# 4.0 ECOLOGICAL RISK CHARACTERIZATION

As discussed in Section 2.1, the ERC builds upon a number of other investigations performed at RMA, particularly the Biota RI and Biota CMP. The objectives of the ERC performed for RMA were the following: (1) characterize the magnitude and spatial extent of potential risks to the diverse aquatic and terrestrial biota from the existing contamination on post, (2) determine whether the projected potential risks are consistent with available data on ecological conditions at RMA, (3) assess the uncertainty associated with these potential risks, and (4) evaluate the spatial relationship of existing contamination and the estimated risks in order to establish a realistic basis for future risk management decisions.

The discussion that follows summarizes the ecological risk assessment conducted to meet the above-listed objectives. Section 4.1 presents the conceptual framework within which the risk evaluations were developed and discusses the computational methodology and features of the assessment. Sections 4.2 and 4.3 address the contaminants evaluated in the ERC and methods used to estimate their exposure concentrations, respectively. Section 4.4 contains a brief description of the model used to derive risk estimates based on toxicological endpoints for target receptors, Section 4.5 summarizes the results of the risk computations, and Section 4.6 presents the summary and conclusions.

## 4.1 CONCEPTUAL FRAMEWORK

The ecological risk evaluations consisted of a number of sequential analytical steps as shown in Figure 4.1-1. These included the following: (1) identifying COCs and ecological assessment and measurement endpoints; (2) calculating contaminant concentrations in exposure media; (3) deriving toxicological threshold values; (4) estimating population mean tissue concentrations in, and doses to target biota receptors; and (5) comparing the predicted risks with current ecological measurement endpoints to assess the implication of contamination to populations and communities of the RMA ecosystem.

There are both known and unknown sources of uncertainty in the ERC. Section 5.0 (particularly Sections 5.2 and 5.3) and Appendix E (Appendix Sections E.9 through E.12) specifically identify and discuss the sources of uncertainty.

### 4.1.1 Toxicological Threshold Values and Ecological Endpoints

The ecological risk evaluations performed for RMA specifically address potential risks to target biota receptors. The two assumptions that are associated with the ERC are the following: (1) risks posed to target receptors in a trophic (i.e., feeding) level within representative food chains of the RMA ecosystem are assumed to create risk to the integrity of the ecosystem represented by healthy populations and communities, and (2) the on-post remediation intended to alleviate risks to target receptors is assumed to be sufficient to also alleviate risks at the ecosystem level.

EPA guidance (1989d) is explicit regarding ecological endpoint definitions and the distinction between assessment endpoints that pertain to environmental values warranting protection and measurement endpoints that are measurable characteristics of the assessment endpoints. In the ERC, the assessment endpoints include structural diversity of the ecosystem, population robustness, and, for the bald eagle, health of the individual. Significant negative effects on the assessment endpoints, as reflected in observations of the measurement endpoints, would indicate a need for remediation (EPA 1989d).

Ecological measurement endpoints were selected to reveal adverse effects at the population and community levels of ecosystem organization and provide perspective on the potential risk indicated by HIs and HQs. The selection process required the simultaneous review of available ecological studies on RMA biota and identification of pertinent measurement endpoints. The ecological measurement endpoints, discussed briefly below and more fully in Appendix Section C.5, were those found in the existing studies on RMA biota and were appropriate for the COCs, their toxic effects on individual organisms, and the consequences of these effects at higher levels of ecosystem organization.

#### 4.1.1.1 Toxicological Threshold Values

The two toxicological threshold values, maximum allowable tissue concentrations (MATCs) and toxicity reference values (TRVs), were derived from the literature as discussed below and in Appendix Sections C.2.4 and C.2.6. The MATCs and TRVs were used to estimate potential risk to target receptors. The MATCs represent maximum whole-body concentrations of bioaccumulative chemicals that are unlikely to cause harmful effects to specific receptors. The MATCs, expressed as the weight of contaminant per unit of body weight (mg/kg/bw), were derived from literature data on tissue concentrations associated with the presence or absence of observed toxicological effects in biological test species. Uncertainty factors (UFs) were applied to the literature-based tissue values for the purpose of ensuring adequate protection of the biota at RMA.

The TRVs represent estimates of a daily dose (mg/kg-bw/day) that are likely to be without an appreciable risk of harmful effects to target receptors. The TRVs computed for the IEA/RC follow an approach that is different from that described in the Off-Post Operable Unit EA/FS for RMA (HLA 1993); however, both RMA approaches are similar to the methodology used by EPA to compute reference doses (RfDs) for assessing risks to human health. Like the MATCs and the human health RfDs, the TRV estimates may have UFs spanning an order of magnitude or more to account for uncertainty associated with extrapolating toxic effects seen in the laboratory to target receptors at RMA.

The toxicological threshold values, MATCs and TRVs, are compared to the site-specific exposure measurements, population mean contaminant tissue concentrations and doses, to estimate potential risk to target biota populations. The toxicological threshold values are intended to be protective of biota populations and individual bald eagles at RMA.

#### 4.1.1.2 Ecological Endpoints

To provide perspective on the presence of potential risk indicated by the MATCs and TRVs, a number of ecological measurement endpoints were evaluated to qualitatively determine whether adverse effects are occurring at the various structural levels of ecosystem organization. These

ecological measurement endpoints were selected based on the following criteria: (1) they address potential adverse effects of RMA contaminants, (2) they are sensitive enough to detect population effects that are outside the normal range of variation, and (3) they are based on RMA ecological studies that examined these measurement endpoints.

Ecological measurement endpoints were selected at the community, population, and individual levels of organization. The community-level measurement endpoints considered were species richness and trophic diversity; these provide information on the assessment endpoint of structural diversity of the ecosystem. Population-level measurement endpoints were relative abundance, reproductive success, and morbidity; these provide information on the assessment endpoint of population robustness. Selected biomarkers (i.e., acetylcholinesterase inhibition and eggshell thinning) were examined at the individual level, but evaluated as measurement endpoints for population effects. Endpoints at the individual level are appropriate for evaluating adverse effects on individuals of threatened or endangered species (e.g., bald eagle), which by definition have populations reduced to the level where individuals are important. EPA concluded that a diversity of endpoints, including tissue concentrations and measurements at a number of ecological levels "provides a holistic examination of the ecosystem, lending greater confidence in risk estimates" (1993, p. 8-17). In Appendix Section C.5, the studies that provide data for evaluation of the ecological endpoints are presented in the context of any limitations or uncertainties they may have. These uncertainties are discussed further in Appendix Section E.12. The comparison of the findings with the results of the risk characterization is presented in Section 4.6.

### 4.1.2 Hazard Quotients and Hazard Indices

Since the ultimate objective of these ecological risk evaluations was to characterize risks to target biota, a quantitative framework for risk estimation was established. The total cumulative risk for all contaminants and exposure routes ($Risk_{total}$) can be expressed as:

$$Risk_{total} = Hazard\ Index \qquad (1)$$

where the HI is defined as:

$$HI = \sum_i \sum_j \frac{Estimated\ Exposure_{i,j}}{Toxicological\ Threshold\ Value_{i,j}} \quad (2)$$

The "i" and "j" refer to contaminant i and exposure route j, respectively. The exposure routes explicitly considered in the ERC are ingestion through terrestrial and aquatic food chains of the RMA food webs. The ratio of the estimated exposure to the toxicological threshold value for a contaminant i is defined as the HQ and is expressed as:

$$HQ_i = \sum_j \frac{Estimated\ Exposure_{i,j}}{Toxicological\ Threshold\ Value_{i,j}} \quad (3)$$

The estimated exposure can be represented as (1) the contaminant concentration in biota tissue, or (2) the estimated contaminant intake rate (dose) at the point of exposure. Similarly, the toxicological threshold value can be represented by the MATC or the TRV.

Therefore, the HQ for contaminant *i* can be expressed as:

$$HQ_i = \frac{Tissue\ Concentration_i}{MATC_i} \quad (4)$$

or

$$HQ_i = \frac{Dose_i}{TRV_i} \quad (5)$$

It can be seen by comparing equations (2) and (3) that the HI is the sum of the contaminant-specific HQs. A more detailed description of tissue concentration and dose estimation methods for the terrestrial and aquatic food chains of the RMA food webs is provided in Section 4.3, and more detailed risk equations are presented and discussed in Section 4.4.

The assumption that the hazards from the different COCs are additive is uncertain. For example, risks may be less than implied by the additivity assumption if the COCs do not induce the same type of effects or do not act by the same mechanism; or more than implied by the additivity assumption if the COCs induce synergistic effects. EPA risk assessment guidance for human health calls for additivity of hazard quotients, but also states that while '...application of the hazard index equation to a number of compounds that are not expected to induce the same type of effects or that do not act by the same mechanism, although appropriate as a screening-level approach, could overestimate the potential for effects.' The guidance goes on to state that '(i)f the HI is greater than unity as a consequence of summing over several hazard quotients of similar value, it would be appropriate to segregate the compounds by effect and by mechanism of action and to derive separate hazard indices for each group' (EPA, RAGS, 1989, Section 8, page 8-15). The segregation of COCs by effect or mechanism of action to derive separate HIs was not done for the ERC because of limited toxicological data on the COCs for the species or trophic boxes of concern or appropriate surrogate animals. Therefore, the Army considered it prudent to sum the individual HQ values and derive species or trophic box HIs, albeit this process probably resulted in an overestimation of potential risks.

### 4.1.3 Special Considerations in Estimating Ecological Risks

In performing this ecological risk assessment, a number of unique concepts and methods were used to enhance the interpretive quality of the assessment and provide useful input to future risk management decisions regarding the eventual cleanup of RMA. Since these concepts and methods focused the subsequent analyses, they are summarized in the sections that follow.

#### 4.1.3.1 Spatial Averaging of Exposure

The large areal extent of biological habitats at RMA necessitated a careful analysis of potential exposure of target receptors to the observed nonuniform contamination of the soils observed across RMA. The spatial distribution of contamination in the RMA lakes was deemed less critical to the ERC because (1) the lakes are relatively small and their waters well mixed; (2) lake biota, the base of the aquatic food chains, attenuate the variability in sediment contamination; and (3) those target biota receptors evaluated for risk that use the aquatic food chains (i.e., water bird,

shorebird, great blue heron, and bald eagle) further average any variability in the sediment contaminant concentrations.

Actual exposure conditions for biota populations are difficult to measure because such measurement requires detailed information about individual organisms' behaviors as well as similar information about prey and the average soil concentration of bioavailable contaminant (the "exposure soil concentration") to which the organisms are directly and indirectly exposed. Exposure soil concentrations were estimated by estimated spatially averaged soil concentrations within "exposure areas," i.e., well defined areas selected to correlate with the foraging range of the target biota receptor for which the risks were estimated. Exposure was assumed to be uniform throughout the exposure area. Through the use of exposure areas, exposures from hot spots and relatively clean areas were taken into account by quantifying contaminant biomagnification throughout the various trophic levels of the food webs selected to represent the RMA ecosystem.

Spatial averaging of potential exposure eliminates the unrealistic assumption that individual biota living in the vicinity of isolated contaminant hot spots are continuously exposed to the high level of contamination found at the hot spot. Failure to take into account spatially distributed exposure erroneously localizes the exposure of biota over their lifetime, ignoring the fundamental ecological principles of mobility and biodispersion. Appendix Sections C.1.4 and E.12.4.2 present a detailed discussion of the quantitative procedures utilized to estimate average exposure. This concept is integral to the determination of the areal distributions of RMA-wide risks.

Estimated exposure area soil concentrations were used in a Geographical Information System (GIS) to model exposure and provide a series of maps that display the areas of potential risks within RMA boundaries. These risk maps facilitate an overview of the target biota affected, the contaminants driving the risks, and the spatial extent to which potential exposure and risks occur. In addition, the use of a GIS enables identification of contaminated areas driving the risks and enhances the environmental manager's ability to make informed decisions regarding risk management, engineering feasibility, and eventual cleanup of RMA.

#### 4.1.3.2 Probabilistic Methodology

In recognition of the variability and uncertainty in literature and field data available for quantifying the parameters used to estimate contaminant biomagnification, probabilistic methodologies were used to develop biomagnification factors (BMFs). This approach improves the risk characterization by clarifying the risk implications of uncertainty, or lack thereof, about ecological parameters and the biomagnification process. This information was then used to develop probabilistic estimates of the BMFs that reflect both data variability and uncertainty. Section 4.5 presents a summary of approaches used to compute BMFs and Appendix Sections C.1 and C.2 provide a detailed description of all methods and procedures. Probabilistic methods were also used to replace measured soil concentration data that fell below certified reporting limits (BCRLs) with numerical "BCRL replacement values" for the estimation of exposure area soil concentrations (ESCs).

## 4.2 CONTAMINANTS EVALUATED FOR POTENTIAL RISK

The contaminants at RMA that pose a potential risk to biota have been extensively discussed in reports from prior RMA programs (see Appendix A). Those that were consistently identified as potentially threatening to biota were also identified as COCs by the ERC. The ERC COCs, selected on the basis of toxicity, persistence in the environment, and production volume and areal extent of contamination, were aldrin, dieldrin, DDT, DDE, endrin, mercury, arsenic, cadmium, copper, chlordane, CPMS, $CPMSO_2$, DBCP, and DCPD. For 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. Throughout the IEA/RC, these combined contaminants are referred to as aldrin/dieldrin and DDT/DDE. This notation does not represent contaminant ratios, but rather the sum of the related contaminants. Of the 14 contaminants evaluated, 6 (i.e., aldrin/dieldrin, endrin, DDT/DDE, and mercury) are known to bioaccumulate substantively and are collectively referred to as the bioaccumulative COCs. Seven of the remaining eight contaminants (i.e., arsenic, cadmium, CPMS, $CPMSO_2$, copper, DBCP, and DCPD) are collectively referred to as nonbioaccumulative COCs. Chlordane, even though it bioaccumulates, is treated like the nonbioaccumulative COCs because biota tissue samples were not analyzed for chlordane; thus, a BMF could not be calculated. This resulted in the potential

underestimation of risk from chlordane. Appendix Section C.1.2 provides further detail on the selection of COCs evaluated in the ERC.

## 4.3 EXPOSURE CONCENTRATION MODELING

The ERC terrestrial food webs assume that soil is the initial source of contaminant exposure for RMA biota. The aquatic food chains assume that aquatic biota are the initial source of contamination to their avian predators. The exposure route in the food-web model is ingestion of contaminated food, soil, sediment, and water. Ingestion of contaminated food is the predominant route of biota exposure at RMA. Exposure to contaminant vapors or particulates via inhalation or dermal absorption and exposure to contaminants in the groundwater are considered insignificant routes for biota relative to others identified and are, therefore, excluded from exposure modeling. Instead, risk estimates use RMA-specific contaminant concentrations in tissue samples. These tissue concentrations reflect actual exposure through ingestion and all other exposure routes. These tissue data also reflect only the bioaccumulative fraction of the total mercury measured in RMA soil, sediment, and water.

### 4.3.1 Terrestrial

As discussed in Section 4.1, exposure concentrations used to estimate risks were spatially averaged to incorporate natural mobility and biodispersion appropriate to each target receptor evaluated. Estimated exposure area soil concentrations (<ESCs>) were used to approximate the exposure concentrations because of uncertainties associated with the interpolated soil concentrations between locations where soil samples occurred and the actual area used by the sampled organism and its prey.

Exposure area dimensions were based on literature information, field observations, and best professional judgment. Exposure area size varied markedly among target receptors and increased as the trophic level increased. For example, the exposure area for insects was estimated to be 0.004 hectare (ha) (0.01 acre), while the exposure area for the bald eagle was more than 2,800 ha (6,919 acres). The large variation in exposure area implies that biota having very small foraging areas will have tissue contaminant concentrations that are more directly proportional to measured

soil contaminant concentrations in the area where the sampled biota were found. Therefore, the tissue concentrations may be highly variable in both relative magnitude and location, reflecting source magnitude and location as well as contaminant migration. Conversely, biota having large foraging areas will have tissue concentrations that reflect an average of the contaminant concentrations of their more localized prey, which may be highly variable. Thus, the severity of the calculated risk will be increased if exposure from hot spots is significant when compared to exposure from relatively uncontaminated areas. In other words, the intensity of hot spot contamination and its spatial patchiness will govern the magnitude of risk and the size of the risk area for these more mobile species.

For soil, average exposure area concentrations were generally estimated as the arithmetic mean (i.e., average) contaminant concentration in the 0- to 1-ft soil depth interval within a receptor's exposure area. To obtain these concentrations, the data from soil borings and surficial soil samples were first used to estimate replacement values for samples that were measured below the CRL (i.e., BCRL). Once these values are estimated, they were used to interpolate estimated soil contaminant concentrations onto an RMA-wide grid with 100-ft intervals. The interpolated concentrations were then averaged within trophic-box-specific exposure areas (see Appendix Section C.1.4.1). For prairie dogs, the three-dimensional average of soil concentrations within their exposure area incorporated contaminants measured down to a depth of 20 ft below the ground surface to take exposure from burrowing into account (see Appendix Section C.1).

### 4.3.2 Aquatic

Aquatic risks were estimated for the bald eagle, great blue heron, shorebird, and water bird. Aquatic risks were not estimated for the strictly aquatic biota because information required to derive toxicological threshold values was not available for those trophic boxes. Exposure concentrations for aquatic biota and their predators are calculated from biota tissue concentration data from RMA lakes, rather than from average sediment and water concentration estimates, for three reasons. First, it was possible to obtain adequately representative biota samples from the lakes, which are smaller and more homogeneous than the terrestrial ecosystem. Second, because it was possible tc adequately sample the lakes, the tissue concentration field data were more

reliable used than the tissue concentration predictions from lake surface sediment field data used as input in a "media-based" biomagnification model. Third, uncertainties in estimating pore water contaminant concentrations, and subsequently the surface water concentrations, from sediment concentrations are high because of the high value for the mass fraction of organic carbon in the RMA lake sediments.

The computational framework and analytical steps of the aquatic modeling and risk characterization process is summarized in Figure 4.3-1. Biota tissue concentration data from strictly aquatic trophic boxes in RMA lakes were used in a "tissue-based" biomagnification model to estimate risk to their predators (i.e., great blue heron and bald eagle) from exposure through aquatic food chains. A sample average tissue concentration was calculated for each aquatic target receptor in each lake where data were available. The sample averages were treated as deterministic averages in the risk characterization. Methods for estimating missing data are described on a case-by-case basis in Appendix Section C.1.4.2.

