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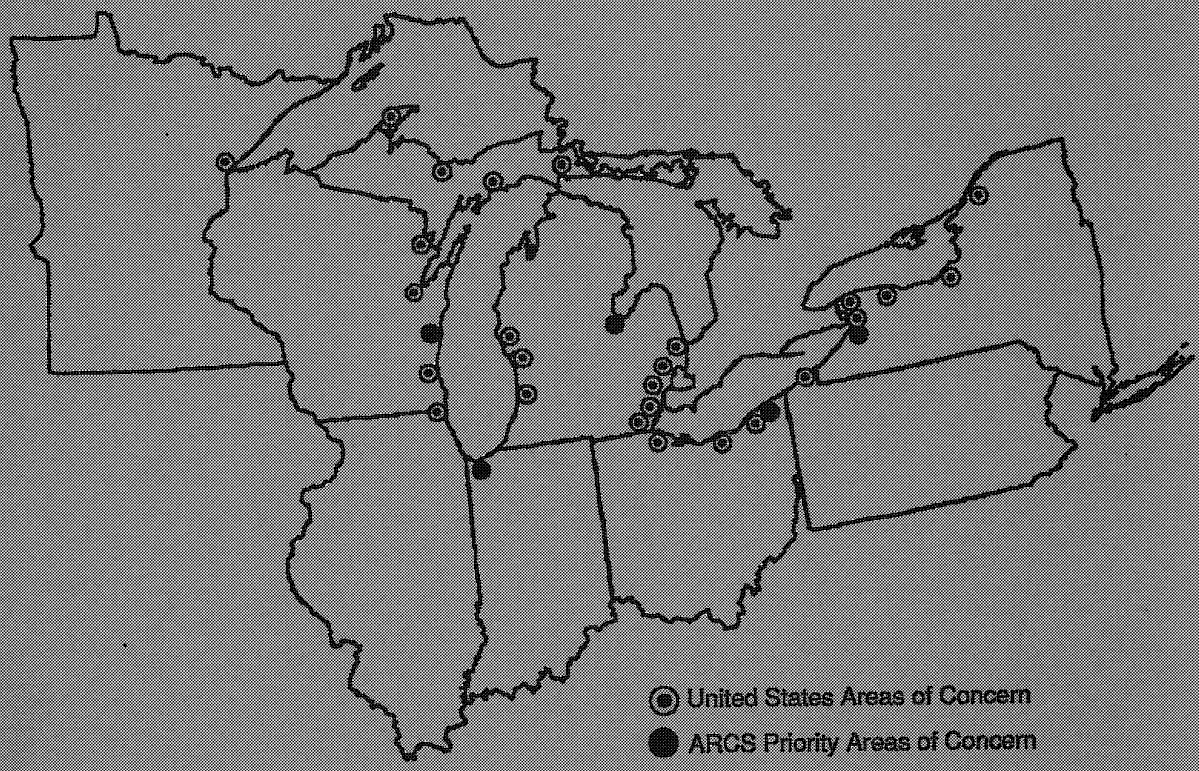
Comparative Human Health and Wildlife Risk Assessment: Buffalo River, New York, Area of Concern



COMPARATIVE HUMAN HEALTH AND WILDLIFE RISK ASSESSMENT: BUFFALO RIVER, NEW YORK, AREA OF CONCERN

by

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CHAPTER 1

EXECUTIVE SUMMARY

1.1 OVERVIEW

The Assessment and Remediation of Contaminated Sediments (ARCS) program, a 5-year study and demonstration project relating to the control and removal of contaminated sediments from the Great Lakes, is being coordinated and conducted by the U.S. Environmental Protection Agency's (EPA) Great Lakes National Program Office (GLNPO). As part of the ARCS program, baseline human health risk assessments have been performed at five Areas of Concern (AOCs) in the Great Lakes region (Crane, 1992a,b; 1993a,b; 1994). In addition, baseline aquatic (Passino-Reader, et al., 1995) and wildlife (Mann-Klager, 1993) risk assessments have been prepared for the Buffalo River, NY.

In this report, exposure and risk assessment guidelines, developed for the EPA Superfund program, have been applied to determine the comparative human health risks associated with direct and indirect exposures to contaminated sediments in the lower Buffalo River under selected remedial alternatives. These risks were estimated for noncarcinogenic (e.g., reproductive toxicity, teratogenicity, liver toxicity) and carcinogenic (i.e., probability of an individual developing cancer over a lifetime) effects, based on currently available data. In addition, noncarcinogenic risks to mink, resulting from the ingestion of PCB-contaminated forage fish (carp), were estimated to give an indication of ecological risks to a piscivorous species.

1.2 STUDY AREA

The Buffalo River AOC is located in Buffalo, NY in western New York State. The river flows from the east and discharges into Lake Erie near the head of the Niagara River. The study area has a history of water quality problems due primarily to point sources of contaminants (i.e., industrial and municipal discharges). The extent of contamination in the Buffalo River led to the International Joint Commission's (IJC) decision to designate this region as a Great Lakes AOC. In response, the New York State Department of Environmental Conservation (NYSDEC) has completed one phase of a remedial action plan (RAP) to identify and implement pollution abatement measures for the Buffalo River AOC (NYSDEC, 1989).

High concentrations of heavy metals, polychlorinated biphenyls (PCBs), polynuclear aromatic hydrocarbons (PAHs), and pesticides have been measured in different compartments of the Buffalo River (e.g., sediments, water column, and fish). Fish advisories have been issued against consuming carp from the Buffalo River because of excessive concentrations of PCBs. The transport of these contaminants into Lake Erie is also of concern. However, it was beyond the scope of this risk

assessment to estimate human health and wildlife risks to people using the nearshore areas of Lake Erie.

1.3 EXPOSURE ASSESSMENT

1.3.1 Modeled Data

In this comparative risk assessment, a modeling exercise was carried out in which estimates of water column and fish contaminant concentrations were made for different remedial alternatives developed by the ARCS Risk Assessment and Modeling (RAM) work group. Remediation was based on upstream and downstream reaches of the AOC and included the following scenarios: no action; no action, no external loading; environmental dredging; no navigational dredging above Hamburg Cove; Hamburg Cove scenario with no external loading. Water column concentrations were modeled for five chemicals [polychlorinated biphenyls (PCBs), benzo(a)anthracene, benzo(a)pyrene, copper, and lead] under five different remediation scenarios. PCBs were the only chemical of interest modeled in the fish bioaccumulation study for carp. Carp were selected because they are a benthic feeder and have a high lipid content. Thus, carp may readily accumulate sediment-derived contaminants through the ingestion and assimilation of contaminated food.

1.3.2 Human Exposure

Water-contact and noncontact recreational activities are limited along the Buffalo River. No public bathing facilities exist along the river, and fish consumption advisories have warned the public not to eat carp from the river. However, there is anecdotal evidence that these activities occur, even near industrial outfalls. This assessment focused on two complete pathways by which residents near the lower Buffalo River could be exposed to sediment-derived contaminants: (1) consumption of contaminated carp and (2) ingestion of surface water while swimming. A third complete pathway of dermal exposure to surface water was assumed to be insignificant in comparison to the risk resulting from the ingestion of contaminated surface water. This assumption was made because contaminants are more easily transported across the gut than the skin. Data for other exposure pathways were determined to be incomplete (e.g., ingestion of sediments).

Noncarcinogenic and carcinogenic risks to humans were estimated for typical, reasonable maximum, and subsistence (fish only) exposure scenarios. Typical (i.e., average) exposures were assumed to occur over a period of 9 years, whereas reasonable maximum (i.e., the maximum exposure that is reasonably expected to occur at a site) and subsistence (i.e., reliance on fish as a major source of protein) exposures were assumed to occur over a period of 30 years (USEPA, 1989a). These exposure durations were extrapolated over a period of 70 years for estimating carcinogenic risks. Exposure intakes were determined for each chemical and added for each exposure pathway.

TABLE 1.1. AMOUNT OF CARP ASSUMED TO BE CONSUMED PER PERSON PER DAY FROM THE BUFFALO RIVER FOR EACH EXPOSURE SCENARIO

Exposure Scenario	Ingestion Rate* (g/day)	X	FI**	Amount of Buffalo R. Carp Consumed per Day (g/day)
Typical	19.2		0.10	1.92
Reasonable Maximum	54		0.25	13.5
Subsistence	132		0.70	92.4

* Sources: Typical (West et al., 1989); Reasonable Maximum and Subsistence (USEPA, 1991a)

** FI = Fraction of fish (i.e., carp) estimated to be ingested from the Buffalo River (study assumption).

For each of the exposure scenarios, different consumption patterns of fish were assumed to take place (Table 1.1). These consumption patterns were based on recommended values given in U.S. EPA Superfund guidance (USEPA, 1989a,b; 1991a), on published literature values, or on study assumptions. Based on an average meal of fish (i.e., 150 g or 0.33 lb), the amount of Buffalo River fish consumed for each exposure scenario (Table 1.1) can also be converted to meals per year using the following equation:

$$\text{Ingestion Rate (meals/yr)} = [\text{Ingestion Rate (g/day)}] \times FI \times (\text{meal}/150\ \text{g}) \times (365\ \text{days/yr})$$

Where FI is the fraction of contaminated fish estimated to be ingested from the Buffalo River. The number of meals of Buffalo River fish consumed over a year-long period for typical, reasonable maximum, and subsistence exposures corresponded to approximately 4.5, 33, and 225 meals, respectively.

1.3.3 Wildlife Exposure

The comparative wildlife risk assessment focused on one exposure pathway: the consumption of contaminated forage fish by mink. Mink were considered due to their occurrence at Tifft Nature Preserve in the AOC and by their inclusion in the baseline wildlife risk assessment (Mann-Klager, 1993). Exposure intakes were not calculated for mink because a simple hazard quotient method was used to screen for risks. A No-Observed-Adverse-Effect-Level (NOAEL) was obtained from the literature for mink exposed to total PCBs through feeding studies. A NOAEL concentration of 0.069 μg/g wet weight was used in the draft baseline wildlife risk assessment (Heaton et al., 1991 cited in Mann-Klager, 1993). This same NOAEL concentration was used in this report.

1.4 RISK ASSESSMENT

1.4.1 Determination of Human Health Risk

Noncarcinogenic effects were evaluated by comparing an exposure level over a specified time period with a reference dose (RfD)[1] derived from a similar exposure period [otherwise known as a hazard quotient (HQ)]. Thus, HQ = exposure level/RfD. An HQ value of less than 1 indicates that exposures are not likely to be associated with adverse noncarcinogenic effects. HQ values between 1 and 10 may be of concern, particularly when additional significant risk factors are present (e.g., other contaminants are present at concentrations of concern) (USEPA, 1989a). HQ values for multiple substances and/or multiple exposure pathways were summed to yield an overall Hazard Index (HI). The HIs were interpreted in the same fashion as the HQs. Summing the HQs did not account for any synergistic or antagonistic effects that may occur among substances.

Carcinogenic risks were estimated as the incremental probability of an individual developing cancer over a lifetime as a result of exposure to potential carcinogens. This risk was computed using average lifetime exposure values that were multiplied by the oral slope factor[2] for a particular chemical. The resulting carcinogenic risk estimate generally represented an upper-bound estimate, because slope factors are usually based on upper 95th percentile confidence limits. Carcinogenic effects were summed for all chemicals in an exposure pathway. This summation of carcinogenic risks assumed that intakes of individual substances were small, that there were no synergistic or antagonistic chemical interactions, and that all chemicals caused cancer. The EPA believes it is prudent public health policy to consider actions to mitigate or minimize exposures to contaminants when estimated, upper-bound excess lifetime cancer risks exceed the 10^{-5} to 10^{-6} range, and when noncarcinogenic health risks are estimated to be significant (USEPA, 1988a).

1.4.1.1 Consumption of Contaminated Fish

The consumption of PCB-contaminated carp resulted in significant noncarcinogenic risks (i.e., Hazard Quotient > 1) and carcinogenic risks [i.e., risk greater than one person in a million (10^{-6})] for all remedial alternatives and fish consumption scenarios (Tables 1.2 and 1.3). The degree of risk increased as local residents of the Area of Concern consumed more locally caught carp. For noncarcinogenic risks, the Hazard Quotient values ranged from 1.7 to 4.2 for typical exposures, 5.6 to 19 for reasonable maximum exposures, and 39 to 130 for subsistence exposures (Table 1.2). For carcinogenic risks, the

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[1] The RfD provides an estimate of the daily contaminant exposure that is not likely to cause harmful effects during either a portion of a person's life or their entire lifetime (USEPA, 1989a).

[2] Slope factors are estimated through the use of mathematical extrapolation models for estimating the largest possible linear slope (within 95% confidence limits) at low extrapolated doses that is consistent with the data (USEPA, 1989a).

TABLE 1.2. NONCARCINOGENIC RISKS ASSOCIATED WITH CONSUMING WHOLE CARP UNDER VARIOUS REMEDIATION AND CONSUMPTION SCENARIOS

Remediation Scenario	Noncarcinogenic Risk		
	Typical	RME	Subsistence
UPSTREAM			
No Action	2.5	13	89
No Action, No Load	2.4	11	76
Environmental Dredging	1.7	10	70
Hamburg Cove	2.1	7.9	54
Hamburg Cove, No Load	1.8	5.6	39
DOWNSTREAM			
No Action	4.2	19	130
No Action, No Load	4.1	18	130
Environmental Dredging	1.7	11	76
Hamburg Cove	4.2	19	130
Hamburg Cove, No Load	4.1	18	120

TABLE 1.3. CARCINOGENIC RISKS ASSOCIATED WITH CONSUMING WHOLE CARP UNDER VARIOUS REMEDIATION AND CONSUMPTION SCENARIOS

Remediation Scenario	Lifetime Cancer Risk		
	Typical	RME	Subsistence
UPSTREAM			
No Action	4.9E-05	8.5E-04	5.8E-03
No Action, No Load	4.7E-05	7.3E-04	5.0E-03
Environmental Dredging	3.4E-05	6.7E-04	4.6E-03
Hamburg Cove	4.2E-05	5.2E-04	3.6E-03
Hamburg Cove, No Load	3.6E-05	3.7E-04	2.5E-03
DOWNSTREAM			
No Action	8.3E-05	1.3E-03	8.8E-03
No Action, No Load	8.1E-05	1.2E-03	8.4E-03
Environmental Dredging	3.4E-05	7.3E-04	5.0E-03
Hamburg Cove	8.3E-05	1.3E-03	8.8E-03
Hamburg Cove, No Load	8.1E-05	1.2E-03	7.9E-03

risks ranged from 3.4×10^{-6} to 8.3×10^{-5} for typical exposures, 3.7×10^{-4} to 1.3×10^{-3} for reasonable maximum exposures, and from 2.5×10^{-3} to 8.8×10^{-3} for subsistence exposures (Table 1.3). A greater degree of risk was observed in the downstream remediation scenarios than the upstream ones.

1.4.1.2 Ingestion of Contaminated Surface Water

The noncarcinogenic and carcinogenic risks resulting from the ingestion of surface water while swimming under typical and reasonable maximum exposure scenarios were estimated to be far below levels of concern for the environmental dredging scenario. These risks were estimated based on a modeled data set for copper, lead, benzo(a)anthracene, benzo(a)pyrene, and PCBs. Noncarcinogenic risks ranged from 0.000006 to 0.00001 for typical and reasonable maximum exposures, respectively. Lifetime cancer risks were on the order of 10^{-10} for both scenarios. Lower risks could be expected for the other remedial alternatives which had equal or lower contaminant concentrations.

Based on these estimated risk values, an assumption was made that insignificant risks would also result from dermal exposure to surface water while swimming. This assumption was made because the direct intake of contaminants into the gut generally results in a greater intake of contaminants than the absorption of contaminants (with varying capacity for penetration) through the skin.

1.4.2 Determination of Wildlife Risk

Wildlife risk was assessed for a representative piscivorous species, mink. A simple hazard quotient method was used to compare the modeled PCB concentration in carp, a representative forage fish, to the target forage concentration (i.e., the NOAEL value). The NOAEL was exceeded by 19 to 46 times for the various remedial alternatives. The estimated degrees of exceedance were very protective because an assumption was made that local mink populations consumed 100% of their diet from contaminated carp.

1.5 UNCERTAINTIES

Several assumptions and estimated values were used in this comparative risk assessment that contributed to the overall level of uncertainty associated with the noncarcinogenic and carcinogenic risk estimates. One of the greatest sources of uncertainty in this risk assessment arose from the use of modeled data. Another large source of uncertainty was using modeled data for whole, raw carp in the exposure assessment for human health; PCB concentrations could be greatly reduced in fish by trimming off the fat and skin and cooking the fish. As with most environmental risk assessments, the uncertainty of the risk estimates probably ranges over an order of magnitude or greater. On a comparison basis, this risk assessment exercise was useful in judging the adequacy of different remedial alternatives in reducing risk to human health and wildlife.

