well as all prairie-dog towns present in April 1993. The shape of this polygon is fixed and is not to be used in formation of a circular estimate of exposure area. The great blue heron exposure area was based on the surface area of each RMA lake plus a terrestrial band equal to 4 percent of the lake surface area as an estimate of the shoreline areas used in foraging. The waterbird exposure area was based only on the surface area of each RMA lake. No upland or shoreline areas were included, even though nests may be found there, since little feeding occurs in these areas at RMA. The shorebird exposure area was a circular area since these species forage in terrestrial areas near the shoreline of the lakes. The exposure areas for great horned owl and American kestrel were the most certain among the six species requiring special attention, because these species exhibit considerable fidelity to their nesting territories when foraging.

C.2.5 TOXICITY REFERENCE VALUES

C.2.5.1 Parameter Definition and Characterization

The dose-based method employed in the IEA/RC to estimate ecological risk and biota criteria is based on the use of a TRV as a benchmark toxicity parameter. Like the reference doses (RfDs) used in human health risk assessment, the TRV represents an estimate, with uncertainty spanning

perhaps an order of magnitude or greater, of a daily dose in mg/kg-bw/day to a receptor that is likely to be without an appreciable risk of deleterious effects. Unlike the MATCs (Section C.2.4), which were identified only for the bioaccumulative COCs, the TRVs were determined for all of the COCs evaluated in the IEA/RC.

The TRV approach developed for the IEA/RC generally follows that developed for the Off-Post EA/Feasibility Study (HLA 1993) and is based on the method used by EPA (1986) to derive human health RfDs. The RfDs represent values protective of human health against systemic toxicity effects. A basic premise of this approach is that homeostatic, compensating, and adaptive mechanisms exist that must be overcome before a toxic endpoint is manifested. With systemic toxicity, therefore, there is a threshold effect. The RfD represents a benchmark dose operationally derived from a NOAEL by the consistent application of UFs that reflect the various types of data sets used to estimate RfDs. That is, the critical toxicity dose from the literature is divided by a value representing the product of all UFs and modifying factors determined to be appropriate on the basis of the quality of the data used to arrive at the NOAEL (HLA 1993). The TRVs used in the IEA/RC were derived in a similar manner, starting with a critical dose obtained from the literature, which was then adjusted by the application of appropriate UFs (including modifying factors) to result in a TRV that is protective of the representative trophic box. The development of the UF values is discussed in Section C.2.4 for the MATC. With one noted exception, the same protocol was used for TRVs.

The development of a TRV was a step-wise process beginning with the review of available toxicological literature for each chemical being evaluated for potential toxic effects to ecological receptors. The available literature was reviewed to identify all experimental and field studies that could be used to establish a critical dose. The preferred studies were those in which the test species was the same as the ecological receptor being evaluated. Unfortunately, the available literature for field studies using avian or wildlife species was very limited, which made reliance on information from laboratory animal studies necessary. The best study located in the literature was then selected to provide the critical dose and the critical toxicological endpoint. If the selected study reported a dietary concentration rather than a dose, the concentration was converted

to a critical study dose rate on the basis of the study animal's feed rate. The critical study dose rate was then divided by the total UF to derive the TRV as described in Section C.2.4.

C.2.5.2 Database Characterization

C.2.5.2.1 Literature Description

In the selection of critical dose values, many variables were considered during review of the toxicological dose data from the literature, including the toxicological endpoints of the study and the characteristics of the test species. The primary toxicological endpoints of interest were effects on reproduction and behavior; toxicological studies reporting potential effects on reproduction were preferred. Other endpoints, such as pathological changes or mortality, were used only if reproduction or behavioral studies were unavailable. Often, toxicity data were unavailable for the species of interest and data from a surrogate species were used, if available. A study with a species of similar size was preferred over a study using a smaller or larger animal. Because the composition of the diet may ultimately influence chemical uptake by the receptor, species with similar dietary habits were selected over species that had different dietary habits. If no data were available for a species in the same class as the species of concern, then no critical value was recommended.

C.2.5.2.2 Parameter Quantification

A summary of the critical study selection process is presented below for the COCs. The critical dose values obtained from the literature sources described below are summarized in Table C.2-14.

Aldrin/dieldrin

Several studies have been conducted documenting the toxicity of aldrin and dieldrin to avian species. A mallard duck study was selected as the most appropriate study to provide a critical dose for the great blue heron and the water bird (Sharma et al. 1976). The dose rate obtained from this study was 0.4 mg/kg-bw/day, which was designated a lowest observed effect level (LOEL) on the basis of a change in aggressive behavior in male birds. LOELs differ from LOAELS in that the observed effects reported may not necessarily be detrimental to an animal's health.

Two raptor studies were used to provide the critical dose for the bald eagle. These were reproductive studies of the American kestrel (Wiemeyer et al. 1986) and barn owl (Mendenhall et al. 1983). Both studies provided a NOAEL for reproductive effects. The critical dose for the eagle derived from these papers was 0.05 mg/kg-bw/day. The Mendenhall et al. reproduction study was also used to provide the critical dose for the great horned owl trophic box as a NOAEL of 0.06 mg/kg-bw/day. The Wiemeyer et al. study provided the critical dose for the American kestrel trophic box as a NOAEL for reproduction of 0.04 mg/kg-bw/day.

A 2-year study of 20 homing pigeons by Robinson and Crabtree (1969) provided the critical dose for the small bird trophic box as a NOAEL of 0.28 mg/kg-bw/day. Toxicological endpoints included mortality, number of birds that failed to return, and tissue residues (e.g., liver, eggs, blood). All of the 18 incubated eggs hatched normally. This study was also used to derive the critical dose for the shorebird, in conjunction with a lethality study of quail by Shellenberger (1978) and a study of quail by Stickel et al. (1969). The critical dose for shorebirds, based on NOAEL values from these three studies, was 0.22 mg/kg-bw/day.

For mammals, a study on reproductive and pathological effects in 440 rats was selected (Harr et al. 1970). The NOAEL for reproduction derived from this study was 0.06 mg/kg-bw/day. Toxicological endpoints included reproductive parameters (dam survival, dam conception, born litter size, and weaned litter size), dieldrin intake, tissue residue, overt signs of toxicity, and lesions. Lethal concentrations, tissue partitioning, and excretion rate were also obtained. Dam survival and conception were lower in the higher dose groups, as was litter size (born and weaned).

Critical dose values for aquatic species were also derived for aldrin/dieldrin. A study by Schuytema et al. (1991) provided a NOAEL of 1.9 micrograms per liter (µg/l) for the leopard frog, which was used as the amphibian critical dose value. Schuytema et al. investigated acute and chronic toxicity, teratogenesis, growth, and bioconcentration in three frog species. The other frog species had NOAEL values of 0.8 µg/l and 11.0 µg/l. On the basis of these observations, the authors concluded that the current EPA AWQC for dieldrin (0.0019 µg/l) is protective of

frogs. A study by Cairns and Scheier (1964) provided a LOAEL of 1.7 μg/l for the pumpkinseed sunfish, which exhibited decreased cruising speed and increased oxygen consumption at this concentration. This critical dose value was used for small fish. The critical dose value for large fish was a NOAEL of 0.12 μg/l, which is based on steelhead trout (Chadwick and Shumway 1969).

DDT/DDE

Several bird studies provided the critical dose values for avian species. A reproductive study of black ducks was used for the great blue heron and water bird trophic box critical doses (Longcore and Stendell 1983). Measurement of tissue residues and eggshell thickness resulted in a LOEL for reproductive effects of 0.06 mg/kg-bw/day.

Two American kestrel studies (Wiemeyer et al. 1986; Lincer 1975) provided the critical dose value used for the bald eagle, kestrel, and the shorebird trophic boxes. These two reproductive studies of the American kestrel resulted in the derivation of a LOAEL of 0.4 mg/kg-bw/day.

A reproductive study of barn owls (Mendenhall et al. 1983) was used to obtain the critical dose for the great horned owl trophic box. The value of 0.3 mg/kg-bw/day was a Frank effect level (FEL) for reproduction and was associated with statistically significant increases in eggshell thinning, egg breakage, embryo mortality, and reduced breeding success per pair of birds.

A reproductive study of finches (Jefferies 1971) provided the critical dose of 0.8 mg/kg-bw/day for the small bird trophic box. This was a LOAEL for reproductive effects.

A reproductive study of rats (Green 1969) provided the critical dose value of 0.35 mg/kg-bw/day for mammals. This was a NOAEL for reproductive effects since chronic exposure to DDT did not change fecundity, fertility growth rate, or litter size in two generations of rats.

Studies of aquatic species provided critical dose values for several trophic boxes. A NOAEL of 0.8 μg/l for behavioral changes in tadpoles (Cooke 1972) was used as the exposure concentration

for the amphibian trophic box. An FEL of 0.74 μg/l for mortality in fathead minnows exposed to DDT in food and water was used as the exposure concentration for the small and large fish trophic boxes (Jarvinen et al. 1977).

Endrin

Two studies for endrin toxicity in avian species provided critical dose values. A reproductive study of ducks (Roylance et al. 1985) provided an NOAEL of 0.05 mg/kg-bw/day as the critical dose for the great blue heron and the waterbird trophic boxes. Egg production, fertility, hatchability, and hatchling survival were unaffected by exposure to endrin at this concentration, although embryo survival was decreased at a higher dose than was selected as the critical value. A reproductive study of screech owls (Fleming et al. 1982) provided a LOAEL of 0.1 mg/kg-bw/day, which was used as the critical dose for the other avian trophic boxes. Reproductive endpoints in this study included number of eggs laid and hatched, number of cracked eggs and eggs laid outside of nest boxes, and number of young birds fledged, which were measured together with carcass and egg residues.

A study of rats and mice provided a critical value dose for mammals (Noda et al. 1972) as a LOAEL of 0.58 mg/kg-bw/day. This value was used for both large and small mammal trophic boxes.

Mercury

One study, a multi-generational reproductive study of mallard ducks, provided a critical mercury dose for birds as a LOAEL of 0.047 mg/kg-bw/day (Heinz 1976). Mercury levels in eggs and tissues were measured together with reproductive endpoints including whole-egg weight and shell thickness, percent of eggs laid outside nest box, percent of cracked eggs, percent of eggs producing normal ducklings, percent of normal hatchlings surviving 1 week, and number of 1-week-old ducklings.

A rat study provided the mercury critical dose for the mammal trophic boxes (Soares et al. 1973) as a NOAEL of 0.17 mg/kg-bw/day. Tissue residues were measured and the endpoints of

lethality and growth were examined in ten weanling rats exposed to mercury in diet for 12 weeks. There was no mortality in this dose group.

Arsenic

A study of ducks provided the critical dose for birds (Van Vleet 1982) as a LOAEL of 18.9 mg/kg-bw/day. A 2-year study of rats was used to obtain the critical dose for mammals as a LOAEL for pathology of 1.5 mg/kg-bw/day (Byron et al. 1967). Body weight was recorded, and hematological measurements (i.e., hemoglobin, hematocrit, and leukocyte counts) were made. Dose-related enlargement of the common bile ducts was noted, and survival was affected at the higher dose levels.

Cadmium

All of the cadmium critical dose values for birds were derived from a single study of doves (Scheuhammer and Templeton 1990). At the NOAEL of 1.8 mg/kg-bw/day, no overt toxicity or mortality was observed. The TRVs developed from this dose differed among the bird trophic boxes due to differences in the UFs applied.

The critical dose values for mammals were derived from a study with rats (Groten et al. 1991) as a NOAEL of 1.8 mg/kg-bw/day. When cadmium accumulation in tissue from this dose was measured over the 4 weeks of exposure, a dose- and time-related increase in liver, kidney, and intestinal concentrations of cadmium was observed, although no effects were reported.

Chlordane

An LD_{50} study of chlordane provided the only toxicity data for birds (Hudson et al. 1984). The LD_{50} was 14.1 mg/kg-bw for quail; an LD_{50} of 1,250 mg/kg-bw was also reported for ducks.

The only critical dose data for mammals was a NOAEL of 1.2 mg/kg-bw/day reported by EPA (1984). This study was summarized in the 1991 Health Effects Assessment Summary Tables.

Copper

Copper toxicity data for birds were limited. A study by Leach et al. (1990) describing growth limitation in chickens provided a critical dose value used for all of the avian trophic boxes. Growth limitation was observed at a dose of 48 mg/kg-bw/day. Chickens were more sensitive to copper toxicity when inadequate calcium was supplied in the diet; at a calcium content of 0.6 percent or less in the diet, body weight decreased in birds consuming a high amount of copper in diet compared to birds consuming a low amount of copper.

The critical dose for copper in mammals was an LD_{50} of 300 milligrams per kilogram (mg/kg) (RTECS 1993). Other data were not found in the literature surveyed.

Chlorophenylmethyl Sulfide

Data for toxicity to avian species were unavailable for this chemical. The only study available to provide the critical dose for mammals was Thake et al. (1979), which was a dietary study of rats. The LOAEL was 14.1 mg/kg-bw/day.

Chlorophenylmethyl Sulfone

Data for toxicity to avian species were unavailable for this chemical. The Thake et al. (1979) dietary study of rats provided a LOAEL of 16.3 mg/kg-bw/day, which is the only value available for mammals.

Dibromochloropropane

Very few studies were available on the toxicity of DBCP to wildlife species. For birds, the only value obtained was an LD_{50} for ducks of 66.8 mg/kg (Hudson et al. 1984). For mammals, a dose rate of 0.6 mg/kg-bw/day was obtained as a NOEL for rats (EPA 1987).

Dicyclopentadiene

A dietary study using bobwhite quail provided the critical dose used for all bird trophic boxes, a LOEL of 400 mg/kg-bw/day (Aulerich et al. 1979). EPA (1991) provided the NOEL dose value for rats of 34 mg/kg-bw/day.

C.2.5.2.3 Data Variability

It is not unusual to observe a wide range in the critical dose values before UFs are applied, because there is inherent intrinsic and extrinsic variability associated with each of the trophic boxes. The variability of the pre-UF dose data is within two orders of magnitude among avian and mammalian species for each COC evaluated. Also, the variability never exceeded two orders of magnitude within the same phylogenetic class for each COC. The data were not segregated based on study type or endpoint, so the variability is partially the result of differences in study type and study endpoint.

The variability was reduced following the application of the total UF to the critical dose values. For mammalian and avian trophic boxes for each COC, the final TRVs were within one order of magnitude. The only exception was for chlordane, for which the small bird TRV was 0.035 mg/kg-bw/day. This value compares to 3.12 mg/kg-bw/day for the water bird.

C.2.5.2.4 Final Value Selection and Assessment

The UF protocol applied to the critical dose values was essentially identical to that described for each UF category in Section C.2.4 for the MATC. The only difference is that the modifying factor subcategory addressing extrapolation from organ or egg to whole body for the MATC was not included in the TRV derivation process.

To calculate TRVs, the critical dose value for a given trophic box/chemical combination was divided by the total UF:

$$\text{TRV} = \frac{\text{Dose}}{\text{Total UF}} \qquad (7)$$

where: TRV = toxicity reference value (mg/kg-bw/day)
Dose = critical study dose value (mg/kg-bw/day)
Total UF = total uncertainty factor

Each of these values, as well as the individual UF values, are listed in Table C.2-15. These TRVs are expected to be protective of the designated trophic boxes through a review and evaluation of the available literature.

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To Install the Ecological Risk Characterization Bibliographic Reference Database (ERCBIBL.dbf)

Summary: These instructions outline a custom Microsoft Windows procedure for installing a dBASE database file into a predetermined directory on your hard disk.