The estimation of the contribution from aquatic food chains to the contaminant body burden ($TC_{aquatic}$) in heron and bald eagle was computed using the food-web models for these species:

$$TC_{aquatic} = BAF * \Sigma(FR * preyTC) \qquad (6)$$

An average tissue concentration estimate was computed for the bald eagle and heron for each of the RMA lakes using the average tissue concentration estimates described above in equation (6) for the aquatic prey. The estimates for all the lakes except East Upper Derby Lakes were then combined as described for the water bird trophic box in Appendix Section C.1, using normalized lake surface areas as weighting factors to characterize the levels of predation at the different lakes, to form an RMA-wide estimate of average tissue concentration from exposure through the aquatic food chains. Similar calculations were used to estimate eagle and heron doses from aquatic food chains:

$$dose_{aquatic} = R * \Sigma(FR * preyTC) \quad (7)$$

Equations (6) and (7) are used to estimate the portion of total risk to the bald eagle and heron from exposure to aquatic food chains, as is described below in Section 4.4.2. Estimates of contributions from aquatic food chains to the contaminant tissue concentrations in, and doses to, heron and the bald eagle are presented in Appendix Figures D.1-16 and D.1-17.

## 4.4 RISK MODELING

Representative food webs for RMA were developed from the information on RMA ecosystems using selection criteria presented in the Biota RI (ESE 1989) and Biota CMP (RLSA 1992) reports. The criteria considered include species abundance, exposure range, distribution, and whether the species is threatened or endangered, is socially or economically important, or is an important component of regional food webs. Most of the species that had been sampled under the Biota RI and Biota CMP were designated as "target biota receptors." The target biota receptors were combined in groups of similar species within the same trophic level (i.e., trophic boxes) representing the trophic diversity on RMA. For example, the small mammal trophic box was represented by deer mouse and thirteen-lined ground squirrel, and the small fish trophic box was represented by bluegill, channel catfish, and bullhead. The generic predator food web (Figure 4.4-1) illustrates the various interrelationships of these trophic boxes. Five specific food webs were constructed to represent different food habits and exposure routes in terrestrial and aquatic food chains at RMA, i.e., one for each of four top predatory birds (bald eagle, great horned owl, American kestrel, and great blue heron), and one for a group of birds significantly exposed to lake sediments (shorebirds). For the trophic boxes in these food webs, information collected during the sampling programs was used to quantify site-specific dietary fractions on the basis of biomass of prey consumed by some of the species at RMA (see Appendix Section C.2.3). The five food webs modeled in the ERC are illustrated in Appendix Figures C.1-2 through C.1-6. The relationships between trophic boxes in these food webs are described mathematically in Sections 4.4.1 and 4.4.2 for the terrestrial and aquatic food chains, respectively. Additional details of the risk calculation method are provided in Appendix Section C.1.6.

In general, BMFs should not be exported from one model and database to another because the BMF is a function of the exposure concentration statistic used to estimate the true exposure soil concentration (i.e., in this case <ESC>, the estimated exposure area soil concentration; a detailed definition of <ESC> is provided in Appendix Section C.1.4.1). The BMFs derived for use in the ERC depend on the manner in which the samples used in the models were collected and measured, and on the manner in which the <ESC> used to derive risk estimates was defined. As such, the numerical BMF values are only valid for use in the IEA/RC, although the derivation methodologies may be applicable to other sites. Even at the same site, if the analysis or the data on which the analysis is based were to significantly change (e.g., if the depth interval used to estimate exposure soil concentration were to change), then the BMFs would have to be recalculated. For example, if the exposure assessment were to change such that the depth profile or some other aspect of the soil sampling protocol changed, then the <ESC> value would change and the BMF value would have to be recalculated accordingly, so that the HQ could be correctly calculated.

### 4.4.1 Terrestrial

The relationship between contaminant-specific risk, based on a tissue-based toxicological threshold value (MATC), and estimated biomagnification can be seen by expanding equation (4) from Section 4.1, i.e., by substituting the equivalent, BMF multiplied by estimated exposure area soil concentration (<ESC>), for tissue concentration (TC):

$$(HQ_i)_{terrestrial} = \frac{BMF * <ESC>}{MATC} \quad \textbf{(8)}$$

Expanding equation (5) yields a similar relationship between the BMF and risk, when risk is estimated by a dose-based toxicological threshold value (TRV):

$$(HQ_i)_{terrestrial} = \frac{R * BMF * <ESC>}{BAF * TRV} \quad \textbf{(9)}$$

Specification of the toxicological threshold values (i.e., MATC and TRV) and the procedure for estimating ESC are discussed in Sections 4.1.1.1 and 4.3, respectively. The development of bioaccumulation factor (BAF) and feed rate (R) distributions is discussed in Appendix Sections C.2.2 and C.2.3. The calculation of BMFs is discussed below and in Appendix Section C.1.6.1.2.

A variety of methods were used to calculate BMFs for use with the current <ESC> database. The Army, EPA, and Shell each have developed a methodology for calculating BMFs directly from tissue and estimated exposure area soil concentration data. All three approaches use the same soil and tissue data. These approaches are described in Appendix Section C.1.6.1.2. The three approaches yield different BMF values (Table 4.4-1) that reflect the uncertainties associated with the data as well as the alternate methods use to derive the BMFs. The Army, in conjunction with the OAS, is developing a supplemental field study to attempt to resolve uncertainties about the spatial extent of potential excess risk to biota.

The Army and Shell BMF calculation approaches involve deriving RMA-wide TC and <ESC> distributions defined by dissociating the pairs of $TC_{obs}$ and <ESC> values and deriving the BMF from these distributions and assumptions about the correlation between $TC_{obs}$, <ESC>, and BMF. These approaches are referred to as the collocated distributions methods. The Army and Shell approaches differ in their correlation assumptions. In addition, the Army approach calibrates its BMF by comparing tissue concentration predictions to the available tissue concentration field data from RMA. Shell and EPA approach chose not to incorporate a step parallel to the Army's calibration step in their approaches, although there is nothing about the Shell and EPA approaches that would prohibit including such a step. The EPA BMF approach differs from the Army and Shell approaches in that it focuses on paired tissue and soil concentration measurements rather than deriving RMA-wide TC and <ESC> data distributions. In other words, the EPA approach estimates BMF by taking the ratio of individual TC to <ESC> values only at the locations where paired tissue and soil concentration data were available when the BMFs were derived. The EPA approach is referred to as the paired data method.

The conceptual difference between the collocated distributions and paired data approaches can be summarized as follows. The collocated distributions approaches assume that the paired data are not representative of the true correlated TC and ESC data distributions at RMA because of systematic and random error in the sampling of paired data. The collocated distributions approaches further assume that the non-paired data and best professional judgment provide information that can be used to derive collocated TC and <ESC> distributions that better represent the true collocated TC and ESC data distributions at RMA. Conversely, the paired data approach assumes that the paired data better represent the true collocated TC and ESC data distributions than do derived collocated distributions, which depend more heavily on data extrapolations and best professional judgment. There is not a consensus about which approach and assumptions give the "best" estimate of BMF for the ERC, so further field studies are being designed to reduce uncertainty in risk estimates on those areas of RMA where the three approaches give conflicting predictions of potential risk.

The BMF model was used by all three approaches (Army, EPA, and Shell) to compute BMFs, and subsequent tissue concentrations, for the top predators in the RMA food webs because measured tissue concentrations were not available for these trophic boxes.

For the BMF model, information about the parameters necessary to compute BMFs was collected from RMA and the literature. These model parameters included bioaccumulation factors (BAFs) and dietary fractions (FRs) for the predators, as well as prey BMFs. The predator's BMF was then calculated as follows:

$$BMF_{predator} = BAF * \Sigma (FR * preyBMF) \qquad \textbf{(10)}$$

In equation (10), the resultant BMF is the estimate for a predator trophic box. The BAF estimates the amplification of a contaminant's concentration from a prey trophic box to its predator's trophic box. The FR term is the estimated mass fraction of the predator's food that comes from a specified prey trophic box. The dietary fractions are estimated from RMA food habits studies or from the literature. The prey BMFs in the summation term are calculated by

the Army, EPA, and Shell approaches, which results in three sets of predator BMFs (Table 4.4-1).

Substituting equation (10) into equations (8) and (9) gives the equations used to calculate contaminant-specific risks to predators from exposure through the terrestrial food chains:

$$(HQ_i)_{terrestrial} = \frac{[BAF * \Sigma(FR * preyBMF)] * <ESC>}{MATC} \quad \textbf{(11)}$$

when risk is estimated by a tissue-based approach, and

$$(HQ_i)_{terrestrial} = \frac{[R * \Sigma(FR * preyBMF)] * <ESC>}{TRV} \quad \textbf{(12)}$$

when risk is calculated by a dose-based approach.

Additional detail on the development of the food web model and on the terms used in equations (11) and (12) is provided in the appendices as follows: <ESC>, Appendix Section C.1.4; prey BMF by the Army, EPA, and Shell methods, Appendix Section C.1.6.1.2; food-web model, Appendix Section C.1.6.2; BAF, Appendix Section C.2.2; FR and R, Appendix Section C.2.3; MATC, Appendix, Section C.2.4; and TRV, Appendix Section C.2.6.

### 4.4.2 Aquatic

The characterization of risk from exposure to bioaccumulative COCs through the aquatic food chains was based on tissue concentration data from the aquatic biota in the RMA lakes. This approach and the rationale for choosing it is discussed in greater detail in Section 4.3.2. The relationship between contaminant-specific risk and aquatic biota tissue concentrations can be seen by substituting equation (6) from Section 4.3.2 into equation (4) from Section 4.1, i.e., by expressing the numerator (predator tissue concentration) in terms of the RMA-wide average tissue concentrations in the aquatic biota:

$$(HQ_i)_{aquatic} = \frac{BAF * \Sigma(FR * preyTC)}{MATC} \quad (13)$$

Substituting equation (7) into equation (5) yields a similar relationship between the average aquatic prey tissue concentration and risk when risk is estimated by a dose-based approach:

$$(HQ_i)_{aquatic} = \frac{R * \Sigma(FR * preyTC)}{TRV} \quad (14)$$

Equation (13) computes the risk to the heron trophic box from exposure to aldrin/dieldrin, DDT/DDE, and endrin through the aquatic food chains, and to the eagle trophic box from exposure to DDT/DDE and endrin through the aquatic food chains. Equation (14) computes the risk to the heron trophic box from exposure to mercury through the aquatic food chains, and to the eagle trophic box from exposure to aldrin/dieldrin and mercury through the aquatic food chains. Equations (13) and (14) were used for all of the bioaccumulative COCs for which the toxicological threshold values were available, and of those, only the most certain (i.e., the threshold value with the lowest total UF value) was chosen for use. Only equation (14) was used for the nonbioaccumulative COCs and chlordane.

The water bird trophic box is a special case in which the risk estimate is based solely on RMA water bird field data because this trophic box is assumed to feed only via aquatic food chains. For the water bird, HQs were calculated by an approach based on tissue concentration for aldrin/dieldrin, DDT/DDE, and mercury:

$$(HQ_i)_{waterbird} = \frac{TC}{MATC} \quad (15)$$

and by a dose-based approach for endrin:

$$(HQ_i)_{waterbird} = \frac{R * TC}{BAF * TRV} \quad (16)$$

where TC is calculated as the weighted average of the water bird tissue concentration sample averages in individual lakes, excluding East Upper Derby Lake. The HQ and HI calculations for the water bird, shorebird, heron, and bald eagle trophic boxes are provided in Appendix Figure D.1-20. Additional detail on the development of terms used in equations (13) through (16) is provided in the appendices as follows: food-web model, Appendix Section C.1.6.2; BAF, Appendix Section C.2.2; FR and R, Appendix Section C.2.3; MATC, Appendix Section C.2.4; TRV, Appendix Section C.2.6; and preyTC, Appendix Section D.1.

## 4.5 RISK CHARACTERIZATION RESULTS

The quantitative results of the analysis of the exposure measurements as compared to the toxicological threshold values discussed below. The degree to which these results are consistent with the results of the analysis of the ecological measurement endpoints that are discussed more fully in Appendix Section C.5 are also noted. As described in Sections 4.1.2 and 4.4, the characterization of potential ecological risks in the IEA/RC is based on a comparison of exposure measurements and toxicological threshold values, i.e., the ratios of tissue concentrations and doses to MATCs and TRVs. The resultant HIs provide a numerical measure of the potential for an adverse effect to an average individual in a target biota population from exposure to RMA COCs, except in the case of the bald eagle, where the individual is considered in addressing the adverse effects of the COCs.

The risk results (i.e., the HQs and HIs) are tacitly assumed to provide a basis for characterizing protection at the population level, which is warranted for the three ecological assessment endpoints defined for the RMA IEA/RC: structural diversity of the ecosystem, population robustness, and, for the bald eagle, health of the individual. These ecological assessment endpoints are defined more specifically for the ecological effects evaluation in terms of reproductive success, survivability, relative abundance, species richness, and density. The results of the qualitative analysis of ecological measurement endpoints provide additional information

that supplements the quantitative risk characterization results, providing the risk manager with a more thorough characterization of ecological risk at RMA.

The HQs and HIs were calculated by comparing exposure measurements to toxicological threshold values for the small mammal, medium mammal, small bird, water bird, shorebird, American kestrel, great horned owl, great blue heron, and bald eagle trophic boxes in the RMA food web. The type of toxicological threshold (i.e., MATC or TRV) selected for each trophic box and bioaccumulative COC (aldrin/dieldrin, DDT/DDE, endrin, and mercury) is shown in Table 4.5-1. The TRVs were used to characterize potential risks from exposure to all other COCs. The process for choosing between MATC and TRV for the bioaccumulative COCs is described in Appendix Section C.1.7.4.1. The HQs were calculated using equations (11) through (16) from Section 4.4.

The nine trophic boxes evaluated for potential risk varied in their food-chain associations. Five trophic boxes (American kestrel, great horned owl, small bird, medium mammal, and small mammal) were assumed to be exposed through terrestrial food chains only. The potential risks to these trophic boxes are shown in maps described below. Three trophic boxes (shorebird, heron, and eagle) were assumed to receive exposure through both aquatic and terrestrial food chains; for these three trophic boxes, the terrestrial and aquatic HQs are computed and reported separately because the potential terrestrial risk results are spatially variable. Aquatic HQs and HIs for these three trophic boxes and for the water bird, which receives all of its exposure through aquatic food chains, are reported in Table 4.5-1. The aquatic HIs are based only on the bioaccumulative COCs because of the absence of direct tissue concentration or dose data for nonbioaccumulative COCs in the four affected bird trophic boxes.

The terrestrial HQs and HIs are reported on maps that show areas of RMA where either HQs or HIs exceeded 1.0 for various groupings of COCs and trophic boxes. Terrestrial HQs and HIs cannot be summarized adequately in a tabular format because they are spatially variable. Four types of maps are included in this section: maps showing the number of trophic boxes that exceed 1.0 (Figures 4.5-1, 4.5-2, 4.5-4, and 4.5-6); maps showing soil concentrations of

individual COCs (Figures 4.5-3 and 4.5-5); maps showing HIs for particularly important trophic boxes (Figures 4.5-7 through 4.5-11); and maps showing the effect of exposure range size and contaminant concentration magnitude on the size of areas of risk (Figures 4.5-12 and 4.5-13).

As noted in Section 4.4.1, three approaches (Army, EPA, and Shell) were used to calculate BMFs. Risk calculation procedures for the three approaches are identical; all calculate potential risks from exposure through the terrestrial food web in the same manner, i.e., using equations (8) and (9). Therefore, the estimated potential risk by the Army, EPA, and Shell approaches differ solely because of dissimilar BMFs. Several of the maps (Figures 4.5-7 through 4.5-13) superimpose the risk estimates as calculated by the three approaches. The maps that portray only one approach (Figures 4.5-1, 4.5-2, 4.5-4, and 4.5-6) represent the intermediate estimate of the areal extent of potential risk. Large differences in risk estimates by the three approaches reflect large differences in the BMFs calculated by the three approaches for the trophic box(es) depicted on that map. Appendix Section C.3 contains the companion maps to the intermediate results depicted in Section 4.5, as well as additional maps.

When reviewing the maps, several important points must be considered:

- The areas depicted on the maps reflect areas of potential risk and do not represent areas delineating the extent of contamination nor areas necessarily requiring cleanup.

- The trophic box number categories may represent different trophic boxes depending on the area depicted. In Figure 4.5-1, for example, the shading for the "1 trophic box" category on the eastern border of RMA represents the bald eagle; however, on the western edge, the shading for this category represents either the great horned owl or the bald eagle, depending on whether the shading includes the Bald Eagle Management Area (BEMA).

- The areas of potential risk to the greatest number of species tend to be smaller and located more toward the center of RMA.

- Because the areas depicted on the maps show potential risk as a function of receptor exposure area, the trophic boxes representing species with larger exposure areas will generally show greater areas of potential risk (Figure 4.5-12), and trophic boxes representing species with smaller exposure areas will tend to show smaller areas of potential risk (Figure 4.5-13), relative to areas of soil contamination (Figure 4.5-3). These maps further illustrate the influence of the size of exposure area on risk projections as discussed in Section 4.1.

- The potential risk areas shown for the trophic boxes in Figures 4.5-12 and 4.5-13 may be driven primarily by point sources of contamination or hot spots (Figure 4.5-3). It is very important to understand that areas having an HI greater than 1.0 for certain receptors may not contain high concentrations of contaminants throughout, but may only pose potential risk because isolated hot spots occur somewhere within the exposure areas. If the high contaminant concentrations in these hot spots were reduced, then the areal extent of potential risk, as well as the magnitude of HQs and HIs, would be reduced proportionately.

- Portions of a map that are not shaded represent areas of no data as well as no exceedance (i.e., HI or HQ less than 1.0). Small square areas surrounded by shading represent sites that purposely were not sampled for the COCs for various reasons. For raptors, these small areas were insignificant relative to the size of a raptor's exposure area circle and so were assigned the average contaminant concentration value of the exposure area circle in which they were contained. Conversely, these small areas were large relative to the size of the exposure area circles for the other trophic boxes and were therefore excluded in the spatial averaging of soil contaminant concentrations over the exposure area.