CHAPTER 2

INTRODUCTION

2.1 BACKGROUND

The 1987 amendments to the Clean Water Act, in Section 118(c)(3), authorized the U.S. Environmental Protection Agency's (EPA) Great Lakes National Program Office (GLNPO) to coordinate and conduct a 5-year study and demonstration project relating to the control and removal of contaminated sediments from recommended areas in the Great Lakes region (USEPA, 1991b). To achieve this task, GLNPO initiated the Assessment and Remediation of Contaminated Sediments (ARCS) program. A number of agencies were involved in conducting toxicology, chemistry, risk assessment, modeling, and engineering studies at the following Areas of Concern (AOCs): Ashtabula River, OH; Buffalo River, NY; Grand Calumet River/Indiana Harbor Canal, IN; Saginaw River, MI; and Sheboygan River, WI. The ARCS program was extended by one year by the Great Lakes Critical Programs Act of 1990.

This report will present the results of a comparative risk assessment for the Buffalo River AOC. Comparative risk assessments are used to estimate and compare the risks that may be associated with various remedial alternatives (including the "no action" alternative). A mass balance modeling approach is used to estimate how contaminant concentrations in the sediment, water column, and biota will vary with remedial alternative. In turn, the exposure and subsequent risk to humans and biota from contaminants are estimated.

The Buffalo River AOC has a history of water quality problems due primarily to point sources of contaminants (i.e., industrial and municipal discharges). The extent of contamination in this area led to the International Joint Commission's (IJC) decision to designate this region as a Great Lakes AOC. In response, the New York State Department of Environmental Conservation (NYSDEC) has completed one phase of a remedial action plan (RAP) to identify and implement pollution abatement measures for the Buffalo River AOC (NYSDEC, 1989).

High concentrations of heavy metals, polychlorinated biphenyls (PCBs), polynuclear aromatic hydrocarbons (PAHs), and pesticides have been measured in different compartments of the Buffalo River (e.g., sediments, water column, and fish). Fish advisories have been issued against consuming carp from the Buffalo River because of high concentrations of PCBs. The benthic macroinvertebrate community has been adversely affected as the fauna are dominated by pollutant-tolerant oligochaetes and some invertebrates frequently display abnormal mouthparts (Diggins and Stewart, 1993). These observations generated concern that organisms higher up in the food chain may be exposed to unhealthy doses of contaminants.

2.2 BASELINE HUMAN HEALTH RISK ASSESSMENT

As part of the ARCS Risk Assessment and Modeling (RAM) work group activities, a baseline human health risk assessment was conducted for an area adjacent to the lower Buffalo River (Figure 2.1). The purpose of this risk assessment was to determine current risk conditions to human health. Baseline risk estimates were determined for both noncarcinogenic effects (chronic or subchronic) and carcinogenic effects (i.e., the probability of an individual developing cancer over a lifetime) resulting from direct and indirect exposures to sediment-related contaminants (Crane, 1993a). These risk estimates were made using conservative assumptions about exposure scenarios when complete data were not available. Thus, the risk estimates were designed to be overprotective of human health. The baseline risk assessment focused on two complete pathways by which residents of the lower Buffalo River could be exposed to sediment-derived contaminants: (1) consumption of contaminated carp and spottail/emerald shiners, and (2) ingestion of surface water while swimming. Carp data were used in the exposure assessment, despite the fish advisories, because these fish are benthic feeders and accumulate contaminants more readily than pelagic species of fish. Thus, the consumption of carp provided a conservative estimate of risk to anglers and their families.

The results of the baseline human health risk assessment are summarized in Appendix A. The ingestion of surface water did not result in a significant risk to swimmers. However, the consumption of carp represented a noncarcinogenic risk to subsistence anglers that consumed over 225 meals of fish per year. The consumption of either carp or spottail/emerald shiners represented a cancer risk (i.e., greater than one in a million) to anglers and their families that consumed 4.5, 33, or 225 meals of fish per year. As given in Appendix A, only a proportion of these meals of fish were composed of contaminated fish from the Buffalo River.

2.3 BASELINE WILDLIFE RISK ASSESSMENT

The draft baseline wildlife risk assessment for the Buffalo River AOC (Mann-Klager, 1993) focused on one exposure pathway: the consumption of contaminated forage food by piscivorous wildlife. Common terns (*Sterna hirundo*), a colonial waterbird, and mink (*Mustela vison*) were selected for the wildlife risk assessment. Common terns were considered due to the availability of data for this species in the upper Niagara River and their documented contaminant sensitivity and usefulness as bioindicators [Hays and Risebrough, 1972; Fasala et al., 1987; and Karwowski, 1991 (references cited in Mann-Klager, 1993)]. In addition, Gilbertson (1974) concluded that the organochlorines found in common terns were ingested within the vicinity of the breeding colony. Karwowski (1991) observed that there was a large percentage of tern chicks that died while pipping (i.e., trying to break open the shell of their egg during hatching) in the upper Niagara River. Mink were considered due to their occurrence at Tifft Nature Preserve. Although the population density of mink at Tifft has not been determined, sightings are not uncommon (Landsittel, personal communication cited in Mann-Klager, 1993). Foley et al. (1988) concluded that PCB and DDE concentrations in minks were influenced by local sources of the contaminants.

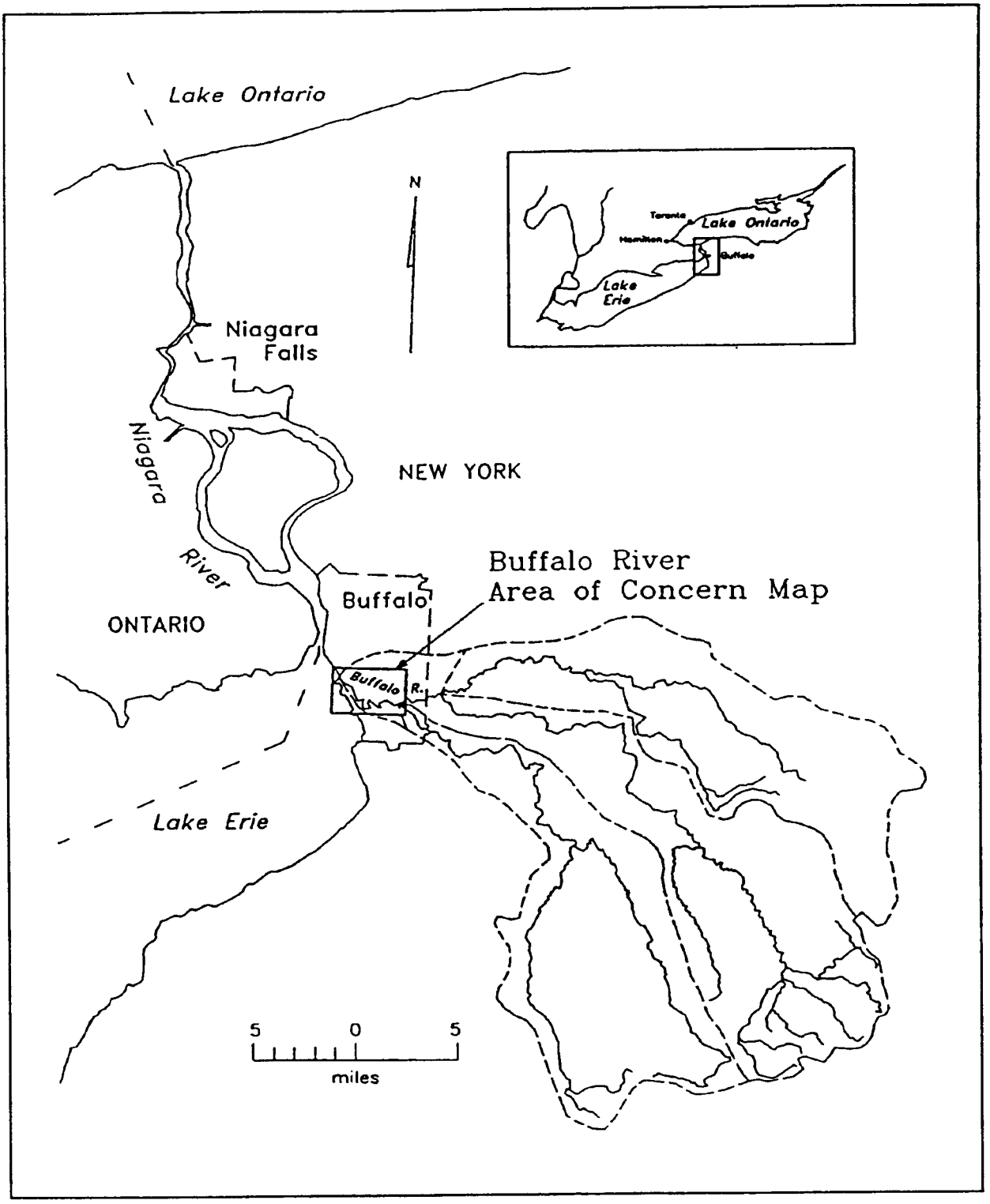

Figure 2.1. Map of Buffalo River Area of Concern (NYSDEC, 1989).

The selection of these species was supported by the Biological Effects Subcommittee to the International Joint Commissions's (IJC) Science Advisory Board. This subcommittee is evaluating the use of the bald eagle (*Haliaeetus leucocephalus*), mink or river otter (*Lutra canadensis*), colonial waterbirds, and lake trout (*Salvelinus namaycush*) as bioindicators of Great Lakes water quality (Kubiak and Best, 1991). Bald eagles were not considered in the draft baseline risk assessment as their occurrence in the area is presently limited to occasional transients. Historic eagle breeding occurred at the northern end of Grand Island on the Niagara River. River otters were also not selected because they are presently rare in the Niagara River area (Newell et al., 1987).

Wildlife risks were estimated for noncarcinogenic (e.g., reproductive toxicity) effects using the simple hazard assessment model developed by Kubiak and Best (1991). The model uses the lowest observed adverse effect levels (LOAEL) and no observed adverse effect levels (NOAEL) of contaminants to assess wildlife health. Preliminary results from the baseline risk assessment indicated the possibility of detrimental effects to common terns and mink. However, a number of assumptions were made in the baseline risk assessment which increased the level of uncertainty with the risk estimates.

2.4 COMPARATIVE RISK ASSESSMENT APPROACH

Another objective of the ARCS RAM work group was to use a mass balance approach at two AOCs (i.e., Buffalo River, NY and Saginaw River, MI) to address management questions concerning the remediation of contaminated sediments. These two AOCs were chosen based upon anticipated impacts from sediments, lack of other on-going activities (such as Superfund remedial activities), and lack of complicating factors (such as complicated groundwater/surface water interactions, multiple sources of contaminant inputs, etc.). The mass balance approach involved an evaluation of the sources, transport, and fate of contaminants in the system. This process follows the law of conservation of mass and requires that the quantities of contaminants entering the system, less quantities stored, transformed, or degraded in the system, must equal the quantities leaving the system.

ARCS RAM participants developed and applied Level 1 (preliminary) models for understanding and predicting the transport and fate of contaminated sediments and the bioaccumulation of persistent sediment contaminants in the affected AOC. The RAM work group used a comparative risk assessment approach (described in Chapter 3) to integrate the results from the baseline risk assessment, field, and mass balance modeling studies to provide estimates of the potential impact of remedial actions on human health and wildlife. Thus the risk, relative to the baseline risk, that would result from the implementation of various sediment remedial alternatives could be evaluated.

The framework for the comparative risk assessment approach is discussed in the following chapter. Subsequent chapters provide the exposure assessment, toxicity assessment, risk characterization, and uncertainty sections of the comparative risk assessment for human health and wildlife at the Buffalo River AOC.

CHAPTER 3

COMPARATIVE RISK ASSESSMENT FRAMEWORK

3.1 OVERVIEW

The comparative risk assessment framework used by the ARCS RAM work group is given in Figure 3.1. The framework (USEPA, 1993a) was developed to: 1) identify existing risks to human health and ecological receptors at sites with contaminated sediments, 2) estimate the potential impact of various sediment remedial alternatives on contaminant concentrations in various media and their associated risks, and 3) compare existing and potential future risks to aid in the selection of sediment remedial alternatives. This framework can be applied to other contaminated sediment sites and should be viewed as a demonstration of the steps one would take to perform a comparative risk assessment.

The main components of the framework are listed below. The components are based on guidance given in USEPA (1988b, 1989a, 1993a) and on discussions generated from the ARCS RAM work group.

- Characterization of the Area of Concern including:
  - collection of existing data to determine extent of contamination in the key sources and media of interest
  - collection of background data
  - examination of QA/QC measures
  - determination of potential contaminants of concern
  - identification of contaminant sources, especially information related to release potential
  - characterization of the environmental setting that may affect the fate, transport, and persistence of contaminants.
- Evaluation of available data including:
  - assessment of data quality
  - identification of data gaps.
- Implementation of field studies including:
  - collection of contaminant data in the media of interest to fill data gaps for risk assessment and modeling components

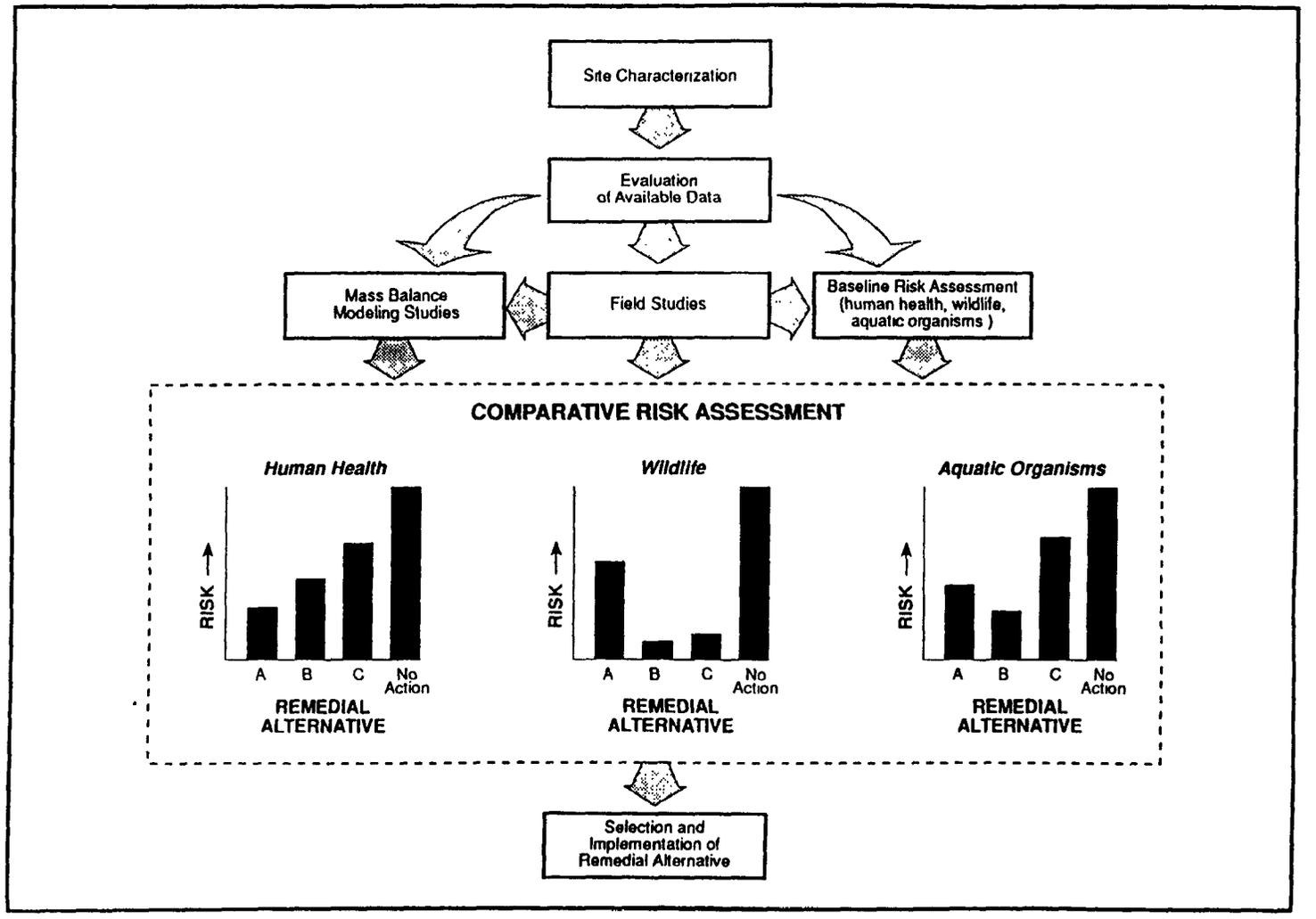

Figure 3.1. Comparative risk assessment in the risk management process (USEPA, 1993a).

- considerations for sampling including:

    - sample size
    - sampling locations
    - types of samples (e.g., grab or composite, whole fish or fillet)
    - temporal and meteorological factors
    - field screening analyses
    - time and cost of sampling
    - QA/QC procedures.

- Evaluation of baseline risks to human health, wildlife, and aquatic organisms including:

    - identification of receptors of concern
    - determination of exposure pathways and exposure intake concentrations
    - identification of toxicity values for contaminants of concern
    - characterization of risks
    - estimation of uncertainties associated with risk estimates.