Instructions:

- To ensure proper installation and program execution, please make sure that your computer has a hard disk with a minimum of 5 megabytes of free space.
- Start the computer and enter Microsoft Windows.
- From the Program Manager Window, choose Run from the File Menu.
- Insert the diskette labeled "RC Media Installation Disk 1 of 3" into the 3.5 inch disk drive.
- Type "<d>:Setup" (without the quotes or angle brackets) where <d> is the drive letter where you have inserted the installation diskette.
- Press ENTER.
- Follow instructions on the screen. You will be prompted to enter the drive letter representing the hard disk where you want to install the database file, ERCBIBL.dbf. (The ERCBIBL.dbf database will be installed in the ERCBIBL directory on the drive of your choice.)

Read the ERCBIBL.txt file installed into the ERCBIBL directory for a summary of the database structure and query information.

The bibliographic reference database for the ecological risk characterization of the Rocky Mountain Arsenal Integrated Endangerment Assessment/Risk Characterization (RMA IEA/RC) project is contained on this disk.

The file ERCBIBL.dbf is the result of the previous Windows installation. Because the ECOREFS.DBF file is about 4 megabytes in size, the file will have to be loaded to a hard drive with at least 4 megabytes of free space.

At the DOS prompt for the hard–drive directory onto which the file was copied, type "DBASE" and hit enter. Hit enter when the DBASE IV logo comes onto the screen. At the DBASE dot prompt (or, if using a Control Center hit Alt-E, then choose to exit to dot prompt), type "USE ECOREFS" and hit enter. At the next dot prompt, type "BROWSE" and hit enter. The bibliographic reference database will come onto the screen. To exit the ECOREFS.DBF file, hit the escape key. Then, type quit at the DBASE dot prompt to return to the DOS prompt. If changes or additions are made to the database, the file name should be changed so the original ECOREFS.DBF can be maintained.

The bibliographic reference database for the ecological risk characterization is organized alphabetically by author and linked to the chronological order of publication year followed by the alphabetical ordering of titles. For example, if there are two references by Ytterbaum, S. and one was published in 1989 called Poisoned Ducks, and the other published in 1989 was called Healthy Ducks, then the latter reference will appear first. The fields most likely to be used in searches are the following (listed from left to right) DOCNO2 (document number), AUTHOR, YEAR, TITLE, OTHERNUM (an alternate document number). Listed below are two of the easiest and most useful search commands that can be used to access particular references.

To locate a particular unique document number, author, year, or title: Type "LOCATE FOR" {fieldname} = "condition" and hit enter. Then type "BROWSE" and hit enter to return to viewing the database.

The fieldname is one of those listed above in capital letters (do not include the brackets), and the condition is the document number, the author's name (last name first, followed by initials or first and middle initial), the year, or the exact title (do include the quotation marks).

To locate a particular reference by subject, or if the exact name of the author or exact title is not known, the following search command can be used: Type "LOCATE FOR" "condition" $ {fieldname} and hit enter. Then type "BROWSE" and hit enter to return to viewing the database. The condition, in this case, can be only the author's last name or one or a string of words from the title.

This bibliographic reference database for the ecological risk characterization may be further updated as additional versions of the RMA IEA/RC report are released. Inquiries may be directed to Michael Jones at Enserch Environmental Corporation, Lakewood, CO (303) 988-2202.

Parameter	Definition (units)
Toxilogical Input Parameters	
BAF	bioaccumulation factor (unitless)
BMF	biomagnification factor (unitless); the final value resulting from the food web model
Ecological Input Parameters	
FR	fraction of item i-1 in diet of level i organism (unitless)
R	feeding rate (grams food per gram body weight per day [g/g bw/d])
Toxicity Threshold Types	
ER	exposure range that represents the average area over which an organism forages or is otherwise consistently exposed (represented as a circle of stated acreage or radius)
MATC	maximum allowable tissue concentration of a contaminant (micrograms contaminant per gram tissue [µg/g])
TRV	toxicity reference value for a contaminant when ingested as a dose (micrograms contaminant per gram prey tissue per day)
Other Measures of Toxicity	
FEL	Frank effect level or that level of exposure producing a significant increase in frequency or severity of unmistakable adverse effects
LD_{xx}	Contaminant dose lethal to stated (xx) percent of test organisms (micrograms contaminant per gram body weight [µg/g])
LOAEL	Lowest observed adverse effects level of a contaminant concentration under chronic exposure conditions (stated as a dose or tissue concentration [µg/g])
LOEL	Lowest observed effects level of a contaminant concentration under chronic exposure conditions (stated as a dose or tissue concentration [µg/g])
NOAEL	No observed adverse effects level of a contaminant concentration under chronic exposure conditions (stated as a dose or tissue concentration [µg/g])
NOEL	No observed effects level of a contaminant concentration under chronic exposure conditions (stated as a dose or tissue concentration [µg/g])
TXCTY	Deleterious effects observed from a contaminant concentration without qualifying as a (stated as a dose or tissue concentration [µg/g])

Table C.2-2 Keywords Used in the DIALOG On-Line Search for the Ecological Risk Characterization

Aldrin
Arsenic
Availability
Bioaccumulation
Biomagnification
Dibromochloropropane
DDE
DDT
Dibromochloropropane
Dichlorodiphenyltrichloroethane
Dichlorodiphenylethane
Dieldrin
Diet
Eating
Endrin
Environment
Feed
Food
Forage
Foraging
Habit
MATC
Mercury
Nonhuman
Partition coefficient
Preference
Prey
Resource
Selection
Soil partition
Soil ingestion
Tissue concentration
Toxic
Uptake
Utilization

DDE	Dichlorodiphenyldichloroethene
DDT	Dichlorodiphenyltrichloroethane
K_d	Soil-water partition coefficient normalized to dry weight
K_{oc}	Soil-water partition coefficient normalized to organic carbon
MATC	Maximum allowable tissue concentration

Table C.2-3 Ecological Risk Characterization Model Input Parameter Values

Parameter = Bioaccumulation Factor (BAF)

Biota	Chemical	Distribution	Mean*	Std. Dev.	LOG Mean	LOG Std Dev.	End Point
Small Bird	Aldrin/Dieldrin	Normal	6.6	1.8			
	Endrin	Lognormal	1.0	1.6	0.000	0.470	
	DDE/DDT	Uniform	NA	NA			7.7, 29
	Arsenic	Uniform	NA	NA			0.3, 3
	Mercury	Triangular	0.33	NA			0.001, 2
Small Mammal	Aldrin/Dieldrin	Uniform	NA	NA			0.64, 1.6
	Endrin	Lognormal	0.08	1.0	-2.526	0.001	
	DDE/DDT	Uniform	NA	NA			0.44, 0.98
	Arsenic	Lognormal	0.19	4.7	-1.684	1.543	
	Mercury	Triangular	22.5	NA			0.001, 50
Medium Mammal	Aldrin/Dieldrin	Uniform	NA	NA			0.64, 3.2
	Endrin	Lognormal	0.16	1.1	-1.833	0.095	
	DDE/DDT	Uniform	NA	NA			0.44, 0.98
	Arsenic	Lognormal	0.19	4.7	-1.684	1.543	
	Mercury	Triangular	22.5	NA			0.001, 50
Water Bird	Aldrin/Dieldrin	Normal	16	5.1			
	Endrin	Lognormal	1.0	1.6	0.000	0.470	
	DDE/DDT	Normal	96	26.2			
	Arsenic	Uniform	NA	NA			0.3, 3
	Mercury	Lognormal	4.1	3.4	1.411	1.224	
Kestrel	Aldrin/Dieldrin	Normal	10.5	1.2			
	Endrin	Lognormal	1.0	1.6	0.000	0.470	
	DDE/DDT	Uniform	NA	NA			7.7, 29
	Arsenic	Uniform	NA	NA			0.3, 3
	Mercury	Triangular	0.33	NA			0.001, 2
Owl	Aldrin/Dieldrin	Normal	21.1	3.4			
	Endrin	Lognormal	1.0	1.6	0.000	0.470	
	DDE/DDT	Lognormal	43.7	2.4	3.777	0.875	
	Arsenic	Uniform	NA	NA			0.3, 3
	Mercury	Triangular	0.33	NA			0.001, 2

* Mean = arithmetic mean for normal distribution, geo. mean for lognormal distribution, and apex for triangular distrib.

Table C.2-3 Ecological Risk Characterization Model Input Parameter Values

Parameter = Bioaccumulation Factor (BAF)

Biota	Chemical	Distribution	Mean*	Std. Dev.	LOG Mean	LOG Std Dev.	End Point
Shorebird	Aldrin/Dieldrin	Normal	13.3	4.2			
	Endrin	Lognormal	1.0	1.6	0.000	0.470	
	DDE/DDT	Uniform	NA	NA			7.7, 29
	Arsenic	Uniform	NA	NA			0.3, 3
	Mercury	Triangular	0.33	NA			0.001, 2
Heron	Aldrin/Dieldrin	Normal	16	5.1			
	Endrin	Lognormal	1.0	1.6	0.000	0.470	
	DDE/DDT	Normal	93.5	20			
	Arsenic	Uniform	NA	NA			0.3, 3
	Mercury	Lognormal	4.1	3.4	1.411	1.224	
Bald Eagle	Aldrin/Dieldrin	Normal	15.9	3.9			
	Endrin	Lognormal	1.0	1.6	0.000	0.470	
	DDE/DDT	Lognormal	27.1	2.4	3.300	0.875	
	Arsenic	Uniform	NA	NA			0.3, 3
	Mercury	Triangular	0.33	NA			0.001, 2

* Mean = arithmetic mean for normal distribution, geo. mean for lognormal distribution, and apex for triangular distrib.

Table C.2-3 Ecological Risk Characterization Model Input Parameter Values

Parameter = Dietary Fractions (FR)

Predator	Prey Item	Biomass Fraction*
Terrestrial Food Chain		
Small Birds	Soil	0.057
	Terrestrial Plants	0.113
	Earthworm	0.116
	Insect	0.714
Small Mammals	Soil	0.020
	Terrestrial Plants	0.866
	Earthworm	0.008
	Insect	0.106
Medium Mammal	Soil	0.074
	Terrestrial Plants	0.926
	Insect	0.000
Kestrel	Soil	0.029
	Insect	0.184
	Small Mammal	0.665
	Small Bird	0.122
Owl	Soil	0.029
	Small Mammal	0.121
	Medium Mammal	0.830
	Small Bird	0.020
Heron	Soil	0.036
	Reptile	0.060
	Small Mammal	0.013
	Water	0.071
	Aquatic Plant	0.000
	Aquatic Invertebrates	0.024
	Small Fish	0.186
	Large Fish	0.604
	Amphibian	0.006
Bald Eagle	Soil	0.029
	Small Mammal	0.000
	Medium Mammal	0.936
	Small Bird	0.003
	Waterbird	0.030
	Large Fish	0.002

* Fractions reported as zero are pathways considered to be relatively inconsequential to model output, due to its small values.

Table C.2-3 Ecological Risk Characterization Model Input Parameter Values

Parameter = Dietary Fractions (FR)

Predator	Prey Item	Biomass Fraction*
Aquatic Food Chain		
Water bird	Water	0.019
	Sediment	0.038
	Aquatic Plant	0.942
	Aquatic Invertebrates	0.001
Shorebird	Terrestrial Plants	0.007
	Insect	0.728
	Sediment	0.160
	Aquatic Invertebrates	0.105

* Fractions reported as zero are pathways considered to be relatively inconsequential to model output, due to its small values.

Parameter = Feed Rate (R) **kg/kg body weight/day**

Biota	Distribution	Mean*	Std. Dev.	LOG Mean	LOG Std Dev.
Water Bird	Normal	0.07602	0.0245		
Small Bird	Fixed	0.0879			
Small Mammal	Fixed	0.12			
Medium Mammal	Fixed	0.096			
Shorebird	Lognormal	0.0879	1.652	-2.4315	0.50189
Kestrel	Normal	0.08913	0.02689		
Owl	Normal	0.08913	0.02689		
Heron	Normal	0.08913	0.02689		
Bald Eagle	Normal	0.08913	0.02689		

* Mean = arithmetic mean for normal distribution, geometric mean for lognormal distribution, and apex for triangular distrib.

Table C.2-3 Ecological Risk Characterization Model Input Parameter Values

Parameter = Maximum Allowable Tissue Concentration (MATC)

Biota	Chemical	Distribution	Value
Small Bird	Aldrin/Dieldrin	Fixed	0.15
	Endrin	Fixed	0.052
	DDE/DDT	Fixed	0.14
	Mercury	Fixed	0.017
Small Mammal	Aldrin/Dieldrin	Fixed	0.19
	Endrin	Fixed	NA
	DDE/DDT	Fixed	0.22
	Mercury	Fixed	NA
Medium Mammal	Aldrin/Dieldrin	Fixed	0.19
	Endrin	Fixed	NA
	DDE/DDT	Fixed	0.22
	Mercury	Fixed	NA
Reptile	Aldrin/Dieldrin	Fixed	NA
	Endrin	Fixed	NA
	DDE/DDT	Fixed	NA
	Mercury	Fixed	NA
Kestrel	Aldrin/Dieldrin	Fixed	0.73
	Endrin	Fixed	0.052
	DDE/DDT	Fixed	4.3
	Mercury	Fixed	0.017
Owl	Aldrin/Dieldrin	Fixed	0.76
	Endrin	Fixed	0.087
	DDE/DDT	Fixed	0.53
	Mercury	Fixed	0.017
Water bird	Aldrin/Dieldrin	Fixed	0.24
	Endrin	Fixed	0.09
	DDE/DDT	Fixed	0.18
	Mercury	Fixed	0.01
Shorebird	Aldrin/Dieldrin	Fixed	0.15
	Endrin	Fixed	0.052
	DDE/DDT	Fixed	1.4
	Mercury	Fixed	0.011
Heron	Aldrin/Dieldrin	Fixed	0.87
	Endrin	Fixed	0.043
	DDE/DDT	Fixed	15
	Mercury	Fixed	0.011
Bald Eagle	Aldrin/Dieldrin	Fixed	0.41
	Endrin	Fixed	0.031
	DDE/DDT	Fixed	2.2
	Mercury	Fixed	0.0083

Table C.2-3 Ecological Risk Characterization Model Input Parameter Values

Parameter = Toxicity Reference Values (TRV)

Biota	Chemical	Distribution	Value
Terrestrial Plant	Arsenic	Fixed	1.9
Small Bird	Aldrin/Dieldrin	Fixed	0.028
	Endrin	Fixed	0.002
	DDE/DDT	Fixed	0.003
	Mercury	Fixed	0.0019
	Arsenic	Fixed	0.38
	Copper	Fixed	0.96
	Cadmium	Fixed	0.24
	DCPD	Fixed	8.9
	Chlordane	Fixed	0.035
	CPMS	Fixed	NA
	CPMSO2	Fixed	NA
	DBCP	Fixed	0.17
Small Mammal	Aldrin/Dieldrin	Fixed	0.004
	Endrin	Fixed	0.010
	DDE/DDT	Fixed	0.029
	Mercury	Fixed	0.0014
	Arsenic	Fixed	0.038
	Copper	Fixed	0.75
	Cadmium	Fixed	0.045
	DCPD	Fixed	2.8
	Chlordane	Fixed	0.10
	CPMS	Fixed	0.24
	CPMSO2	Fixed	0.27
	DBCP	Fixed	0.05
Medium Mammal	Aldrin/Dieldrin	Fixed	0.004
	Endrin	Fixed	0.010
	DDE/DDT	Fixed	0.029
	Mercury	Fixed	0.0014
	Arsenic	Fixed	0.038
	Copper	Fixed	0.75
	Cadmium	Fixed	0.045
	DCPD	Fixed	2.8
	Chlordane	Fixed	0.10
	CPMS	Fixed	0.24
	CPMSO2	Fixed	0.27
	DBCP	Fixed	0.05