The "shading" of Basin F on many of the HI maps as an area of potential risk for certain trophic boxes is a result of the concepts described in the last two items listed above. Because of the Basin F IRA, a layer of 18 inches of clean soil was placed over and now covers the former basin, thus eliminating exposure to contaminants at this location. However, because of existing soil

contamination east of Basin F and the large exposure area for some raptors (e.g., great horned owl), potential risks are depicted as occurring over Basin F when the basin is included within such an exposure area due to the averaging of contaminant concentrations.

### 4.5.1 Terrestrial Ecosystems

Figure 4.5-1 shows how many of the seven trophic boxes that were mapped (bald eagle, great horned owl, American kestrel, small bird, shorebird, small mammal, and medium mammal) have an HI greater than 1.0 for all COCs combined on the basis of exposure to soil. The great blue heron risk was not mapped because its potential for exposure to soil was considered as being constrained to a very narrow band around each lake.

Areas of potential risk to biota occupy most of the interior sections of RMA and include South Plants; Basins A, B, C, D, and F; the Toxic Storage Yard, and the northernmost terrestrial areas adjacent to Lake Mary, Lake Ladora, Upper Derby Lake, and Lower Derby Lake. These are areas where all trophic boxes have HIs greater than 1.0 (Figure 4.5-1). The spatial arrangement of the areas of potential risk is related to the magnitude of chemical concentrations in the soil in areas of historical releases such as testing areas, storage yards, basins, and trenches. For example, the circular area of potential risk in the center of Section 24 (Figure 4.5-2) is due to high aldrin/dieldrin concentrations (Figure 4.5-3) related to the operation of a sewage treatment plant. Similarly, the circular area of potential risk in the southeast quarter of Section 34 (Figure 4.5-4) represents an area of elevated mercury levels in soil in an overflow basin for a sewer line (Figure 4.5-5).

Most of RMA presents a potential risk (HI greater than 1.0) to at least one trophic box (Figure 4.5-1). Much of the potential risk can be attributed to OCPs (Figure 4.5-2), although the metals (Figure 4.5-4) contribute significantly to the potential risk in many areas. Total potential risk is presented because an animal's dose-response threshold for these metals could be just above the level of the background metal concentrations. Thus, any addition of these metals to the habitat of an animal could be toxic. A decrease in the areal extent of potential risk for various numbers of trophic boxes occurs when only the concentrations of metals that are above the background

(indicator) levels are used to estimate risk (Figure 4.5-6). This map is intended to illustrate the magnitude of potential risks from concentrations associated with human activities at RMA. Careful evaluation of the metals maps indicates that the indicator-level concentrations are responsible for the size of the area showing potential risk. It should be noted that the potential risks associated with metal concentrations assumed to be related to RMA activities (i.e., those above the indicator level) are confined to the central region of RMA and are primarily caused by elevated levels of mercury and arsenic (Appendix Section C.3). The potential risks depicted may be overestimated because the trophic-box-specific HQs calculated for mercury assumed that all detected mercury concentrations were the more bioavailable and toxic form, methylmercury. This is a conservative assumption that resulted in an overestimation of the actual mercury risks present at the site because the predominant form of mercury in the soil is most likely inorganic, which is less bioavailable (Ellenhorn and Barceloux 1988; Fergusson 1990; Klaassen et al. 1986). The other COCs, namely chlordane, DBCP, DCPD, CPMS, and $CPMSO_2$, did not contribute significantly to ecological risk, although hot spots exist for some of these COCs. The contribution of these COCs to risk can be seen in maps provided in Appendix C.3 (Figures C.3-9 and C.3-22).

Maps depicting the HIs show the relative magnitude of the potential risk areas on the basis of an HI of 1.0 or 10. Based on best professional judgment, an 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 HI greater than 1.0, the potential for adverse effects increases as the HI value increases. The range of uncertainty in these statements regarding HIs spans at least one order of magnitude. This uncertainty exists in both directions; hence, some risk may occur at values of HI as low as 0.1 and no risk may occur at values of HI as high as 10.

### 4.5.2 Aquatic Ecosystems

The HQs and HIs for exposure to the bioaccumulative COCs through the aquatic food chains (i.e., aquatic HQs and HIs) are reported (Table 4.5.1) for the water bird, shorebird, great blue heron, and bald eagle trophic boxes. The procedure for calculating these risk estimates is

presented in Sections 4.3.2 and 4.4.2. All four trophic boxes exhibit a potential risk from aquatic food chain exposure to the bioaccumulative COCs. The great blue heron HI exceeds 10 (HI equals 13), while the HIs for water bird, shorebird and bald eagle marginally exceed 1.0 (water bird HI equals 1.4; shorebird HI equals 1.1; and bald eagle HI equals 1.9). The HQs indicate that potential risk to the great blue heron is due largely to mercury, although the OCPs contribute significantly as well. The HQs indicate that mercury is also an important contributor to potential risk for the water bird, shorebird, and bald eagle, and that each of the OCPs contributes significantly to the potential risk for at least one of these trophic boxes.

The COC input to the aquatic risk calculations are tissue concentration data, so the contaminants are not explicitly attributed to a source medium (i.e., soil, sediment, or water). Attributing the contaminants to a primary source was avoided because water contaminant concentration values were BCRL values and conversions between water and sediment concentrations were complicated by the relatively high organic carbon content in RMA lake sediments. The average percent of organic carbon in Lower Derby Lake and Lake Ladora sediment grab samples (five per lake) is about 8 percent. At the level of 8 percent, the dynamics of hydrophobic contaminant partitioning between lake sediments and waters is complex and difficult to quantify. Ultimately, however, soils from shoreline and surrounding upland areas are the source of contamination found in RMA lake sediments and waters, so biota contamination through the aquatic food chains is attributable to soil contamination in the vicinity of the lakes. This is qualitatively supported by the observation of highly elevated aldrin, dieldrin, and endrin sediment concentrations near the north inlet of Upper Derby Lake. These concentrations are orders of magnitude above sediment concentrations elsewhere in Upper Derby Lake, many of which are BCRL values (Appendix Figures C.3-116 through C.3-118). The inlet sediment concentrations are similar to nearby soil concentrations that are on the order of 0.1-10 ppm. Although the mechanisms of soil contaminant influx into the RMA lakes were not quantified in the ERC, these mechanisms may include soil erosion, surface-water runoff and deposition of airborne particulates.

### 4.5.3 Target Receptors

Figures 4.5-7 through 4.6-9 illustrate the potential risk from all of the COCs combined to the three raptors (i.e., bald eagle, great horned owl, and American kestrel) based on the Army, EPA, and Shell approaches. The bald eagle has a single exposure area, the BEMA as well as the prairie dog towns existing at the time of the risk characterization. The bald eagle's entire exposure area is equivalent to a single circular exposure area for one of the other trophic boxes. The potential risk to the bald eagle exceeds 1.0 for all three approaches throughout the entire exposure area (Figure 4.5-7). The potential risk from all COCs combined also exceeded 1.0 over most of RMA, regardless of approach, for the great horned owl (Figure 4.5-8) and the American kestrel (Figure 4.5-9). The three raptors have exposure areas that are much larger than other trophic boxes. Sizable areas of potential risk are created for the raptors by averaging very high contaminant concentrations in hot spots around the manufacturing plants and basins over their large exposure areas.

The potential risk from all COCs combined exceeds 1.0 over most of RMA, regardless of approach, for the medium mammal (Figure 4.5-11). Because the exposure area for the prairie dog is relatively small, the large areas of potential risk are due to significant contributions to the HI value from several different COCs and relatively high BMFs for some of the COCs. See Appendix Section C.3 for soil concentration maps and COC-specific HQ maps of the medium mammal.

The influence of exposure area size on delineating areas of potential risk is portrayed in the aldrin/dieldrin HQ maps of the great horned owl (Figure 4.5-12) and the medium mammal (Figure 4.5-13). The hot spots of high aldrin/dieldrin contamination around South Plants, North Plants, Basins A and C, the area east of Former Basin F, Toxic Storage Yard, the Motor Pool/Railyard area (Section 4), Sand Creek Lateral (Section 35), and the Sewage Treatment Plant (Section 24), as depicted in the soil contamination map for aldrin/dieldrin (Figure 4.5-3), are easily noted in the medium mammal map as the origination of areas of potential risk. These hot spots that are also driving the exceedances of larger exposure areas are not apparent in Figure 4.5-12, yet they remain the cause.

Additional receptor-specific risk maps for the various COCs, as well as for all COCs combined, can be found in Appendix Section C.3.

Because terrestrial exposure areas for the great blue heron and shorebirds are limited to lake shorelines and adjacent upland areas around each of the lakes due to habitat preferences, their areas of potential risk from soil contamination are small relative to those of the other trophic boxes. The potential risk to bald eagles from aquatic food chains also entailed considering lake shorelines and adjacent upland areas since these areas were assumed to be the source of the contaminants that enter the lakes. Shorebirds have potential risks from terrestrial food chains for all COCs combined, regardless of approach, for the shoreline and adjacent upland areas of all of the lakes (Figure 4.5-10). Although not mapped, a potential risk (HI greater than 1.0) also exists in these areas for the great blue heron and bald eagle.

### 4.5.4 Ecological Measurement Endpoints

The results of the ecological risk assessment indicate that potential risks occur in areas of RMA having elevated soil concentrations of the contaminants evaluated, and that geometric mean concentrations of contaminants in tissues from apparently healthy individuals exceeded the recommended whole-body MATC values for ten trophic box/COC combinations, four of which were for mercury where the one-half CRL value was greater than the MATC value. Average tissue concentrations that did exceed the MATC for a particular trophic box varied by COC. The dieldrin MATC was exceeded by 4 to 11 fold, the mercury MATC by 2 to 6 fold, and the DDE MATC by 3 fold. The available data on ecological measurement endpoints do not reveal adverse effects of chemical contamination on trophic diversity at RMA. Parties to RMA evaluated these ecological studies for their applicability in the ERC and their positions are presented in Appendix Sections C.5-4 and C.5-5. Data from studies on reproduction do not reveal adverse reproductive effects for birds or mammals, with the possible exception of mallards at Lower Derby Lake in 1986; Appendix Section C.5 provides more current data on waterfowl breeding success.

Species richness was difficult to evaluate because of the anthropogenic disturbance of many areas of contamination. Contamination effects on species abundance may have occurred, but analyses

indicate that habitat quality and diversity account for most of the differences observed relative to areas off post. Comparison of various measures of reproductive success for American kestrel, great horned owl, and burrowing owl with <ESC> values (see Appendix C.5) generally revealed no trends. Studies of reproductive success for small mammals had ambiguous results that are apparently related in part to habitat quality. Comparison of small mammal species abundance and species diversity with ESC revealed no trends. Studies indicated adverse reproductive effects for some bird species in the past. Moreover, there is recent evidence of potentially harmful levels of contamination in some individuals of several bird species, but no evidence of significant population effects. The effects that have been documented with ecological measurement endpoints are consistent with the identified exposure routes, endpoints, and areas of potential risk. Likely effects of RMA contamination have been observed in individual animals collected at RMA, such as tissue concentrations above MATC values in individuals that appeared healthy when collected as intentional specimens, or behavioral symptoms or necropsy results indicating contaminants caused or contributed to the death of raptors and carnivores. These effects were not apparent in the available data collected on measurement endpoints involving wildlife populations at RMA (Appendix Section C.5).

## 4.6 BIOTA SOIL CRITERIA

Tables 4.6-1 and 4.6.2 present a set of soil COC concentrations that provide an alternative means of summarizing RMA Ecological Risk Characterization results. These values, referred to as "biota soil criteria," are soil concentrations that would, if attained on average over a home range, yield a predicted COC tissue concentration or dose- in a (hypothetical) average trophic box individual with that home range- equal to the COC-specific MATC or TRV for that trophic box. Stated another way, these values represent the soil COC concentrations that, if achieved on average over a home range, would yield HQs equal to one for their respective trophic box/COC combinations at the grid point at the center of that home range. The equations for calculating biota soil criteria are derived from the terrestrial HQ equations (Section 4.4.1, equations 8 and 9); the derivation is presented in Appendix C.1, Section C.1.8.

The biota soil criteria reported in Tables 4.6-1 and 4.6-2, though risk-based, are not cleanup criteria. As described above, these values are soil COC concentrations that, if achieved on average over a home range, would yield HQs equal to one for that home range. Misinterpreting the biota soil criteria as maximum (rather than average) allowable soil concentrations would lead to overestimation of the remediation volume necessary to achieve risk-based cleanup objectives. In addition, the biota soil criteria are based on terrestrial HQs, and therefore do not provide for consideration of total risk (i.e., HI due to exposure through both terrestrial and aquatic food webs). Based on the assumption of additivity, consideration of only the HQs would underestimate risk and related remediation.

In general, the biota soil criteria are insufficient for quantifying the amount that soil concentrations need to be reduced at specific points to meet risk-based management objectives. What the biota soil criteria do tell us is the level to which a soil COC concentration (for a single species, single chemical, single pathway, and single medium) needs to be reduced, on average over a home range, to reduce the predicted population mean dose below the TRV, or the predicted population mean tissue concentration below the MATC. If risk management decisions were to be based on a single COC, then the biota soil criteria could be considered as cleanup criteria for those trophic boxes with sufficiently small home ranges. Of the trophic boxes analyzed for risk at RMA, this would include small and medium mammals. If multiple COCs were to be considered in risk management decisions, then the biota soil criteria would have to be reduced to assure that the resulting HI, instead of the individual HQs, was sufficiently low if the HQs were assumed to be additive. The amount that each COC's biota soil criterion would have to be reduced would depend on the pattern of COCs in RMA soils. For example, if risk management decisions were based on the bioaccumulative OCPs (aldrin/dieldrin, DDT/DDE, and endrin), and the objective were to achieve a bioaccumulative OCP HI of 0.80, then any combination of OCP HQs that summed to 0.80 would meet the objective. The particular set of HQs that would have to be achieved at a particular RMA location would depend on the relative abundances and, for trophic boxes with large home ranges (all except small and medium mammal), the spatial distribution of the bioaccumulative OCPs.

When home ranges are large, and/or risk management decisions are to be based on multiple COCs, spatially explicit, risk-based soil cleanup criteria for RMA biota can be determined using ERC results (specifically, BAF, BMF, <ESC>, R, MATC, and TRV values, and the tissue- and dose-based HQ equations) in a GIS-based investigation to evaluate whether possible remediation scenarios are expected to achieve acceptable post-remediation HIs.

## 4.7 SUMMARY AND CONCLUSIONS

An initial inspection of the HI maps that provide the major results of the risk characterization indicates potential risk to target receptors over most of RMA. However, closer inspection shows that areas of highest potential risk are located in the central portion of RMA and are associated with major chemical manufacturing or disposal areas (e.g., South Plants, the basins, North Plants). For species with exposure areas large enough to overlap this central area, its higher concentrations also contribute disproportionately (relative to area) to defining risk outside the central area.

The maps of HIs for groups of selected COCs and HQs for individual COCs (Section 4 and Appendix Section C.3) show that the potential risks in these areas are driven primarily by elevated soil concentrations of aldrin/dieldrin, endrin, DDT/DDE, and mercury. The bald eagle had HQs of 1.0 to 10 for aldrin/dieldrin based on the Army and EPA approaches, and for mercury based on all three approaches. The great horned owl and American kestrel had HQs greater than 10 based on all three approaches for the same COCs. The American kestrel also had an HQ greater than 10 for endrin based on all three approaches, and the great horned owl had an HQ greater than 10 for DDT/DDE based on all three approaches. The medium mammal, a prey item for the great horned owl and bald eagle, had HQs greater than 10 for aldrin/dieldrin, endrin, DDT/DDE, and mercury based on all three approaches. The Army approach was the only one used to evaluate the potential risk from arsenic to medium mammals and resulted in an HQ greater than 10 for some small areas in the central portion of RMA.

The difference in the areal extent of potential risk depicted by the three approaches is due solely to the difference in the BMF values used and reflects both uncertainty in the data and conceptual

differences regarding the derivation of the BMFs. The BMF values developed by the Army, EPA, and Shell approaches are specific to conditions as defined at RMA and should not be applied in food-web models, nor be used to evaluate potential risks at other contaminated sites. However, the methodologies developed by the Army, EPA, and Shell for deriving BMFs from a given data set may be applicable elsewhere.

The size of the potential risk areas depicting the range of HQ or HI values, regardless of approach, is largely driven by hot spots of contamination as well as the size of exposure area. This is particularly true for the driver contaminants. The areal extent conveying potential risk to biota usually does not contain high concentrations of contaminants throughout, and potential risk may be elevated only because of isolated hot spots within the exposure area. If the level of contamination associated with these hot spots were reduced, then the HQ and HI values would be lower and the areal extent of risk should also be reduced.

Potential risks to the bald eagle, great blue heron, shorebird, and water bird from exposures through aquatic food chains exist, but are of relatively low magnitude ($2.0 > HI > 1.0$) for all but the great blue heron (HI equals 13). This potential risk can be attributed primarily to mercury contamination in the soils of lake shorelines and adjacent upland areas. Risk associated with mercury is overestimated because of the very conservative assumption that all mercury at RMA is in the most toxic and bioavailable form, methyl mercury. The primary form of mercury in soil at RMA is inorganic, not methyl mercury.

The quantitative results of the risk analysis characterize the magnitude and spatial extent of potential risks to the diverse aquatic and terrestrial biota posed by the existing contamination on post (i.e., the first objective of the ERC). They indicate that potential risks occur in areas of RMA having elevated soil concentrations of the contaminants evaluated and that, for species with larger exposure areas, risk extends beyond these areas.

To determine whether the projected potential risks are consistent with available data on ecological conditions at RMA (i.e., the second objective of the ERC), previously conducted RMA studies

with data pertinent to the ecological health and status of biota were evaluated and summarized (Appendix Section C.5). This summary assesses the ecological measurement endpoints of reproductive success, survivability, relative abundance, species richness, and density, and together with Appendix Section C.4, also provides the tissue concentration data used in the IEA/RC. The available data on ecological measurement endpoints do not reveal adverse effects on reproductive success for birds or mammals, with the possible exception of mallards at Lower Derby Lake in 1986. Furthermore, they do not reveal that survivability, relative abundance, species richness/trophic diversity, or density of species studied at RMA are outside their normal range of variability. For those studies that compared ecological measurement endpoints in contaminated and uncontaminated areas at RMA, or whose data were retrospectively evaluated against tissue concentrations or <ESC> values, no trends were apparent that reliably associated high chemical concentrations with values for any of the ecological measurement endpoints that would be detrimental to wildlife populations.