- Implementation of mass balance modeling studies including:

    - identification of management questions, such as:

        - How long will it take for contamination to be reduced to an acceptable level through natural processes (e.g., sedimentation)?
        - What will happen if the sediments are dredged to a particular depth?
        - Will the sediments become recontaminated following remediation?

    - selection of contaminants of concern
    - selection of sediment remedial alternatives
    - development of screening models
    - determination of appropriate scales
    - implementation of modeling framework including:

        - hydrodynamic modeling
        - sediment transport modeling
        - contaminant transport and fate modeling
        - food chain modeling

    - calibration/verification of models.

- Evaluation of comparative risks to human health, wildlife, and aquatic life including:
  - adaptation of the baseline risk assessment framework
    - site characterization
    - exposure assessment
    - toxicity/hazard assessment
    - risk characterization
  - integration of the modeling results in the exposure assessment
  - derivation of risk values for selected remedial alternatives
  - comparison of risk values
  - evaluation of uncertainty.
- Selection and implementation of remedial alternative(s).

A multidisciplinary team is needed to carry out the aforementioned components of the framework. It is very important before work commences on a comparative risk assessment that the team agrees on the management questions to be addressed, the level of effort necessary to complete work, and the timelines for completion of tasks. Regular meetings are essential to evaluate progress and to reassess priorities. Interested stakeholders should be informed of progress/results of the comparative risk assessment through an effective risk communication effort.

The results of the comparative risk assessment are evaluated by the risk manager. The risk manager must also take into consideration social, economic, and political factors which would influence the selection of a remedial option. A final remedial action plan can then be developed and implemented.

For additional information on the comparative risk assessment framework, refer to the "Risk Assessment and Modeling Overview Document" (USEPA, 1993a) completed for the ARCS program. Specific details on the methodology used for the ARCS baseline human health risk assessments are given by Crane (1992a,b; 1993a,b; 1994). These assessments followed exposure and risk assessment guidelines established by the U.S. EPA for use at Superfund sites (USEPA, 1988b; 1989a,b; 1991a).

The following sections describe the activities of the modelers from the RAM work group for the Buffalo River AOC. This modeling effort was done as a demonstration project. Therefore, the same approach, level of effort, and data needs pertinent to the Buffalo River AOC may not apply to other contaminated sites.

3.2 MASS BALANCE MODELING

3.2.1 Selection of Remedial Action Scenarios

An important component of the comparative risk assessment was the use of modeled data to estimate changes in contaminant concentrations in sediment, water, and fish tissue based on different remediation scenarios. The RAM work group selected the following remedial action scenarios for the Buffalo River AOC, of which the first three were the most important (DePinto et al., 1994).

1. **No Action Scenario.** This scenario focused on the system response over time under existing external loadings and continued navigational dredging. No additional actions on the river were simulated.

2. **Hamburg Cove Scenario.** This scenario examined the impacts of discontinuing navigational dredging above Hamburg Cove (about halfway from the river mouth to the AOC upstream boundary). This scenario permitted this portion of the river to fill in with "clean" sediments from upstream. The potential for flooding existed as a result of this option.

3. **Environmental Dredging Scenario.** This scenario examined the impact of nearshore dredging along the entire river within the AOC. This option would remove several "hot spots" along the banks.

In order to determine the importance of resuspension on water column contamination, Scenarios 1 and 2 were also evaluated with no external loadings. All other factors were kept the same. Thus, any contamination of the water column would be strictly from sediment resuspension.

4. **No Action - No Loading Scenario.**

5. **Hamburg Cove - No Loading Scenario.**

Two additional scenarios were modeled by DePinto's group (DePinto et al., 1994) to aid in the interpretation of the other remedial action scenarios.

6. **Zero Initial Conditions Scenario.** Similar to Scenario 3, the initial conditions in the top two layers of sediments (depositional and erosional) were set to zero contaminant concentrations. This effectively nullified currently contaminated sediments as a source of water column contamination. Thus, the sole impact would be from external loading.

7. **Flow Switching Scenario.** Two years of actual flow data were switched with each other to evaluate the effect on cumulative export and concentrations in the no action

scenario. The flows from 1978-79, which contained several high flow events, were switched with those from 1970-71, which had no events, and vice versa. The results showed the importance of the sequence of flow events in altering the final results.

3.2.2 Mass Balance Modeling Framework

The application of the mass balance modeling approach involved the quantification of the sources, transport, and fate of contaminants (Figure 3.2). Specific detail on the components of mass balance models can be found in the ARCS "Risk Assessment and Modeling Overview Document" (USEPA, 1993a). The typical steps in a mass balance modeling study are to:

- predict water and sediment transport
- use the predicted transport, along with estimates of contaminant loadings from point and nonpoint sources, to estimate the changes in chemical concentrations in water and sediments
- use the predicted contaminant concentrations in water and sediments to estimate the transfer of contaminants through the food chain and their accumulation in fish.

A mass balance modeling framework (using a modified version of WASP4/TOXI4) was developed for use as a management tool for the Buffalo River AOC (DePinto et al., 1994). Contaminant exposure and benthic food chain bioaccumulation were analyzed based on the response to various remedial alternatives. The model also simulated the export from the Buffalo River to Lake Erie resulting from these remediation actions.

Joseph DePinto of the State University of New York-Buffalo (SUNY-Buffalo) led most of the modeling work for the Buffalo River comparative risk assessment (Atkinson et al., 1994; DePinto et al., 1994). A different application of sediment and contaminant transport was done by Joseph Gailani and coworkers (Gailani et al., 1994) to predict the transport of fine-grained sediments and associated contaminants in the Buffalo River. Gailani's team was especially interested in the transport of these sediments from the river to Lake Erie. The SUNY-Buffalo studies are briefly described below. Each of these studies made use of data collected from field sampling programs conducted for the ARCS program. Refer to the modeler's reports for a description of their data needs and sources of data. The only modeled data used in this comparative risk assessment came from the SUNY-Buffalo modelers.

3.2.2.1 SUNY Modeling Effort

The modeling work by DePinto's group involved several components. One component was to develop estimates of mass loading rates for the following contaminants: total PCBs, chlordane, dieldrin, DDT, benzo(a)anthracene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, chrysene, lead and

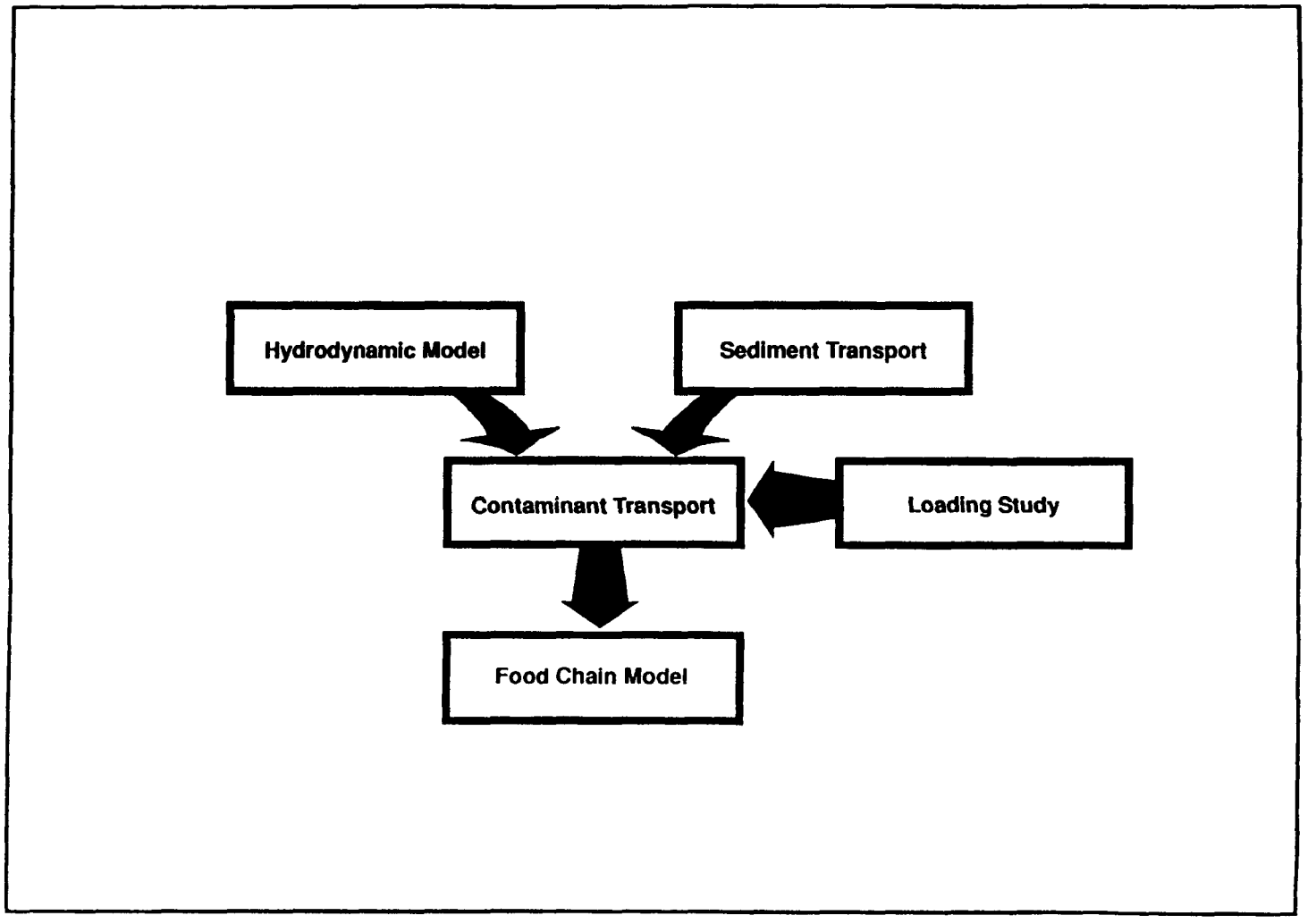

Figure 3.2. Components of the mass balance modeling study (USEPA, 1993a).

copper (Atkinson et al., 1994). The aforementioned contaminants were chosen based on fish advisories, concerns cited in the Remedial Action Plan, and results obtained from Toxicity Identification Evaluation work (USEPA, 1994). Total suspended solids (TSS) loading was also calculated. The annual loading calculations indicated relatively small loadings for most of the contaminants of interest.

The major source for all the contaminants of interest was found to be the upstream tributary flows (Atkinson et al., 1994). Upstream loadings were calculated on the basis of average daily flows and TSS, along with measured contaminant concentrations. Groundwater and combined sewer overflow (CSO) loadings were estimated on the basis of separate model calculations and industrial loadings taken from the Buffalo River Remedial Action Plan (NYSDEC, 1989). Results were presented for use in water quality mass balance models which could be used to simulate the time history of contaminant concentrations in the water column, sediments, and biota of the river as a function of source inputs. This information was used to evaluate system response to selected remedial action scenarios.

Another component of this work was to calculate several parameters needed to develop and apply general water quality and contaminant transport models to the river (Atkinson et al., 1994). These parameters included primary distribution (partition) coefficients for each of the contaminants of interest, as well as data for a number of conventional parameters. Annual and monthly average flows were presented and data were provided for specifying upstream and downstream boundary conditions.

Additional modeling was done by DePinto et al. (1994) to develop water quality mass balance and bioaccumulation models for the Buffalo River by simulating a time-history of contaminant concentrations in the water column, sediments, and biota of the river as a function of source inputs. The overall modeling framework followed Figure 3.2, and a modified version of WASP4/TOXI4 was used for the contaminant mass balance model. The contaminants that were modeled included total PCBs, benzo(a)anthracene, benzo(a)pyrene, copper, and lead. The parameterization, segmentation, and calibration of the models are described in detail in DePinto et al. (1994).

The following conclusions were made by DePinto et al. (1994):

- On days of average or low flow, resuspension of contaminated sediment was not a significant factor in water column concentrations.

- Sediment remediation **will not** have a significant impact on reducing water column contaminant exposure. Environmental or full dredging of bottom sediments will not alleviate water column concerns for the five chemicals examined. Also, the potential to exacerbate the water column problem still exists with these dredging options by exposing higher contaminated sediments in deeper layers.

- Sediment remediation **will be** a potentially important action for reducing direct sediment exposure, especially in "hot spots." Environmental dredging of nearshore "hot spots" could be beneficial to the benthic community and corresponding food web.

- The contaminant body burdens of bottom-dwelling and bottom-feeding organisms, such as carp, will improve in response to sediment remediation actions. On a river-wide basis, environmental dredging in the nearshore depositional areas lead to the largest reduction in PCB concentrations in carp. However, the cessation of navigational dredging above Hamburg Cove proved to be the best alternative for that portion of the modeled river.

3.2.2.2 Limitations of Modeling Effort

Specific limitations and data needs noted by the SUNY modelers (Atkinson et al., 1994; DePinto et al., 1994) included:

- Year-round TSS data were not available, especially during high flow events, for calculating loading estimates.

- Uniform sample collection and analytical protocols were not applied by all groups involved with data collection.

- Finer resolution of vertical profiles in sediment cores and a more uniform distribution of the horizontal location of sediment cores are needed to improve quantification of initial conditions for model runs.

- A more accurate description of erosional/depositional areas of the river would enhance the model simulation.

- Sediment transport could be characterized more accurately if deposition rates and other physical and chemical properties of resuspended and upstream sediments were measured as a function of flow.

- Profiles of sediment concentration data before and after dredging would be useful to examine the effect of sediment sloughing.

The water column and carp data from the modeling effort of DePinto et al. (1994) were used in the exposure assessment described in the following chapter. The uncertainty associated with these data cannot be quantitatively calculated but will be qualitatively assessed in Chapter 8.

CHAPTER 4

EXPOSURE ASSESSMENT

4.1 EXPOSURE PATHWAYS

In this exposure assessment, the magnitude, frequency, duration, and route of direct and indirect exposures of people to sediment-derived contaminants from the Buffalo River AOC will be determined for different remediation scenarios. In addition, the exposure to mink residing within the AOC will be evaluated.

4.1.1 Human Exposure Pathways

Human exposure to contaminants in the Buffalo River AOC can potentially occur via three pathways: dermal contact, inhalation, and ingestion. Dermal contact involves direct contact of the skin with either contaminated sediments, riverplain soils, or overlying water. Inhalation of airborne vapors or dust may introduce chemicals of potential concern into the respiratory system. Ingestion of contaminants through the consumption of contaminated soils, sediment, or food (e.g., fish) is potentially significant because of the direct transfer of contaminants across the gut.

The potential pathways by which people may be exposed to contaminants from the Buffalo River AOC were given in the baseline risk assessment (Crane, 1993a). Although four exposure pathways were considered complete (i.e., exposure could occur through each of those routes) in the baseline risk assessment (Table 4.1), not all of those exposure pathways may result in substantial human health risks. In addition, for humans inhaling airborne contaminants, it would be difficult to separate out the contributions of contaminants from the river (if any) and those from industrial, municipal, and background sources. Thus, although the inhalation exposure pathway may be complete, the currently available data set for atmospheric contaminants in the Buffalo River AOC are inadequate to quantitatively assess the risks to human health.

TABLE 4.1. COMPLETE EXPOSURE PATHWAYS IN THE BUFFALO RIVER AOC

• Ingestion of Contaminated Fish
• Ingestion of Surface Water while Swimming or Playing in the Water
• Dermal Contact with Water while Boating, Fishing, Swimming, Water Skiing, etc.
• Inhalation of Airborne Contaminants

In the baseline risk assessment, the only substantial risk resulted from the consumption of fish (Crane, 1993a). For this comparative risk assessment, it was recognized that contaminant concentrations in the water might change enough under different remediation scenarios to pose a potential risk to people swimming in the river. Thus, an exposure assessment was conducted for swimmers ingesting small amounts of surface water under the environmental dredging scenario; this scenario was selected because it resulted in the highest estimated surface water contaminant concentrations. If this risk was determined to be substantial, then the ingestion of surface water pathway would be examined for all remediation scenarios. However, the risk was insignificant for both carcinogenic and noncarcinogenic effects (see Chapter 6).

Dermal exposure to surface water was judged to result in an insignificant risk based on the low frequency with which these exposures would take place and because the direct intake of contaminants into the gut is usually greater than the absorption of contaminants (with varying capacities to penetrate) across the skin interface. Thus, risks from dermal exposure should be less than those from ingesting surface water.

The only complete exposure pathways considered for this risk assessment were the consumption of fish (for all remedial action scenarios) and ingestion of surface water while swimming (for the environmental dredging scenario). Noncarcinogenic and carcinogenic risks were determined for both typical (i.e., average) and reasonable maximum exposures (i.e., the maximum exposure that is reasonably expected to occur at a site). In addition, risks were calculated for subsistence anglers that relied on the consumption of fish for their main source of protein. The subsistence exposure scenario was chosen because of economic problems in the area which might contribute to an underemployed/unemployed person consuming large amounts of locally caught fish.