NA = data not available to calculate a TRV

Table C.2-3 Ecological Risk Characterization Model Input Parameter Values

Parameter = Toxicity Reference Values (TRV)

Biota	Chemical	Distribution	Value
Kestrel	Aldrin/Dieldrin	Fixed	0.01
	Endrin	Fixed	0.002
	DDE/DDT	Fixed	0.04
	Mercury	Fixed	0.0019
	Arsenic	Fixed	0.38
	Copper	Fixed	0.96
	Cadmium	Fixed	0.24
	DCPD	Fixed	8.9
	Chlordane	Fixed	0.035
	CPMS	Fixed	NA
	CPMSO2	Fixed	NA
	DBCP	Fixed	0.17
Owl	Aldrin/Dieldrin	Fixed	0.004
	Endrin	Fixed	0.003
	DDE/DDT	Fixed	0.008
	Mercury	Fixed	0.0019
	Arsenic	Fixed	0.38
	Copper	Fixed	0.96
	Cadmium	Fixed	0.24
	DCPD	Fixed	8.9
	Chlordane	Fixed	0.035
	CPMS	Fixed	NA
	CPMSO2	Fixed	NA
	DBCP	Fixed	0.17
Water brid	Aldrin/Dieldrin	Fixed	0.027
	Endrin	Fixed	0.003
	DDE/DDT	Fixed	0.004
	Mercury	Fixed	0.00094
	Arsenic	Fixed	0.38
	Copper	Fixed	0.96
	Cadmium	Fixed	0.24
	DCPD	Fixed	3.2
	Chlordane	Fixed	3.1
	CPMS	Fixed	NA
	CPMSO2	Fixed	NA
	DBCP	Fixed	0.17

NA = data not available to calculate a TRV

Table C.2-3 Ecological Risk Characterization Model Input Parameter Values

Parameter = Toxicity Reference Values (TRV)

Biota	Chemical	Distribution	Value
Shorebird	Aldrin/Dieldrin	Fixed	0.022
	Endrin	Fixed	0.002
	DDE/DDT	Fixed	0.008
	Mercury	Fixed	0.00094
	Arsenic	Fixed	0.38
	Copper	Fixed	0.96
	Cadmium	Fixed	0.24
	DCPD	Fixed	8.9
	Chlordane	Fixed	0.035
	CPMS	Fixed	NA
	CPMSO2	Fixed	NA
	DBCP	Fixed	0.17
Heron	Aldrin/Dieldrin	Fixed	0.03
	Endrin	Fixed	0.003
	DDE/DDT	Fixed	0.004
	Mercury	Fixed	0.00094
	Arsenic	Fixed	0.38
	Copper	Fixed	0.96
	Cadmium	Fixed	0.24
	DCPD	Fixed	8.9
	Chlordane	Fixed	0.035
	CPMS	Fixed	NA
	CPMSO2	Fixed	NA
	DBCP	Fixed	0.17
Bald Eagle	Aldrin/Dieldrin	Fixed	0.002
	Endrin	Fixed	0.001
	DDE/DDT	Fixed	0.005
	Mercury	Fixed	0.00063
	Arsenic	Fixed	0.19
	Copper	Fixed	0.48
	Cadmium	Fixed	0.10
	DCPD	Fixed	5.3
	Chlordane	Fixed	0.035
	CPMS	Fixed	NA
	CPMSO2	Fixed	NA
	DBCP	Fixed	0.17

NA = data not available to calculate a TRV

Table C.2-4 Importance Categories for Parameters in Aquatic Food Chains

Parameters	Bald Eagle Importance Category	Great Blue Heron Importance Category
MATC	1	1
Fraction Water Bird Aquatic Plant	1	4
Fraction Water Bird Aquatic Invertebrate	4	4
Fraction Generic Predator Large Fish	1	1
Fraction Generic Predator Aquatic Plant	4	4
Fraction Generic Predator Aquatic Invertebrate	4	4
Fraction Generic Predator Amphibian	4	4
Feed Rate Water Bird	1	4
Feed Rate Generic Predator	1	1

@	Assimilation rate
MATC	Maximum allowable tissue concentration
K_{OC}	Soil-water partition coefficient normalized to organic carbon
K2	Depuration (loss) rate
F_{OC}	Fraction of organic carbon content
BCF	Bioconcentration factor

Table C.2-5 Importance Categories for Parameters in Terrestrial Food Chains

Parameter	Bald Eagle Importance Categories	Great Blue Heron Importance Categories	Great Horned Owl Importance Categories	American Kestrel Importance Categories
MATC	1	1	1	1
Fraction Water Bird	4	4	4	4
Fraction Sm. Mammal Ter. Plant	4	4	4	3
Fraction Sm. Mammal Soil	4	4	4	4
Fraction Sm. Mammal Insect	4	1	4	1
Fraction Sm. Mammal Earthworm	4	4	4	4
Fraction Sm. Bird Ter. Plant	2	4	2	3
Fraction Sm. Bird Soil	4	4	4	4
Fraction Sm. Bird Insect	2	4	2	4
Fraction Sm. Bird Earth	2	4	2	2
Fraction Shore Bird Sediment	4	4	4	4
Fraction Shore Bird Aqu. Invert.	4	4	4	4
Fraction Md. Mammal Ter. Plant	4	4	4	4
Fraction Md. Mammal Soil	2	4	2	4
Fraction Md. Mammal Insect	4	4	4	4
Fraction Gen. Pred. Water Bird Ter.	4	4	4	4
Fraction Gen. Pred. Water	4	3	4	4
Fraction Gen. Pred. Soil	3	4	4	4
Fraction Gen. Pred. Sm. Mammal	4	1	4	1
Fraction Gen. Pred. Sm. Bird	1	4	1	1
Fraction Gen. Pred. Md. Mammal	2	4	2	4
Fraction Gen. Pred. Insect	4	4	4	4
Fraction Gen. Pred. Herp.	4	4	4	4
Fraction Earthworm Ter. Plant	4	4	4	4
Fraction Earthworm Soil	2	3	2	2
BAF Water Bird	4	4	4	4
BAF Small Mammal	4	1	4	1
BAF Small Bird	1	4	1	1
BAF Shore Bird	4	4	4	4
BAF Medium Mammal	2	4	2	4
BAF Generic Predator	1	1	1	1

MATC	Maximum allowable tissue concentration	Inverb	Invertebrate
BAF	Bioaccumulation Factor	Ter	Terrestrial
Sm	Small	Md	Medium
Aqy	Aquatic	Gen	Generic
Pred	Predator		

Study Species	Value (s)	Literature Cited	Comments on How BAFs Were Calculated	Distribution of BAF Values
Aldrin/Dieldrin				
Trophic Box—Bald Eagle				
Herring gull	16	Braune and Norstrom 1989, p.961	Whole body wet-weight herring gull BAF, diet of alewife from Lake Ontario.	Normal; am = 15.9; sd = 3.9; kestrel sd from birds that died, combined with owl sd and gull sd.
Kestrel	10.5	Wiemeyer et al. 1986, p.14	Wet-wt; chronic; nonlethal doses; exposed to dieldrin and DDT; lab study; converted to whole-body.	
Barn owl	21.1	Mendenhall et al. 1983, p.237	Wet-wt carcass residues, converted to whole-body, ave. of males and females, dose = 0.58 ppm.	
Trophic Box—American Kestrel				
Kestrel	10.5	Wiemeyer et al. 1986, p.14	Wet-wt; chronic; nonlethal doses; exposed to dieldrin and DDT; lab study; converted to whole-body.	Normal; am = 10.5; sd = 1.2; kestrel sd from birds that died.
Trophic Box—Great Horned Owl				
Barn owl	21.1	Mendenhall et al. 1983, p.237	Wet-wt carcass residues, converted to whole-body; ave. of males and females, dose = 0.58 ppm.	Normal; am = 21.1; sd = 3.4; from data used to derive the BAF.

am = arithmetic mean
sd = standard deviation
wt = weight

Study Species	Value (s)	Literature Cited	Comments on How BAFs Were Calculated	Distribution of BAF Values
Aldrin/Dieldrin (cont.)				
Trophic Box—Great Blue Heron				
Herring gull	16	Braune and Norstrom 1989, p.961	Whole-body wet-wt herring gull BAF, diet of alewife from Lake Ontario.	Normal; am = 16; sd = 5.1; given in paper.
Trophic Box—Shorebird				
Kestrel	10.5	Wiemeyer et al 1986, p.14	Wet-wt; chronic; nonlethal doses; exposed to dieldrin and DDT; lab study; converted to whole-body.	Normal; am = 13.3; sd = 4.2; sd from data in papers.
Herring gull	16	Braune and Norstrom 1989, p.961	Whole body wet-wt herring gull BAF, diet of alewife from Lake Ontario.	
Trophic Box—Water Bird				
Herring gull	16	Braune and Norstrom 1989, p.961	Whole body wet-wt herring gull BAF, diet of alewife from Lake Ontario.	Normal; am = 16; sd = 5.1; given in paper.
Trophic Box—Small Bird				
Quail	2.7	Stickel et al. 1969, p.185	Consensus value; paper supplied by EPA.	Normal; am = 6.6; sd = 1.8; sd from data in papers.
Kestrel	10.5	Wiemeyer et al. 1986, p.14	Wet-wt; chronic; nonlethal doses; exposed to dieldrin and DDT; lab study; converted to whole-body.	

am = arithmetic mean
sd = standard deviation
wt = weight

Study Species	Value (s)	Literature Cited	Comments on How BAFs Were Calculated	Distribution of BAF Values
Endrin				
Trophic Box—Bald eagle				
Chicken	1.5	Cummings et al. 1967, p.421	Endrin in fat of hens (n=15), fed endrin-fortified feed for 14 wks, converted to whole-body (20% lipids); data from Fig. 3.	Lognormal; gm = 1.0; gsd = 1.6; sd based on data from papers; type based on professional judgment (normal gives negative values).
Screech owl	0.6	Fleming et al. 1982, p.465	Screech owls fed 0.75 ppm endrin (n=14) for 83 days, wet-wt carcass data from Table 2 converted to whole-body.	
Mallard	1.2	Spann et al. 1986, p.758	Nonlethal; carcass (no feet, bill, GI, feathers); ave. both sexes; wet-wt; get same result on lipid-weight with 15% lipid in whole body.	
Trophic Box—American Kestrel				
Chicken	1.5	Cummings et al. 1967, p.421	Endrin in fat of hens (n=15), fed endrin-fortified feed for 14 wks, converted to whole-body (20% lipids); data from Fig. 3.	Lognormal; gm = 1.0; gsd = 1.6; sd based on data from papers; type based on professional judgment (normal gives negative values).
Screech owl	0.6	Fleming et al. 1982, p.465	Screech owls fed 0.75 ppm endrin (n=14) for 83 days, wet-wt carcass data from Table 2 converted to whole-body.	
Mallard	1.2	Spann et al. 1986, p.758	Nonlethal; carcass (no feet, bill, GI, feathers); ave. both sexes; wet-wt; get same result on lipid-weight with 15% lipid in whole body.	

am = arithmetic mean
sd = standard deviation
wt = weight

Study Species	Value (s)	Literature Cited	Comments on How BAFs Were Calculated	Distribution of BAF Values
Endrin (cont.)				
Trophic Box—Great Horned Owl				
Chicken	1.5	Cummings et al. 1967, p.421	Endrin in fat of hens (n=15), fed endrin-fortified feed for 14 wks, converted to whole-body (20% lipids); data from Fig. 3.	Lognormal; gm = 1.0; gsd = 1.6; sd based on data from papers; type based on professional judgment (normal gives negative values).
Screech owl	0.6	Fleming et al. 1982, p.465	Screech owls fed 0.75 ppm endrin (n=14) for 83 days, wet-wt carcass data from Table 2 converted to whole-body.	
Mallard	1.2	Spann et al. 1986, p.758	Nonlethal; carcass (no feet, bill, GI, feathers); ave. both sexes; wet-wt; get same result on lipid-weight with 15% lipid in whole body.	
Trophic Box—Great Blue Heron				
Chicken	1.5	Cummings et al. 1967, p.421	Endrin in fat of hens (n=15), fed endrin-fortified feed for 14 wks, converted to whole-body (20% lipids); data from Fig. 3.	Lognormal; gm = 1.0; gsd = 1.6; sd based on data from papers; type based on professional judgment (normal gives negative values).
Screech owl	0.6	Fleming et al. 1982, p.465	Screech owls fed 0.75 ppm endrin (n=14) for 83 days, wet-wt carcass data from Table 2 converted to whole-body.	
Mallard	1.2	Spann et al. 1986, p.758	Nonlethal; carcass (no feet, bill, GI, feathers); ave. both sexes; wet-wt; get same result on lipid-weight with 15% lipid in whole body.	

am = arithmetic mean
sd = standard deviation
wt = weight

Study Species	Value (s)	Literature Cited	Comments on How BAFs Were Calculated	Distribution of BAF Values
Endrin (cont.)				
Trophic Box—Shorebird				
Chicken	1.5	Cummings et al. 1967, p.421	Endrin in fat of hens (n=15), fed endrin-fortified feed for 14 wks, converted to whole-body (20% lipids); data from Fig. 3.	Lognormal; gm = 1.0; gsd = 1.6; sd based on data from papers; type based on professional judgment (normal gives negative values).
Screech owl	0.6	Fleming et al. 1982, p.465	Screech owls fed 0.75 ppm endrin (n=14) for 83 days, wet-wt carcass data from Table 2 converted to whole-body.	
Mallard	1.2	Spann et al. 1986, p.758	Nonlethal; carcass (no feet, bill, GI, feathers); ave. both sexes; wet-wt; get same result on lipid-weight with 15% lipid in whole body.	
Trophic Box—Water Bird				
Chicken	1.5	Cummings et al. 1967, p.421	Endrin in fat of hens (n=15), fed endrin-fortified feed for 14 wks, converted to whole-body (20% lipids); data from Fig. 3.	Lognormal; gm = 1.0; gsd = 1.6; sd based on data from papers; type based on professional judgment (normal gives negative values).
Screech owl	0.6	Fleming et al. 1982, p.465	Screech owls fed 0.75 ppm endrin (n=14) for 83 days, wet-wt carcass data from Table 2 converted to whole-body.	
Mallard	1.2	Spann et al. 1986, p.758	Nonlethal; carcass (no feet, bill, GI, feathers); ave. both sexes; wet-wt; get same result on lipid wt with 15% lipid in whole body.	

am = arithmetic mean
sd = standard deviation
wt = weight

Study Species	Value (s)	Literature Cited	Comments on How BAFs Were Calculated	Distribution of BAF Values
Endrin (cont.)				
Trophic Box—Small Bird				
Chicken	1.5	Cummings et al. 1967, p.421	Endrin in fat of hens (n=15), fed endrin-fortified feed for 14 wks, converted to whole-body (20% lipids); data from Fig. 3.	Lognormal; gm = 1.0; gsd = 1.6; sd based on data from papers; type based on professional judgment (normal gives negative values).
Screech owl	0.6	Fleming et al. 1982, p.465	Screech owls fed 0.75 ppm endrin (n=14) for 83 days, wet-wt carcass data from Table 2 converted to whole-body.	
Mallard	1.2	Spann et al. 1986, p.758	Nonlethal; carcass (no feet, bill, GI, feathers); ave. both sexes; wet-wt; get same result on lipid wt with 15% lipid in whole body.	
Trophic Box—Small Mammal				
Dog	0.08	Richardson et al. 1967, p.217	Consensus value; used with A.I.T. Walker 1969; converted to whole-body from fat; papers and value supplied by EPA.	Lognormal; gm = 0.08; gsd = 1.0; sd based on data in paper; type based on professional judgment.
Trophic Box—Medium Mammal				
Dog	0.16	Richardson et al. 1967, p.217	Consensus value; used with A.I.T. Walker 1969; converted to whole-body from fat; papers and value supplied by EPA.	Lognormal; gm = 0.16; gsd = 1.1; sd based on data in paper; type based on professional judgment.