Among trophic boxes for which literature-based MATC values and intentional on-post tissue samples were obtained, few have geometric mean values that exceed whole-body MATC values. Geometric mean concentrations of contaminants in tissues from apparently healthy individuals exceeded the recommended whole-body MATC values for ten trophic box/COC combinations, four of which were for mercury, where the one-half CRL value is greater than the MATC value. Geometric mean values for all other trophic box/COC combinations were below their respective MATCs; maximum concentrations were exceeded more frequently. This information further supports the potential for risk to individuals from chemical contamination at RMA even as it indicates the general absence of population-level impacts.

In summary, the results of the ERC are not inconsistent with available data on ecological conditions at RMA. Both types of data indicate potential risk to individuals, but generally not to populations of wildlife, and although both are subject to uncertainties (as discussed in Section 5.0 and Appendix E), both types of data serve to establish a realistic basis for future risk management decisions.

Table 4.4-1. Predator Biomagnification Factors Using the Army, EPA, and Shell Approaches

| | Aldrin/Dieldrin | DDT/DDE | Endrin | Mercury |
|---|---|---|---|---|
| Army BMF Approach | | | | |
| Trophic Box | | | | |
| American Kestrel | 2.6 | 9.9 | 0.19 | 0.32 |
| Great Horned Owl | 8 | 32 | 0.088 | 0.26 |
| Great Blue Heron | 2.9 | 11.4 | 0.11 | 0.68 |
| Bald Eagle | 6.1 | 19 | 0.067 | 0.23 |
| EPA BMF Approach | | | | |
| Trophic Box | | | | |
| American Kestrel | 23 | 55 | 1.3 | 0.18 |
| Great Horned Owl | 41 | 340 | 1.4 | 4.8 |
| Great Blue Heron | 8.6 | 42 | 0.16 | 0.76 |
| Bald Eagle | 29 | 220 | 1.3 | 5.4 |
| Shell BMF Approach | | | | |
| Trophic Box | | | | |
| American Kestrel | 4.9 | 14 | 0.26 | 0.068 |
| Great Horned Owl | 6.9 | 17 | 0.40 | 0.24 |
| Great Blue Heron | 3.0 | 18 | 0.10 | 0.72 |
| Bald Eagle | 4.5 | 120 | 0.40 | 0.26 |

Table 4.5-1 Hazard Quotients and Hazard Indices for Exposure through Aquatic Food Chains

| Trophic Box | Hazard Quotients for COCs | | | | Hazard Index |
|---|---|---|---|---|---|
| | Aldrin/Dieldrin | DDT/DDE | Endrin | Mercury | |
| Water bird | 2.87 | 1.66 | 0.63 | 6.75 | 11.91 |
| Shorebird | 0.19 | 2.60 | 1.17 | 8.30 | 12.26 |
| Great Blue Heron | 2.28 | 1.06 | 0.63 | 15.63 | 19.60 |
| Bald Eagle | 0.93 | 0.17 | 0.03 | 0.21 | 1.34 |

| Trophic Box | Toxicological Threshold Selection | | | |
|---|---|---|---|---|
| | Adlrin/Dieldrin | DDT/DDE | Endrin | Mercury |
| Water bird | MATC | MATC | TRV | MATC |
| Shorebird | TRV | MATC | MATC | TRV |
| Great Blue Heron | MATC | MATC | MATC | TRV |
| Bald Eagle | TRV | MATC | MATC | TRV |

## Table 4.6-1 Biota Soil Criteria for the Bioaccumulative COCs[1, 2]

A. Using Army BMFs

| | Ald/Dld | DDT/DDE | Endrin | Mercury |
|---|---|---|---|---|
| Small Bird | 2.3E+00 | 4.1E-02 | 4.7E-01 | 1.5E-01 |
| Small Mammal | 7.7E-01 | 3.5E-01 | 5.3E-01 | 2.1E-01 |
| Medium Mammal | 4.7E-01 | 4.4E-01 | 5.0E-01 | 1.4E-01 |
| Kestrel | 4.5E-01 | 4.4E-01 | 2.7E-01 | 5.2E-02 |
| Great Horned Owl | 1.2E-01 | 1.7E-02 | 9.9E-01 | 6.4E-02 |
| Shorebird | 4.1E+00 | 2.9E-02 | 5.2E-02 | 1.1E+00 |
| Great Blue Heron | 3.0E-01 | 1.3E+00 | 3.9E-01 | 1.3E-01 |
| Bald Eagle | 5.8E-02 | 1.1E-01 | 4.6E-01 | 2.4E-02 |

B. Using EPA BMFs

| | Ald/Dld | DDT/DDE | Endrin | Mercury |
|---|---|---|---|---|
| Small Bird | 4.8E-01 | 7.9E-03 | 5.7E-02 | 5.7E-02 |
| Small Mammal | 1.4E-01 | 4.9E-02 | 7.0E-02 | 3.6E-02 |
| Medium Mammal | 1.7E-01 | 6.2E-02 | 8.1E-02 | 4.0E-02 |
| Kestrel | 5.2E-02 | 7.8E-02 | 3.9E-02 | 9.0E-02 |
| Great Horned Owl | 2.3E-02 | 1.5E-03 | 6.3E-02 | 3.5E-03 |
| Shorebird | 7.2E-01 | 9.1E-03 | 4.6E-02 | 4.7E-02 |
| Great Blue Heron | 1.0E-01 | 3.6E-01 | 2.7E-01 | 1.2E-01 |
| Bald Eagle | 1.3E-02 | 9.9E-03 | 2.4E-02 | 1.0E-03 |

C. Using Shell BMFs

| | Ald/Dld | DDT/DDE | Endrin | Mercury |
|---|---|---|---|---|
| Small Bird | 1.3E+00 | 3.4E-02 | 4.1E-01 | 1.0E-01 |
| Small Mammal | 3.7E-01 | 2.6E-01 | 4.0E-01 | 6.7E-02 |
| Medium Mammal | 3.2E-01 | 3.3E-01 | 3.9E-01 | 6.7E-02 |
| Kestrel | 2.4E-01 | 3.2E-01 | 2.0E-01 | 2.4E-01 |
| Great Horned Owl | 1.4E-01 | 3.1E-03 | 2.2E-01 | 6.9E-02 |
| Shorebird | 3.1E+00 | 2.3E-02 | 8.7E-02 | 9.5E-02 |
| Great Blue Heron | 2.9E-01 | 8.2E-01 | 4.1E-01 | 1.3E-01 |
| Bald Eagle | 8.0E-02 | 1.9E-02 | 7.7E-02 | 2.1E-02 |

[1] This table presents the soil concentrations (ppm), that, if measured using sampling and laboratory procedures as in the RMA-IEA/RC, and if achieved on average over a home range, would yield HQ = 1 for the trophic box/COC in question at the grid point at the center of that home range.

[2] These values are not cleanup criteria, i.e., they do not represent the maximum soil concentrations that could be allowed without exceeding risk-based cleanup objectives. Proper interpretation of the biota soil criteria is provided in the body of Section 4.6.

Table 4.6-2 Biota Soil Criteria for the Nonbioaccumulative COCs[1,2]

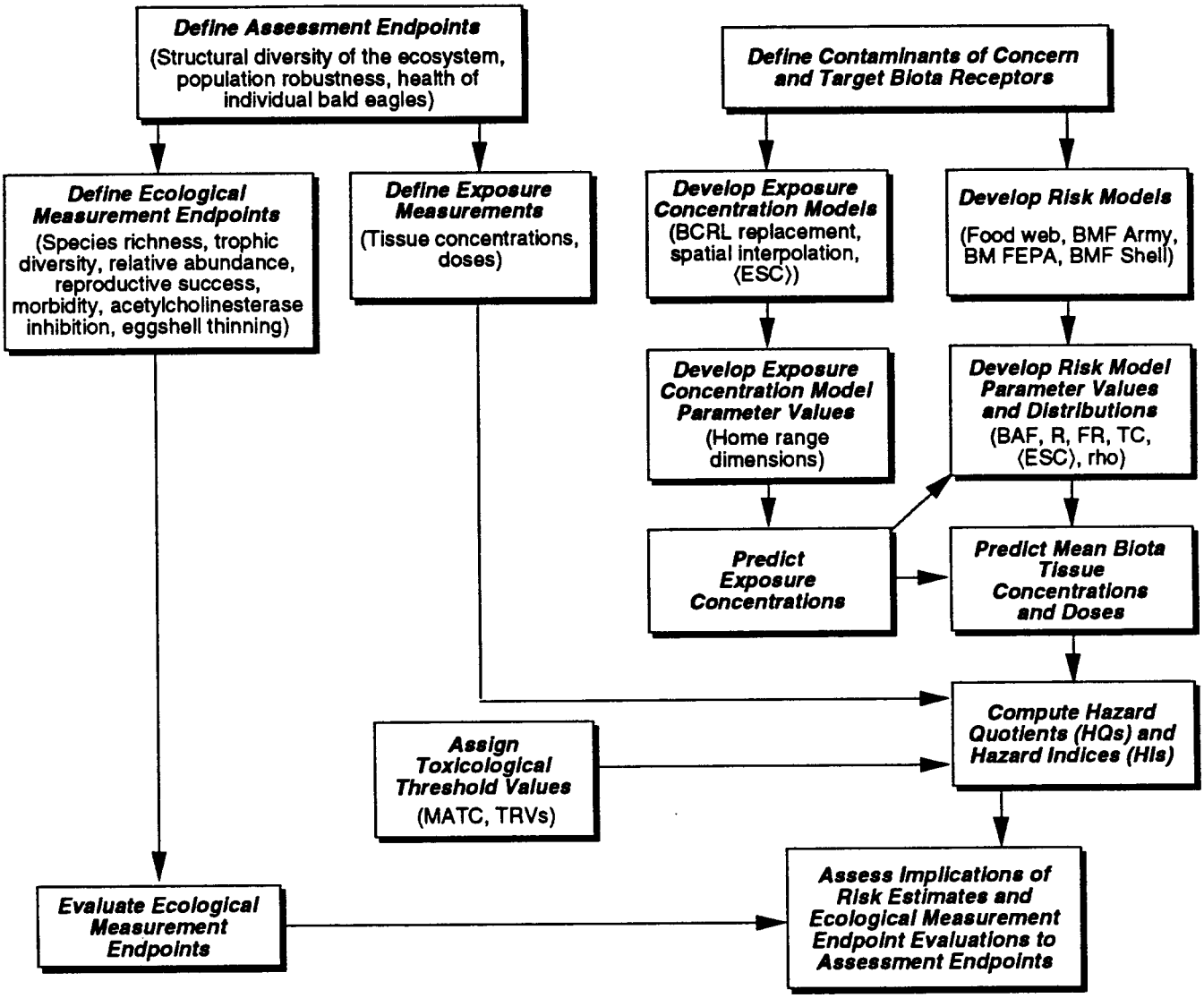


| Trophic Box | As | Cu | Cd | DCPD | Chlordane | CPMS | $CPMSO_2$ | DBCP |
|---|---|---|---|---|---|---|---|---|
| Bald Eagle | 74.0 | 186 | 39.0 | 2050 | 14.0 | NA | NA | 66.0 |
| Great Blue Heron | 118 | 299 | 75.0 | 2774 | 11.0 | NA | NA | 53.0 |
| Shorebird | NA | NA | NA | NA | NA | NA | NA | NA |
| Great Horned Owl | 147 | 371 | 93.0 | 3443 | 14.0 | NA | NA | 66.0 |
| Kestrel | 147 | 371 | 93.0 | 3443 | 13.5 | NA | NA | 66.0 |
| Herptile | NA | NA | NA | NA | NA | NA | NA | NA |
| Small Bird | 74.8 | 189 | 47.2 | 1752 | 6.89 | NA | NA | 33.5 |
| Small Mammal | 15.8 | 313 | 18.8 | 1180 | 42.0 | 100 | 113 | 20.8 |
| Medium Mammal | 5.0 | 106 | 6.0 | 399 | 14.0 | 34.0 | 38.0 | 7.0 |

[1] This table presents the soil concentrations (ppm), that, if measured using sampling and laboratory procedures as in the RMA-IEA/RC, and if achieved on average over a home range, would yield HQ = 1 for the trophic box/COC in question at the grid point at the center of that home range.

[2] These values are not cleanup criteria, i.e., they do not represent the maximum soil concentrations that could be allowed without exceeding risk-based cleanup objectives. Proper interpretation of the biota soil criteria is provided in the body of Section 4.6.