4.1.2 Wildlife Exposure Pathways

At least twelve wildlife species in the Great Lakes basin have experienced reproductive or other problems and/or population decreases since the 1960s that have been associated with chemical contaminants (Government of Canada, 1991 cited in Fox, 1993). The list includes two mammals, nine species of birds, and one reptile. All are long-lived, fish-eating species. A number of studies in the Great Lakes area have been conducted to study impairments to wildlife using such biomarkers as induction of mixed function oxidases, alterations in heme biosynthesis, retinol homeostasis, thyroid function, DNA integrity, and various manifestations of reproductive and developmental toxicity (Fox, 1993).

Several species of biota have been used as biomonitors of general ecosystem health in the Great Lakes. In 1975, the Ontario Ministry of the Environment adopted the use of young-of-the-year spottail shiners (*Notropis hudsonius*) as biomonitors for the nearshore waters of the Great Lakes (Suns et al., 1993). Contaminant residue data from spottail shiner surveys have been used to identify areas of concern for contaminant inputs and trend assessment over time. Results of annual collections of these fish indicate

that, at most sites sampled, PCB residues continued to decline during the 1980s. Shiners are an important forage fish in the Great Lakes, and contaminant concentrations in the fish could cause/contribute to detrimental effects in other biota, such as waterfowl.

Colonial fish-eating birds have been used to study the impact of chronic exposure to complex mixtures of hydrophobic organic contaminants (HOCs) within the Great Lakes ecosystem (Fox et al., 1991a). Fox et al. (1991a) suggest that double-crested cormorants (a totally piscivorous species) be used as a biomarker due to their long life, abundance, and mostly ground nesting. The occurrence of bill malformations in double-crested cormorant chicks have been documented from colonies in Green Bay and elsewhere in the Great Lakes and in reference areas off the Great Lakes, in the years 1979 through 1987 (Fox et al., 1991b). Although the severity of waterfowl impairment effects have generally decreased between the early 1970s and late 1980s, these studies confirm the continued presence of sufficient amounts of PCBs and other HOCs in forage fish to cause physiological impairments in these birds over much of the Great Lakes basin (Fox, 1993). Tillitt et al. (1992 cited in Fox, 1993) suggest that PCBs are the major contaminant influencing cormorant reproductive success in the Great Lakes. Fox and coworkers (1991b) suggest that monitoring reproductive outcomes in fish-eating birds is a cost-effective and sensitive method of detecting biologically significant concentrations of developmental toxins in Great Lakes fish.

In terms of estimating risks to piscivorous wildlife at the Buffalo River AOC, the baseline risk assessment looked at two species: mink and common terns (Mann-Klager, 1993). Mann-Klager assumed that the majority of contaminant uptake by these species would be from the consumption of fish. For the comparative risk assessment, the ingestion of fish pathway was the only exposure pathway examined. Mink was the only piscivorous wildlife species considered here, because carp was the only forage fish for which contaminant concentrations were modeled for different remedial action scenarios. Common terns were not included in this exposure assessment because they would not be consuming carp. Instead, they are shoreline feeders that would be consuming small fish such as minnows and shiners.

The majority of habitat available for wildlife utilization within or near the AOC is at Tifft Nature Preserve and Times Beach confined disposal facility (CDF) (Figure 4.1). The Times Beach CDF was constructed by the Buffalo District U.S. Army Corps of Engineers in 1971 for the containment of dredged materials from the Federal navigation channel. The CDF is approximately 19 ha in size and is located west of the mouth of the Buffalo River. The site received dredged material from 1972 to 1976. In response to a request by the Buffalo Ornithological Society, the site was designated a native preserve and left only partially filled (Stafford et al., 1991 cited in Mann-Klager, 1993).

Little wildlife habitat is available along the river, outer harbor, and Erie basin due to shoreline development. Included in Appendix B is a list of endangered, threatened, and special concern species of New York State observed near the Buffalo River AOC as well as wildlife species observed at the Times Beach CDF and Tifft Nature Preserve.

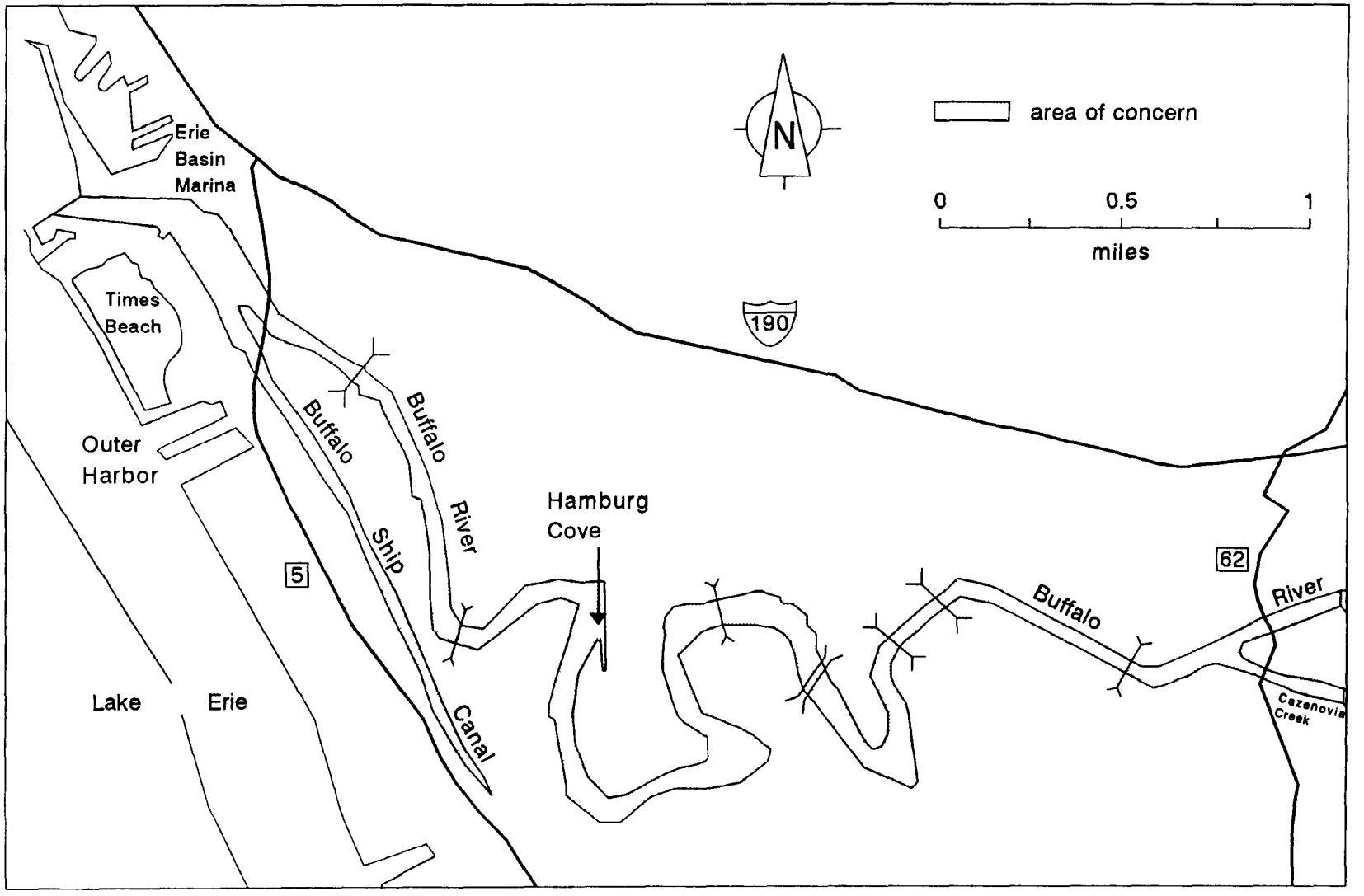

Figure 4.1. Location of Times Beach Confined Disposal Facility (NYSDEC, 1989).

4.2 MODELED DATA USED IN THE EXPOSURE ASSESSMENT

4.2.1 Carp Data

Three different age classes of carp (i.e., young, middle, and old) were collected from the Buffalo River by the Great Lakes Laboratory, State University College at Buffalo. The fish were collected specifically for the ARCS program. Each age class included three composite samples of five whole fish. The samples were received on November 13, 1991 by Battelle, Pacific Northwest Division, and were analyzed for eighty individual PCB congeners and eight chlorinated pesticides. Total PCB concentrations are given in Table 4.2. The samples were analyzed using Battelle Standard Operating Procedures (SOP) MSL-042 and MSL-044. The data underwent a QA/QC review by Lockheed Engineering and Sciences Company (Lockheed-ESC) under a contract with the EPA Environmental Monitoring Systems Laboratory in Las Vegas, NV. The collection and analysis of these carp had to comply with a detailed QA/QC plan, and these data have been approved for use by the ARCS program (USEPA, 1994). The results were reported as ng/g on a dry weight basis and were converted to wet weight for use in the exposure assessment.

The carp data were used by DePinto et al. (1994) to calibrate a bioaccumulation model which had originally been developed for Green Bay, Lake Michigan (Connolly et al., 1992). The model describes the major features of the predator-prey relationship, seasonal movement of fish, and species bioenergetic parameters, as well as the dependency of the transfer of PCBs across the gill and gut of the animals on water column and sediment PCB concentrations (Connolly et al., 1992). It used the estimates of water column and sediment concentrations from the physical-chemical mass balance model to compute the time-variable concentration of PCBs in benthic biota of the Buffalo River. From the latter, the food chain accumulation of PCBs in carp could be estimated for each remedial alternative. Refer to DePinto et al. (1994) for a more detailed explanation of the assumptions used for the various parameters in the model.

Ten year predictive runs were chosen for model simulations of each scenario. Navigational dredging was simulated every two years when estimating sediment and water column concentrations. Bioaccumulation modeling was done for the following scenarios:

1. no action
2. Hamburg Cove
3. environmental dredging
4. no action/no load
5. Hamburg Cove/no load.

The scenarios were run using modeled water column and sediment data obtained for selected segments of the river located upstream and downstream of the AOC boundary. Bioaccumulation results for each scenario are listed in Tables 4.3 and 4.4. The values shown represent mean PCB concentrations

TABLE 4.2. AGE CLASS DATA ON CARP COLLECTED FROM THE BUFFALO RIVER*

Age Class	Age, Years	Wet Weight (kg)	PCB Concentration (μg/g wet weight)
YOUNG AGE CLASS			
BRF Y W-1	4.2	0.944	1.89
BRF Y W-2	4.0	0.972	1.8
BRF Y W-3	4.6	0.927	2.2
Mean	4.3	0.948	1.96
MIDDLE AGE CLASS			
BRF M W-1	6.4	1.633	2.76
BRF M W-2	6.0	1.667	2.34
BRF M W-3	5.4	1.61	3.7
Mean	5.9	1.637	2.93
OLD AGE CLASS			
BRF O W-1	10.0	4.552	5.9
BRF O W-2	10.8	4.257	3.1
BRF O W-3	10.0	4.45	3.4
Mean	10.3	4.42	4.13

* Each age class has three groups associated with it, each group containing five fish. Values shown are mean values for the five fish (Irvine et al., 1992 cited in DePinto et al., 1994).

of the young, middle, and old age classes of carp used in the model (DePinto et al., 1994). The upstream scenario data from Table 4.3 are plotted in Figure 4.2, whereas the downstream scenario data from Table 4.4 are plotted in Figure 4.3.

In the upstream scenario (Figure 4.2), environmental dredging resulted in the most rapid decline of PCBs in carp tissue until stabilizing at approximately 1 mg/kg after Day 1280. The Hamburg Cove, no load scenario resulted in the greatest overall decline of PCBs to 0.2 mg/kg in carp at Day 3660. The no action scenario resulted in the highest PCB concentrations in carp, ending at 1.1 mg/kg on Day 3660.

TABLE 4.3. MODELED PCB CONCENTRATIONS IN CARP FOR ALL UPSTREAM SCENARIOS (DEPINTO ET AL., 1994)

Day	Modeled PCB Concentrations (mg/kg wet weight) in Carp				
	No Action	No Action, No Load	Environmental Dredging	Hamburg Cove	Hamburg Cove, No Load
10	2.82	2.82	2.80	2.82	2.82
190	2.74	2.71	2.24	2.74	2.71
370	2.89	2.84	2.01	2.89	2.84
550	2.76	2.70	1.65	2.76	2.70
730	2.90	2.82	1.52	2.90	2.82
920	2.62	2.53	1.26	2.61	2.52
1100	2.41	2.31	1.15	2.39	2.29
1280	2.11	1.99	1.07	1.96	1.82
1470	2.05	1.89	1.12	1.78	1.60
1650	1.83	1.66	1.04	1.49	1.30
1830	1.81	1.63	1.08	1.41	1.18
2020	1.64	1.45	1.02	1.19	0.957
2200	1.65	1.44	1.06	1.15	0.894
2380	1.49	1.29	1.00	0.977	0.722
2560	1.43	1.21	0.994	0.908	0.635
2750	1.26	1.04	0.916	0.769	0.500
2930	1.24	1.01	0.927	0.738	0.454
3110	1.16	0.934	0.914	0.617	0.347
3300	1.20	0.964	0.982	0.587	0.307
3480	1.11	0.891	0.950	0.502	0.236
3660	1.12	0.895	0.983	0.479	0.208
Average	1.9	1.8	1.3	1.6	1.4
S.D.*	0.65	0.72	0.49	0.89	0.98

*S.D. = standard deviation

Note that at Day 3660, the final range of PCB concentrations for all scenarios was fairly narrow (0.2 - 1.1 mg/kg).

For the downstream scenario (Figure 4.3), environmental dredging resulted in similar results to the upstream scenario. However, the other remediation actions all resulted in an almost two-fold increase in PCB concentrations from Day 10 to Day 730. After Day 730, the modeled PCB concentrations in carp declined for the other four remediation actions, ending slightly greater than the environmental dredging scenario at Day 3660. Thus, environmental dredging resulted in the most rapid decrease of PCBs in fish tissue, but the end result at Day 3660 was about the same for all remediation actions.

TABLE 4.4. MODELED PCB CONCENTRATIONS IN CARP FOR ALL DOWNSTREAM SCENARIOS (DEPINTO ET AL., 1994)

Day	Modeled PCB Concentrations (mg/kg wet weight) in Carp				
	No Action	No Action, No Load	Environmental Dredging	Hamburg Cove	Hamburg Cove, No Load
10	2.86	2.86	2.80	2.86	2.86
190	3.62	3.60	2.24	3.62	3.60
370	4.45	4.41	2.01	4.45	4.41
550	4.74	4.69	1.65	4.74	4.69
730	5.35	5.28	1.52	5.35	5.28
920	5.07	5.00	1.27	5.07	5.00
1100	4.80	4.71	1.18	4.80	4.71
1280	4.11	4.00	1.11	4.11	4.00
1470	3.87	3.74	1.17	3.87	3.74
1650	3.40	3.27	1.10	3.40	3.27
1830	3.31	3.15	1.15	3.31	3.15
2020	2.94	2.78	1.09	2.94	2.78
2200	2.91	2.74	1.14	2.91	2.74
2380	2.58	2.41	1.08	2.58	2.41
2560	2.42	2.23	1.09	2.42	2.23
2750	2.09	1.90	1.01	2.09	1.90
2930	2.02	1.82	1.03	2.02	1.82
3110	1.78	1.59	1.01	1.78	1.59
3300	1.77	1.56	1.08	1.76	1.56
3480	1.59	1.39	1.06	1.58	1.39
3660	1.59	1.39	1.13	1.58	1.38
Average	3.2	3.1	1.3	3.2	3.1
S.D.*	1.2	1.2	0.46	1.2	1.2

*S.D. = standard deviation

A summary table of modeled PCB concentrations in carp for a ten-year predictive run are given in Table 4.5. These results were also extrapolated over a 30-year period for use in the exposure assessment. The 30-year mean was derived using the PCB concentration at Day 3660 as the representative concentration for the next 20 years. Thus, a weighted mean could be derived for a 30-year period by using the following equation:

$$\text{30-year mean} = \frac{\text{(10-year mean)} + 2\text{(Day 3660 concentration)}}{3}$$