am = arithmetic mean
sd = standard deviation
wt = weight

Study Species	Value (s)	Literature Cited	Comments on How BAFs Were Calculated	Distribution of BAF Values
DDE/DDT				
Trophic Box—Bald Eagle				
Kestrel	7.7	Rudolph et al. 1983, p.128	Controlled study, kestrels fed 6 ppm DDE wet-wt for 4 mos., converted to whole-body.	Lognormal; gm = 27.1; gsd = 2.4; standard deviation based on statistical information from papers, type based on Statgraphics and professional judgment.
Kestrel	29	Wiemeyer et al. 1986, p.17	Controlled study, wet-wt, converted to whole-body, n=23, dose = 2.8 ppm for 1 yr, Table 8 data.	
Barn owl	43.7	Mendenhall et al. 1983, p.237	Captive barn owls fed 2.83 ppm DDE for 2 yrs, mean of males and females, converted to whole-body.	
Peregrine	55	Enderson and Berger 1968, p.150	Field estimated BAF from 11 prey items, wet-wt egg residues converted to whole-body with info from Wiemeyer et al. 1986.	
Trophic Box—American Kestrel				
Kestrel	7.7	Rudolph et al. 1983, p.128	Controlled study, kestrels fed 6 ppm DDE wet-wt for 4 mos., converted to whole-body.	Uniform; endpoints = 7.7, 29; standard deviation based on data in papers gives negative values, so uniform chosen based on professional judgment.
Kestrel	29	Wiemeyer et al. 1986, p.17	Controlled study, wet-wt, converted to whole-body, n=23, dose = 2.8 ppm for 1 yr, Table 8 data.	

am = arithmetic mean
sd = standard deviation
wt = weight

Study Species	Value (s)	Literature Cited	Comments on How BAFs Were Calculated	Distribution of BAF Values
DDE/DDT (cont.)				
Trophic Box—Great Horned Owl				
Barn owl	43.7	Mendenhall et al. 1983, p.237	Captive barn owls fed 2.83 ppm DDE for 2 yrs, mean of males and females, converted to whole-body.	Lognormal; gm = 43.7; gsd = 2.4; used sd from bald eagle (data from kestrel, owl, and peregrine papers), mean from owl paper.
Trophic Box—Great Blue Heron				
Herring gull	102	Kozie and Anderson 1991, p.41	Field study, wet-wt basis for DDE, carcass converted to whole-body, fish and gull remains from bald eagle nests.	Normal; mean = 93.5; sd = 20; sd based on information in paper.
Herring gull	85	Braune and Norstrom 1989, p.961	Wet-wt DDE residues in whole body of herring gulls eating alewife, n= 10.	
Trophic Box—Shorebird				
Kestrel	7.7	Rudolph et al. 1983, p.128	Controlled study, kestrels fed 6 ppm DDE wet-wt for 4 mos., converted to whole-body.	Uniform; endpoints = 7.7, 29; standard deviation based on data in papers gives negative values, so uniform chosen based on professional judgment.
Kestrel	29	Wiemeyer et al. 1986, p.17	Controlled study, wet-wt, converted to whole-body, n=23, dose = 2.8 ppm for 1 yr, Table 8 data.	

am = arithmetic mean
sd = standard deviation
wt = weight

Study Species	Value (s)	Literature Cited	Comments on How BAFs Were Calculated	Distribution of BAF Values
DDE/DDT (cont.)				
Trophic Box—Waterbird				
Black duck	101	Longcore and Stendell 1977, p.295	Controlled study, dose = 2 ppm wet-wt for 7 mos., mean of male and female carcass converted to whole-body.	Normal; mean = 96, sd = 26.2; statistical information from E113 and D1; type based on professional judgment.
Herring gull	102	Kozie and Anderson 1991, p.41	Field study, wet-wt basis for DDE, carcass converted to whole-body, fish and gull remains from bald eagle nests.	
Herring gull	85	Braune and Norstrom 1989, p.961	Wet-wt DDE residues in whole body of herring gulls eating alewife, n= 10.	
Trophic Box—Small Bird				
Kestrel	7.7	Rudolph et al. 1983, p.128	Controlled study, kestrels fed 6 ppm DDE wet-wt for 4 mos., converted to whole-body.	Uniform; endpoints = 7.7, 29; standard deviation based on data in papers gives negative values, so uniform chosen based on professional judgment.
Kestrel	29	Wiemeyer et al. 1986, p.17	Controlled study, wet-wt, converted to whole-body, n=23, dose = 2.8 ppm for 1 yr, Table 8 data.	
Trophic Box—Small Mammal				
Mouse	0.44	Tomatis et al. 1974, p.886	Lab study, fat levels from 7 mice fed 250 ppm DDE, converted to whole-body (20%) lipids.	Uniform; endpoints = 0.44, 0.98; statistical computations from data in papers very complicated, as both values close to zero uniform chosen.
Vole	0.98	Forsyth et al. 1983, p.1629	6th year of field study, diet-weighted mean of 75% grass, 20% forbs, 5% roots.	

am = arithmetic mean
sd = standard deviation
wt = weight

Study Species	Value (s)	Literature Cited	Comments on How BAFs Were Calculated	Distribution of BAF Values
DDE/DDT (cont.)				
Trophic Box—Medium Mammal				
Mouse	0.44	Tomatis et al. 1974, p.886	Lab study, fat levels from 7 mice fed 250 ppm DDE, converted to whole-body (20%) lipids.	Uniform; endpoints = 0.44, 0.98; statistical computations from data in papers very complicated, as both values close to zero uniform chosen.
Rabbit	0.98	Forsyth et al. 1983, p.1629	6th year of field study, diet weighted mean of 75% grass, 20% forbs, 5% roots; authors state rabbit levels same as those of voles.	
Mercury				
Trophic Box—Bald Eagle				
Red-tailed hawk	0.3	Fimreite and Karstad 1971, p.296	Wet-wt ave. of 3 tissues (brain, breast muscle, liver), n=12, 12 wk exposure, dose = 3.9 ppm.	Triangular; mode = 0.3; endpoints = 0.001, 2.1; based on stats from paper and professional judgment.
Trophic Box—American Kestrel				
Red-tailed hawk	0.3	Fimreite and Karstad 1971, p.296	Wet-wt ave. of 3 tissues (brain, breast muscle, liver), n=12, 12 wk exposure, dose = 3.9 ppm.	Triangular; mode = 0.3; endpoints = 0.001, 2.1; based on stats from paper and professional judgment.
Trophic Box—Great Horned Owl				
Red-tailed hawk	0.3	Fimreite and Karstad 1971, p.296	Wet-wt ave. of 3 tissues (brain, breast muscle, liver), n=12, 12 wk exposure, dose = 3.9 ppm.	Triangular; mode = 0.3; endpoints = 0.001, 2.1; based on stats from paper and professional judgment.

am = arithmetic mean
sd = standard deviation
wt = weight

Study Species	Value (s)	Literature Cited	Comments on How BAFs Were Calculated	Distribution of BAF Values
Mercury (cont.)				
Trophic Box—Great Blue Heron				
Mallard	0.6	Vermeer et al. 1973, p.60	Wet-wt, breast muscles, weighted mean of food items from Table 4, 60% fish, 30% crayfish, 10% dragonfly nymphs.	Lognormal; gm = 4.1; gsd = 3.4; borrowed BAF and its distribution from water bird.
Duck	1.7	Gardiner 1972, p.422	Data from treatment group 2, wet-wt, 35-day exposure to 0.33 ppm, organic form, ave. of heart, breast muscle, liver, kidney.	
Black duck	4	Finley and Stendell 1978, p.60	Wet-wt, ave. of liver, kidney, brain, muscle, and feathers, dose = 3 ppm for 28 wks.	
Wood duck	7.4	Lindsay and Dimmick 1983, p.115	Juvenile wood ducks, n= 50; ave. tissue conc. based on liver, breast muscle, fat, data from Tables 1 & 2, wet-wt basis.	
Mallard	11.9	Heinz 1979, p.396	Wet-wt basis, ave. of liver, kidney, breast muscle, brain, dose = 0.1 ppm wet-wt in food.	
Mallard	14	Heinz 1980, p.384	Wet-wt basis, ave. of liver, kidney, brain, breast muscle, male and female, wild strain and farm-raised, dose = 0.1 ppm wet-weight.	

am = arithmetic mean
sd = standard deviation
wt = weight

Study Species	Value (s)	Literature Cited	Comments on How BAFs Were Calculated	Distribution of BAF Values
Mercury (cont.)				
Trophic Box—Shorebird				
Red-tailed hawk	0.3	Fimreite and Karstad 1971, p.296	Wet-wt ave. of 3 tissues (brain, breast muscle, liver), n=12, 12 wk exposure, dose = 3.9 ppm.	Triangular; mode = 0.3; endpoints = 0.001, 2.1; based on stats from paper and professional judgment.
Trophic Box—Water Bird				
Mallard	0.6	Vermeer et al. 1973, p.60	Wet-wt, breast muscles, weighted mean of food items from Table 4, 60% fish, 30% crayfish, 10% dragonfly nymphs.	Lognormal; gm = 4.1; gsd = 3.4; based on Statgraphics and professional judgment.
Duck	1.7	Gardiner 1972, p.422	Data from treatment group 2, wet-wt, 35-day exposure to 0.33 ppm, organic form, ave. of heart, breast muscle, liver, kidney.	
Black duck	4	Finley and Stendell 1978, p.60	Wet-wt, ave. of liver, kidney, brain, muscle, and feathers, dose = 3 ppm for 28 wks.	
Wood duck	7.4	Lindsay and Dimmick 1983, p.115	Juvenile wood ducks, n= 50; ave. tissue conc. based on liver, breast muscle, fat, data from Tables 1 and 2, wet-wt basis.	
Mallard	11.9	Heinz 1979, p.396	Wet-wt basis, ave. of liver, kidney, breast muscle, brain, dose = 0.1 ppm wet-wt in food.	
Mallard	14	Heinz 1980, p.384	Wet-wt basis, ave. of liver, kidney, brain, breast muscle, male and female, wild strain and farm-raised, dose = 0.1 ppm wet.	

am = arithmetic mean
sd = standard deviation
wt = weight

Study Species	Value (s)	Literature Cited	Comments on How BAFs Were Calculated	Distribution of BAF Values
Mercury (cont.)				
Trophic Box—Small Bird				
Red-tailed hawk	0.3	Fimreite and Karstad 1971, p.296	Wet-wt ave. of 3 tissues (brain, breast muscle, liver), n=12, 12 wk exposure, dose = 3.9 ppm.	Triangular; mode = 0.3; endpoints = 0.001, 2.1; based on stats from paper and professional judgment.
Trophic Box—Small Mammal				
Mink	22.5	Wren et al. 1987, p.444	Liver BAF, fed MeHg, sublethal 0.5 ppm dose for 8 months, assume wet-wt basis.	Triangular; mode = 22.5; endpoints = 0.001, 51.3; based on stats from pater and professioal judgment.
Trophic Box—Medium Mammal				
Mink	22.5	Wren et al. 1987, p.444	Liver BAF, fed MeHg, sublethal 0.5 ppm dose for 8 months, assume wet-wt basis.	Triangular; mode = 22.5; endpoints = 0.001, 51.3; based on stats from paper and professional judgment.
Trophic Box—Reptile/Terretrial Amphibian				
Bullsnake, Toad	1.5	RMA biota/small mammal data	RMA data; snake/mouse	Lognormal; gm = 1.5; gsd = 1.3; Cmedia data used to make professional judgment.

am = arithmetic mean
sd = standard deviation
wt = weight

Trophic Box	Value(s)*	Literature Cited	Comments	Distribution
Bald Eagle	0.0500, 0.110, 0.0911	Palmer, 1988	Lit (bald eagle): 0.0500, 0.110, 0.0911; 0.102, 0.123; 0.0911; 0.0567; n = 7	Normal
	0.102, 0.123, 0.0911	Stalmaster & Gessaman 1984		
	0.0567	Swies 1986		
Great Blue Heron	0.0500, 0.110, 0.0911	Palmer 1988	Lit (bald eagle): 0.0500, 0.110, 0.0911; 0.102, 0.123; 0.0911; 0.0567; n = 7	Normal
	0.102, 0.123, 0.0911	Stalmaster & Gessaman 1984		
	0.0567	Swies 1986		
Great Horned Owl	0.0500, 0.110, 0.0911	Palmer, 1988	Lit (bald eagle): 0.0500, 0.110, 0.0911; 0.102, 0.123; 0.0911; 0.0567; n = 7	Normal
	0.102, 0.123, 0.0911	Stalmaster & Gessaman 1984		
	0.0567	Swies 1986		
American Kestrel	0.0500, 0.110, 0.0911	Palmer, 1988	Lit (bald eagle): 0.0500, 0.110, 0.0911; 0.102, 0.123; 0.0911; 0.0567; n = 7	Normal
	0.102, 0.123, 0.0911	Stalmaster & Gessaman 1984		
American Kestrel (cont'd)	0.0567	Swies 1986		

* Values reported in kilograms per kilogram of body weight per day
Lit = Literature values

Trophic Box	Value(s)*	Literature Cited	Comments	Distribution
Medium Mammal	0.096	Tileston and Lechleitner 1966	Value from white-tailed prairie dogs.	Fixed
Small Mammal	0.12	Sax 1984		Fixed
Small Bird	0.0879	Shuman et al. 1988	Lit (mourning dove): grams offered per day; n = 8; 10, 10, 16, 7.5, 7.5, 16, 7.5, 7.5; and Lit (mourning dove): grams of body weight; n = 2; 120.5, 128.4.	Lognormal GM = 0.0879; GSD = 1.652
Shorebird	0.0879	Shuman et al. 1988	Lit (mourning dove): grams offered per day; n = 8; 10, 10, 16, 7.5, 7.5, 16, 7.5, 7.5; and Lit (mourning dove): grams of body weight; n = 2; 120.5, 128.4.	Lognormal; GM = 0.0879; GSD = 1.652.
Water Bird	0.0565, 0.0522 0.100, 0.0954	Miller 1975 Sax 1984 White & Finley 1978	Lit (mallard, coot): 0.0565, 0.0522; 0.100; 0.0954; n = 4.	Normal

* Values reported in kilograms per kilogram of body weight per day
Lit = Literature values