NA = not available

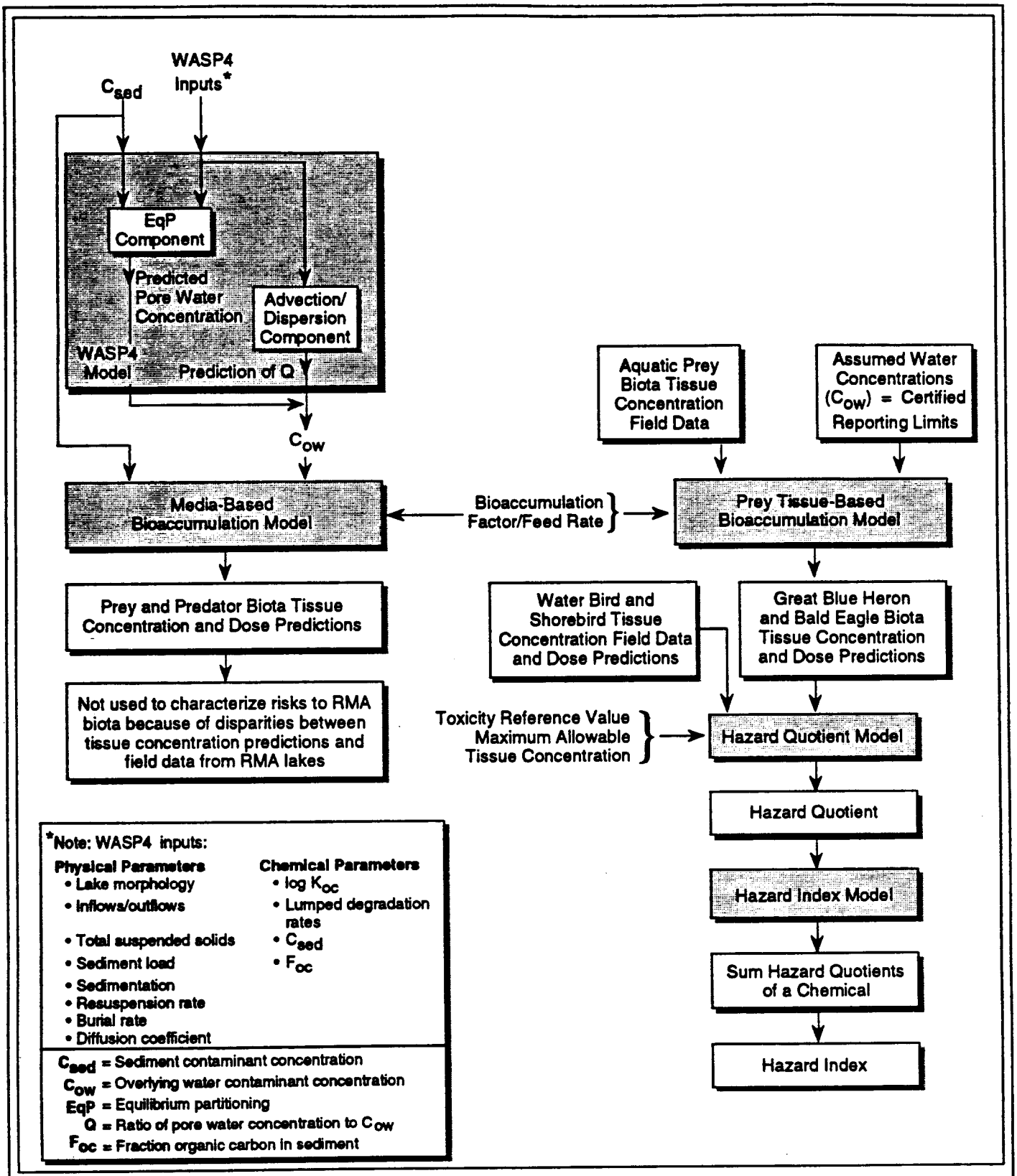



RMA IEA/RC 4 6.94 jb

**Figure 4.1.1**

**Rocky Mountain Arsenal Ecological Risk Characterization Flow Chart**

Rocky Mountain Arsenal
Prepared by: Enserch Environmental Corporation

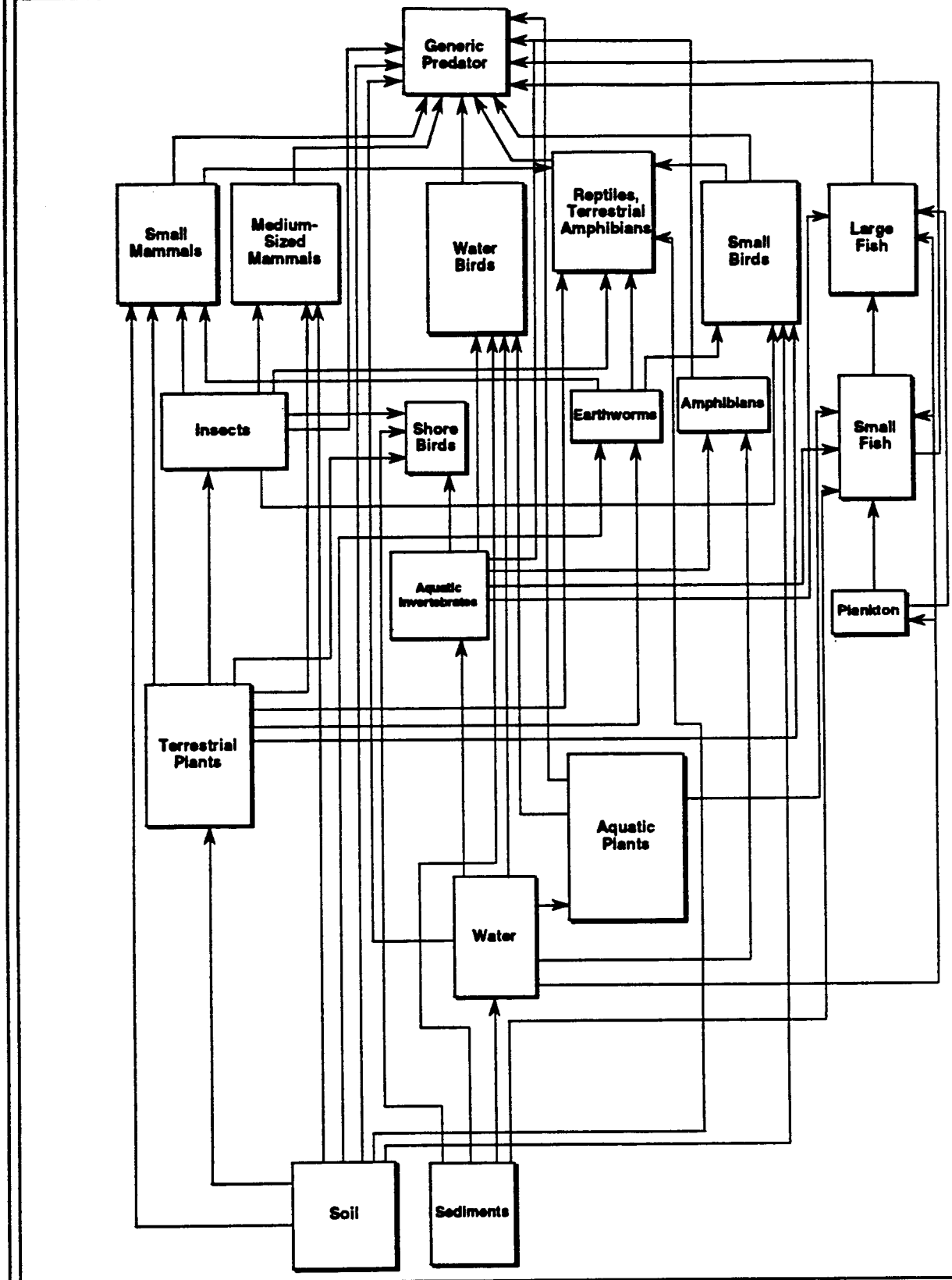



RMA IEA/RC 4 2.94 jb

**Figure 4.3.1**

**Aquatic Modeling Flow Chart for Ecological Risk Characterization at Rocky Mountain Arsenal**

Rocky Mountain Arsenal
Prepared by: Enserch Environmental Corporation

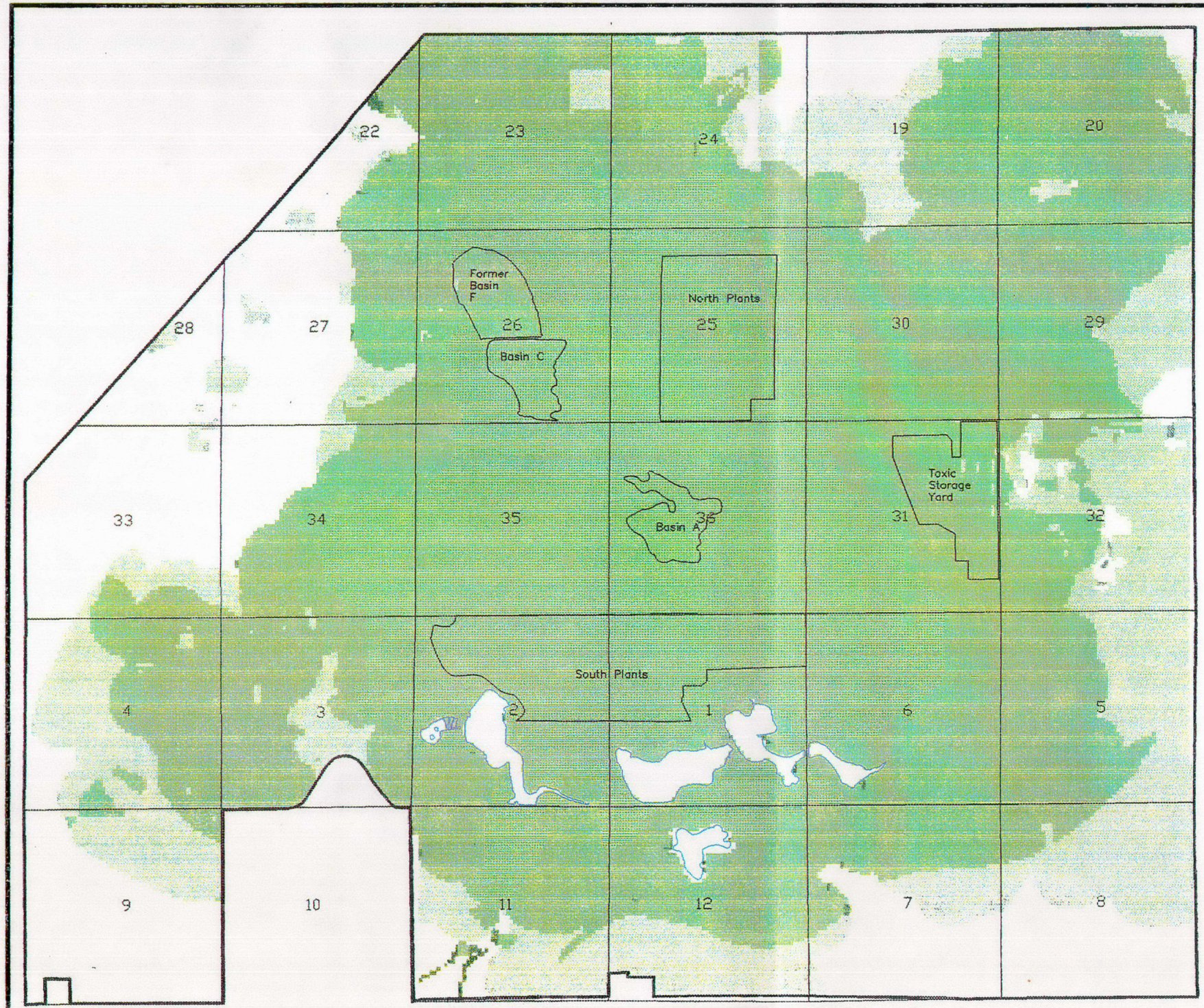



RMA IEA/RC 4 2.94.jb

**Figure 4.4-1**

**Generic Predator Food Web**

Rocky Mountain Arsenal
Prepared by: Ebasco Services Incorporated




## Legend

- 1 trophic box with HI > 1
- 2-4 trophic boxes with HI > 1
- 5-7 trophic boxes with HI > 1

Lake

31 Section Number

Section Line

N

0 2000 4000

Scale in Feet

Prepared for:

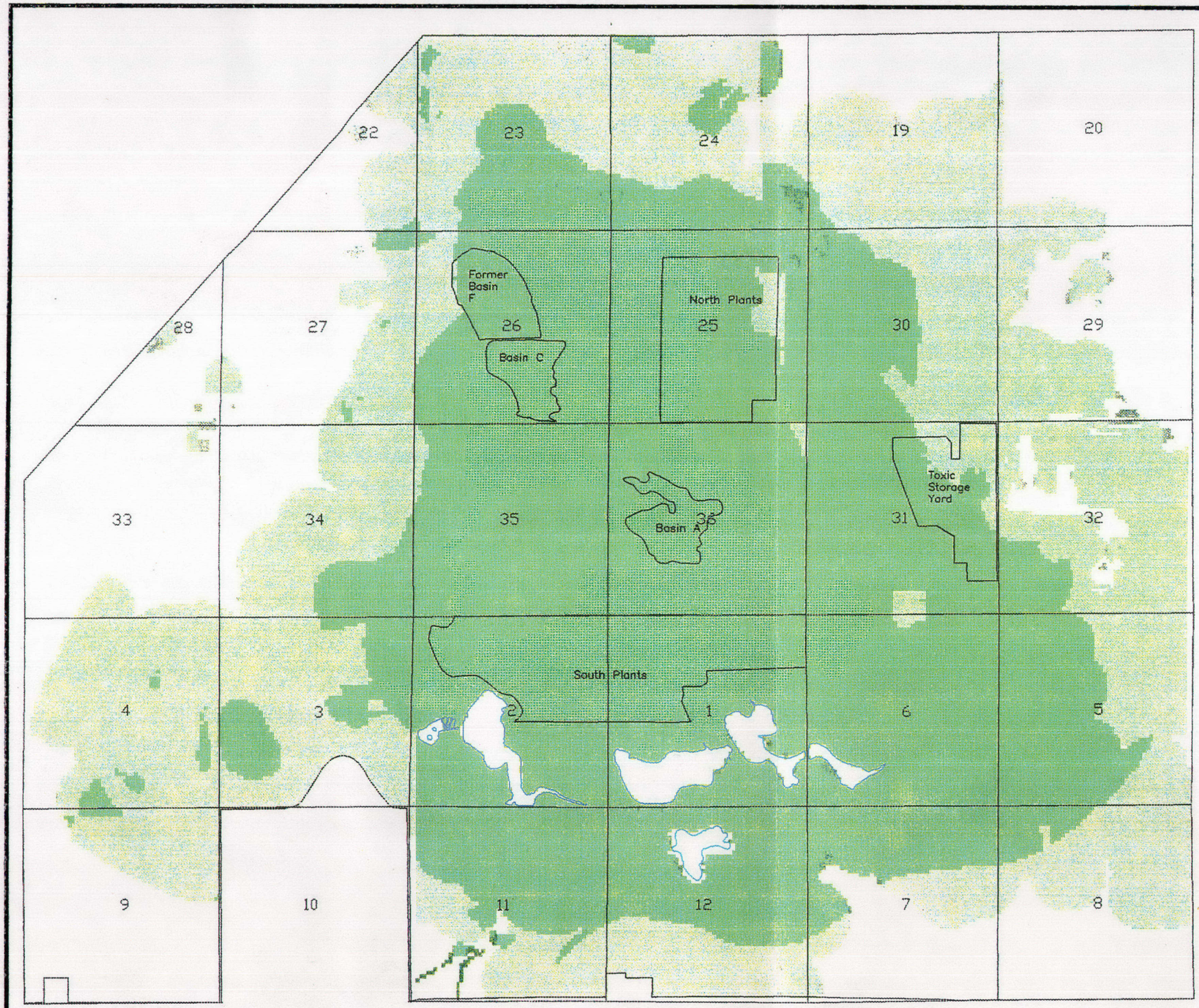
U.S. Army Program Manager
for Rocky Mountain Arsenal

February 1994

Figure 4.5-1

Number of Trophic Boxes with Soil Hazard Indices Greater than 1.0 for All COCs Combined Based on the Shell Approach

Rocky Mountain Arsenal
Prepared by: ENSERCH Environmental Corp




## Legend

- 1 trophic box with HI > 1
- 2-4 trophic boxes with HI > 1
- 5-7 trophic boxes with HI > 1
- Lake
- 31 Section Number
- Section Line

N

0 2000 4000

Scale in Feet

Prepared for:

U.S. Army Program Manager
for Rocky Mountain Arsenal

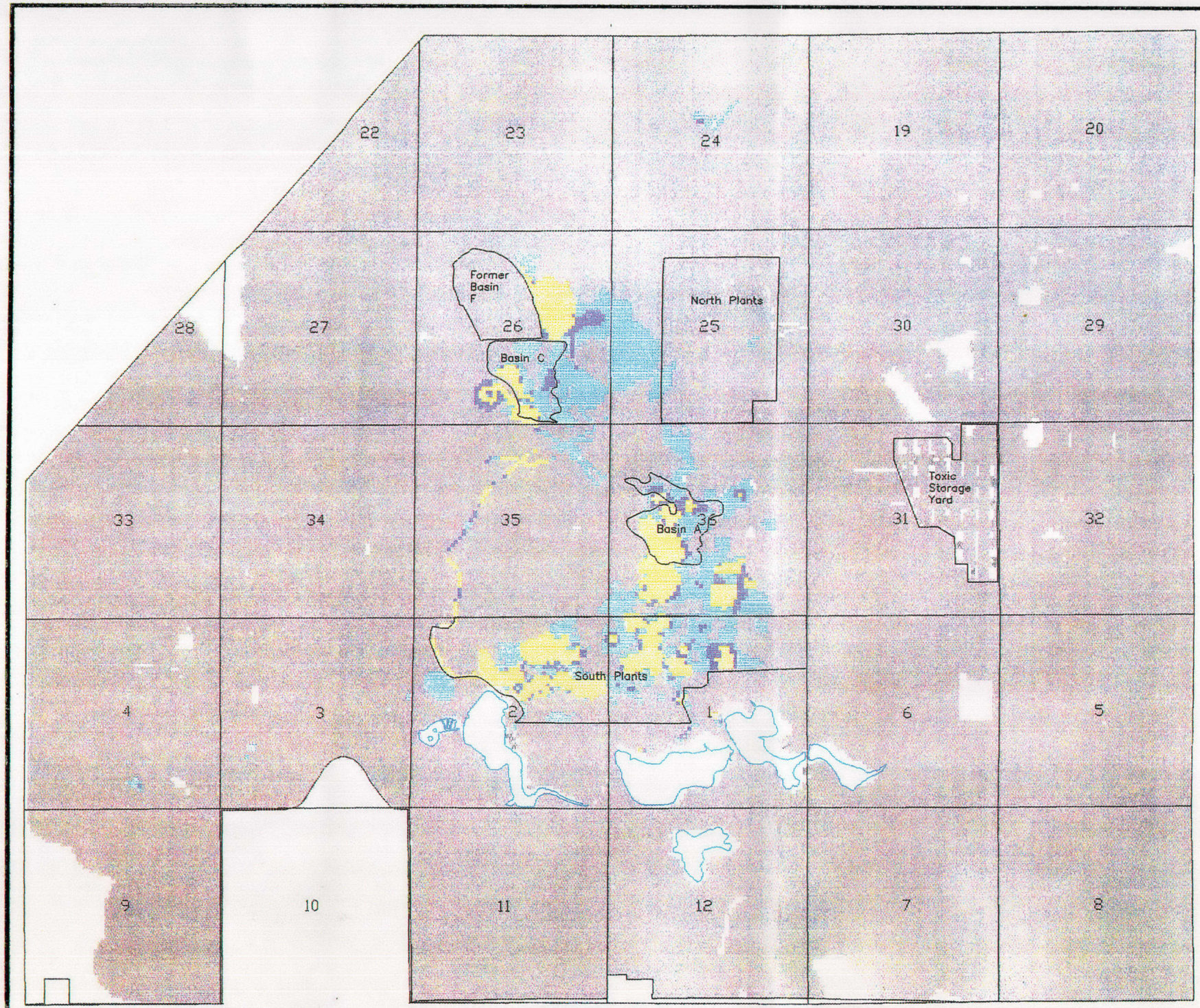
February 1994

Figure 4.5-2

Number of Trophic Boxes with Soil Hazard Indices Greater than 1.0 for Aldrin/Dieldrin, DDT/DDE, and Endrin Combined Based on the Shell Approach

Rocky Mountain Arsenal

Prepared by: ENSERCH Environmental Corp




## Legend

Soil Concentrations

- 0 - 1 ppm
- 1 - 5 ppm
- 5 - 10 ppm
- > 10 ppm

Lake

31 Section Number

Section Line

N

0 2000 4000

Scale in Feet

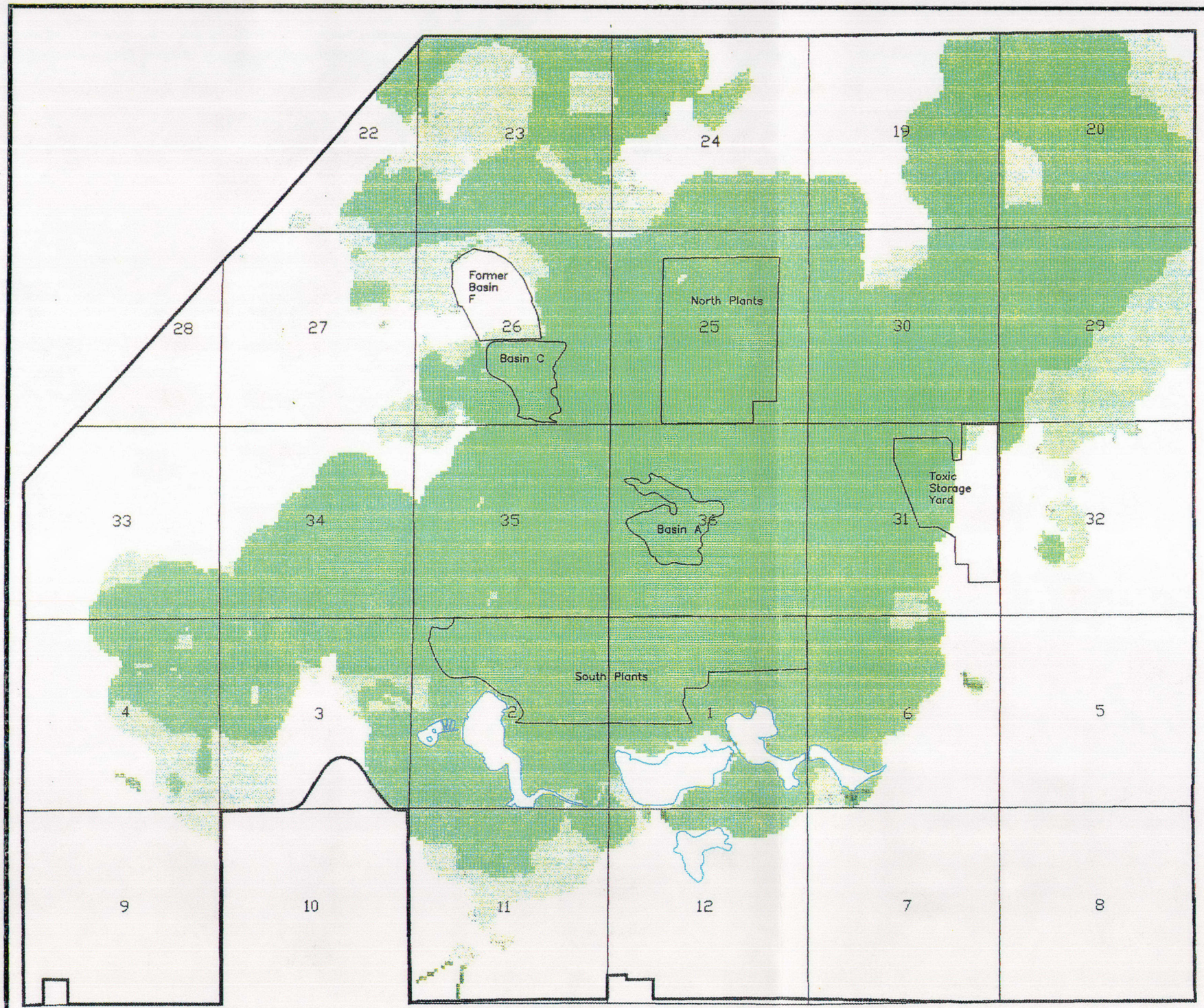
Prepared for:
U.S. Army Program Manager
for Rocky Mountain Arsenal