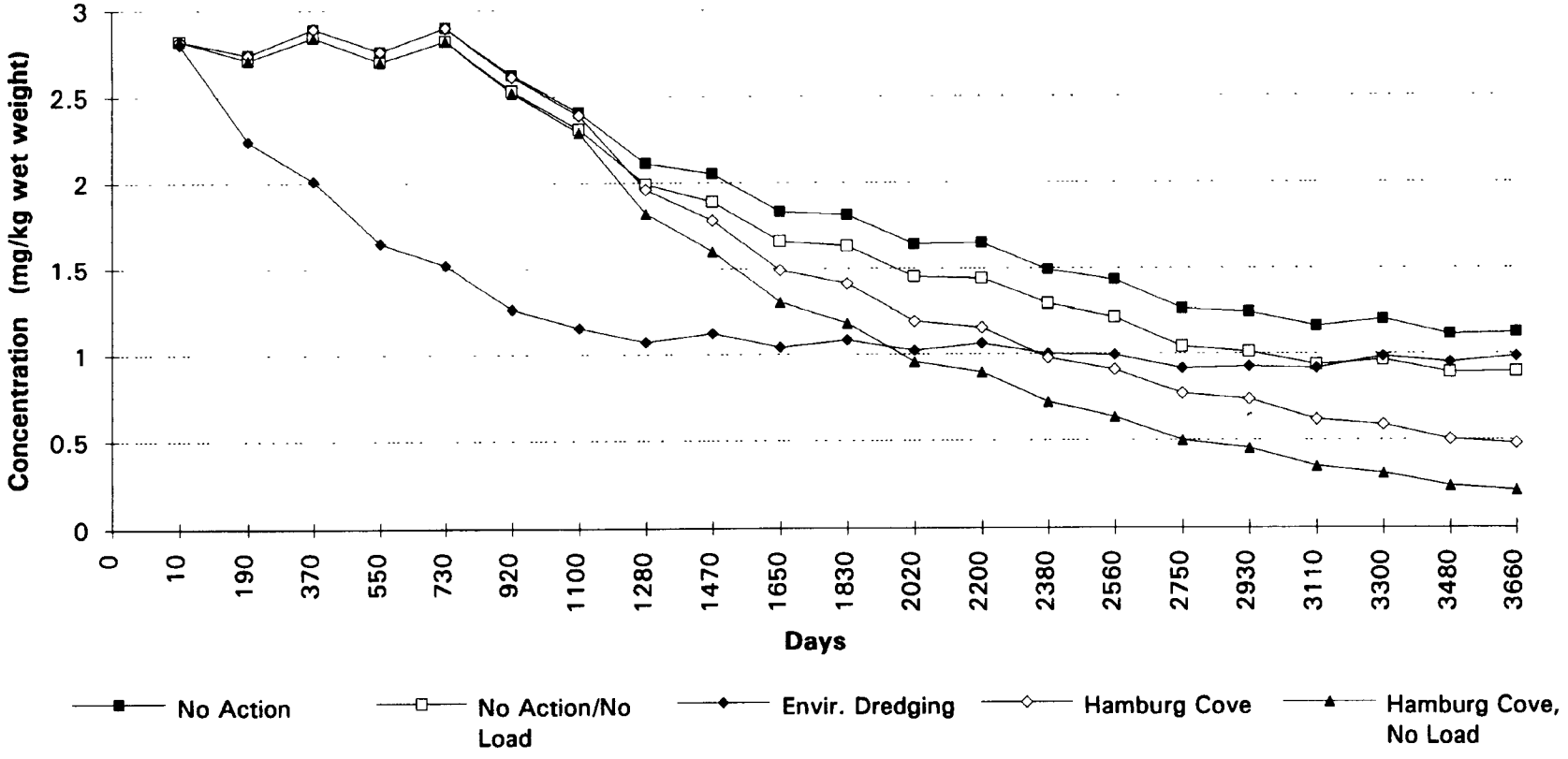

Figure 4.2. Modeled PCB concentrations in carp: Upstream scenario.

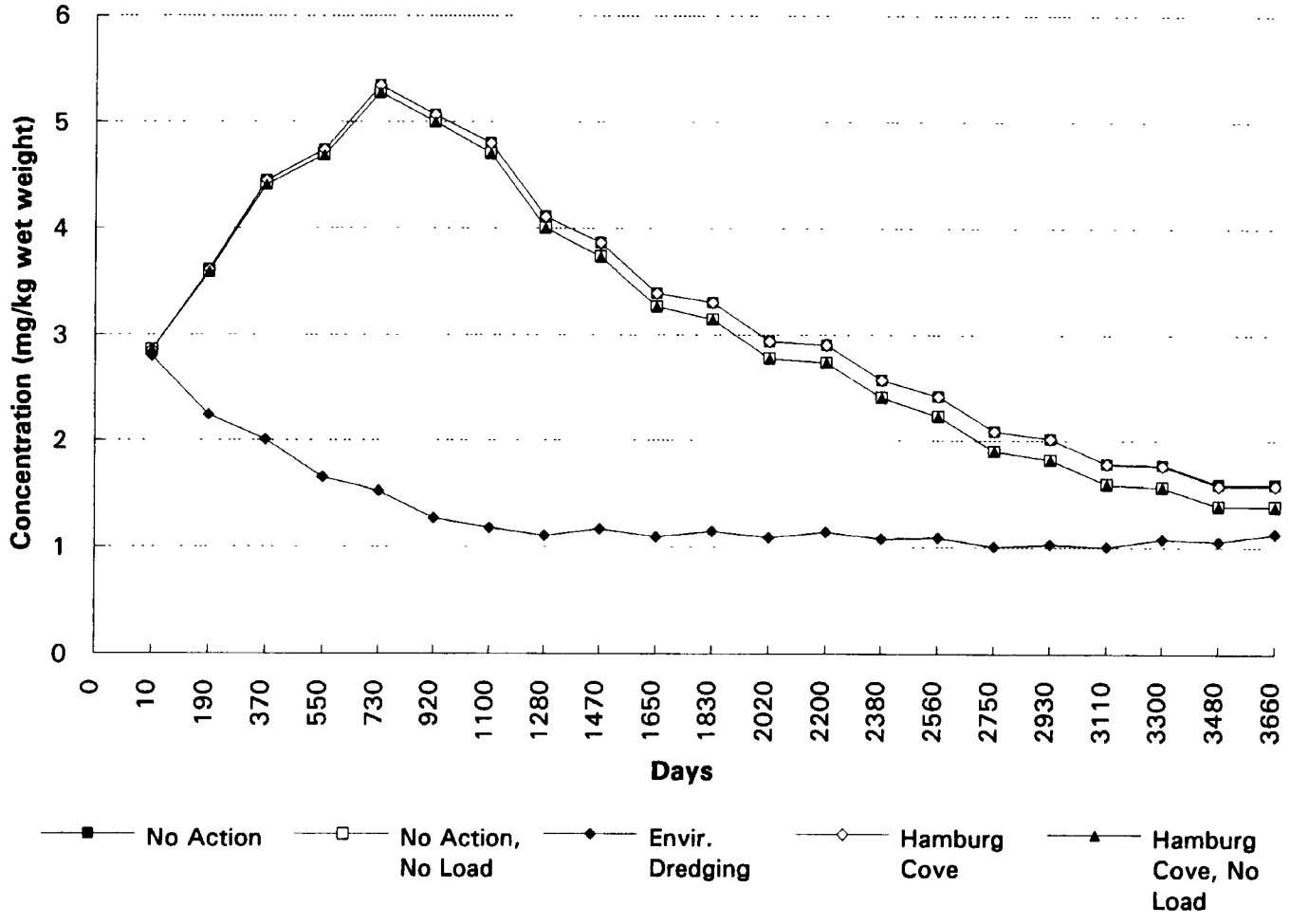

Figure 4.3. Modeled PCB concentrations in carp: Downstream scenario.

TABLE 4.5. SUMMARY OF MODELED PCB CONCENTRATIONS IN BUFFALO RIVER CARP FOR 10-YEAR AND 30-YEAR SCENARIOS

Remediation Scenario	Modeled PCB Concentration (mg/kg wet weight)			
	10 Year		30 Year	
	Mean	S.D.	Mean	S.D.
UPSTREAM				
No Action	1.9	0.65	1.4	0.53
No Action, No Load	1.8	0.72	1.2	0.59
Environmental Dredging	1.3	0.49	1.1	0.32
Hamburg Cove	1.6	0.89	0.85	0.74
Hamburg Cove, No Load	1.4	0.98	0.61	0.81
DOWNSTREAM				
No Action	3.2	1.2	2.1	1.0
No Action, No Load	3.1	1.2	2.0	1.1
Environmental Dredging	1.3	0.46	1.2	0.28
Hamburg Cove	3.2	1.2	2.1	1.0
Hamburg Cove, No Load	3.1	1.2	1.9	1.1

This calculation assumed that the PCB concentrations in carp reached a steady state by Day 3660 for each remediation scenario.

4.2.2 Surface Water Data

Water quality data from the Buffalo River were obtained for the following contaminants: total PCBs, α-chlordane, γ-chlordane, dieldrin, DDT, benzo(a)anthracene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, chrysene, lead, and copper. Samples were collected from six sites during two primary sampling periods, each covering a few weeks during the fall of 1990 and spring of 1992. Specific sampling dates were October 18, 22, 27, 31, and November 5, 9, 13, 1990 and April 4, 18, 22, 1992 (Atkinson et al., 1994). Detailed descriptions of analytical techniques and results are under preparation by researchers at Buffalo State College (as cited in Atkinson et al., 1994).

For the contaminant transport model, the initial water column conditions were taken as the average concentrations from six different sites collected on October 18, 1990. The model was run for total PCBs, benzo(a)anthracene, benzo(a)pyrene, copper, and lead. The modeled data used in each remediation scenario came from selected segments of the river. The modeled upstream water column

data were derived from Segment 9, whereas the downstream data were derived from Segment 25 (Figure 4.4) (DePinto et al., 1994).

The modeled water column concentrations are given in Tables 4.6 and 4.7. The mean contaminant concentrations for the no action, Hamburg Cove, environmental dredging, zero initial conditions, and flow switching scenarios were quite similar. Contaminant concentrations decreased by over an order of magnitude when upstream loads were eliminated under the no action and Hamburg Cove scenarios. For the human health exposure assessment, only the data from the downstream environmental dredging scenario were used to estimate the risk from consuming water while swimming. Since the risk posed by this scenario was insignificant (see Chapter 6), it would thus follow that the risk would also be insignificant for the other remedial alternative scenarios.

4.3 EXPOSURE ASSESSMENT FOR HUMAN HEALTH

4.3.1 General Determination of Chemical Intakes

Once the complete exposure pathways were identified and modeled contaminant concentrations for fish and surface water were obtained, the human health exposure assessment was conducted. The period of exposure was assumed to take place after the remedial alternative was completed. This approach was used so that the increase or reduction in risk, compared to the no action alternative, could be determined. A similar assumption was made for the wildlife risk assessment.

Exposures were normalized for time and body weight to determine chemical "intakes," expressed in units of mg chemical/kg body weight-day. For the ingestion of contaminated fish and water, intakes represent the amount of chemical available for absorption in the gut. The general equation for calculating chemical intakes is given in Table 4.8. Several variables were used to determine intakes, including specific information about the exposed population and the period over which the exposure was averaged. Noncarcinogenic effects were averaged over the same time period as the exposure duration [i.e., 9 years for typical exposures and 30 years for reasonable maximum (RME) and subsistence exposures]. Carcinogenic effects were averaged over a lifetime (i.e., 70 years). Intake variable values were selected so that the combination of all values resulted in an estimate of either the typical, reasonable maximum, or subsistence exposure intakes.

Modeled water column and fish data that were averaged over a ten-year period were used as the contaminant concentrations for a typical duration scenario of nine years. This averaging was done to take into account the movement of people into the Buffalo River AOC one year after the remedial alternative was implemented.

Chemical intakes were not calculated for the wildlife exposure assessment as a different approach was used to estimate risks (see Chapter 7).

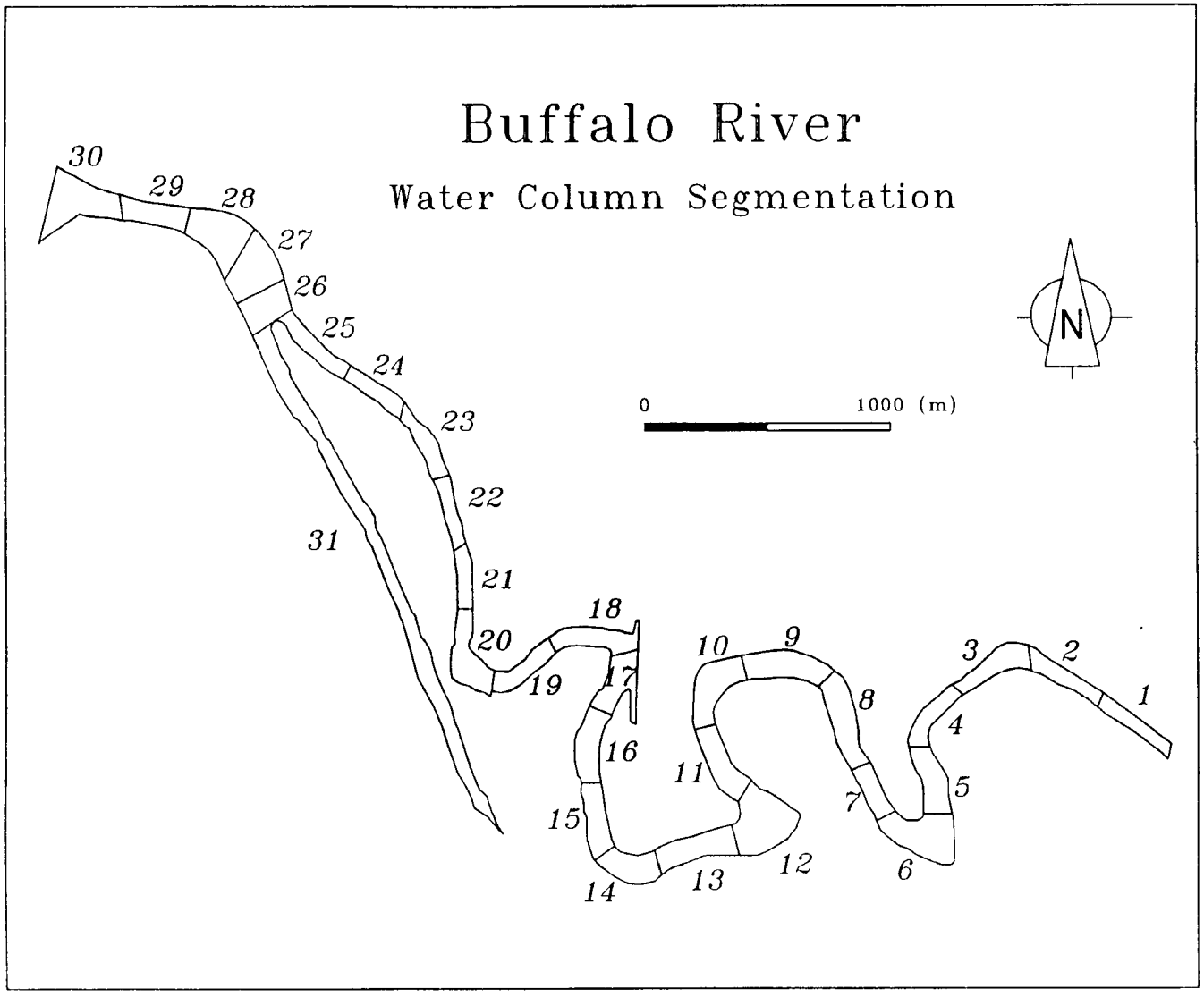

Figure 4.4. Water column segmentation.

TABLE 4.6. MODELED UPSTREAM WATER COLUMN CONCENTRATIONS FOR VARIOUS REMEDIAL ALTERNATIVES

Remediation Scenario	PCBs (ng/L)	Benzo(a)-anthracene (ng/L)	Benzo(a)-pyrene (ng/L)	Copper (ug/L)	Lead (ug/L)
No Action					
Mean	1.8E+00	1.1E+01	4.2E+01	2.6E+00	1.9E+00
S.D.	4.8E-01	6.5E+00	5.3E+01	1.3E+00	9.8E-01
Variance	2.3E-04	7.2E-02	2.8E+00	1.7E+00	9.5E-01
Hamburg Cove					
Mean	1.8E+00	1.1E+01	4.2E+01	2.6E+00	1.9E+00
S.D.	4.7E-01	6.4E+00	5.3E+01	1.3E+00	9.8E-01
Variance	2.2E-04	7.0E-02	2.8E+00	1.7E+00	9.5E-01
Environmental Dredging					
Mean	1.8E+00	1.1E+01	4.2E+01	2.6E+00	1.9E+00
S.D.	4.8E-01	6.5E+00	5.3E+01	1.3E+00	9.8E-01
Variance	2.3E-04	7.2E-02	2.8E+00	1.7E+00	9.5E-01
No Action - No Load					
Mean	2.3E-01	1.5E-01	1.6E-01	1.5E-01	1.3E-01
S.D.	1.1E-01	1.7E-01	1.6E-01	9.4E-02	7.9E-02
Variance	1.2E-05	2.9E-05	2.5E-05	8.8E-03	6.3E-03
Hamburg Cove - No Load					
Mean	2.2E-01	4.6E-02	7.2E-02	1.5E-01	1.3E-01
S.D.	1.1E-01	5.0E-02	8.8E-02	9.4E-02	8.0E-02
Variance	1.3E-05	2.5E-06	7.7E-06	8.9E-03	6.3E-03
Zero Initial Conditions					
Mean	1.8E+00	1.1E+01			
S.D.	4.8E-01	6.5E+00	(no data)	(no data)	(no data)
Variance	2.3E-04	7.2E-02			
Flow Switching					
Mean	1.8E+00	1.1E+01			
S.D.	4.7E-01	6.4E+00	(no data)	(no data)	(no data)
Variance	2.2E-04	7.0E-02			

4.3.2 Ingestion of Contaminated Fish

The equation used to estimate intakes of contaminants due to the ingestion of contaminated fish is provided in Table 4.9. The parameter values used in that equation are given in Table 4.10. Parameter values were obtained mostly from recommended EPA sources. The exposure parameters used in the typical fishing scenario were assumed to be applicable to the general angling population of Buffalo, whereas the reasonable maximum exposure scenario applied to recreational anglers and their families. The subsistence exposure scenario was chosen for a sensitive subpopulation of people who would be