Trophic Box	Prey Group	Value(s)	Distribution	Literature Cited	Comments
Bald Eagle	Soil	0.03	Fixed	Beyer et.al. 1991	Literature (turkey scats)/Professional judgment: based on 0.062, N = 6
	Small Mammal	0.0-0.036; 0.0		ESE 1988 USFWS 1989	RMA data (bald eagle castings): based on the sum of weighted averages of three winter studies [0.0 and 0.036 (ESE 1988), 0.0, (USFWS 1989) and adjusted for inclusion of soil, N = 364
	Medium Mammal	0.863-0.900; 0.949		ESE 1988 USFWS 1989	RMA data (bald eagle castings): based on the sum of weighted averages of three winter studies [0.863 and 0.900 (ESE 1988), 0.949 (USFWS 1989)], and adjusted for inclusion of soil, N = 364
	Small Bird	0.018-0.030; 0.007		ESE 1988 USFWS 1989	RMA data (bald eagle castings): based on the sum of weighted averages of three winter studies [0.018 and 0.030 (ESE 1988), 0.007 (USFWS 1989)], and adjusted for inclusion of soil, N = 364
	Waterbird	0.054; 0.035; 0.014		ESE 1988 USFWS 1989	RMA data (bald eagle castings): based on the sum of weighted averages of "unknown" category of birds [0.054 and 0.035 (ESE 1988)] and waterfowl data [0.014 (USFWS 1989)] from three winter studies, and adjusted for inclusion of soil, N = 364
	Large Fish	0.0; 0.005; 0.0		ESE 1988 USFWS 1989	RMA data (bald eagle castings): based on sum of weighted averages of three winter studies [0.0 and 0.005 (ESE 1988); 0.0 (USFWS 1989)], and adjusted for inclusion of soil, N = 364
Great Blue Heron	Soil	0.04	Fixed	Beyer et al. 1991	Literature (Canada goose scats)/Professional judgment: based on 0.082, N = 23
	Reptile	0.02		Palmer 1962	Literature (great blue heron): proportioned from 0.0425, N = 1
	Small Mammal	0.04		Palmer 1962	Literature (great blue heron): proportioned from 0.0466, N = 1
	Water	0.08		Palmer 1962	Professional judgment

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	Aquatic Plant	0.02		Palmer 1962	Literature (great blue heron): proportioned from 0.0248, adjusted for inclusion of water, N = 1
	Aquatic Invert.	0.16		Palmer 1962	Literature (great blue heron): proportioned from 0.171, adjusted for inclusion of water, N = 1
	Small Fish	0.37		Palmer 1962	Literature (great blue heron): proportioned from 0.4316, adjusted for inclusion of water, N = 1
	Large Fish	0.24		Palmer 1962	Literature (great blue heron): proportioned from 0.266, adjusted for inclusion of water, N =1
	Amphibian	0.03		Palmer 1962	Literature (great blue heron): proportioned from 0.0425, adjusted for inclusion of water, N =1
Great Horned Owl	Soil	0.03	Fixed	Beyer et al. 1991	Literature (turkey scats)/Professional judgment: based on 0.062
	Small Mammal	0.665		EBASCO 1991d	RMA data (great horned owl nests): based on 0.685 (combined data), and adjusted for inclusion of soil
	Medium Mammal	0.25		EBASCO 1991d	RMA data (great horned owl nests): based on 0.26 (combined data), and adjusted for inclusion of soil
	Small Bird	0.055		EBASCO 1991d	RMA data (great horned owl nests): based on 0.0548 (combined data)
American Kestrel	Soil	0.03	Fixed	Beyer et al. 1991	Literature (turkey scats)/Professional judgment: based on 0.062
	Insect	0.86		EBASCO 1991c	RMA data (American kestrel boxes): based on 0.89 (combined data), and adjusted for inclusion of soil
	Small Mammal	0.093		EBASCO 1991c	RMA data (American kestrel boxes): 0.093, combined data
	Small Bird	0.017		EBASCO 1991c	RMA data (American kestrel boxes): 0.017, combined data

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Medium Mammals	Soil	0.077; 0.063	Fixed	Beyer et al. 1991; Arthur and Gates 1991	Literature (prairie dog scats [12(1)^], jackrabbit scats [>84])/Professional judgment: based on 0.077 (Beyer et al 1991), 0.063 (Arthur and Gates 1991), N=2
	Terrestrial Plant	0.88			Professional judgment
	Insect	0.04			Professional judgment
Small Mammals	Soil	0.03; 0.024; <0.02	Fixed	Garten, 1980; Beyer et al. 1991	Literature (mouse, rat, vole)/Professional judgment: averaged from 0.03 (Garten 1980), 0.024 (Beyer et al 1991), <0.02 (F-904), N = 3
	Terrestrial Plant	0.47			Professional judgment
	Earthworm	0.03			Professional judgment
	Insect	0.48			Professional judgment
Small Birds	Soil	0.06	Fixed	Beyer et al. 1991	Literature (turkey scats): 0.062, N = 1
	Terrestrial Plant	0.17		EBASCO 1991b	RMA data (vesper sparrow): based on 0.20 (combined data), and adjusted for inclusion of earthworms and soil, N = 1
	Earthworm	0.05			Professional judgment
	Insect	0.72		EBASCO 1991b	RMA data (vesper sparrow): based on 0.80 (combined data), and adjusted for inclusion of earthworms and soil, N = 1
Shorebird	Terrestrial Plant	0.02	Fixed		Professional judgment
	Insect	0.69			Professional judgment

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Sediment	0.17; 0.30; 0.073; 0.18; 0.091; 0.35	Beyer et al. 1991 Reeder 1951	Literature (shorebirds, sandpipers, woodcock): averaged from 0.17, 0.30, 0.073, 0.18, 0.091 (Beyer et al 1991); 0.35 (Reeder 1951), N = 6
Aquatic Invert.	0.10		Professional judgment

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Waterbirds	Water	0.02	Fixed		Professional judgment
	Sediment	0.04		Beyer et al.. 1991; Phillips 1922; Cottam 1939	Literature (Canada goose scats [23(1)^], shoveler stomachs [70], ruddy duck observations)/Professional judgment: based on 0.082 (Beyer et al 1991) and qualitative literature information (Phillins 1992, Cottam 1939), N = 23
	Aquatic Plant	0.93		EBASCO 1991a	RMA data (mallard, coot): based on 0.93, averaged from 1.0, 1.0, 0.92, 1.0, 0.83, 1.0, 0.90, 0.80, 0.95, 0.95, 0.90, and adjusted for inclusion of water and sediment, N = 11
	Aquatic Invert.	0.068	Fixed	EBASCO 1991a	RMA data (mallard, coot)/Professional judgment: based on 0.068, averaged from 0.01, 0.17, 0.07, 0.10, 0.20, 0.0, 0.0, 0.0, 0.05, 0.05, 0.10, N = 11

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Table C.2-9 Dietary Fraction Consensus for ERC Model Based on Biomass

MODEL COMPONENTS

Trophic Box	Occurrence \1	Represent. Weight. (g)	Converted Biomass (g)\2	Apportioned Values \3	Biomass Fraction \4	Comments
Bald Eagle						
Soil	0.03	NA	0.030	NA	0.029	Soil likely from prey contact during feeding
Small Mammal	0.005	33	0.165	0.0003	0.000	RMA data vole to deer mice size; Marti 1974
Medium Mammal	0.915	500	457.500	0.9640	0.936	RMA weighted field data p. dog @434g and cottontail @658.5g
Small Bird	0.019	65	1.235	0.0026	0.003	Ave. small and med. passerine (30–100g; Ryder)
Waterbird	0.029	506	14.674	0.0309	0.030	RMA field data for mallards
Large Fish	0.002	500	1.000	0.0021	0.002	RMA probable prey: bass @400g; carp @900g
Great Blue Heron						
Soil	0.04	NA	0.040	NA	0.036	Soil likely from prey contact during feeding
Reptile	0.02	300	6.000	0.0676	0.060	RMA field data for bull snakes
Small Mammal	0.04	33	1.320	0.0149	0.013	RMA data vole to deer mice size; Marti 1974
Water	0.08	NA	0.080	NA	0.071	Professional judgment
Plant Aquatic	0.02	0.1	0.002	0.0000	0.000	Plant likely from prey contact during feeding
Aquatic Invertebrate	0.16	15	2.400	0.0270	0.024	Crayfish most important from RMA data
Small Fish	0.37	50	18.500	0.2083	0.186	RMA weighted data and probable prey about 2 oz.
Large Fish	0.24	250	60.000	0.6755	0.604	RMA probable prey: bass @400g; carp @900g
Amphibian	0.03	20	0.600	0.0068	0.006	RMA probable prey: leopard frog est. @16.6g
Great Horned Owl						
Soil	0.03	NA	0.030	NA	0.029	Soil likely from prey contact during feeding
Small Mammal	0.665	33	21.945	0.125	0.121	RMA data vole to deer mice size; Marti 1974
Medium Mammal	0.25	600	150.000	0.855	0.830	RMA weighted field data P. Dog @434 g and cottontail @658.6 g
Small Bird	0.055	65	3.575	0.020	0.020	Ave. small and med. passerine (30–100g; Ryder)
American Kestrel						
Soil	0.03	NA	0.030	NA	0.029	Soil likely from prey contact during feeding
Insect	0.86	0.6	0.516	0.190	0.184	RMA probable prey:large grasshopper aver. weight. @0.6g
Small Mammal	0.093	20	1.860	0.685	0.665	RMA data vole to deer mice size; Marti 1974
Small Bird	0.017	20	0.340	0.125	0.122	Ave. passerine birds(30–100g; Ryder), downsized for consensus
Medium Mammal						
Soil	0.08	NA	0.080	NA	0.074	Soil likely from prey contact during feeding
Plant Terrain	0.88	60	52.800	0.9995	0.926	Ten percent of live weight. of animal
Insect	0.04	0.6	0.024	0.0005	0.000	RMA probable prey: Large grasshopper aver. weight. @0.6g

Table C.2-9 Dietary Fraction Consensus for ERC Model Based on Biomass

MODEL COMPONENTS

Trophic Box	Occurrence \1	Represent. Weight. (g)	Converted Biomass (g)\2	Apportioned Values \3	Biomass Fraction \4	Comments
Small Mammals						
Soil	0.02	NA	0.020	NA	0.020	Soil likely from adsorption to plant tissue
Plant Terrain	0.47	5	2.350	0.884	0.866	Fifteen percent of live weight. of animal
Earthworm	0.03	0.7	0.021	0.008	0.008	RMA field sample data
Insect	0.48	0.6	0.288	0.108	0.106	RMA probable prey: large grasshopper aver. weight. @0.6g
Small Birds						
Soil	0.06	NA	0.060	NA	0.057	Soil likely from prey contact during feeding
Plant Terrain	0.17	0.2	0.034	0.119	0.113	Weighted est. for seeds in the diet (professional judgment)
Earthworm	0.05	0.7	0.035	0.123	0.116	RMA field sample data
Insect	0.72	0.3	0.216	0.758	0.714	RMA probable prey: grasshopper aver. weight. @0.3g
Shorebirds						
Plant Terrain	0.02	0.1	0.002	0.008	0.007	Plant likely from prey contact during feeding
Insect	0.69	0.3	0.207	0.866	0.728	RMA probable prey: grasshopper aver. weight. @0.3g
Sediment	0.19	NA	0.190	NA	0.160	Sediment likely from prey contact during feeding
Aquatic Invertebrate	0.1	0.3	0.030	0.126	0.105	Coleoptera and Orthoptera most important from RMA data
Waterbirds						
Water	0.02	NA	0.020	NA	0.019	Professional estimate
Sediment	0.04	NA	0.040	NA	0.038	Sediment likely from prey contact during feeding
Plant Aquatic	0.84	30	25.200	0.999	0.942	Six percent of live weight of animal
Aquatic Invertebrate	0.1	0.3	0.030	0.001	0.001	Coleoptera and Orthoptera most important from RMA data

1 = Values used in draft final report (brown cover) based on percent occurrence in diets.
2 = Occurrence times representative values.
3 = Values are re-proportioned from prey items only to equal 100%.
4 = Values to be used in final report (gray cover) based on biomass of dietary items.
ave = average
g = grams
NA = Not applicable because values already in percent mass format.

DDE/DDT

Trophic Box	Value	Source(s)	Endpoint
Kestrel	12.8	Wiemeyer et al.. 1986	NOAEL-r
Bald Eagle	6.5	Wiemeyer et al.. 1984, Kaiser et al.. 1980, Prouty et al.. 1977	NOAEL-r
Great Horned Owl	6.4	Mendenhall et al.. 1983	NOAEL-r
Great Blue Heron	30	Fitzner et al.. 1988	NOAEL-r
Shorebird	11	Fox 1976	NOAEL-r
Water Bird	4.4	Longcore and Stendell 1977	LOAEL-r
Small Bird	54	Jefferies 1971	LOAEL-r
Mammal	13.4	Laug et al.. 1949	NOAEL-p

DDE/DDT

Trophic Box	Comments
Kestrel	Whole-body value, converted from carcass with factor of 1.3; modifying factor needed for co-contamination.
Bald Eagle	Converted from critical level of 3 ppm in egg to carcass (0.6) then to whole body (1.3).
Great Horned Owl	Based on 10% shell thinning, converted to egg level then to carcass from information provided in paper, then to whole body (1.3).
Great Blue Heron	14 ppm wet-weight in eggs assoc. with 10% eggshell thinning, converted to carcass (0.6) and then to whole body (1.3).
Shorebird	4 ppm wet-weight in eggs critical level for adverse reproductive effects, converted to carcass (0.6) then to whole body (1.3).
Water Bird	Egg level of 6.2 ppm wet-weight after adults on clean feed for 2 years., carcass level of female 3.4, converted to whole body (1.3).
Small Bird	Lower fertility, fewer hatched, female fed 12 ppm wet-weight DDE, converted from 15.8 ppm in liver of hatched birds.
Mammal	Fat level after 27 weeks expos. to 5 ppm in diet (dry weight) converted to whole body based on 20% lipids.

Aldrin/Dieldrin

Trophic Box	Value	Source(s)	Endpoint
Kestrel	2.9	Weimeyer et al.. 1986	NOAEL-r
Bald Eagle	12.2	Mendenhall et al.. 1983	NOAEL-r
Great Horned Owl	12.2	Mendenhall et al.. 1983	NOAEL-r
Great Blue Heron	1.3	Ohlendorf et al.. 1981, 1979	NOAEL-m
Shorebird	2.9	Weimeyer et al.. 1986	NOAEL-r
Water Bird	7.1	Sharma et al.. 1976	LOAEL-b
Small Bird	2.9	Weimeyer et al.. 1986	NOAEL-r
Mammal	4.5	Harr et al.. 1970, Walker et al.. 1969	NOEL-r

Aldrin/Dieldrin Trophic Box	Comments
Kestrel	Average of birds that survived, low dose; modifying factors for co-contaminants and unclear endpoint.
Bald Eagle	Based on owl data; modifying factors for threatened and endangered, endpoint unclear.
Great Horned Owl	Strong paper; modifying factor for endpoint unclear.
Great Blue Heron	Field study, value calculated based on 85th percentile (FWS); modifying factor for co-contaminant.
Shorebird	Kestrel value, strong paper; modifying factors for co-contaminants and endpoint unclear.
Water Bird	Based on mallard data, assume 20% lipids for conversion to whole body; modifying factor for endpoint unclear.
Small Bird	Value from kestrel data; modifying factors for co-contaminants and endpoint unclear.
Mammal	Diet of 1.25 ppm wet-wt. multiplied by BAF of 18, then converted from fat to whole body based on 20% lipids.

Endrin

Trophic Box	Value	Source(s)	Endpoint
Kestrel	0.78	Fleming et al.. 1982	NOAEL-r
Bald Eagle	0.78	Fleming et al.. 1982	NOAEL-r
Great Horned Owl	0.78	Fleming et al.. 1982	NOAEL-r
Great Blue Heron	0.13	Ohlendorf et al.. 1979	NOAEL-l
Shorebird	0.78	Fleming et al.. 1982	NOAEL-r
Water Bird	1.4	Ohlendorf et al.. 1979	NOAEL-r
Small Bird	0.78	Fleming et al.. 1982	NOAEL-r
Mammal	–		

Endrin Trophic Box	Comments
Kestrel	Based on screech owl data, n = 1, 0.3 ppm wet-wt. critical level in eggs converted to carcass (0.6) then to whole body (1.3).
Bald Eagle	Based on screech owl data, n = 1, 0.3 ppm wet-wt. critical level in eggs converted to carcass (0.6) then to whole body (1.3).
Great Horned Owl	Based on screech owl data, n = 1, 0.3 ppm wet-wt. critical level in eggs converted to carcass (0.6) then to whole body (1.3).
Great Blue Heron	Field data for herons; modifying factors for co-contaminants and unclear endpoint.
Shorebird	Based on screech owl data, n = 1, 0.3 ppm wet-wt. critical level in eggs converted to carcass (0.6) then to whole body (1.3).
Water Bird	1.1 ppm in mallard carcasses from diet of 1 ppm, carcass:whole body conversion of 1.3, data p. 758.
Small Bird	Based on screech owl data, n = 1, 0.3 ppm wet-wt. critical level in eggs converted to carcass (0.6) then to whole body (1.3).
Mammal	No appropriate data found.