February 1994

Figure 4.5-3

Aldrin/Dieldrin Soil Concentrations

Rocky Mountain Arsenal
Prepared by: ENSERCH Environmental Corp




## Legend

- 1 trophic box with HI > 1
- 2-4 trophic boxes with HI > 1
- 5-7 trophic boxes with HI > 1
- Lake
- 31 Section Number
- Section Line

N

0 2000 4000

Scale in Feet

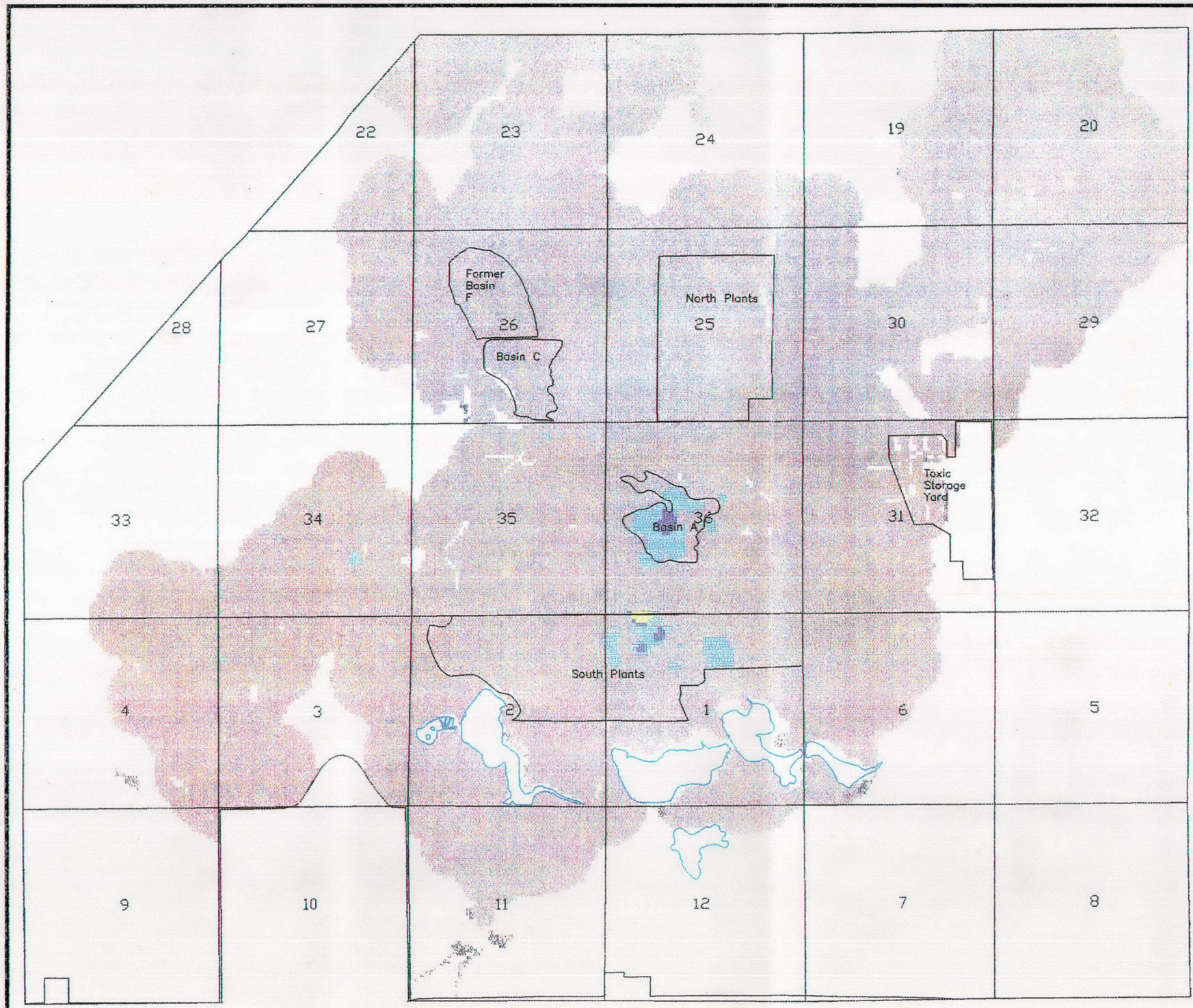
Prepared for:

U.S. Army Program Manager
for Rocky Mountain Arsenal

February 1994

Figure 4.5-4
Number of Trophic Boxes with Soil Hazard Indices Greater than 1.0 for Arsenic, Mercury, Cadmium, and Copper Combined Based on the Army Approach

Rocky Mountain Arsenal
Prepared by: ENSERCH Environmental Corp




## Legend

Soil Concentrations

- 0 – 1 ppm
- 1 – 10 ppm
- 10 – 100 ppm
- > 100 ppm

Background Mercury = 0.10 ppm

- Lake
- 31 Section Number
- Section Line

N

0 2000 4000

Scale in Feet

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

February 1994

Figure 4.5–5

Mercury Soil Concentrations
(Including Background)

Rocky Mountain Arsenal
Prepared by: ENSERCH Environmental Corp

Former Basin F

Basin C

North Plants

Basin A

Toxic Storage Yard

South Plants

22 23 24 19 20

28 27 26 25 30 29

33 34 35 36 31 32

4 3 2 1 6 5

9 10 11 12 7 8

## Legend

1 trophic box with HI > 1

2-4 trophic boxes with HI > 1

5-7 trophic boxes with HI > 1

Lake

31 Section Number

Section Line

N

0 2000 4000

Scale in Feet

Prepared for:

U.S. Army Program Manager
for Rocky Mountain Arsenal

February 1994

Figure 4.5-6

Number of Trophic Boxes with Soil Hazard Indices Greater than 1.0 for Arsenic, Mercury, Cadmium, and Copper Concentrations Above Indicator Levels Based on the Shell Approach

Rocky Mountain Arsenal

Prepared by: ENSERCH Environmental Corp


## Legend

Army and EPA Approaches

Army and Shell and EPA Approaches

Colors show which approaches result in HI > 1 in the areas indicated

Lake

31 Section Number

Section Line

N

0 2000 4000

Scale in Feet

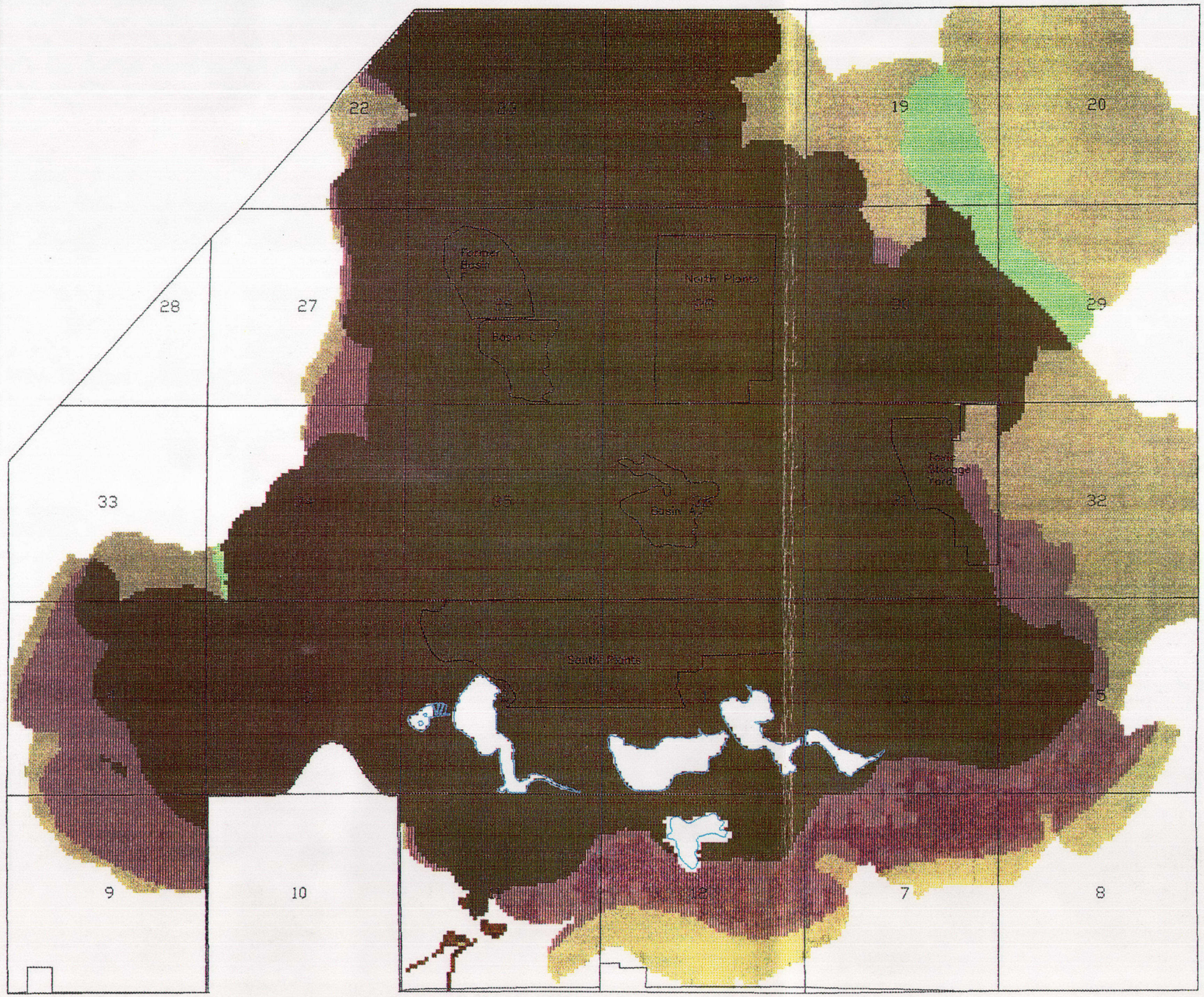
Prepared for:
U.S. Army Program Manager
for Rocky Mountain Arsenal

February 1994

Figure 4.5–7

Hazard Index Map (HI>1) for the Bald Eagle Trophic Box Based on the Army, EPA, and Shell Approaches

Rocky Mountain Arsenal
Prepared by: ENSERCH Environmental Corp




## Legend

- EPA Approach
- Army and EPA Approaches
- Shell and EPA Approaches
- Army and Shell and EPA Approaches

Colors show which approaches result in HI > 1 in the areas indicated

Lake

31 Section Number

Section Line

-N-

0 2000 4000

Scale in Feet

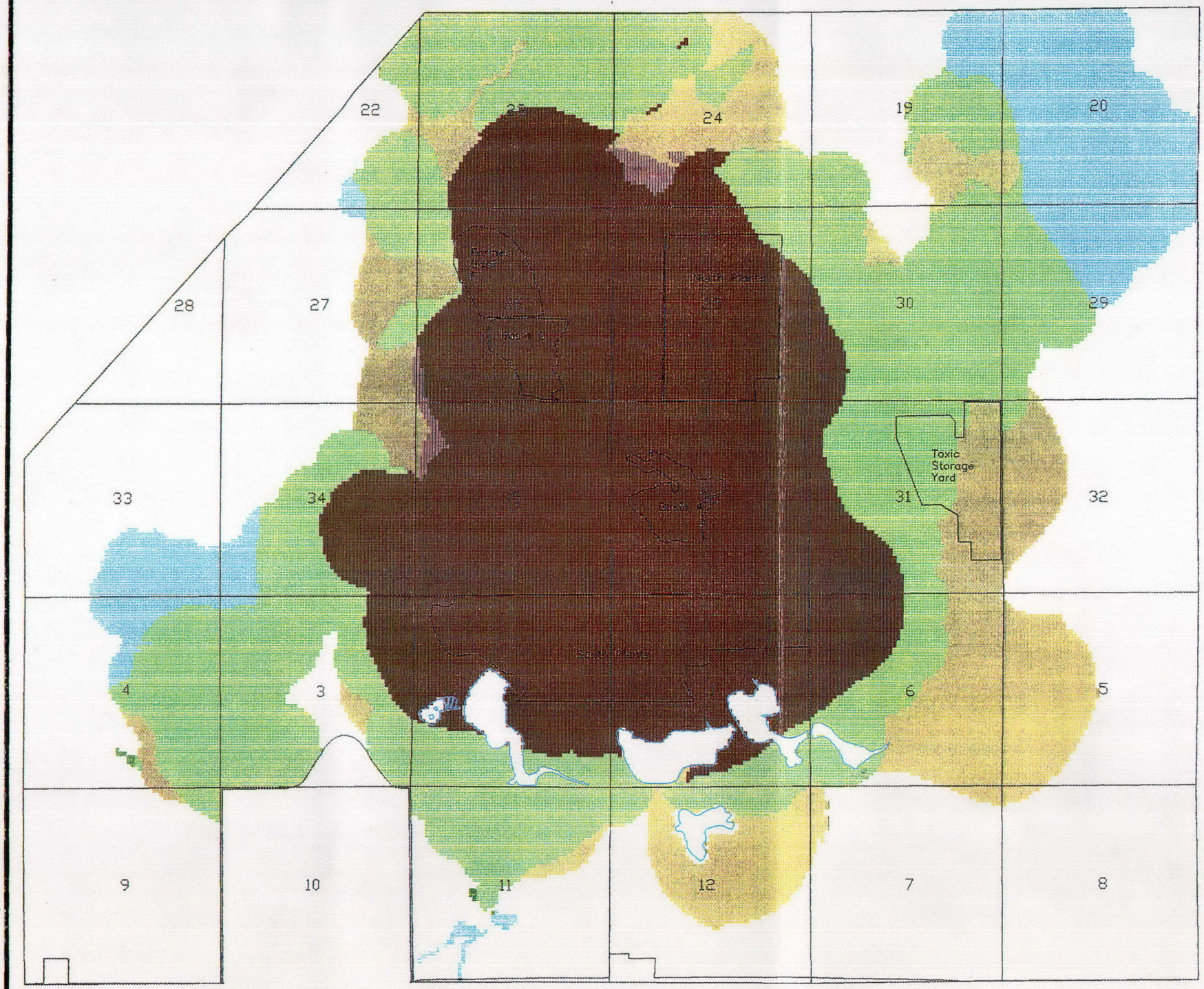
Prepared for:
U.S. Army Program Manager
for Rocky Mountain Arsenal

February 1994

Figure 4.5-8

Hazard Index Map (HI>1) for the Great Horned Owl Trophic Box Based on the Army, EPA, and Shell Approaches

Rocky Mountain Arsenal
Prepared by: ENSERCH Environmental Corp




## Legend

- Army Approach
- EPA Approach
- Army and EPA Approaches
- Shell and EPA Approaches
- Army and Shell and EPA Approaches

Colors show which approaches result in HI > 1 in the areas indicated

Lake

31 Section Number

Section Line

N

0 2000 4000

Scale in Feet

Prepared for:

U.S. Army Program Manager
for Rocky Mountain Arsenal

February 1994

Figure 4.5–9

Hazard Index Map (HI>1) for the American Kestrel Trophic Box Based on the Army, EPA, and Shell Approaches

Rocky Mountain Arsenal
Prepared by: ENSERCH Environmental Corp

22 23 24 19 20

28 27 Former Basin F 26 Basin C North Plants 25 30 29

33 34 35 Basin A 36 31 Toxic Storage Yard 32

4 3 South Plants 6 5

9 10 11 12 7 8

## Legend

- EPA Approach
- Army and EPA Approaches
- Shell and EPA Approaches
- Army and Shell and EPA Approaches

Colors show which approaches result in HI > 1 in the areas indicated

Lake

31 Section Number

Section Line

N

0 2000 4000

Scale in Feet

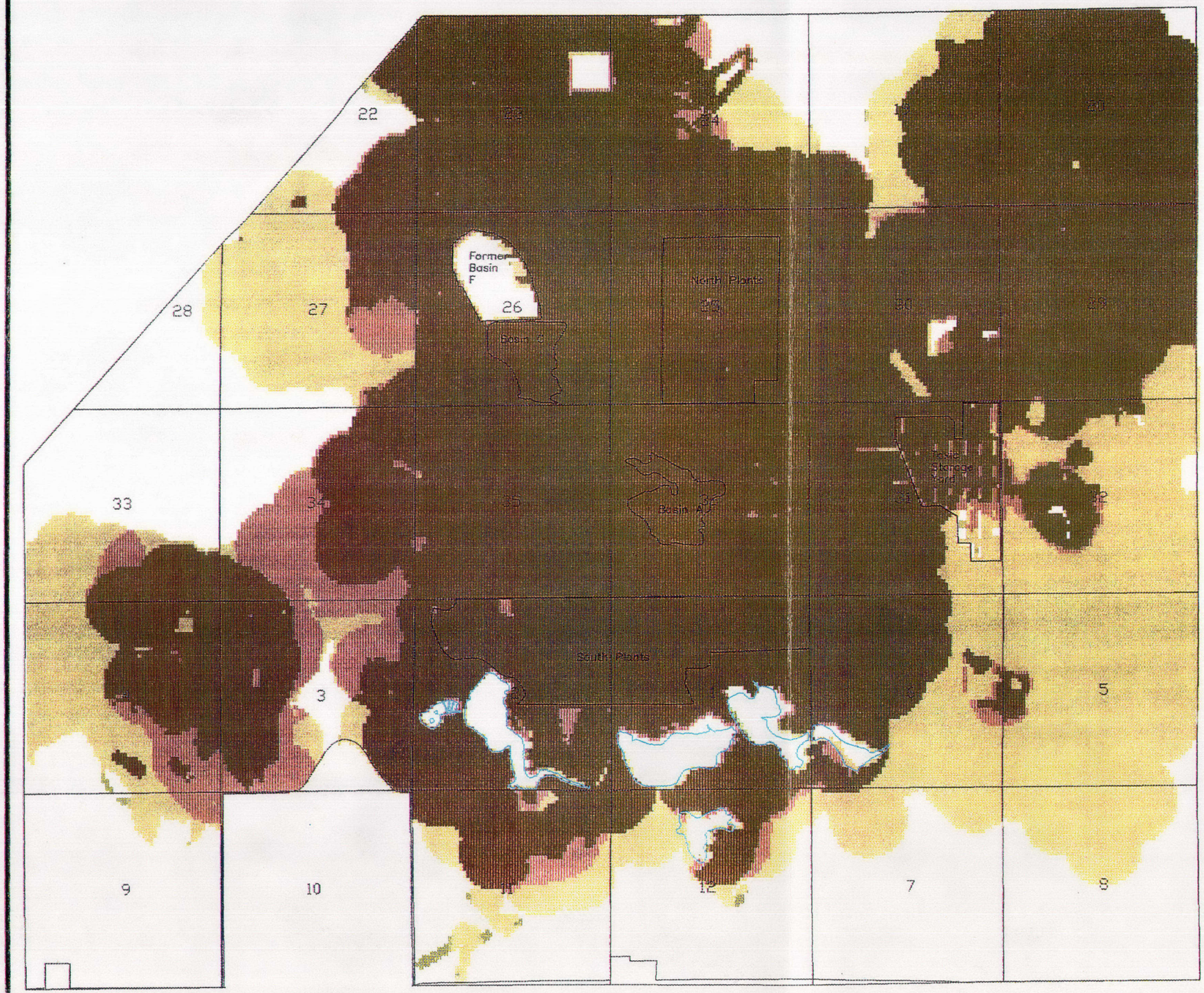
Prepared for:
U.S. Army Program Manager
for Rocky Mountain Arsenal