TABLE 4.7. MODELED DOWNSTREAM WATER COLUMN CONCENTRATIONS FOR VARIOUS REMEDIAL ALTERNATIVES

Remediation Scenario	PCBs (ng/L)	Benzo(a)-anthracene (ng/L)	Benzo(a)-pyrene (ng/L)	Copper (ug/L)	Lead (ug/L)
No Action					
Mean	2.0E+00	9.9E+00	2.0E+01	2.6E+00	2.3E+00
S.D.	3.6E-01	9.6E+00	2.3E+01	1.2E+00	1.0E+00
Variance	1.3E-04	9.2E-02	5.4E-01	1.5E+00	1.1E+00
Hamburg Cove					
Mean	1.9E+00	9.8E+00	2.0E+01	2.6E+00	2.3E+00
S.D.	3.4E-01	9.4E+00	2.3E+01	1.2E+00	1.0E+00
Variance	1.1E-04	8.8E-02	5.4E-01	1.5E+00	1.1E+00
Environmental Dredging					
Mean	2.0E+00	9.9E+00	2.0E+01	2.6E+00	2.3E+00
S.D.	3.7E-01	9.6E+00	2.3E+01	1.2E+00	1.0E+00
Variance	1.3E-04	9.2E-02	5.4E-01	1.5E+00	1.1E+00
No Action - No Load					
Mean	7.8E-01	3.8E-01	4.2E-01	4.7E-01	4.0E-01
S.D.	3.5E-01	6.0E-01	6.3E-01	2.5E-01	2.1E-01
Variance	1.2E-04	3.6E-04	4.0E-04	6.4E-02	4.5E-02
Hamburg Cove - No Load					
Mean	7.6E-01	2.4E-01	3.0E-01	4.7E-01	4.0E-01
S.D.	3.6E-01	2.9E-01	4.3E-01	2.6E-01	2.1E-01
Variance	1.3E-04	8.2E-05	1.9E-04	6.5E-02	4.6E-02
Zero Initial Conditions					
Mean	1.9E+00	9.8E+00			
S.D.	3.5E-01	9.6E+00	(no data)	(no data)	(no data)
Variance	1.2E-04	9.1E-02			
Flow Switching					
Mean	2.0E+00	1.0E+01			
S.D.	3.5E-01	9.4E+00	(no data)	(no data)	(no data)
Variance	1.2E-04	8.8E-02			

relying on locally caught fish for a large proportion of their diet. The ingestion rates used for each of those scenarios are listed in Table 4.10; the rationale for selecting these values has been discussed in detail in Crane (1993a). An assumption was made that the ingestion rate included both "clean" and contaminated fish. Only fish consumed from the Buffalo River were assumed to be contaminated. In addition, only modeled PCB data for uncooked, whole carp were available for use in the exposure assessment. Thus, the contaminant intakes may overestimate risk since PCB concentrations can be reduced in fish by trimming off the fat and cooking the fish (Zabik et al., 1979).

TABLE 4.8. GENERIC EQUATION FOR CALCULATING CHEMICAL INTAKES (USEPA, 1989a)

$$Intake = \frac{C \times CR \times EFD}{BW \times AT}$$

where:	
Intake	Intake = the amount of chemical at the exchange boundary (mg/kg body weight-day)
	Chemical-Related Variables
C	Chemical Concentration = the average concentration contacted over the exposure period (e.g., mg/L)
	Variables that Describe the Exposed Population
CR	Contact Rate = the amount of contaminated medium contacted per unit time or event (e.g., L/day)
EFD	Exposure Frequency and Duration = how long and how often exposure occurs. Often calculated using two terms, EF and ED, where EF = exposure frequency (days/year) ED = exposure duration (years)
BW	Body Weight = the average body weight over the exposure period (kg)
	Assessment-Determined Variables
AT	Averaging Time = period over which exposure is averaged (days)

Because there was no quantitative information available on the fraction of fish ingested from the Buffalo River (i.e., FI), conservative estimates were made. Based on an average meal of fish (150 g or 0.33 lb), the amount of Buffalo River fish consumed for each exposure scenario could also be converted to meals per year using the following equation:

$$\text{Ingestion Rate (meals/yr)} = [\text{Ingestion Rate (g/day)}] \times FI \times (\text{meal}/150\ \text{g}) \times (365\ \text{days/yr})$$

The number of meals of Buffalo River fish consumed over a year-long period for typical, reasonable maximum, and subsistence exposures corresponded to approximately 4.5, 33, and 225 meals, respectively.

TABLE 4.9. EQUATION USED TO ESTIMATE CONTAMINANT INTAKES DUE TO INGESTION OF FISH

$$Intake = \frac{C \times IR \times FI \times EF \times ED}{BW \times AT}$$

where:	
Intake	Intake Rate (mg/kg-day)
C	Contaminant Concentration in Fish (mg/kg)
IR	Ingestion Rate (kg/day)
FI	Fraction of Fish Ingested from Contaminated Area (unitless)
EF	Exposure Frequency (days/yr)
ED	Exposure Duration (yr)
BW	Body Weight (kg)
AT	Averaging Time (days)

Chemical intake rates for carp are given in Tables 4.11 and 4.12. Both noncarcinogenic and carcinogenic intake rates were calculated for typical, reasonable maximum, and subsistence exposures.

4.3.3 Ingestion of Surface Water While Swimming

Ingestion of surface water occurs naturally during swimming. The equation used in computing this exposure is provided in Table 4.13, and the corresponding exposure are given in Table 4.14. Where possible, site-specific exposure values were selected following consultation with local residents and agencies. Where values were taken from the literature, the sources of the values are provided. The typical exposure scenario assumed that someone went swimming three days per year, whereas six days per year was the frequency of swimming under reasonable maximum exposure conditions. The estimated intake rates are listed in Table 4.15.

TABLE 4.10. PARAMETERS USED IN ESTIMATING CONTAMINANT INTAKES DUE TO CONSUMPTION OF FISH FROM THE BUFFALO RIVER AOC

Var.	Units	Value Used	Comment
IR	kg/day	0.0192	Typical: West et al. (1989)
		0.054	Reasonable Maximum Exposure (RME): USEPA (1991a)
		0.132	Subsistence: used the 95th percentile daily intakes averaged over 3 days for consumers of fin fish [Pao et al. (1982) cited in USEPA (1989a)]
FI	-	0.1	Typical: study assumption
		0.25	RME: study assumption
		0.7	Subsistence: study assumption
EF	day/yr	350	USEPA (1991a)
ED	yrs	9	Typical: USEPA (1989a)
		30	RME and Subsistence: USEPA (1989a)
BW	kg	70	50th percentile average for adult men and women (USEPA, 1989b)
AT	days	3285	9 yrs x 365 days/yr (typical noncarcinogenic risk)
		10950	30 yrs x 365 days/yr (RME and subsistence noncarcinogenic risk)
		25550	70 yrs x 365 days/yr (carcinogenic risk)

TABLE 4.11. PCB INTAKE RATES RESULTING FROM THE TYPICAL CONSUMPTION OF CARP UNDER DIFFERENT REMEDIATION SCENARIOS

Remediation Scenario	Mean Modeled PCB Conc.* wet wt. (mg/kg)	Noncarcinogenic Intake (mg/kg-day)	Carcinogenic Intake (mg/kg-day)
UPSTREAM			
No Action	1.9	5.0E-05	6.4E-06
No Action, No Load	1.8	4.7E-05	6.1E-06
Environmental Dredging	1.3	3.4E-05	4.4E-06
Hamburg Cove	1.6	4.2E-05	5.4E-06
Hamburg Cove, No Load	1.4	3.7E-05	4.7E-06
DOWNSTREAM			
No Action	3.2	8.4E-05	1.1E-05
No Action, No Load	3.1	8.2E-05	1.0E-05
Environmental Dredging	1.3	3.4E-05	4.4E-06
Hamburg Cove	3.2	8.4E-05	1.1E-05
Hamburg Cove, No Load	3.1	8.2E-05	1.0E-05

* Based on a 10-year modeling duration

TABLE 4.12. PCB INTAKE RATES RESULTING FROM THE REASONABLE MAXIMUM AND SUBSISTENCE CONSUMPTION OF CARP UNDER DIFFERENT REMEDIATION SCENARIOS

Remediation Scenario	Mean Modeled PCB Conc.* wet wt. (mg/kg)	Noncarcinogenic Intake (mg/kg-day)		Carcinogenic Intake (mg/kg-day)	
		RME	Subsistence	RME	Subsistence
UPSTREAM					
No Action	1.4	2.6E-04	1.8E-03	1.1E-04	7.6E-04
No Action, No Load	1.2	2.2E-04	1.5E-03	9.5E-05	6.5E-04
Environmental Dredging	1.1	2.0E-04	1.4E-03	8.7E-05	6.0E-04
Hamburg Cove	0.85	1.6E-04	1.1E-03	6.7E-05	4.6E-04
Hamburg Cove, No Load	0.61	1.1E-04	7.7E-04	4.8E-05	3.3E-04
DOWNSTREAM					
No Action	2.1	3.9E-04	2.7E-03	1.7E-04	1.1E-03
No Action, No Load	2.0	3.7E-04	2.5E-03	1.6E-04	1.1E-03
Environmental Dredging	1.2	2.2E-04	1.5E-03	9.5E-05	6.5E-04
Hamburg Cove	2.1	3.9E-04	2.7E-03	1.7E-04	1.1E-03
Hamburg Cove, No Load	1.9	3.5E-04	2.4E-03	1.5E-04	1.0E-03

* Based on a 30-year modeling duration

RME = reasonable maximum exposure

TABLE 4.13. EQUATION USED TO ESTIMATE CONTAMINANT INTAKE RATES DUE TO INGESTION OF SURFACE WATER WHILE SWIMMING

$$Intake = \frac{CW \times CR \times ET \times EF \times ED}{BW \times AT}$$

where:	
Intake	Lifetime Average Intake Rate (mg/kg/day)
CW	Chemical Concentration in Water (mg/L)
CR	Ingestion Rate (L/hour)
ET	Exposure Time (hours/day)
EF	Exposure Frequency (days/year)
ED	Exposure Duration (years)
BW	Body Weight (kg)
AT	Period of Exposure (days)

TABLE 4.14. PARAMETERS USED FOR COMPUTING INGESTION OF SURFACE WATER WHILE SWIMMING

Variable	Units	Exposure Scenario	Value Used	Reference
CR	L/hr	Typical, RME	0.05	USEPA (1989a)
ET	hr/day	Typical, RME	0.5	Study assumption
EF	day/yr	Typical	3	Study Assumption
		RME	6	Study Assumption
ED	yrs	Typical	9	USEPA (1989a)
		RME	30	USEPA (1989a)
BW	kg	Typical, RME	70	50th percentile average for adult men and women (USEPA, 1989b)
AT	days	Typical	3285	9 yrs x 365 days/yr (noncarcinogenic risk)
		RME	10950	30 yrs x 365 days/yr (noncarcinogenic risk)
		Typical, RME	25550	70 yrs x 365 days/yr (carcinogenic risk)

TABLE 4.15. EXPOSURE INTAKE RATES ASSOCIATED WITH INGESTING CONTAMINATED SURFACE WATER WHILE SWIMMING

Chemical	Mean Water Conc. (mg/L)	Noncarcinogenic Intake (mg/kg-day)		Carcinogenic Intake (mg/kg-day)	
		Typical	RME	Typical	RME
METALS					
Copper	2.6E-03	7.5E-09	1.5E-08	9.7E-10	6.5E-09
Lead	2.3E-03	6.7E-09	1.3E-08	8.6E-10	5.7E-09
ORGANIC COMPOUNDS					
Benzo(a)anthracene	9.9E-06	2.9E-11	5.8E-11	3.7E-12	2.5E-11
Benzo(a)pyrene	2.0E-05	5.8E-11	1.2E-10	7.5E-12	5.0E-11
PCBs	2.0E-06	5.9E-12	1.2E-11	7.5E-13	5.0E-12

CHAPTER 5

TOXICITY/HAZARD ASSESSMENT

5.1 HUMAN HEALTH TOXICITY VALUES

Two types of toxicity values were used in combination with the chemical intake rates to calculate noncarcinogenic and carcinogenic health risks to humans. One toxicity value, the reference dose (RfD), provides an estimate of the daily contaminant exposure that is not likely to cause harmful effects during either a portion of a persons' life or his/her entire lifetime. The RfD is the toxicity value used in evaluating noncarcinogenic effects. The other toxicity value, the slope factor, is used in risk assessments to estimate an upper-bound lifetime probability of an individual developing cancer as a result of exposure to a particular concentration of a potential carcinogen. In addition, the EPA weight-of-evidence classification scheme indicates the strength of evidence that the contaminant is a human carcinogen (Table 5.1). Slope factors are typically calculated for potential carcinogens in Classes A, B1, and B2, as well as for Class C on a case-by-case basis. A more detailed description of these toxicity values, summarized from "Risk Assessment Guidance for Superfund. Volume 1. Human Health Evaluation Manual (Part A)" (USEPA, 1989a), is given in Appendix C.

TABLE 5.1. EPA WEIGHT-OF-EVIDENCE CLASSIFICATION SYSTEM FOR CARCINOGENICITY (USEPA, 1989a)

Group	Description
A	Human carcinogen
B1 or B2	Probable human carcinogen
	B1 indicates that limited human data are available
	B2 indicates sufficient evidence in animals and inadequate or no evidence in humans
C	Possible human carcinogen
D	Not classifiable as to human carcinogenicity
E	Evidence of noncarcinogenicity for humans

Chronic oral RfD values and oral slope factors were used for the fish ingestion and surface water ingestion pathways examined in this risk assessment. Toxicity values, which had undergone an EPA review process, were obtained from the EPA's Integrated Risk Information System (IRIS) data base. For chemicals lacking a "verified value," interim toxicity values were obtained from the Health Effects

Assessment Summary Tables (HEAST), if available (USEPA, 1989c). Table 5.2 lists the toxicity data used for the chemicals of interest. Although RfDs are provided for known carcinogens, it does not imply that these doses are protective against carcinogenicity. This table also includes the form in which the chemical was administered to the test animal or patient (e.g., drinking water, diet, or gavage) for determination of the oral RfD. The endpoints of concern for evaluating noncarcinogenic risks are listed in Appendix C. Toxicity profiles of the chemicals of interest are listed in Appendix D.

5.2 WILDLIFE HAZARD ASSESSMENT

A No Observed Adverse Effect Level (NOAEL) was obtained from the literature for mink exposed to total PCBs through feeding studies. Mann-Klager (1993) used a NOAEL value of 0.069 μg/g wet weight for the baseline wildlife risk assessment. This value was used in the comparative risk assessment.

5.3 LIMITATIONS

This risk assessment was limited by the current availability of toxicity information for the select group of chemicals examined in the modeling exercise. Toxicity values were not available for lead because age, health, nutritional state, body burden, and exposure duration influence the absorption, release, and excretion of lead. These factors make it difficult to estimate noncarcinogenic and carcinogenic toxicity values for lead. Another limitation was that some toxicity values were only available for a certain form of chemical. For example, the RfD value for PCBs applies only to Aroclor 1254, whereas the oral slope factor applies to Aroclor 1260. At the present time, toxicity values are not available for all PCB Aroclors.

TABLE 5.2. HUMAN HEALTH RISK TOXICITY DATA FOR CHEMICALS OF INTEREST IN THE BUFFALO RIVER

Chemical	Oral RfD (mg/kg/day)	Form	Source	Carcinogenic Weight-of-Evidence Class	Source	Oral Slope Factor 1/(mg/kg/day)	Source
METALS							
Copper	1.3E-03		b	D	a		
Lead				B2	a		
PAHs							
Benzo(a)anthracene				B2	a	1.15E+01	c
Benzo(a)pyrene				B2	a	7.30E+00	a
CHLORINATED HYDROCARBONS							
PCBs	2.0E-05	diet	a	B2	a	7.70E+00	a

Sources:
- a: IRIS (current as of 12/09/94)
- b: USEPA (1989c)
- c: Interim guidance, relative to benzo(a)pyrene, suggested by OERR (USEPA, 1989d)

Blank spaces denote a lack of information for the chemical of interest.

CHAPTER 6

COMPARATIVE RISK CHARACTERIZATION: HUMAN HEALTH

6.1 PURPOSE OF THE RISK CHARACTERIZATION STEP

The purpose of the risk characterization step is to combine the exposure and toxicity estimates into an integrated expression of human health risk. This section presents the calculated potential human health risks associated with the consumption of contaminated fish and surface water from the Buffalo River AOC under alternative remediation scenarios. It is important to recognize that these calculated risk estimates are not intended to be used as actual values. Risk assessment is a regulatory process that provides risk managers with quantitative estimates that are to be used for comparative purposes only. These risk estimates must be interpreted in the context of all the uncertainties associated with each step in the process. Some of the major uncertainties in this risk assessment are addressed in Chapter 8.