Mercury

Trophic Box	Value	Source(s)	Endpoint
Kestrel	0.83	Heinz 1976	LOAEL-r, b
Bald Eagle	0.83	Heinz 1976	LOAEL-r, b
Great Horned Owl	0.83	Heinz 1976	LOAEL-r, b
Great Blue Heron	0.83	Heinz 1976	LOAEL-l, b
Shorebird	0.83	Heinz 1976	LOAEL-r, b
Water Bird	0.83	Heinz 1976	LOAEL-r, b
Small Bird	0.83	Heinz 1976	LOAEL-r, b
Mammal	–		

Mercury Trophic Box	Comments
Kestrel	Liver value (0.89ppm) for mallards, converted to whole body with regression from Stickel et al. 1977.
Bald Eagle	Liver value (0.89ppm) for mallards, converted to whole body with regression from Stickel et al. 1977.
Great Horned Owl	Liver value (0.89ppm) for mallards, converted to whole body with regression from Stickel et al. 1977.
Great Blue Heron	Liver value (0.89ppm) for mallards, converted to whole body with regression from Stickel et al. 1977.
Shorebird	Liver value (0.89ppm) for mallards, converted to whole body with regression from Stickel et al. 1977.
Water Bird	Liver value (0.89ppm) for mallards, converted to whole body with regression from Stickel et al. 1977.
Small Bird	Liver value (0.89ppm) for mallards, converted to whole body with regression from Stickel et al. 1977.
Mammal	No appropriate data found.

b = behavior
m = mortality
p = pathology
r = reproduction

Basis for Uncertainty	Uncertainty Value Assigned
Intertaxon Variability Extrapolation Category—	
Same species	1
Same genus, different species	2
Same family, different genus	3
Same order, different family	4
Same class, different order	5
Study Duration Extrapolation Category—	
Chronic studies where contaminants attained equilibrium	1
Chronic studies where equilibrium not attained or possibly not attained, including subchronic studies	5
Acute studies	20

Study Endpoint Extrapolation Category—

	Nonlethal	Lethal
No observed effects level	(NOEL): 1	NOEL: 3
No observed adverse effects level	(NOAEL): 1	NOAEL: 3
Lowest observed effects level	(LOEL): 3	LOEL: 10
Lowest observed adverse effect level	(LOAEL): 5	LOAEL: 10
Frank effects level	(FEL): 10	FEL: 15

Modifying Factor Category—	
Threatened and endangered species	0 or 2
Relevance of endpoint to ecological health	-1 to 0
Extrapolating lab to field	0 to 2
Study had co-contaminants	-1 to +1
Endpoint was unclear	-2 to +2
Study species was obviously highly sensitive	-2 to +2
Ratios used to get from organ or egg to whole body	0 to 2*
Intraspecific variability	0 to 2

* Used only for MATC (not TRV) uncertainty factor development.

Table C.2-12 Post-Uncertainty MATC

Aldrin/Dieldrin	Critical Value (mg/kg bw)	Intertaxon (1)	Study Duration (Q2)	Study Endpoints (Q3)	Modifying Factor* (U)	T&E	Endpoint Relevance	Lab to Field	Co-Contam.	Unclear Endpoint	ID. Sensitive Species	Tissue to Whole-Body Ratio	Intraspecific Variability
American Kestrel	2.9	1	1	1	4			1		2			1
Bald Eagle	12.2	5	1	1	6	2		1		2			1
Great Horned Owl	12.2	4	1	1	4			1		2			1
Great Blue Heron	1.3	1	1	3	0.5			0	-1				0
Shorebird	2.9	5	1	1	4			1		2			1
Waterbird	7.1	5	1	3	2		-1	1				1	1
Small Bird	2.9	5	1	1	4			1		2			1
Mammal	4.5	4	1	1	6			2		2		1	1

Trophic Box	Total UF	Final MATC
Amercian Kestrel	4	0.73
Bald Eagle	30	0.41
Great Horned Owl	16	0.76
Great Blue Heron	1.5	0.87
Shorebird	20	0.15
Waterbird	30	0.24
Small Bird	20	0.15
Mammal	24	0.19

Table C.2-12 Post-Uncertainty MATC

DDT/DDE	Critical Value (mg/kg bw)	Intertaxon (1)	Study Duration (Q2)	Study Endpoints (Q3)	Modifying Factor* (U)	T&E	Endpoint Relevance	Lab to Field	Co-Contam.	Unclear Endpoint	ID. Sensitive Species	Tissue to Whole-Body Ratio	Intraspecific Variability
American Kestrel	12.8	1	1	1	3			1				1	1
Bald Eagle	6.5	1	1	1	3	2		0				1	0
Great Horned Owl	6.4	4	1	1	3			1				1	1
Great Blue Heron	30	1	1	1	2			0				1	1
Shorebird	11	4	1	1	2			0				1	1
Waterbird	4.4	5	1	5	1			1			-2		1
Small Bird	54	5	5	5	3			1				1	1
Mammal	13.4	4	5	1	3		-1	2				1	1

Trophic Box	Total UF	Final MATC
American Kestrel	3	4.27
Bald Eagle	3	2.17
Great Horned Owl	12	0.53
Great Blue Heron	2	15.00
Shorebird	8	1.38
Waterbird	25	0.18
Small Bird	375	0.14
Mammal	60	0.22

Endrin	Critical Value (mg/kg bw)	Intertaxon (1)	Study Duration (Q2)	Study Endpoints (Q3)	Modifying Factor* (U)	T&E	Endpoint Relevance	Lab to Field	Co-Contam.	Unclear Endpoint	ID. Sensitive Species	Tissue to Whole-Body Ratio	Intraspecific Variability
American Kestrel	0.78	5	1	1	3			1				1	1
Bald Eagle	0.78	5	1	1	5	2		1				1	1
Great Horned Owl	0.78	3	1	1	3			1				1	1
Great Blue Heron	0.13	1	1	3	0.5								0.5
Shorebird	0.78	5	1	1	3			1				1	1
Waterbird	1.4	5	1	1	3			1				1	1
Small Bird	0.78	5	1	1	3			1				1	1
Mammal	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

Trophic Box	Total UF	Final MATC
American Kestrel	15	0.05
Bald Eagle	25	0.03
Great Horned Owl	9	0.09
Great Blue Heron	1.5	0.09
Shorebird	15	0.05
Waterbird	15	0.09
Small Bird	15	0.05
Mammal	NA	NA

Table C.2-12 Post-Uncertainty MATC

Mercury	Critical Value (mg/kg bw)	Intertaxon (1)	Study Duration (Q2)	Study Endpoints (Q3)	Modifying Factor* (U)	T&E	Endpoint Relevance	Lab to Field	Co-Contam.	Unclear Endpoint	ID. Sensitive Species	Tissue to Whole-Body Ratio	Intraspecific Variability
American Kestrel	0.83	5	1	5	2			1	-1			1	1
Bald Eagle	0.83	5	1	5	4	2		1	-1			1	1
Great Horned Owl	0.83	5	1	5	2			1	-1			1	1
Great Blue Heron	0.83	5	1	5	3			1				1	1
Shorebird	0.83	5	1	5	3			1				1	1
Waterbird	0.83	5	1	5	3			1				1	1
Small Bird	0.83	5	1	5	2			1	-1			1	1
Mammal	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

Trophic Box	Total UF	Final MATC
American Kestrel	50	0.02
Bald Eagle	100	0.01
Great Horned Owl	50	0.02
Great Blue Heron	75	0.01
Shorebird	75	0.01
Waterbird	75	0.01
Small Bird	50	0.02
Mammal	NA	NA

Total UF = I * Q2 * Q3 * U
U = Sum of factors to right
Final TRV = Critical value/total UF

* Note: If 0<=U<1, replaced with 1; If U<0, replaced with 0.5

Study Species	Value (s) (acres)	Literature Cited	Comments
Trophic Box*—Top Predator			
Bald eagle	8277	Recommendation from USFWS	Bald eagle management unit plus all prairie dog towns present on RMA in April 1993.
Great blue heron	Lake size + 4%	OAS professional judgment	Lake Mary = 12 acres Lake Ladora = 72 acres Lower Derby = 91 acres Upper Derby = 69 acres East Upper Derby = 23 acres Rod and Gun Club Pond = 28 acres
Great horned owl	512	Zeiner 1990	Study best describes owl home range at RMA.
American kestrel	270	Balgooyen 1976	Study best describes kestrel home range at RMA.
Trophic Box*—Medium Mammal			
Black-tailed prairie dog	0.5	Tileston 1966	Also recommended by USFWS.
Cottontail	8.6	USFWS HEP manual	Value recommended from USFWS Habitat Evaluation Procedures Manual for cottontails
Trophic Box*—Small Mammal			
Deer mouse	0.5	Blair 1942	Also OAS professional judgment based on deer mouse foraging range.
13-lined ground squirrel	1.0	Clark 1981, Evans 1951	Mean of 3 values; 0.86, 0.59, 1.63.

Study Species	Value (s) (acres)	Literature Cited	Comments
Trophic Box*—Small Bird			
Mourning Dove	54	Schoener 1968	Study best describes mourning dove home range at RMA.
Meadowlark	15	Bent 1965, Schroeder 1968, Welty 1979, personal communication, Bury USFWS 1992	Studies best describe meadowlark home range at RMA.
Vesper sparrow	2.5	Reed 1985 plus OAS professional judgment	Study best describes vesper sparrow home range at RMA.
Trophic Box*—Reptiles/Amphibians			
Gopher (bull) snake	11	Stickel and Cope 1947 plus OAS professional judgment	Study best describes bull snake home range at RMA.
Trophic Box*—Insects			
Grasshopper	0.010	Personal communication, Kimberly A. with Zoology Department, Colorado State University	Grasshopper nymph movements in a fractal analysis study.
Trophic Box*—Earthworm			
Earthworm	0.022	OAS professional judgment	Based on 5.5m radius.

Study Species	Value (s) (acres)	Literature Cited	Comments
Trophic Box*—Terrestrial Plants			
All annual plants	0.0007	Personal communication, Dr. Leslie Fraley, Radiology Department, Colorado State University	Study using radioisotopes to determine 3-D maps of root distributions.
Trophic Box*—Shorebird			
Killdeer	12	USFWS recommendation plus OAS professional judgment	Defended territory of killdeer from Great Salt Lake Study by Fellows recommended by USFWS consensus reached at 12 acres.
Trophic Box*—Water Bird			
Mallard Blue-winged teal American coot	Lake area where sample was collected	OAS professional judgment	Lake Mary = 12 ac Lake Ladora = 69 ac Lower Derby = 89 ac Upper Derby = 67 ac East Upper Derby = 22 ac Rod and Gun Club Pond = 27 ac

* Exposure ranges for all other species representing aquatic trophic boxes were not calculated. The aquatic environment is relatively homogenous and home range is inappropriate.

ac acres

OAS Organizations and State Subcommittee

USFWS United States Fish and Wildlife Service

Table C.2-14 Critical Dose Values (Pre-UF Toxicity Reference Values)

COC	TROPHIC BOX	SPECIES OF CONCERN	TEST SPECIES	CITATION	ENDPOINT/ STUDY TYPE	PPM IN DIET	DOSE (mg/kg-bw/day)
Aldrin	Heron	great blue heron	mallard	Sharma et al.'76	LOEL[1]	4	0.4
Dieldrin	Eagle	bald eagle	kestrel/owl	Wiemeyer et al.. '86/Mendenhall et al. '83	NOAEL-r[1]	0.28/0.58	0.05 Weimeyer et al.. '86/Mendenhall et al.. '83
	Owl	great horned owl	owl	Mendenhall et al. '83	NOAEL-r[1]	0.58	0.06 Mendenhall et al.. '83
	Kestrel	American kestrel	kestrel	Wiemeyer et al.'86	NOAEL-r[1]	0.28	0.04 Weimeyer et al. '86
	Small bird	mourning dove	domestic pigeon	Robinson & Crabtree '69	NOAEL-r	3	0.28 Robinson and Crabtree '69
	Shorebird	killdeer	quail/pigeon	Shellenberger '78/Stickel et al. '69/Robinson & Crabtree '69	NOAEL-r,l	2	0.22 Shellenberger '78/Stickel et al. '69/Robinson and Crabtree '69
	Water bird	mallard	mallard	Sharma et al.'83	LOEL[1]	4	0.4
	Sm. mammal	deer mouse	rat	Harr et al. '70	NOAEL-r	1.25	0.06
	Med. mammal	prairie dog	rat	Harr et al. '70	NOAEL-r	1.25	0.06
	T. plants	plants	ND	ND	ND	ND	ND
	Insects	insects	ND	ND	ND	ND	ND
	Earthworm	earthworm	ND	ND	ND	ND	ND
	Reptile	toad/snake	ND	ND	ND	ND	ND
Arsenic	Heron	great blue heron	duck	Van Vleet '82	LOAEL	ND	18.9
	Eagle	bald eagle	duck	Van Vleet '82	LOAEL	ND	18.9
	Owl	great horned owl	duck	Van Vleet '82	LOAEL	ND	18.9
	Kestrel	American kestrel	duck	Van Vleet '82	LOAEL	ND	18.9
	Small bird	mourning dove	duck	Van Vleet '82	LOAEL	ND	18.9
	Shorebird	killdeer	duck	Van Vleet '82	LOAEL	ND	18.9
	Water bird	mallard	duck	Van Vleet '82	LOAEL	ND	18.9
	Sm. mammal	deer mouse	rat	Byron et al. '67	LOAEL-p	31	1.5
	Med. mammal	prairie dog	rat	Byron et al. '67	LOAEL-p	31	1.5
	T. plants	plants	grasses	ESE tox. file	field data	NA	19
	Insects	insects	ND	ND	ND	ND	ND
	Earthworm	earthworm	ND	ND	ND	ND	ND
	Reptile	toad/snake	ND	ND	ND	ND	ND
Chlordane	Heron	great blue heron	quail	Hudson et al. '84	LD50	NA	14.1
	Eagle	bald eagle	quail	Hudson et al. '84	LD50	NA	14.1
	Owl	great horned owl	quail	Hudson et al. '84	LD50	NA	14.1
	Kestrel	American kestrel	quail	Hudson et al. '84	LD50	NA	14.1
	Small bird	mourning dove	quail	Hudson et al. '84	LD50	NA	14.1
	Shorebird	killdeer	quail	Hudson et al. '84	LD50	NA	14.1
	Water bird	mallard	duck	Hudson et al. '84	LD50	NA	1250
	Sm. mammal	deer mouse	rat	EPA '84	NOEL	NA	1.2
	Med. mammal	prairie dog	rat	EPA '84	NOEL	NA	1.2
	T. plants	plants	ND	ND	ND	NA	ND
	Insects	insects	ND	ND	ND	ND	ND
	Earthworm	earthworm	ND	ND	ND	ND	ND
	Reptile	toad/snake	ND	ND	ND	ND	ND

1. A modifying factor will be applied for calculation of the total UF.
2. The barn owl may be an unusually sensitive species; this will be taken into account when assigning UFs.
3. The total UF will reflect the fact this is a NOAEL.