February 1994

Figure 4.5–10

Hazard Index Map (HI>1) for the Shorebird Trophic Box Based on the Army, EPA, and Shell Approaches

Rocky Mountain Arsenal
Prepared by: ENSERCH Environmental Corp




## Legend

- EPA Approach
- Army and EPA Approaches
- Shell and EPA Approaches
- Army and Shell and EPA Approaches

Colors show which approaches result in HI > 1 in the areas indicated

Lake

31 Section Number

Section Line

N

0 2000 4000

Scale in Feet

Prepared for:

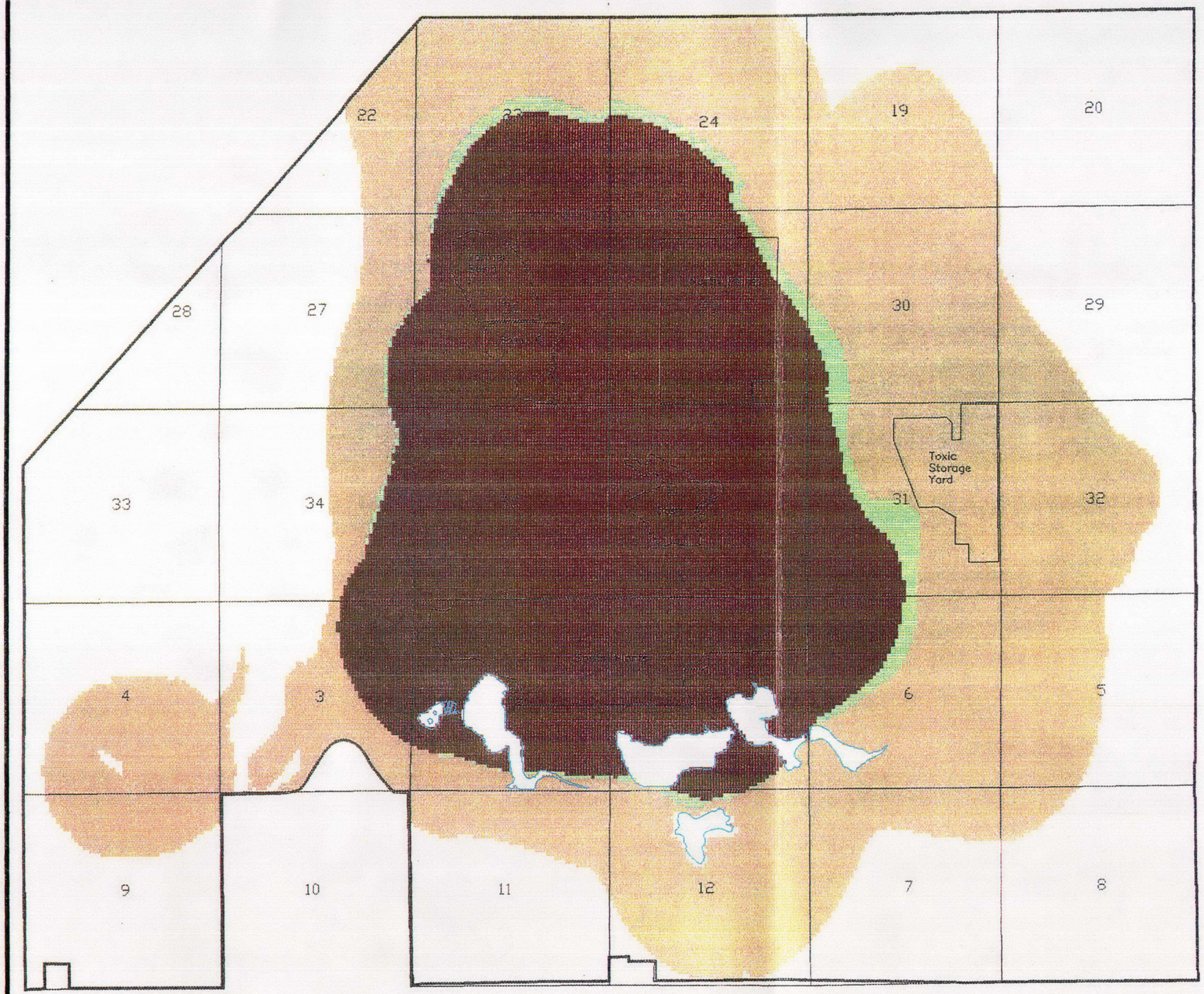
U.S. Army Program Manager
for Rocky Mountain Arsenal

February 1994

Figure 4.5–11

Hazard Index Map (HI>1) for the Medium Mammal Trophic Box Based on the Army, EPA, and Shell Approaches

Rocky Mountain Arsenal
Prepared by: ENSERCH Environmental Corp




## Legend

- EPA Approach
- Army and EPA Approaches
- Army and Shell and EPA Approaches

Colors show which approaches result in HQ > 1 in the areas indicated

- Lake
- 31 Section Number
- Section Line

N

0 2000 4000

Scale in Feet

Prepared for:

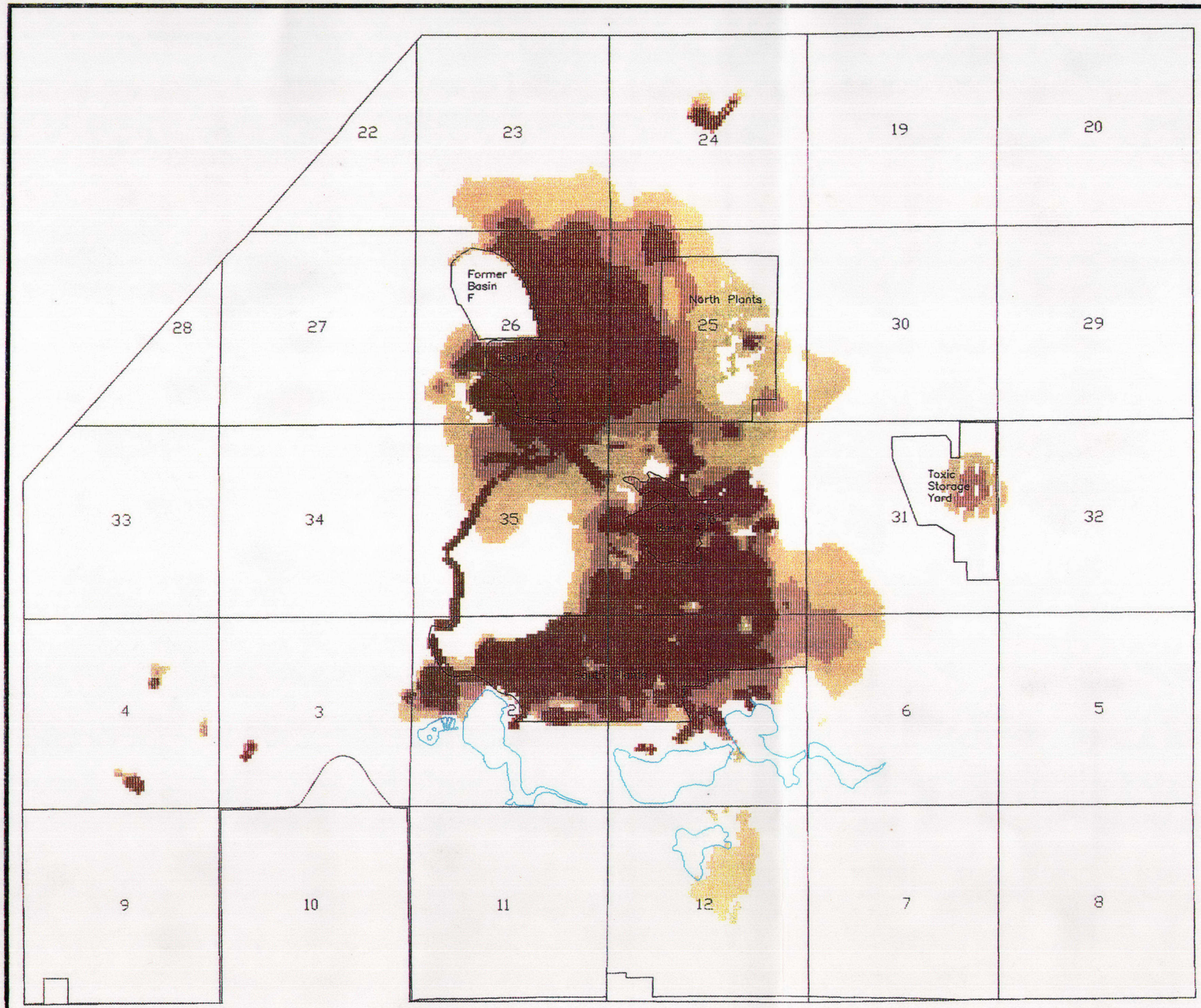
U.S. Army Program Manager
for Rocky Mountain Arsenal

February 1994

Figure 4.5–12

Aldrin/Dieldrin Hazard Quotient Map (HQ>1) for the Great Horned Owl Trophic Box Based on the Army, EPA, and Shell Approaches

Rocky Mountain Arsenal
Prepared by: ENSERCH Environmental Corp




## Legend

- EPA Approach
- Shell and EPA Approaches
- Army and Shell and EPA Approaches

Colors show which approaches result in HQ > 1 in the areas indicated

Lake

31 Section Number

Section Line

N

0 2000 4000

Scale in Feet

Prepared for:

U.S. Army Program Manager
for Rocky Mountain Arsenal

February 1994

Figure 4.5–13

Aldrin/Dieldrin Hazard Quotient Map (HQ>1) for the Medium Mammal Trophic Box Based on the Army, EPA, and Shell Approaches

Rocky Mountain Arsenal
Prepared by: ENSERCH Environmental Corp

# 5.0 FACTORS INFLUENCING THE CHARACTERIZATION OF POTENTIAL RISKS

This section discusses and summarizes factors influencing the characterization of risks at RMA, including uncertainty about human exposure scenarios and toxicity estimates (Section 5.1); uncertainty about toxicological threshold values, ecological exposure, biomagnification, and risk estimates (Section 5.2); and limitations associated with the RMA chemical database (Section 5.3). In general, the consideration of uncertainty provided insights that lent clarity and defensibility to the analyses. Appendix E provides additional details on the assumptions, limitations, and uncertainties identified in this section.

## 5.1 UNCERTAINTIES ASSOCIATED WITH HUMAN HEALTH RISK CHARACTERIZATION

The primary sources of uncertainty for human health are related to exposure scenarios and toxicity estimates. These sources are discussed separately below.

### 5.1.1 Human Health Exposure Scenario Uncertainty

Several potential uncertainties are associated with the human health exposure scenarios evaluated in the IEA/RC. The following paragraphs summarize these uncertainties, which are related to land-use scenarios, target receptors, exposure pathways, exposure parameters, BCRL replacement, spatial exposure patterns, additivity assumptions regarding risk estimates, the soil vapor inhalation model, and the Latin Hypercube sampling protocol.

There are considerable uncertainties regarding the likelihood of most of the land-use scenarios and target receptors projected to occur under a future development scenario at RMA. Therefore, risks that were quantified for future land-use scenarios, some of which may not occur on RMA after final land-use decisions are made, may be overstated. If future land uses eliminate exposure for some receptors, then no risks associated with those particular scenarios would be expected. For example, given the goal stated in the FFA to preserve large portions of RMA for open space uses, including the passage of the Refuge Act, the risks associated with the commercial and

industrial workers under an economic development option may not occur.

Uncertainties also result because several exposure pathways were not considered in the IEA/RC given restrictions specified in the FFA. Pathways that were excluded from the quantitative risk evaluation on this basis include consumptive pathways (meat, dairy, vegetable, and fish ingestion) and groundwater ingestion/vapor inhalation pathways. Exclusion of these pathways is expected to have no impact on estimated risks, assuming that the restrictions specified in the FFA continue to apply. Little impact on the risk projections is expected as a result of the exclusion of a dermal contact pathway for metals due to low absorption. Similarly, the impacts on site risks associated with the lack of evaluation of dermal contact with surface water is not expected to be significant given that previous evaluations of this exposure pathway indicated negligible risk (EBASCO 1990).

Uncertainties are also associated with exposure parameters including soil ingestion rates, dermal absorption factors, and the time-dependent variables (TDVs). A scenario for a child with pica (a child who ingests atypical substances such as lead paint or soils) was not quantitatively evaluated in the assessment (and pica data were not used to develop soil ingestion rate distributions); hence, risks to children exhibiting pica behavior may be underestimated. However, conservative assumptions were generally made in assigning values or distributions to soil ingestion rates, dermal absorption factors, and TDVs. Ages and activities associated with the visitor population under the open space land-use option were characterized using available empirical data and best professional judgment. Although survey data were used to characterize time and activity patterns for the refuge subpopulations and biological worker population to improve the confidence in the analysis, the representativeness of the resulting distributions for current or future exposed populations at RMA remains uncertain. The data sets compiled for these populations or subpopulations may underrepresent exposures for some portion of the future RMA population and overrepresent exposures for some other portion. However, it is expected that most parameter distributions and the resultant PPLVs were estimated with conservative

biases and are, therefore, likely to overestimate site risks.

Uncertainty is associated with the presence of BCRL values in the data used for the calculation of $C_{rep}$, the arithmetic mean of a site's soil concentration data. This uncertainty is associated with potential errors in the replacement values assigned to BCRL soil concentration data by the robust estimation method. However, uncertainty due to treatment of BCRL data is expected to have little impact on $C_{rep}$ because the upper and lower confidence limits estimated for $C_{rep}$ (summarized in Appendix Section B.4.5 for the driver chemicals) reflect the variability associated with this mean value, and this variability is large compared to the uncertainty associated with BCRL replacement.

The probabilistic model parameters used to quantify human health risks are, by definition, variable and the true values of deterministic parameters can only be estimated. The variation may be spatial, temporal, physiological, or may occur due to the dependence of a parameter on other unknown and variable factors such as environmental conditions (e.g., temperature) or behavior (e.g., activity patterns). The probabilistic distributions or fixed values assigned to parameters used in calculating PPLVs are influenced by several types of uncertainty including data representation error, extrapolation error, uncertainty due to small data sets, and uncertainty associated with parameter correlation. Appendix Section E.7 details these uncertainties.

Additional uncertainties associated with human exposure settings stem from the following factors and assumptions: (1) that exposure conditions are constant over time for a given individual; (2) that simplifying assumptions are used in the PPLV model construction; (3) that intentional and unintentional biases are incorporated in the models (e.g., the use of conservative assumptions for those parameters with limited data); (4) that assumptions are used in the open and enclosed space vapor models; and (5) that additivity assumptions for both carcinogens and noncarcinogens are used, especially for those chemicals with different modes of toxicity and different weights of evidence for potential carcinogenicity.

Uncertainty is also associated with the Latin Hypercube simulation sample size since increasing sample sizes narrows confidence intervals (i.e., the PPLV sample values chosen for the upper and lower bounds are closer together for larger sample sizes). The impact of simulation sample size on confidence interval width is described in Appendix Section E.8.2. The analysis found that increasing sample size above 100 did not reduce confidence interval widths significantly and that larger sample sizes were not justified in light of the resulting impact on simulation time. HHRC simulations were performed using sample sizes of 100, for which a single run of the model took between 3 and 4 hours. Doubling the sample size would double this simulation time. Increasing the sample size by much more than a factor of 2 would result in a greater than proportional slowdown because of computer memory limitations.

### 5.1.2 Human Toxicity Estimate Uncertainties

The uncertainties in toxicity estimates for human health vary depending upon the endpoint considered (i.e., whether the COC is a carcinogen or a noncarcinogen). A significant degree of uncertainty is also associated with the toxicity endpoint values used to estimate risks to human health. Uncertainties are associated with the use of animal data in the absence of human toxicity data, variations within animal test species, extrapolation from high experimental doses to low doses, extrapolation from oral to other routes of exposure, and the use of upper-bound toxicity values for carcinogenic COCs.

For carcinogens, the level of confidence is reflected in the weight-of-evidence carcinogenicity classification. For noncarcinogens, the level of confidence in the reference toxicity study is reflected in the magnitude of the uncertainty factors (UFs) applied to calculate the RfDs. The following paragraphs summarize the uncertainties associated with the toxicity estimates used to calculate risks for the carcinogenic and noncarcinogenic COCs, which are discussed in detail in Appendix Section E.6.

5.1.2.1 Carcinogens

As indicated in the EPA Guidelines for Carcinogen Risk Assessment (1986b), the CSFs generated from the linearized multistage extrapolation procedure lead to what is considered a "plausible upper limit to the risk that is consistent with some proposed mechanisms of carcinogenesis. Such an estimate, however, does not necessarily give a realistic prediction of the [cancer] risk. The true value of the risk is unknown, and may be as low as zero." With the exception of arsenic and benzene, whose slope factors were generated using a different extrapolation model, all of the slope factors used to estimate PPLVs for carcinogenic COCs and potential cancer risks were derived using the linearized multistage dose-response model and thus constitute upper-bound estimates. Therefore, cancer risks associated with these chemicals are not likely to be underestimated, but may be substantially overestimated.