Two means of expressing the carcinogenic and noncarcinogenic risks of adverse health effects are presented in this chapter. First, chemical-specific risks were estimated for each exposure pathway. Secondly, chemical specific risks were added to estimate a cumulative pathway-specific risk.

6.2 QUANTIFYING RISKS

6.2.1 Determination of Noncarcinogenic Risks

Noncarcinogenic effects are evaluated by comparing an exposure level over a specified time period with a RfD derived from a similar exposure period [otherwise known as the hazard quotient (HQ)]. Thus,

$$HQ = \text{exposure level (or intake)}/RfD.$$

Hazard quotients are expressed to one significant figure in a nonprobabilistic way. In this risk assessment, HQ values were expressed to two significant figures for each chemical; this was done to reduce rounding errors when HQ values were summed for each pathway. An HQ value of less than 1 indicates that exposures are not likely to be associated with adverse noncarcinogenic effects (e.g., reproductive toxicity, teratogenicity, or liver toxicity). As the HQ approaches or exceeds 10, the likelihood of adverse effects is increased to the point where action to reduce human exposure should be considered. Owing to the uncertainties involved with these estimates, HQ values between 1 and 10 may be of concern, particularly when additional significant risk factors are present (e.g., other contaminants are present at concentrations of concern). However, the level of concern does not

increase linearly as the RfD is approached or exceeded because RfDs do not have equal accuracy or precision; nor are RfDs based on the same severity of toxic effects (USEPA, 1989a).

In assessing health risks, all HQ values are representative of long-term exposures (i.e., exposures assumed to occur over a period of 9 or 30 years). The sum of more than one HQ value for multiple substances and/or multiple exposure pathways is the Hazard Index (HI). Adding the HQs does not account for any synergistic or antagonistic effects that may occur among chemicals. For this risk assessment, no attempt was made to distinguish between risk endpoints (e.g., target organs and related effects) when calculating the HI. Thus, this expression of total risk may be extremely conservative; it would be better to refine the HI to specific endpoints for HQ values greater than one. Additional limitations of HQ values and the segregation of hazard indexes have been described elsewhere (USEPA, 1989a).

6.2.2 Determination of Carcinogenic Effects

Unlike noncarcinogenic effects, carcinogenic substances are thought to pose some degree of risk at all exposure levels. These effects are estimated as the incremental probability of an individual developing cancer over a lifetime as a result of exposure to the potential carcinogen. This risk is computed using average lifetime exposure values that are multiplied by the oral slope factor for a particular chemical. Slope factors are used to convert estimated daily intakes averaged over a lifetime of exposure directly to the incremental risk of an individual developing cancer. The resulting carcinogenic risk estimate is generally an upper-bound estimate, because slope factors are usually based on upper 95th percentile confidence limits. The EPA believes it is prudent public health policy to consider actions to mitigate or minimize exposures to contaminants when estimated excess lifetime cancer risks exceed the 10^{-5} to 10^{-6} range, and when noncarcinogenic health risks are estimated to be significant (USEPA, 1988a).

Carcinogenic effects were summed for all chemicals in an exposure pathway. This summation of carcinogenic risks assumed that intakes of individual substances were small, that there were no synergistic or antagonistic chemical interactions, and that all chemicals produced the same effect (i.e., cancer). The limitations to this approach are discussed in detail elsewhere (USEPA, 1989a).

6.3 COMPARATIVE HUMAN HEALTH RISKS IN THE BUFFALO RIVER

6.3.1 Consumption of Contaminated Fish

The consumption of PCB-contaminated carp resulted in substantial noncarcinogenic risks, with the HQ > 1, and carcinogenic risks [i.e., greater than one person in a million (10^{-6})] for all remedial alternatives and fish consumption scenarios (Tables 6.1 and 6.2). The degree of risk increased as local residents of the Area of Concern consumed more locally caught carp. For noncarcinogenic risks, the HQ values ranged from 1.7 to 4.2 for typical exposures, 5.6 to 19 for reasonable maximum exposures, and 39 to 130 for subsistence exposures. For carcinogenic risks, the risks ranged from 3.4×10^{-5} to 8.3×10^{-5}

TABLE 6.1. HAZARD QUOTIENTS FOR NONCARCINOGENIC RISKS ASSOCIATED WITH CONSUMING WHOLE CARP UNDER VARIOUS REMEDIATION AND CONSUMPTION SCENARIOS

Remediation Scenario	Noncarcinogenic Risk		
	Typical	RME	Subsistence
UPSTREAM			
No Action	2.5	13	89
No Action, No Load	2.4	11	76
Environmental Dredging	1.7	10	70
Hamburg Cove	2.1	7.9	54
Hamburg Cove, No Load	1.8	5.6	39
DOWNSTREAM			
No Action	4.2	19	130
No Action, No Load	4.1	18	130
Environmental Dredging	1.7	11	76
Hamburg Cove	4.2	19	130
Hamburg Cove, No Load	4.1	18	120

TABLE 6.2. CARCINOGENIC RISKS ASSOCIATED WITH CONSUMING WHOLE CARP UNDER VARIOUS REMEDIATION AND CONSUMPTION SCENARIOS

Remediation Scenario	Lifetime Cancer Risk		
	Typical	RME	Subsistence
UPSTREAM			
No Action	4.9E-05	8.5E-04	5.8E-03
No Action, No Load	4.7E-05	7.3E-04	5.0E-03
Environmental Dredging	3.4E-05	6.7E-04	4.6E-03
Hamburg Cove	4.2E-05	5.2E-04	3.6E-03
Hamburg Cove, No Load	3.6E-05	3.7E-04	2.5E-03
DOWNSTREAM			
No Action	8.3E-05	1.3E-03	8.8E-03
No Action, No Load	8.1E-05	1.2E-03	8.4E-03
Environmental Dredging	3.4E-05	7.3E-04	5.0E-03
Hamburg Cove	8.3E-05	1.3E-03	8.8E-03
Hamburg Cove, No Load	8.1E-05	1.2E-03	7.9E-03

for typical exposures, 3.7×10^{-4} to 1.3×10^{-3} for reasonable maximum exposures, and from 2.5×10^{-3} to 8.8×10^{-3} for subsistence exposures. A greater degree of risk was observed in the downstream remediation scenarios than the upstream ones.

For the downstream remediation scenarios, some reduction in noncarcinogenic and carcinogenic risk was observed with environmental dredging, whereas the risk for the other four remediation scenarios was about the same. Slightly more variability was observed in the upstream scenarios; the lowest risk generally occurred for no upstream loading of the Hamburg Cove scenario. However, all of the risk estimates were still above levels of concern. On a comparison basis, this risk assessment exercise was useful in judging the adequacy of different remedial alternatives in reducing risk. For example, environmental dredging resulted in the quickest way by which risks to public health could be reduced during the first 3.5 years (i.e., 1280 days) of the upstream and downstream scenarios. However, the final modeled output at Day 3660 did not vary much among remediation actions for downstream and upstream scenarios. A risk manager would need to evaluate the costs and benefits from these results before deciding which remedial action, if any, to implement.

PCBs were the only chemical included in the exposure and risk assessment because of modeling limitations. There is a possibility that people who ingest, inhale, or have dermal contact with certain PCB mixtures may have a greater chance of incurring liver cancer; however, this statement is based on suggestive evidence rather than on verified data. Studies with three strains of rats and two strains of mice have verified the carcinogenicity of PCBs through the occurrence of hepatocellular carcinomas. This evidence was used to classify PCBs as a probable human carcinogen. Monkeys that ingested 0.005-0.08 mg/kg-day doses of Aroclor 1254 showed the following noncarcinogenic effects: ocular exudate, prominence and inflammation of the eyelid Meibomian glands, and distortion in nail bed formation. Similar changes have been documented in humans for accidental oral ingestion of PCBs (IRIS data base retrieval for PCBs, 1994).

This risk assessment assumed that all of the human health risk was attributable to the direct and indirect (e.g., food chain transfer) exposure of fish to contaminants in the sediments. Since carp are mostly benthic feeders that generally reside in a localized area, they were used as an indicator of local contamination problems. In addition, carp have a high lipid content which may readily accumulate contaminants through the ingestion and assimilation of contaminated food and possibly through the consumption of contaminated sediment while feeding.

These risk estimates are likely to be overly conservative because they are based on the consumption of whole fish, rather than fillets. The modeling exercise could not be conducted on fillets, and at the present time, extrapolations of contaminant concentrations from whole fish to fillets cannot be accurately made. In addition, these risk estimates were based on raw fish. At the present time, contaminant concentrations in raw fish cannot be accurately extrapolated to concentrations in cooked products. For the past 20 years, Mary Zabik and coworkers from Michigan State University have been investigating whether cooking methods can reduce pesticide and PCB residues in meat and fish (Smith et al., 1973; Stachiw et al., 1988; Zabik, 1974, 1990; Zabik et al., 1979, 1982). However, their

results have not been consistent between and within species of fish. In one instance, different cooking methods did not result in significant changes in the concentrations of PCBs, DDE, or DDT in cooked carp fillets (Zabik et al., 1982). In another case, cooking resulted in reductions of TCDD in restructured, deboned carp fillets (Stachiw et al., 1988).

The Michigan Department of Public Health and Michigan State University have conducted a joint investigation to further assess how cooking techniques may alter the concentrations of contaminants in fish (H. Humphrey, Michigan Department of Public Health, personal communication, 1994). Studies have been performed on a variety of sport fish, including chinook salmon, carp, and walleye, in the Great Lakes area for skin-on and skin-off fillets. Preliminary results indicate that:

- removal of skin produced up to a 50% reduction in contaminant concentrations in uncooked fillets due to removal of the fat layer below the skin

- cooking produced a 30 to 50% reduction in organic contaminant concentration for chemicals such as total PCBs, DDT, TCDD, and several pesticides

- the choice of cooking method made no significant difference in the reduction of contaminant concentrations in the tissues.

The resultsu of the Michigan study will be useful for future human health risk assessments for determining better estimates of contaminant concentrations in cooked fish. For the present time, anglers can use the following cooking techniques to reduce their risk to contaminants: (1) trim fatty areas, (2) puncture or remove skin before cooking so that fats drain away, or (3) deep-fry trimmed fillets in vegetable oil and discard the oil.

6.3.2 Consumption of Contaminated Surface Water

The noncarcinogenic and carcinogenic risks resulting from the ingestion of surface water while swimming under typical and reasonable maximum exposure scenarios were estimated to be far below levels of concern for the environmental dredging scenario (Table 6.3). These risks were estimated based on a modeled data set for copper, lead, benzo(a)anthracene, benzo(a)pyrene, and PCBs. Hazard Indices for noncarcinogenic risks ranged from 0.000006 to 0.00001 for typical and reasonable maximum exposures, respectively. Lifetime cancer risks were on the order of 10^{-10} for both scenarios.

Based on these estimated risks, an assumption was made that insignificant risks would also result from dermal exposure to surface water while swimming. This assumption was made because the direct intake of contaminants into the gut generally results in a greater intake of contaminants than the absorption of contaminants (with varying capacity for penetration) through the skin. This assumption has been supported by dermal exposure estimates at more contaminated sites in the Great Lakes region (Crane, 1994) which have shown negligible carcinogenic risk.

TABLE 6.3. NONCARCINOGENIC AND CARCINOGENIC RISKS ASSOCIATED WITH INGESTING CONTAMINATED SURFACE WATER WHILE SWIMMING IN THE BUFFALO RIVER: ENVIRONMENTAL DREDGING SCENARIO

Chemical	Mean Water Conc. (mg/L)	Hazard Index (Intake/RfD) Typical	Hazard Index (Intake/RfD) RME	Lifetime Cancer Risk (Intake*Slope Factor) Typical	Lifetime Cancer Risk (Intake*Slope Factor) RME
METALS					
Copper	2.6E-03	5.8E-06	1.2E-05		
Lead	2.3E-03				
ORGANIC COMPOUNDS					
Benzo(a)anthracene	9.9E-06			4.3E-11	2.9E-10
Benzo(a)pyrene	2.0E-05			5.5E-11	3.6E-10
PCBs	2.0E-06	2.9E-07	5.9E-07	5.8E-12	3.9E-11
CUMULATIVE RISK		0.000006	0.00001	1E-10	7E-10

Blank spaces denote chemicals lacking toxicity data.

6.3.3 Additive Risks

Risks may be added among exposure pathways to yield an overall estimate of risk to the human population. For the Buffalo River AOC, the risk associated with consuming fish far outweighed that of surface water ingestion. Therefore, the additive risks correspond to the fish consumption risks.

CHAPTER 7

COMPARATIVE RISK CHARACTERIZATION: WILDLIFE

7.1 INTRODUCTION

A simple hazard assessment model, developed by Kubiak and Best (1991), was used to assess comparative risks to mink in the Buffalo River AOC. The same model was used in the baseline wildlife risk assessment (Mann-Klager, 1993). The model basically uses a quotient method by which the forage contaminant concentration is compared to either the lowest observed adverse effect level (LOAEL) or no observed adverse effect level (NOAEL) for contaminants of concern. To obtain a reasonable amount of protection, this ratio should not exceed one. Two other techniques which incorporate either sensitive lifestage information or a biomagnification factor are discussed by Kubiak and Best (1991); these latter techniques were not relevant to this hazard assessment.

The model provides a way to determine the exceedance over NOAEL by using the derived dietary NOAEL/LOAEL as the target environmental forage concentration (Kubiak and Best, 1991). This target forage concentration is divided into the environmental contaminant concentration measured in the locally caught forage to determine the exceedance over NOAEL.

$$\text{Exceedance over NOAEL} = \frac{\text{Measured Forage}}{\text{Target Forage}} \qquad \text{eqn. 1}$$

7.2 COMPARATIVE RISKS TO MINK

Mink was used as a representative piscivorous species inhabiting the Buffalo River AOC. Thus, any adverse effects observed in mink may signal a hazard to other piscivorous species.

The amount by which potential mink forage exceeded the NOAEL was determined using equation 1 and the 10-year modeled carp data from Table 4.5. The only contaminant of concern included in this assessment was total PCBs. The NOAEL was exceeded by 19 to 46 times for the various remedial alternatives (Table 7.1). Environmental dredging resulted in the least risk to mink, whereas the other downstream remediation scenarios were almost identical in risk. In the upstream remediation scenarios, there was little difference between the environmental dredging and Hamburg Cove - no loading scenarios. A greater exceedance of the NOAEL was observed for the other upstream remediation

TABLE 7.1. COMPARATIVE RISKS TO MINK RESULTING FROM THE CONSUMPTION OF CONTAMINATED CARP FOR VARIOUS REMEDIATION ALTERNATIVES

Remediation Scenario	Exceedance over NOAEL*
UPSTREAM	
No Action	28
No Action – No Load	26
Environmental Dredging	19
Hamburg Cove	23
Hamburg Cove – No Load	20
DOWNSTREAM	
No Action	46
No Action – No Load	45
Environmental Dredging	19
Hamburg Cove	46
Hamburg Cove – No Load	45

* NOAEL for mink exposed to PCBs = 0.069 ug/g wet weight

scenarios. The exceedance values in Table 7.1 are overly conservative because an assumption was made that local mink populations consumed 100% of their diet from contaminated carp.

PCBs are known to cause reproductive and behavioral impairments in mammals. Mink have been found to be one of the most sensitive mammals to PCBs (Eisler, 1986). The status and health of the mink population in the Buffalo River system is not known. The results of this comparative risk assessment warrant additional work to gain a better idea of the local diet of mink in the Buffalo River AOC. In addition, a more detailed ecological risk assessment should be done to assess the exposure and potential risks of all contaminants of concern to mink and other indicator species.

CHAPTER 8

CHARACTERIZATION OF QUALITATIVE UNCERTAINTIES

8.1 INTRODUCTION

A number of assumptions and estimated values were used in the comparative risk assessments that contributed to the level of uncertainty about the risk estimates. For most environmental risk assessments, the uncertainty of the risk estimates ranges over an order of magnitude or greater (USEPA, 1989a). A qualitative listing of the uncertainties associated with each step in the risk assessment process will be given below to determine the impact of these uncertainties on the final risk assessment results.

8.2 QUALITATIVE LIST OF UNCERTAINTIES: HUMAN HEALTH

8.2.1 Exposure Assessment

The exposure assessment was based on modeled data. Although the models were verified, uncertainty still exists about the assumptions used to derive the contaminant concentrations in fish and water under different remedial alternatives. Specific limitations of the modeling effort were given in Section 3.2.2.3. The modeling effort was hampered by the limited data set used to verify the models. A medium to high level of uncertainty was probably associated with using these data in the exposure assessment.

Additional uncertainties associated with the data and assumptions used in the human health exposure assessment are given below.

- **An adequate assessment of complete and incomplete exposure pathways was made.** There is a low uncertainty that some exposure pathways were either not identified or else were incorrectly classified as a complete or incomplete exposure pathway.

- **The exclusion of the complete exposure pathways of dermal exposure to surface water from the exposure assessment was justifiable because of the low probability that these pathways would result in substantial human health risks.** There is low uncertainty associated with this assumption.

- **The complete exposure pathways chosen for the exposure assessment represent the primary pathways by which people in the Buffalo River are exposed to contaminants.** The pathways chosen were based primarily on observed activities and on available data. Thus the level of uncertainty is low.