COC	TROPHIC BOX	SPECIES OF CONCERN	TEST SPECIES	CITATION	ENDPOINT/ STUDY TYPE	PPM IN DIET	DOSE (mg/kg-bw/day)
CPM-Sulfide	Heron	great blue heron	ND	ND	ND	ND	ND
	Eagle	bald eagle	ND	ND	ND	ND	ND
	Owl	great horned owl	ND	ND	ND	ND	ND
	Kestrel	American kestrel	ND	ND	ND	ND	ND
	Small bird	mourning dove	ND	ND	ND	ND	ND
	Shorebird	killdeer	ND	ND	ND	ND	ND
	Water bird	mallard	ND	ND	ND	ND	ND
	Sm. mammal	deer mouse	rat	Thake et al. '79	LOAEL	281	14.1
	Med. mammal	prairie dog	rat	Thake et al. '79	LOAEL	281	14.1
	T. plants	plants	grasses	field data	NA	NA	0.7
	Insects	insects	ND	ND	ND	ND	ND
	Earthworm	earthworm	ND	ND	ND	ND	ND
	Reptile	toad/snake	ND	ND	ND	ND	ND
CPM-Sulfone	Heron	great blue heron	ND	ND	ND	ND	ND
	Eagle	bald eagle	ND	ND	ND	ND	ND
	Owl	great horned owl	ND	ND	ND	ND	ND
	Kestrel	American kestrel	ND	ND	ND	ND	ND
	Small bird	mourning dove	ND	ND	ND	ND	ND
	Shorebird	killdeer	ND	ND	ND	ND	ND
	Water bird	mallard	ND	ND	ND	ND	ND
	Sm. mammal	deer mouse	rat	Thake et al. '79	LOAEL	325	16.3
	Med. mammal	prairie dog	rat	Thake et al. '79	LOAEL	325	16.3
	T. plants	plants	ND	ND	ND	NA	ND
	Insects	insects	ND	ND	ND	ND	ND
	Earthworm	earthworm	ND	ND	ND	ND	ND
	Reptile	toad/snake	ND	ND	ND	ND	ND
DBCP	Heron	great blue heron	duck	Hudson et al. '84	LD50	NA	66.8 mg/kg
	Eagle	bald eagle	duck	Hudson et al. '84	LD50	NA	66.8 mg/kg
	Owl	great horned owl	duck	Hudson et al. '84	LD50	NA	66.8 mg/kg
	Kestrel	American kestrel	duck	Hudson et al. '84	LD50	NA	66.8 mg/kg
	Small bird	mourning dove	duck	Hudson et al. '84	LD50	NA	66.8 mg/kg
	Shorebird	killdeer	duck	Hudson et al. '84	LD50	NA	66.8 mg/kg
	Water bird	mallard	duck	Hudson et al. '84	LD50	NA	66.8 mg/kg
	Sm. mammal	deer mouse	rat	EPA 1987	NOEL	12.5	0.6
	Med. mammal	prairie dog	rat	EPA 1987	NOEL	12.5	0.6
	T. plants	plants	ND	ND	ND	NA	ND
	Insects	insects	ND	ND	ND	ND	ND
	Earthworm	earthworm	ND	ND	ND	ND	ND
	Reptile	toad/snake	ND	ND	ND	ND	ND
Dicyclo-penta-diene (DCPD)	Heron	great blue heron	BW quail	Aulerichetal'79	LOEL	4000	400
	Eagle	bald eagle	BW quail	Aulerichetal'79	LOEL	4000	400
	Owl	great horned owl	BW quail	Aulerichetal'79	LOEL	4000	1010.0
	Kestrel	American kestrel	BW quail	Aulerichetal'79	LOEL	4000	1010.0
	Small bird	mourning dove	BW quail	Aulerichetal'79	LOEL	4000	1010.0
	Shorebird	killdeer	BW quail	Aulerichetal'79	LOEL	4000	1010.0
	Water bird	mallard	duck	Aulerichetal'79	NOEL-r	320	32
	Sm. mammal	deer mouse	rat	EPA '91	NOEL	690	34
	Med. mammal	prairie dog	rat	EPA 91	NOEL	690	34
	T. plants	plants	grasses	ND	ND	NA	ND
	Insects	insects	ND	ND	ND	ND	ND

1. A modifying factor will be applied for calculation of the total UF.
2. The barn owl may be an unusually sensitive species; this will be taken into account when assigning UFs.
3. The total UF will reflect the fact this is a NOAEL.

Table C.2-14 Critical Dose Values (Pre-UF Toxicity Reference Values)

COC	TROPHIC BOX	SPECIES OF CONCERN	TEST SPECIES	CITATION	ENDPOINT/ STUDY TYPE	PPM IN DIET	DOSE (mg/kg-bw/day)
	Earthworm	earthworm	ND	ND	ND	ND	ND
	Reptile	toad/snake	ND	ND	ND	ND	ND
DDE/DDT	Heron	great blue heron	black duck	Longc&Stend'83	LOEL-r	0.6	0.06 Longcore and Stendell '83
	Eagle	bald eagle	kestrel	Wiemeyer et al.. '86, Lincer '75	LOAEL-r	2.8/3	0.4Wiemeyer et al.. '86/Linar '72
	Owl	great horned owl	owl	Mendenhall '83	FEL-r[3]	2.8	0.3 Mendenhall et al.. '83
	Kestrel	American kestrel	kestrel	Wiemeyer et al.. '86, Lincer '75	LOAEL-r	2.8/3	0.4Wiemeyer et al.. '86/Lincer '72
	Small bird	mourning dove	finch	Jefferies '71	LOAEL-r	4	0.8
	Shorebird	killdeer	kestrel	Wie'86/Lin'72,5	LOAEL-r	2.8/3	0.4 Wiemeyer et al.. '86/Lincer '75
	Water bird	mallard	black duck	Longc&Stend'83	LOEL-r	0.6	0.06 Longcore and Stendell '83
	Sm. mammal	deer mouse	rat	Green '69	FEL-r	7	0.35 Green '69
	Med. mammal	prairie dog	rat	Green '69	FEL-r	7	0.35 Green '69
	T. plants	plants	ND	ND	ND	NA	ND
	Insects	insects	ND	ND	ND	ND	ND
	Earthworm	earthworm	ND	ND	ND	ND	ND
	Reptile	toad/snake	ND	ND	ND	ND	ND
Endrin	Heron	great blue heron	duck	Roylanceetal'85	NOAEL-r[4]	0.5	0.05
	Eagle	bald eagle	owl	Fleming etal'82	LOAEL-r	0.75	0.1
	Owl	great horned owl	owl	Fleming etal'82	LOAEL-r	0.75	0.1
	Kestrel	American kestrel	owl	Fleming etal'82	LOAEL-r	0.75	0.1
	Small bird	mourning dove	owl	Fleming etal'82	LOAEL-r	0.75	0.1
	Shorebird	killdeer	owl	Fleming etal'82	LOAEL-r	0.75	0.1
	Water bird	mallard	duck	Roylanceetal'85	NOAEL-r[4]	0.5	0.05
	Sm. mammal	deer mouse	mouse	Noda et al. '72	LOAEL	ND	0.58
	Med. mammal	prairie dog	mouse	Noda et al. '72	LOAEL	ND	0.58
	T. plants	plants	grasses	ND	ND	ND	ND
	Insects	insects	ND	ND	ND	ND	ND
	Earthworm	earthworm	ND	ND	ND	ND	ND
	Reptile	toad/snake	ND	ND	ND	ND	ND
Mercury	Heron	great blue heron	mallard	Heinz '76	LOAEL-r	0.5	0.047
	Eagle	bald eagle	mallard	Heinz '76	LOAEL-r	0.5	0.047
	Owl	great horned owl	mallard	Heinz '76	LOAEL-r	0.5	0.047
	Kestrel	American kestrel	mallard	Heinz '76	LOAEL-r	0.5	0.047
	Small bird	mourning dove	mallard	Heinz '76	LOAEL-r	0.5	0.047
	Shorebird	killdeer	mallard	Heinz '76	LOAEL-r	0.5	0.047
	Water bird	mallard	mallard	Heinz '76	LOAEL-r	0.5	0.047
	Sm. mammal	deer mouse	rat	Soares et al.'73	NOAEL-l,g	3	0.17
	Med. mammal	prairie dog	rat	Soares et al.'73	NOAEL-l,g	3	0.17
	T. plants	plants	ND	ND	ND	ND	ND
	Insects	insects	ND	ND	ND	ND	ND
	Earthworm	earthworm	ND	ND	ND	ND	ND
	Reptile	toad/snake	ND	ND	ND	ND	ND
	Aquatic invert.	aquatic invert.	ND	EPA	AWQC-c	NA	0.012 µg/l
	Aquatic plants	aquatic plants	ND	EPA	AWQC-c	NA	0.012 µg/l
	Plankton	plankton	ND	EPA	AWQC-c	NA	0.012 µg/l

1. A modifying factor will be applied for calculation of the total UF.
2. The barn owl may be an unusually sensitive species; this will be taken into account when assigning UFs.
3. The total UF will reflect the fact this is a NOAEL.

COC	TROPHIC BOX	SPECIES OF CONCERN	TEST SPECIES	CITATION	ENDPOINT/ STUDY TYPE	PPM IN DIET	DOSE (mg/kg-bw/day)
Cadmium	Heron	great blue heron	mallard	Scheuhammer et al. and Templeton '90	NOAEL	NA	1.8
	Eagle	bald eagle	mallard	Scheuhammer et al. and Templeton "90	NOAEL	NA	1.8
	Owl	great horned owl	mallard	Scheuhammer et al. and Templeton "90	NOAEL	NA	1.8
	Kestrel	American kestrel	mallard	Scheuhammer et al. and Templeton "90	NOAEL	NA	1.8
	Small bird	mourning dove	mallard	Scheuhammer et al. and Templeton "90	NOAEL	NA	1.8
	Shorebird	killdeer	mallard	Scheuhammer et al. and Templeton "90	NOAEL	NA	1.8
	Water bird	mallard	mallard	Scheuhammer et al. and Templeton "90	NOAEL	NA	1.8
	Sm. mammal	deer mouse	rat	Groten et al.'91	NOEL	30	1.8
	Med. mammal	prairie dog	rat	Groten et al.'91	NOEL	30	1.8
	T. plants	plants	rhizobia/clover	Chaudri etal'92	growth limit	NA	7.1 µg/g soil
	Insects	insects	ND	ND	ND	ND	ND
	Earthworm	earthworm	ND	ND	ND	ND	ND
	Reptile	toad/snake	ND	ND	ND	ND	ND
Copper	Heron	great blue heron	chicken	Leach et al. '90	growth limit	480	48
	Eagle	bald eagle	chicken	Leach et al. '90	growth limit	480	48
	Owl	great horned owl	chicken	Leach et al. '90	growth limit	480	48
	Kestrel	American kestrel	chicken	Leach et al. '90	growth limit	480	48
	Small bird	mourning dove	chicken	Leach et al. '90	growth limit	480	48
	Shorebird	killdeer	chicken	Leach et al. '90	growth limit	480	48
	Water bird	mallard	chicken	Leach et al. '90	growth limit	480	48
	Sm. mammal	deer mouse	rat	ND	LD50	NA	300
	Med. mammal	prairie dog	rat	ND	LD50	NA	300
	T. plants	plants	Potomogetan	Greger, Kaut. '91	decr. biomass	NA	875 mg/kg sed.
	Insects	insects	ND	ND	ND	ND	ND
	Earthworm	earthworm	ND	ND	ND	ND	ND
	Reptile	toad/snake	ND	ND	ND	ND	ND

FEL	Frank-effect-level
LOAEL	Lowest observed adverse effect level
LOEL	Lowest observed effect level
NA	Not applicable
ND	No data
NOAEL	No observed adverse effect level
NOEL	No observed effect level
LC50	Lethal concentration affecting 50% of the test population
-r	Reproductive impairment
-g	Growth limiting
-b	Behavioral
-s	Survival
-p	Pathology
-l	Lethality

1. A modifying factor will be applied for calculation of the total UF.
2. The barn owl may be an unusually sensitive species; this will be taken into account when assigning UFs.
3. The total UF will reflect the fact this is a NOAEL.

Table C.2-15 Toxicity Reference Value (Post-UF)

Aldrin/Dieldrin	Critical Value (mg/kg bw/day)	Intertaxon (1)	Study Duration (Q2)	Study Endpoints (Q3)	Modifying Factor* (U)	T&E	Endpoint Relevance	Lab to Field	Co-Contam.	Unclear Endpoint	ID. Sensitive Species	Intraspecific Variability
American Kestrel	0.04	1	1	1	4			1		2		1
Bald eagle	0.05	5	1	1	6	2		1	0	2		1
Great horned owl	0.06	4	1	1	4			1	0	2		1
Great blue heron	0.4	5	1	3	1		-1	1				1
Shorebird	0.22	5	1	1	2			1				1
Water Bird	0.4	5	1	3	1		-1	1				1
Small Bird	0.28	5	1	1	2			1				1
Sm. Mammal	0.06	4	1	1	4			2		1		1
Med. Mammal	0.06	4	1	1	4			2		1		1
Reptile	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

Trophic Box	Total UF	Final TRV
American Kestrel	4	0.010
Bald eagle	30	0.002
Great horned owl	16	0.004
Great blue heron	15	0.027
Shorebird	10	0.022
Water Bird	15	0.027
Small Bird	10	0.028
Sm. Mammal	16	0.004
Lg. Mammal	16	0.004
Reptile	NA	NA

Final TRV	Critical value/total UF	TRV	Toxicity Reference Value
NA	not available	U	Sum of factors to right
Total of UF	1* Q2* Q3 *3	UF	Uncertainty Factor

*Note: If 0< = U<1, replaced with 1; If U<0, replaced with 0.5

DDT/DDE	Critical Value (mg/kg bw/day)	Intertaxon (1)	Study Duration (Q2)	Study Endpoints (Q3)	Modifying Factor* (U)	T&E	Endpoint Relevance	Lab to Field	Co-Contam.	Unclear Endpoint	ID. Sensitive Species	Intraspecific Variability
American Kestrel	0.4	1	1	5	2			1				1
Bald eagle	0.4	4	1	5	4	2		1				1
Great horned owl	0.3	4	1	10	1			1		-1		1
Great blue heron	0.06	5	1	3	1			1			-2	1
Shorebird	0.4	5	1	5	2			1				1
Water Bird	0.06	5	1	3	1			1			-2	1
Small Bird	0.8	5	5	5	2			1				1
Sm. Mammal	0.35	4	1	1	3			2				1
Lg. Mammal	0.35	4	1	1	3			2				1
Reptile	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

Trophic Box	Total UF	Final TRV
American Kestrel	10	0.040
Bald eagle	80	0.005
Great horned owl	40	0.008
Great blue heron	15	0.004
Shorebird	50	0.008
Water Bird	15	0.004
Small Bird	250	0.003
Sm. Mammal	12	0.029
Med. Mammal	12	0.029
Reptile	NA	NA

Final TRV	Critical value/total UF
NA	not available
Total of UF	1* Q2* Q3 *3

*Note: If 0< = U<1, replaced with 1; If U<0, replaced with 0.5

TRV	Toxicity Reference Value
U	Sum of factors to right
UF	Uncertainty Factor

Endrin	Critical Value (mg/kg bw/day)	Intertaxon (1)	Study Duration (Q2)	Study Endpoints (Q3)	Modifying Factor* (U)	T&E	Endpoint Relevance	Lab to Field	Co-Contam.	Unclear Endpoint	ID. Sensitive Species	Intraspecific Variability
American Kestrel	0.1	5	1	5	2			1				1
Bald eagle	0.1	5	1	5	4	2		1				1
Great horned owl	0.1	3	1	5	2			1				1
Great blue heron	0.05	5	1	1	4			1		2		1
Shorebird	0.1	5	1	5	2			1				1
Water Bird	0.05	5	1	1	4			1		2		1
Small Bird	0.1	5	1	5	2			1				1
Sm. Mammal	0.58	4	1	5	3			2				1
Med. Mammal	0.58	4	1	5	3			2				1
Reptile	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