Additionally, with the exception of arsenic, benzene, and chromium, the weight-of-evidence classifications for carcinogenic COCs are predicated on animal data that may not be representative of the potential carcinogenic response induced by the chemical in humans. For example, the weight-of-evidence assessments of aldrin and dieldrin, which are significant contributors to total cancer risks estimated for both the site- and boring-specific risk evaluations, are complicated by several areas of uncertainty. As discussed in Appendix Section E.6.1.1, these uncertainties, which stem from the lack of evidence of carcinogenicity in animal species other than mice, are further complicated by the lack of compelling evidence reported in more recent epidemiological studies. If aldrin and dieldrin were assumed not to be human carcinogens, total estimated cancer risks would be reduced significantly at selected sites (see Figures 3.2-11 and 3.2-12). Appendix Section E.6.1 provides a more detailed discussion of the weight-of-evidence uncertainties associated with carcinogenic COCs.

5.1.2.2 Noncarcinogens

A potentially significant degree of uncertainty is associated with the EPA noncarcinogenic dose-response values (i.e., RfDs) used in this assessment. EPA has acknowledged that these

uncertainties generally imply a range of variability of at least one order of magnitude. Appendix Section B.1 and Appendix Section E.6.2 provide a review of the critical effects for each RfD, the magnitude of the UFs incorporated in their derivation, and the level of confidence in the RfD so derived.

Uncertainty was considered high for those COCs for which there is low confidence in the established RfD, regardless of the magnitude of the assigned UF. The COCs meeting these criteria include chlordane, chromium, dieldrin, and HCCPD. Uncertainty was also considered high for those COCs for which verified EPA RfDs or Health Effects Assessment Summary Tables values were not available. Chemicals in this category include isodrin and lead.

Uncertainty was considered moderate for those COCs for which there is a medium level of confidence in the established RfD and UFs ranging between 100 and 1,000. The COCs in this category include aldrin, carbon tetrachloride, chlorobenzene, chloroform, DDT, 1,2-dichloroethylene, endrin, methylene chloride, tetrachloroethylene, and toluene.

Uncertainty was considered low for those COCs for which there is high confidence in the established RfD and UFs ranging between 1 and 100. Only cadmium met these criteria. The uncertainties associated with the remaining COCs could not be determined because no evaluation of confidence in the database, RfD, or critical study was available.

## 5.2 UNCERTAINTIES ASSOCIATED WITH THE ECOLOGICAL RISK CHARACTERIZATION

The ERC for RMA utilized a number of unique concepts and methodologies, including spatially distributed exposure models and probabilistic evaluation of spatially distributed population mean tissue concentrations and doses. The design of the ERC, which incorporated spatial information and accounted for uncertainty, was necessitated by the difficulty of adequately sampling an area as large and diverse as RMA, particularly for the large number of trophic boxes evaluated. This

section identifies and describes how sources of uncertainty were treated in the ERC. A more detailed discussion of uncertainty can be found in Appendix E.

The ERC was based on a comparison of exposure measurement endpoints (average tissue concentrations and doses) to toxicological threshold values (MATCs and TRVs). This comparison yielded HQs and HIs, which provide a spatially explicit measure of average risks to biota populations that is based on toxicological endpoints. The treatment of uncertainty about MATCs and TRVs is discussed in Section 5.2.1. Uncertainty in terrestrial risk estimates is discussed in Section 5.2.2, and uncertainties associated with the aquatic risk estimates are discussed in Section 5.2.3. Finally, Section 5.2.4 summarizes the rationale for the use of uncertain ecological measurement endpoints to enhance the ERC.

### 5.2.1 Uncertainty Associated with MATCs and TRVs Used in the Ecological Risk Characterization

Uncertainty about toxicity was treated by applying UFs to account for various extrapolations of the literature toxicity values to conditions at RMA. First, literature-based estimates were made of tissue concentrations and doses likely to be without an appreciable risk of deleterious effects. These literature-based ("pre-UF") MATCs and TRVs were divided by the UF (numbers greater than one; greater magnitude corresponds to greater uncertainty) to yield "final" MATCs and TRVs.

The UFs were developed to account for uncertainty due to intertaxon variability in toxicological responses, extrapolation from experimental study duration to chronic exposure, extrapolation from experimental study's toxicity endpoint to a no observed adverse effects level (NOAEL), and eight additional modifying factors that are identified in Appendix Section C.2.4.2.5. The degree to which the UFs under- or overcompensated for these factors has not been quantified, nor were other sources of uncertainty (e.g., species differences in responses to toxicity within the trophic box and differences between the test individuals and test conditions among studies and from

those at RMA) quantified. The protocol for determining UFs is presented in Appendix Table C.2-11.

Further discussion of the MATCs and TRVs used in the ERC is provided in Section 4.1.1.1 and Appendix Sections C.2.4 and C.2.6. The final MATCs and TRVs are reported together with an itemized accounting for the UFs in Appendix Tables C.2-12 and C.2-15. The pre-UF MATCs and TRVs and citations of studies used to define the pre-UF values are reported in Tables C.2-10 and C.2-14.

### 5.2.2 Uncertainty Associated with Terrestrial Risk Estimates

For the terrestrial portion of the RMA food web, average tissue concentrations and doses were estimated from estimated exposure area soil concentrations (ESC values) and BMFs. The uncertainty associated with these calculations and with overall estimates of terrestrial risk is discussed below.

#### 5.2.2.1 Exposure Concentration Uncertainty

As discussed in detail in Appendix Sections E.9.4 and E.12.4, the characterization and estimation of exposure soil concentration is the most important source of parameter uncertainty in estimating terrestrial risks for the ERC. The discussion of uncertainty about exposure soil concentrations utilizes the following formal definitions to relate the estimate of the measurable quantity ESC to theoretical quantities describing true soil exposure as they indicate assumptions and uncertainties in using ESC to estimate risk:

- Exposure activity (theoretical)—The behaviors of an individual pertaining to its exposure to soil contamination (e.g., feeding, digging, lifetime mobility patterns).

- Exposure soil concentration (theoretical)—The concentrations in soil that are available to and accessed by an individual during exposure activity.

- Exposure area soil concentration (ESC)—The average soil concentration in a specified depth profile within a circular species-specific exposure area. The exposure area is representative of typical exposure activity for a given species as defined in the literature.

- Estimated exposure area soil concentration (<ESC>)—The statistical estimate of ESC derived from the RMA soil concentration database.

Tissue concentration, dose, and risk, depend on the theoretical exposure soil concentration defined above, a quantity which, for all practical purposes, cannot be known because of the difficulty of accurately knowing the average exposure activity of each RMA population. Therefore, exposure soil concentration is represented by a simple areal average referred to as exposure area soil concentration, which is in turn estimated across RMA.

The definitions provided above help to illustrate that there are two types of uncertainty in estimates of exposure soil concentration. "Representation uncertainty" refers to the uncertainty in representing the exposure soil concentration by ESC. "Estimation uncertainty" refers to the uncertainty in estimating the ESC value for a given location when that value is used to calculate risk or, when paired with an actual sample individual, when that value is used to estimate BMF. "Overall ESC uncertainty" refers to the net difference between the true exposure concentration and <ESC>.

Representation uncertainty explains the difference between true exposure soil concentration for an individual and the ESC for a typical (average) individual. This uncertainty includes the following components: (1) differences between the actual available exposure soil concentration, which is affected by contaminant availability and individual exposure activities, and the measured soil concentrations in the sample units (e.g., 1-ft bores, 0- to 2-inch surficial samples); and (2) differences between the true spatial/temporal pattern for a given individual and the assumed mean spatial/temporal pattern of exposure. Unfortunately, representation uncertainty is for all practical

purposes unquantifiable and irreducible because the detailed information on individual organisms (and their prey) required for its calculation cannot be reasonably obtained.

Estimation uncertainty is thought to be a smaller contributor to overall ESC uncertainty than representation uncertainty. Estimation uncertainty explains differences between the (unknown) true ESC and its estimated value, <ESC>. Estimation uncertainty includes the following components: (1) the estimation of the spatial distributions of RMA soil contaminants; (2) the replacement of BCRL concentration data; (3) differences between the defined exposure range radius for an average individual of a species and the most appropriate mean radius for that species, individual, and location; and (4) differences between the collection location for an individual tissue sample and the true central location associated with this individual, i.e., the location that implies the true exposure area soil concentration. The latter uncertainty pertains to <ESC> when it is used to estimate BMF, but does not influence the ESC estimate for a hypothetical average individual at a specified location at RMA (i.e., it influences the BMF, but not the <ESC>, in the equation used to predict tissue concentration in an average individual: TC = <ESC> * BMF).

Even complete elimination of estimation uncertainty (i.e., a complete, error-free soil concentration map of RMA) would leave uncertainty about biota exposure to the known pattern of soil concentrations unresolved. This implies that any soil-based ecological risk assessment for RMA can at best provide a semiempirical characterization of the spatial variation in average risk. This is the basis for the assertion that <ESC> uncertainty is the most important source of uncertainty in the ERC. The semiempirical nature of the ERC implies that <ESC> maps and BMFs should be judged by how well the observed tissue concentrations at RMA can be predicted from the ERC model, keeping in mind irreducible representation uncertainty and uncertainty in the individual tissue samples as estimators of the tissue concentration in an average individual.

5.2.2.2 BMF Uncertainty

BMFs are the modeling coefficients used in the terrestrial risk characterization to represent the average ratio of tissue concentration to <ESC>. Terrestrial risk was estimated based on three sets of BMF estimates reflecting approaches maintained by the Army, EPA, and Shell. Uncertainties in estimating BMFs are discussed in detail in Appendix E.12 and summarized below.

The three BMFs depended heavily or entirely on the RMA field data, i.e., the measured tissue concentrations and <ESC>s for each biota sample. The primary sources of uncertainty associated with the tissue concentrations were low sample sizes (for some species) and the interpretation of BCRL data. As discussed above, <ESC> uncertainty arises from many sources and is believed to be the most important uncertainty in the risk estimates since it helped diminish the correlation between the measured tissue concentrations and their associated <ESC> values. The low correlations led to varied statistical approaches for deriving BMF estimates from field data.

5.2.2.3 Risk Estimate Uncertainty

Terrestrial HQs and HIs were calculated by comparing tissue concentrations or doses estimated by the model to MATC or TRV toxicological threshold values, respectively. Uncertainties in terrestrial HQs and HIs arise from the uncertainties in MATC, TRV, <ESC>, and BMF described above.

Because the estimation of BMF is dependent on the field data, <ESC> is used in both the numerator and denominator of the risk equation. Therefore, some of the potential statistical bias in <ESC> may cancel so that it does not effect the final risk estimate. For example, if assumptions regarding exposure range for a given species are such that <ESC> tends to be biased upward by a factor of 2, then the BMF estimate should be biased downward by a factor of approximately 2, so that the estimate of tissue concentration as BMF*<ESC> does not change. Therefore, the error in predicting risk can be assessed by comparing predicted tissue concentrations in a given area to measured tissue concentrations in that area. Although this

comparison provides little information on the accuracy of the <ESC> and BMF components (one may be biased high while the other is biased low), it indicates the accuracy of their product as an estimator of average tissue concentrations.

Irreducible sources of uncertainty limit the quality of the fit between model predictions and field observations of biota tissue concentrations. Specifically these irreducible sources of uncertainty are the following: (1) uncertainty about the deviation of the measured individual tissue concentration at a particular location from the theoretical true population mean tissue concentration for that point, and (2) representation uncertainty in estimating the theoretical true exposure soil concentration at each point on RMA. Despite these irreducible uncertainties, the appropriate matching of semiempirical BMFs and <ESC>s can eliminate some of the potential statistical bias in risk estimates based on the ERC model.

5.2.3 Uncertainty Associated with Aquatic Risk Estimates

As described in Section 4, estimates were derived of potential risk to water bird, shorebird, great blue heron, and bald eagle from exposure to bioaccumulative COCs through the aquatic portion of the RMA food web. These estimates were based on tissue concentration data for the trophic box itself (in the case of water bird and shorebird) or for the prey trophic boxes (in the case of great blue heron and bald eagle). The tissue concentration data were used to estimate population mean tissue concentrations and doses for the bird trophic boxes. Dose estimates for water bird and shorebird were derived from the trophic box tissue data, feeding rate coefficient (R), and bioaccumulation factors (BAFs) as described in Section 4.4.2. Tissue concentration and dose estimates for great blue heron and bald eagle were calculated from the prey tissue concentration data, dietary fractions (FR), BAFs (for tissue concentration estimation), and R (for dose estimation), described in Section 4.3.1. The development of and uncertainty in consensus parameter values (BAF, FR, R, MATC, and TRV) is discussed extensively elsewhere in this report (particularly in Appendix Section C.2). Uncertainty in the average prey and predator tissue concentration estimates used to characterize aquatic risk may result from unrepresentative

sampling of the RMA lakes, errors in sample means, correspondence of sampled biota of the true prey and predator populations they are meant to represent, and errors in procedures used to replace missing data (these methods are described in Appendix Section C.1.4.1). The source of uncertainty was investigated in the example described below.

Lake sampling methods for the ERC field sampling program are described in Appendix Section C.4.1.2.1. Additional data were collected during the Biota RI and Biota CMP. In all three of these programs, sampling stations were set up at dispersed locations that provided good coverage of the lake areas. In the ERC field sampling program, the five established sampling stations in Lake Ladora and Lower Derby Lake (Figures C.4-2 and C.4-3) were sampled. Fish species collected in the ERC field sampling program were bluegill, largemouth bass, and northern pike. In addition to these fish species, the Biota CMP also sampled black bullhead, brown bullhead, and channel catfish; most of these species were also sampled by the Biota RI. These species were assumed to provide representative tissue concentrations for estimating contaminant dose from small and large fish to great blue heron.

A sensitivity analysis of the estimates of potential risk to uncertainty about prey tissue concentrations was performed for the great blue heron using data from the ERC field sampling program. The results of the analysis of errors in sample means indicate that at least for heron, which has an aquatic HI of 20, the prediction of HI exceedence is insensitive to uncertainty about the sample average tissue concentration in its prey. Aldrin/dieldrin standard errors are roughly 20 percent of the mean and mercury standard errors approximately 10 percent of the mean for both small and large fish. Small and large fish comprise an estimated 78 percent of the heron's diet, and aldrin/dieldrin and mercury HQs comprise together more than 90 percent of the heron's aquatic HI.

Sensitivity analyses were also performed to determine the impact on great blue heron risk estimates of procedures used to estimate missing water OCP concentrations, aquatic invertebrate DDT/DDE concentrations, and amphibian DDT/DDE concentrations. Direct ingestion of water comprises about 7 percent by mass of the heron's diet. A range sensitivity analysis was performed, varying OCP water concentrations from zero to the CRL; heron risk predictions were not changed by this range of variation of the water concentrations. The analysis of risk sensitivity to DDT/DDE replacement values, reported in Appendix Table C.1-3, also show insensitivity to this source of uncertainty. The combined amphibian and aquatic invertebrate trophic boxes combined are estimated to contribute only 3 percent by mass to the great blue heron's diet.

### 5.2.4 Use of Uncertain Data on Ecological Status and Health

The data on ecological status and health used to evaluate ecological measurement endpoints (discussed in Appendix Section C.5) are also subject to uncertainty. In this context, uncertainty results from the short-term nature of many of the studies, estimation of parameters with precision that may not be statistically significant, and study designs that did not precisely correlate ecological parameters with contaminant concentrations or quantify all parameters that might have affected the ecological data. However, EPA (1993, pg. 8-17) concluded that the diversity of endpoints used at RMA, including tissue concentrations and measurements at a number of ecological levels "... provides a holistic examination of the ecosystem, lending greater confidence in risk estimates." In Appendix C.5, the studies that provide data for evaluation of the ecological endpoints are presented in the context of any limitations they may have. These uncertainties are discussed further in Appendix Section E.9.

## 5.3 LIMITATIONS ASSOCIATED WITH THE RMA CHEMICAL DATABASE

Several limitations are associated with the chemical database used to characterize potential risks to human and ecological receptors. These include limitations associated with soil sample collection, tentatively identified compounds, and Army agent contamination.

### 5.3.1 Soil Sample Collection

Soil boring placement was intentionally biased toward "designated sites," i.e., areas most likely to contain or concentrate contaminants (depressions, scarred areas, and similar features). This could lead to an overestimation of boring-specific risks in the HHRC (Figure 3.2-17). In addition, outside of samples taken at designated sites, samples taken from a near-surface depth interval were often composited with deeper soils. This underestimates near-surface contaminant concentrations if lower contaminant concentrations occur in the deeper interval. To compensate for this potential underestimation, detected concentrations in the composited samples were doubled in both the HHRC and ERC, effectively attributing all contamination detected to the near-surface depth interval. For the same reason, CRLs were also doubled for composited samples in the HHRC; this was not done in the ERC.

Other limitations associated with soil sample collection include the intentional avoidance of areas that were known to be highly contaminated and/or that presented special safety concerns. This could lead to an underestimation of human health risks, as is discussed in the qualitative risk assessment (Section 3.3). Because many highly contaminated areas had substantial physical disturbance (e.g., South Plants), exposure pathways and adverse effects of contamination were difficult to evaluate. The heterogeneity of the materials disposed at RMA adds additional uncertainty to inferences based on soil samples. In the ERC, which computes risks based on spatially averaged soil concentrations, risk estimation uncertainty due to uncertainty about the spatial distribution of contaminants is thought to be small compared to uncertainty about the actual spatial and temporal pattern of exposure to soil contaminants.

### 5.3.2 Tentatively Identified Compounds

Tentatively identified compounds were considered in the EA, but not in the risk characterization. The potential underestimation of risks due to the presence of these compounds is not thought to be significant in light of uncertainties associated with other aspects of the current evaluations.

### 5.3.3 Army Agent Sampling

The detections of Army chemical agent reported in the RMA database were not quantitatively evaluated in the IEA/RC because many of the detections were considered analytical artifacts. As discussed in Section 3.3, some sites with potential agent presence (e.g., secondary basins and ditches/drainage areas) were excluded from evaluation because the only indications of potential agent were detections of fluoroacetic acid. Subsequent analyses have ruled out fluoroacetic acid as a COC. Nonetheless, 23 sites are currently identified as areas of potential agent presence; consequently, risks for these sites may be underestimated in the IEA/RC.