Trophic Box	Total UF	Final TRV
American Kestrel	50	0.002
Bald eagle	100	0.001
Great horned owl	30	0.003
Great blue heron	20	0.003
Shorebird	50	0.002
Water Bird	20	0.003
Small Bird	50	0.002
Sm. Mammals	60	0.010
Med. Mammal	60	0.010
Reptile	NA	NA

Final TRV	Critical value/total UF
NA	not available
Total of UF	1* Q2* Q3 *3

*Note: If 0< = U<1, replaced with 1; If U<0, replaced with 0.5

TRV	Toxicity Reference Value
U	Sum of factors to right
UF	Uncertainty Factor

Table C.2-15 Toxicity Reference Value (Post-UF)

Mercury	Critical Value (mg/kg bw/day)	Intertaxon (1)	Study Duration (Q2)	Study Endpoints (Q3)	Modifying Factor* (U)	T&E	Endpoint Relevance	Lab to Field	Co-Contam.	Unclear Endpoint	ID. Sensitive Species	Intraspecific Variability
American Kestrel	0.047	5	1	5	1			1	-1			1
Bald eagle	0.047	5	1	5	3	2		1	-1			1
Great horned owl	0.047	5	1	5	1			1	-1			1
Great blue heron	0.047	5	1	5	2			1				1
Shorebird	0.047	5	1	5	2			1				1
Water Bird	0.047	5	1	5	2			1				1
Small Bird	0.047	5	1	5	1			1	-1			1
Sm. Mammal	0.17	4	5	3	2			2	-1			1
Med. Mammal	0.17	4	5	3	2			2	-1			1
Reptile	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

Trophic Box	Total UF	Final TRV
American Kestrel	25	0.002
Bald eagle	75	0.001
Great horned owl	25	0.002
Great blue heron	50	0.001
Shorebird	50	0.001
Water Bird	50	0.001
Small Bird	25	0.002
Sm. Mammal	120	0.001
Med. Mammal	120	0.001
Reptile	NA	NA

Final TRV	Critical value/total UF
NA	not available
Total of UF	1* Q2* Q3 *3

*Note: If 0< = U<1, replaced with 1; If U<0, replaced with 0.5

TRV	Toxicity Reference Value
U	Sum of factors to right
UF	Uncertainty Factor

Arsenic	Critical Value (mg/kg bw/day)	Intertaxon (1)	Study Duration (Q2)	Study Endpoints (Q3)	Modifying Factor* (U)	T&E	Endpoint Relevance	Lab to Field	Co-Contam.	Unclear Endpoint	ID. Sensitive Species	Intraspecific Variability
American Kestrel	18.9	5	1	5	2			1				1
Bald eagle	18.9	5	1	5	4	2		1				1
Great horned owl	18.9	5	1	5	2			1				1
Great blue heron	18.9	5	1	5	2			1				1
Shorebird	18.9	5	1	5	2			1				1
Water Bird	18.9	5	1	5	2			1				1
Small Bird	18.9	5	1	5	2			1				1
Sm. Mammal	1.5	4	1	5	2		-1	2				1
Med. Mammal	1.5	4	1	5	2		-1	2				1
T. Plants	19				10							
Reptile	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

Trophic Box	Total UF	Final TRV
American Kestrel	50	0.378
Bald eagle	100	0.189
Great horned owl	50	0.378
Great blue heron	50	0.378
Shorebird	50	0.378
Water Bird	50	0.378
Small Bird	50	0.378
Sm. Mammal	40	0.038
Lg. Mammal	40	0.038
T. plants	10	1.900
Reptile	NA	NA

Final TRV	Critical value/total UF
NA	not available
Total of UF	1* Q2* Q3 *3

*Note: If 0< = U<1, replaced with 1; If U<0, replaced with 0.5

TRV	Toxicity Reference Value
U	Sum of factors to right
UF	Uncertainty Factor

Copper	Critical Value (mg/kg bw/day)	Intertaxon (1)	Study Duration (Q2)	Study Endpoints (Q3)	Modifying Factor* (U)	T&E	Endpoint Relevance	Lab to Field	Co-Contam.	Unclear Endpoint	ID. Sensitive Species	Intraspecific Variability
American Kestrel	48	5	1	5	2		-1	2				1
Bald eagle	48	5	1	5	4	2	-1	2				1
Great horned owl	48	5	1	5	2		-1	2				1
Great blue heron	48	5	1	5	2		-1	2				1
Shorebird	48	5	1	5	2		-1	2				1
Water Bird	48	5	1	5	2		-1	2				1
Small Bird	48	5	1	5	2		-1	2				1
Sm. Mammal	300	4	20	15	3			2				1
Lg. Mammal	300	4	20	15	3			2				1
T. plants	875											
Reptile	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

Trophic Box	Total UF	Final TRV	
American Kestrel	50	0.960	
Bald eagle	100	0.480	
Great horned owl	50	0.960	
Great blue heron	50	0.960	
Shorebird	50	0.960	
Water Bird	50	0.960	
Small Bird	50	0.960	
Sm. Mammal	3600	0.750	UF capped at 400
Med. Mammal	3600	0.750	UF capped at 400
Reptile	NA	NA	

Final TRV	Critical value/total UF
NA	not available
Total of UF	1* Q2* Q3 *3

*Note: If 0< = U<1, replaced with 1; If U<0, replaced with 0.5

TRV	Toxicity Reference Value
U	Sum of factors to right
UF	Uncertainty Factor

Cadmium	Critical Value (mg/kg bw/day)	Intertaxon (1)	Study Duration (Q2)	Study Endpoints (Q3)	Modifying Factor* (U)	T&E	Endpoint Relevance	Lab to Field	Co-Contam.	Unclear Endpoint	ID. Sensitive Species	Intraspecific Variability
American Kestrel	1.8	5	1	1	1.5			1				0.5
Bald eagle	1.8	5	1	1	3.5	2		1				0.5
Great horned owl	1.8	5	1	1	1.5			1				0.5
Great blue heron	1.8	5	1	1	1.5			1				0.5
Shorebird	1.8	5	1	1	1.5			1				0.5
Water Bird	1.8	5	1	1	1.5			1				0.5
Small Bird	1.8	5	1	1	1.5			1				0.5
Sm. Mammal	1.8	4	5	1	2		-1	2				1
Med. Mammal	1.8	4	5	1	2		-1	2				1
T. Plants	7.1											
Reptile	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

Trophic Box	Total UF	Final TRV
American Kestrel	7.5	0.240
Bald eagle	17.5	0.103
Great horned owl	7.5	0.240
Great blue heron	7.5	0.240
Shorebird	7.5	0.240
Water Bird	7.5	0.240
Small Bird	7.5	0.240
Sm. Mammals	40	0.045
Med. Mammal	40	0.045
Reptile	NA	NA

Final TRV	Critical value/total UF
NA	not available
Total of UF	1* Q2* Q3 *3

*Note: If 0< = U<1, replaced with 1; If U<0, replaced with 0.5

TRV	Toxicity Reference Value
U	Sum of factors to right
UF	Uncertainty Factor

Table C.2-15 Toxicity Reference Value (Post - UF)

DCPD	Critical Value (mg/kg bw/day)	Intertaxon (1)	Study Duration (Q2)	Study Endpoints (Q3)	Modifying Factor* (U)	T&E	Endpoint Relevance	Lab to Field	Co-Contam.	Unclear Endpoint	ID. Sensitive Species	Intraspecific Variability
American Kestrel	400	5	1	3	3			2				1
Bald eagle	400	5	1	3	5	2		2				1
Great horned owl	400	5	1	3	3			2				1
Great blue heron	400	5	1	3	3			2				1
Shorebird	400	5	1	3	3			2				1
Water Bird	32	5	1	1	2			1				1
Small Bird	400	5	1	3	3			2				1
Sm. Mammal	34	4	1	1	3			2				1
Med. Mammal	34	4	1	1	3			2				1
T. Plants	NA											
Reptile	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

Trophic Box	Total UF	Final TRV
American Kestrel	45	8.889
Bald eagle	75	5.333
Great horned owl	45	8.889
Great blue heron	45	8.889
Shorebird	45	8.889
Water Bird	10	3.200
Small Bird	45	8.889
Sm. Mammal	12	2.833
Med. Mammal	12	2.833
Reptile	NA	NA

Final TRV	Critical value/total UF
NA	not available
Total of UF	1* Q2* Q3 *3

*Note: If 0< = U<1, replaced with 1; If U<0, replaced with 0.5

TRV	Toxicity Reference Value
U	Sum of factors to right
UF	Uncertainty Factor

Table C.2-15 Toxicity Reference Value (Post - UF)

Chlordane	Critical Value (mg/kg bw/day)	Intertaxon (1)	Study Duration (Q2)	Study Endpoints (Q3)	Modifying Factor* (U)	T&E	Endpoint Relevance	Lab to Field	Co-Contam.	Unclear Endpoint	ID. Sensitive Species	Intraspecific Variability
American Kestrel	14.1	5	20	15	3			2				1
Bald eagle	14.1	5	20	15	5	2		2				1
Great horned owl	14.1	5	20	15	3			2				1
Great blue heron	14.1	5	20	15	3			2				1
Shorebird	14.1	5	20	15	3			2				1
Water Bird	1250	5	20	15	2			1				1
Small Bird	14.1	5	20	15	3			2				1
Sm. Mammal	1.2	4	1	1	3			2				1
Med. Mammal	1.2	4	1	1	3			2				1
T. Plants	NA											
Reptile	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

Trophic Box	Total UF	Final TRV	
American Kestrel	4500	0.035	UF capped at 400
Bald eagle	7500	0.035	UF capped at 400
Great horned owl	4500	0.035	UF capped at 400
Great blue heron	4500	0.035	UF capped at 400
Shorebird	4500	0.035	UF capped at 400
Water Bird	3000	3.125	UF capped at 400
Small Bird	4500	0.035	UF capped at 400
Sm. Mammal	12	0.100	
Med. Mammal	12	0.100	
Reptile	NA	NA	

Final TRV	Critical value/total UF	TRV	Toxicity Reference Value
NA	not available	U	Sum of factors to right
Total of UF	1* Q2* Q3 *3	UF	Uncertainty Factor

*Note: If 0< = U<1, replaced with 1; If U<0, replaced with 0.5

Table C.2-15 Toxicity Reference Value (Post - UF)

CPMS	Critical Value (mg/kg bw/day)	Intertaxon (1)	Study Duration (Q2)	Study Endpoints (Q3)	Modifying Factor* (U)	T&E	Endpoint Relevance	Lab to Field	Co-Contam.	Unclear Endpoint	ID. Sensitive Species	Intraspecific Variability
American Kestrel	ND	NA	NA	NA	NA							
Bald eagle	ND	NA	NA	NA	NA							
Great horned owl	ND	NA	NA	NA	NA							
Great blue heron	ND	NA	NA	NA	NA							
Shorebird	ND	NA	NA	NA	NA							
Water Bird	ND	NA	NA	NA	NA							
Small Bird	ND	NA	NA	NA	NA							
Sm. Mammal	14.1	4	1	5	3		-1	2		1		1
Med. Mammal	14.1	4	1	5	3		-1	2		1		1
T. Plants	0.7											
Reptile	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

Trophic Box	Total UF	Final TRV
American Kestrel	NA	ND
Bald eagle	NA	ND
Great horned owl	NA	ND
Great blue heron	NA	ND
Shorebird	NA	ND
Water Bird	NA	ND
Small Bird	NA	ND
Sm. Mammal	60	0.235
Med. Mammal	60	0.235
Reptile	NA	NA

Final TRV	Critical value/total UF
NA	not available
Total of UF	1* Q2* Q3 *3

*Note: If 0< = U<1, replaced with 1; If U<0, replaced with 0.5

TRV	Toxicity Reference Value
U	Sum of factors to right
UF	Uncertainty Factor

Table C.2-15 Toxicity Reference Value (Post - UF)

CPMSO2	Critical Value (mg/kg bw/day)	Intertaxon (I)	Study Duration (Q2)	Study Endpoints (Q3)	Modifying Factor* (U)	T&E	Endpoint Relevance	Lab to Field	Co-Contam.	Unclear Endpoint	ID. Sensitive Species	Intraspecific Variability
American Kestrel	ND	NA	NA	NA	NA							
Bald eagle	ND	NA	NA	NA	NA							
Great horned owl	ND	NA	NA	NA	NA							
Great blue heron	ND	NA	NA	NA	NA							
Shorebird	ND	NA	NA	NA	NA							
Water Bird	ND	NA	NA	NA	NA							
Small Bird	ND	NA	NA	NA	NA							
Sm. Mammal	16.3	4	1	5	3		-1	2		1		1
Med. Mammal	16.3	4	1	5	3		-1	2		1		1
T. Plants	ND	NA	NA	NA	NA							
Reptile	ND	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

Trophic Box	Total UF	Final TRV
American Kestrel	NA	ND
Bald eagle	NA	ND
Great horned owl	NA	ND
Great blue heron	NA	ND
Shorebird	NA	ND
Water Bird	NA	ND
Small Bird	NA	ND
Sm. Mammal	60	0.272
Med. Mammal	60	0.272
Reptile	NA	NA

Final TRV	Critical value/total UF
NA	not available
Total of UF	1* Q2* Q3 *3

*Note: If 0< = U<1, replaced with 1; If U<0, replaced with 0.5

TRV	Toxicity Reference Value
U	Sum of factors to right
UF	Uncertainty Factor

Table C.2-15 Toxicity Reference Value (Post - UF)

DBCP	Critical Value (mg/kg bw/day)	Intertaxon (1)	Study Duration (Q2)	Study Endpoints (Q3)	Modifying Factor* (U)	T&E	Endpoint Relevance	Lab to Field	Co-Contam.	Unclear Endpoint	ID. Sensitive Species	Intraspecific Variability
American Kestrel	66.8	5	20	15	2			1				1
Bald eagle	66.8	5	20	15	4	2		1				1
Great horned owl	66.8	5	20	15	2			1				1
Great blue heron	66.8	5	20	15	2			1				1
Shorebird	66.8	5	20	15	2			1				1
Water Bird	66.8	5	20	15	2			1				1
Small Bird	66.8	5	20	15	2			1				1
Sm. Mammal	0.6	4	1	1	3			2				1
Med. Mammal	0.6	4	1	1	3			2				1
T. Plants	ND	NA	NA	NA	NA							
Reptile	ND	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

Trophic Box	Total UF	Final TRV	
American Kestrel	3000	0.167	UF capped at 400
Bald eagle	6000	0.167	UF capped at 400
Great horned owl	3000	0.167	UF capped at 400
Great blue heron	3000	0.167	UF capped at 400
Shorebird	3000	0.167	UF capped at 400
Water Bird	3000	0.167	UF capped at 400
Small Bird	3000	0.167	UF capped at 400
Sm. Mammal	12	0.050	
Med. Mammal	12	0.050	
Reptile	NA	NA	

Final TRV	Critical value/total UF	TRV	Toxicity Reference Value
NA	not available	U	Sum of factors to right
Total of UF	1* Q2* Q3 *3	UF	Uncertainty Factor

*Note: If 0< = U<1, replaced with 1; If U<0, replaced with 0.5