- **Modeled data were only available for a 10-year duration.** The data were extrapolated over 30 years for use in the reasonable maximum and subsistence exposure scenarios. The uncertainties associated with doing this, beyond those resulting from the model

itself, are low to medium because the contaminant concentrations from the model leveled off after 10 years.

- **Carp was the only species of fish for which modeled PCB data were available.** A medium to high level of uncertainty was associated with using carp data because other species of fish (e.g., walleye and perch) would more likely be consumed by people.

- **The number of chemicals included in the modeled data were limited. Other chemicals (e.g., dieldrin, chlordane) not included may contribute to risk.** However, PCBs caused the majority of carcinogenic risk in the baseline risk assessment (Crane, 1993a), and the uncertainty associated with not including other chemicals is probably low. PCBs also contributed to most of the noncarcinogenic risk in this risk assessment, and the uncertainty of not including other chemicals is probably low as well.

- **The fish consumption exposure assessment was based on raw, whole carp.** In reality, most fish would be cut into fillets and cooked. This would reduce the contaminant burden in the cooked fish. The uncertainty associated with using raw, whole carp is medium to high.

- **The selection of exposure frequencies and durations, body weight, life expectancy, and population characteristics were appropriate.** The values for body weight, life expectancy, and exposure frequency were based on EPA guidance (USEPA, 1989a,b; 1991a) and have a low to moderate level of uncertainty associated with them. Similar levels of uncertainty may be attributed to professional judgements about the fraction of fish ingested from contaminated sources and the number of times someone would go swimming in the Buffalo River.

- **The exposure assessment only estimated contaminant intakes after a remedial alternative was implemented and not prior to implementation.** This was used so that the effectiveness of the remedial alternatives could be judged more easily. As such, this risk assessment provided a paper exercise to assist risk managers with making decisions concerning remediation. If one of the remedial alternatives presented in this report were ever implemented, it would be useful to collect follow-up data on contaminant concentrations in fish to compare with the modeled predictions.

8.2.2 Toxicity Values

The oral RfDs and oral slope factors used in this risk assessment were either verified values obtained from IRIS or interim values obtained from other sources. RfDs and slope factors are subject to change as result of new information and updates of the IRIS data base. In addition, chemicals will be added to IRIS in the future to expand the data base. Thus, this risk assessment is "dated" to the toxicity values available at the time it was prepared. Listed below are the uncertainties associated with using these toxicity values.

- **RfD values and oral slope factors have uncertainty associated with them.** Uncertainty and modifying factors are incorporated into the calculation of RfDs (see Appendix C) and take into consideration factors such as extrapolating data from long-term animal studies to humans. In general, RfD values have an uncertainty range of about one order of magnitude. Since oral slope factors represent an estimate of an upper-bound lifetime

probability of an individual developing cancer, these values are already conservative. Thus, the amount of uncertainty associated with oral slope factor values is minimal.

- **Toxicity values were not available for all of the chemicals detected in the Buffalo River.** A risk characterization could not be done for lead because the U.S. EPA's Carcinogen Assessment Group recommends that a numerical estimate not be used for an oral slope factor. This recommendation was made because quantifying lead's cancer risk involves many uncertainties, some of which may be unique to lead. The uncertainty of not being able to include lead in the risk assessment is unknown.

8.2.3 Risk Characterization

The uncertainties associated with the risk characterization step are listed below.

- **Exposure intakes and toxicity values will remain the same over the exposure duration.** This assumes that human activities and contaminant levels will remain the same over the exposure duration, and that toxicity values will not be updated. A moderate to high level of uncertainty is probably associated with this assumption since toxicity values are frequently updated in the IRIS data base as new information becomes available. The level of uncertainty will probably increase with longer exposure durations.

- **Health risks are additive for both noncarcinogenic and carcinogenic effects.** The uncertainty associated with this assumption is unknown for the ingestion of surface water pathway. The toxicity exhibited by a mixture of chemicals may involve synergistic and antagonistic effects. However, no guidelines are available to judge the complex interactions a mixture of contaminants may possess in terms of its potential toxicity to humans. At the present time, standard risk assessment guidance assumes that health risks are additive.

- **The risk characterization only included substances for which data were available.** The potential contribution of other substances expected to be present is probably low.

8.3 QUALITATIVE LIST OF UNCERTAINTIES: WILDLIFE

The comparative risk assessment for wildlife was limited to mink and used the ecological risk assessment techniques common to a screening assessment. As such, the wildlife assessment was designed to be very protective. A similar range of uncertainties, as for human health, can be identified for:

- accuracy of modeled carp data
- selection of exposure pathways
- choice of carp, as opposed to other fish species, for use in the exposure assessment
- exclusion of other chemicals in the exposure assessment.

In addition, some uncertainties are specific to the wildlife risk assessment including:

- **Selection of a NOAEL from the literature.** The uncertainty associated with this NOAEL is probably low to moderate.

- **Exclusion of other species (e.g., common terns) from the wildlife risk assessment due to the lack of modeled contaminant data for other species of fish.** A high degree of uncertainty is associated with this assumption.

- **Exclusion of specific information about habitat utilization in the AOC.** The uncertainty associated with this lack of information is medium to high.

8.4 SUMMARY

Based on the current information available, a complete description of the level of uncertainty associated with all of the assumptions and data used in this risk assessment cannot be made. This comparative human health and wildlife risk assessment was based on data and assumptions that, in reality, represent a snapshot in time. One of the greatest sources of uncertainty in this risk assessment arose from the use of modeled data. The overall uncertainty of these risk estimates varies by over an order of magnitude.

This risk assessment was useful for illustrating the process by which a comparative risk assessment could be conducted. This same methodology could be applied to other contaminated sediment sites.

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APPENDIX A

EXECUTIVE SUMMARY OF THE BASELINE HUMAN HEALTH RISK ASSESSMENT

A.1 OVERVIEW

The Assessment and Remediation of Contaminated Sediments (ARCS) program, a 5-year study and demonstration project relating to the control and removal of contaminated sediments from the Great Lakes, is being coordinated and conducted by the U.S. Environmental Protection Agency's (EPA) Great Lakes National Program Office (GLNPO). As part of the ARCS program, baseline human health risk assessments have been performed at five Areas of Concern (AOCs) in the Great Lakes region. The Buffalo River, located in western New York State, is one of these AOCs.

In this report, exposure and risk assessment guidelines, developed for the EPA Superfund program, have been applied to determine the baseline human health risks associated with direct and indirect exposures to contaminated sediments in the lower Buffalo River. These risks were estimated for noncarcinogenic (e.g., reproductive toxicity, teratogenicity, liver toxicity) and carcinogenic (i.e., probability of an individual developing cancer over a lifetime) effects, based on currently available data. The risk estimates were not extrapolated to potential future scenarios.

A.2 STUDY AREA

This baseline risk assessment covers an area adjacent to the lower Buffalo River as it passes through Buffalo, NY before entering Lake Erie. This area has a history of water quality problems due primarily to point sources of contaminants (i.e., industrial and municipal discharges). The extent of contamination in the Buffalo River led to the International Joint Commission's (IJC) decision to designate this region as a Great Lakes AOC. In response, the New York State Department of Environmental Conservation (NYSDEC) has completed one phase of a remedial action plan (RAP) to identify and implement pollution abatement measures for the Buffalo River AOC (NYSDEC, 1989).

High concentrations of heavy metals, polychlorinated biphenyls (PCBs), polynuclear aromatic hydrocarbons (PAHs), and pesticides have been measured in different compartments of the Buffalo River (e.g., sediments, water column, and fish). Fish advisories have been issued against consuming carp from the Buffalo River because of excessive levels of PCBs. The transport of these contaminants into Lake Erie is also of concern. However, it was beyond the scope of this risk assessment to estimate human health risks to people using the nearshore areas of Lake Erie.

A.3 EXPOSURE ASSESSMENT

Contact and noncontact recreational activities are limited along the Buffalo River. No public bathing facilities exist along the river, and fish consumption advisories have warned the public not to eat carp from the river. However, there is anecdotal evidence that these activities occur, even near industrial outfalls. This assessment focused on two complete pathways by which residents of the lower Buffalo River could be exposed to sediment-derived contaminants: (1) consumption of contaminated carp and spottail/emerald shiners, and (2) ingestion of surface water while swimming. A third complete pathway of dermal exposure to surface water was assumed to be insignificant in comparison to the risk resulting from the ingestion of contaminated surface water. This assumption was made because contaminants are more easily transported across the gut than the skin. Data for other exposure pathways were determined to be incomplete (e.g., ingestion of sediments).

A limited data set of fish contaminant concentrations was available for use in the exposure assessment. Carp from three different age classes (i.e., young, middle, and old), collected as part of the ARCS program, were used. Carp generally represent the most contaminated fish in water bodies due to their benthic feeding habits and high lipid content. Data from young-of-the-year spottail/emerald shiners were used to represent another type of fish. Young-of-the-year fish are an important food source for a variety of fish species consumed by humans. If young-of-the-year fish were the sole food source of piscivores, they could be used as an indicator of chemical contaminants that may be present in fish consumed by humans.

Since many species of fish travel between the river and Lake Erie, there is some uncertainty as to where the fish accumulated their contaminant burden. For the purpose of this risk assessment, it was assumed that fish collected from the mouth of the river accumulated most of their contaminant burden from the lower Buffalo River.

Noncarcinogenic and carcinogenic risks were estimated for typical, reasonable maximum, and subsistence (fish only) exposure scenarios. Typical (i.e., average) exposures were assumed to occur over a period of 9 years, whereas reasonable maximum (i.e., the maximum exposure that is reasonably expected to occur at a site) and subsistence (i.e., reliance on fish as a major source of protein) exposures were assumed to occur over a period of 30 years (USEPA, 1989a). These exposure durations were extrapolated over a period of 70 years for estimating carcinogenic risks. Exposure intakes were determined for each chemical and added for each exposure pathway.

For each of the fish exposure scenarios, different consumption patterns of fish were assumed to take place (Table A.1). These consumption patterns were based, in part, on recommended values given in EPA Superfund guidance (USEPA, 1989a,b; 1991a), on published literature values, or on study assumptions. Based on an average meal of fish (i.e., 150 g or 0.33 lb), the amount of Buffalo River fish consumed for each exposure scenario (Table A.1) can also be converted to meals per year using the following equation:

TABLE A.1. AMOUNT OF FISH ASSUMED TO BE CONSUMED PER PERSON PER DAY FROM THE BUFFALO RIVER FOR EACH EXPOSURE SCENARIO

Exposure Scenario	Ingestion Rate* (g/day)	X	FI**	Amount of Buffalo R. Fish Consumed (g/day)
Typical	19.2		0.10	1.92
Reasonable Maximum	54		0.25	13.5
Subsistence	132		0.70	92.4

* Sources: Typical (West et al., 1989); Reasonable Maximum and Subsistence (USEPA, 1991a)

** FI = Fraction of fish (i.e., carp or spottail/emerald shiner) estimated to be ingested from the Buffalo River (study assumption).

Ingestion Rate (meals/yr) = [Ingestion Rate (g/day)] x FI x (meal/150 g) x (365 days/yr)

The number of meals of Buffalo River fish consumed over a year-long period for typical, reasonable maximum, and subsistence exposures corresponded to approximately 4.5, 33, and 225 meals, respectively.

A number of heavy metals and organic compounds were included in the exposure assessment. Toxicity values for the chemicals detected in the media of interest were obtained from the EPA's Integrated Risk Information System (IRIS) data base.

A.4 RISK ASSESSMENT

A.4.1 Determination of Risk

Noncarcinogenic effects were evaluated by comparing an exposure level over a specified time period with a reference dose (RfD)[1] derived from a similar exposure period [otherwise known as a hazard quotient (HQ)]. Thus, HQ = exposure level/RfD. An HQ value of less than 1 indicates that exposures are not likely to be associated with adverse noncarcinogenic effects. HQ values between 1 and 10 may be of concern, particularly when additional significant risk factors are present (e.g., other contaminants are present at concentrations of concern) (USEPA, 1989a). HQ values for multiple substances and/or multiple exposure pathways were summed to yield an overall Hazard Index (HI). The HIs are interpreted

[1] The RfD provides an estimate of the daily contaminant exposure that is not likely to cause harmful effects during either a portion of a person's life or their entire lifetime (USEPA, 1989a).

in the same fashion as the HQs. Summing the HQs does not account for any synergistic or antagonistic effects that may occur among substances.

Carcinogenic risks were estimated as the incremental probability of an individual developing cancer over a lifetime as a result of exposure to potential carcinogens. This risk was computed using average lifetime exposure values that were multiplied by the oral slope factor[2] for a particular chemical. The resulting carcinogenic risk estimate generally represented an upper-bound estimate, because slope factors are usually based on upper 95th percentile confidence limits. Carcinogenic effects were summed for all chemicals in an exposure pathway. This summation of carcinogenic risks assumed that intakes of individual substances were small, that there were no synergistic or antagonistic chemical interactions, and that all chemicals caused cancer. The EPA believes it is prudent public health policy to consider actions to mitigate or minimize exposures to contaminants when estimated, upper-bound excess lifetime cancer risks exceed the 10^{-5} to 10^{-6} range, and when noncarcinogenic health risks are estimated to be significant (USEPA, 1988a).

A.4.2 Noncarcinogenic Risks

A summary of noncarcinogenic risks, as represented by the Hazard Indices, is given in Table A.2. Noncarcinogenic risks were below levels of concern (i.e., HI < 1) for typical and reasonable maximum exposure levels for the fish consumption and surface water ingestion pathways. An assumption was made that dermal exposure to surface water while swimming would also be insignificant. The risk levels were of concern (i.e., HI ranged from 2 to 4) for subsistence anglers and their families who consumed carp from the Buffalo River. Most of the risk was attributable to dieldrin contamination.

Because some of the chemicals detected in the fish do not presently have EPA approved RfD values (e.g., PCBs), this assessment may underestimate the noncarcinogenic risks from consuming fish from the lower Buffalo River area. The noncarcinogenic risk reported here is an estimated risk based on currently available data and toxicity information and should not be construed as an absolute risk.

A.4.3 Carcinogenic Risks

The estimated, upper-bound carcinogenic risks for all fish consumption exposure scenarios were at or above levels of concern (i.e., 10^{-5} to 10^{-6} range) (Table A.3). The carcinogenic risk increased with the age class of carp, and the risk increased by about an order of magnitude for each exposure scenario from typical to reasonable maximum to subsistence exposures. Spottail/emerald shiners presented less risk to consumers by at least an order of magnitude, perhaps because of their young age and limited time for accumulating contaminants.

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[2] Slope factors are estimated through the use of mathematical extrapolation models for estimating the largest possible linear slope (within 95% confidence limits) at low extrapolated doses that is consistent with the data (USEPA, 1989a).

TABLE A.2. SUMMARY OF NONCARCINOGENIC RISKS ASSOCIATED WITH TWO EXPOSURE PATHWAYS IN THE BUFFALO RIVER AOC*

Exposure Pathway	Age Class	Exposure Scenario		
		Typical	RME	Subsistence
Fish Consumption				
Carp	Young	0.04	0.3	2
Carp	Middle	0.05	0.4	2
Carp	Old	0.08	0.6	4
Surface Water Ingestion	-	0.002	0.005	-

* Non-carcinogenic risks were averaged over the same period as the exposure duration.

TABLE A.3. SUMMARY OF CARCINOGENIC RISKS ASSOCIATED WITH TWO EXPOSURE PATHWAYS IN THE BUFFALO RIVER AOC*

Exposure Pathway	Age Class	Exposure Scenario		
		Typical	RME	Subsistence
Fish Consumption				
Carp	Young	5E-05	1E-03	9E-03
Carp	Middle	8E-05	2E-03	1E-02
Carp	Old	1E-04	3E-03	2E-02
Spottail/Emerald Shiners	Young-of-the-Year	4E-06	9E-05	6E-04
Surface Water Ingestion	-	6E-08	4E-07	-

* Carcinogenic risks were averaged over a period of 70 years (i.e., average lifetime of an individual).

PCBs accounted for most of the carcinogenic risk from fish consumption. There is a possibility that people who ingest, inhale, or have dermal contact with certain PCB mixtures may have a greater chance of incurring liver cancer; however, this statement is based on suggestive evidence rather than on verified data (IRIS database retrieval for PCBs, 1993).

The carcinogenic risk associated with ingesting surface water while swimming ranged from 6×10^{-8} to 4×10^{-7} for typical and reasonable maximum exposures, respectively. Because these risk estimates were below levels of concern, it was also assumed that dermal exposure to surface water would also result in an insignificant carcinogenic risk.

A.4.4 Uncertainties

Several assumptions and estimated values were used in this baseline risk assessment that contributed to the overall level of uncertainty associated with the noncarcinogenic and carcinogenic risk estimates. As with most environmental risk assessments, the uncertainty of the risk estimates probably ranges over an order of magnitude or greater. The uncertainties were addressed in a qualitative way for the parameters and assumptions that appeared to contribute the greatest degree of uncertainty. One of the greatest sources of uncertainty was the assumption that exposure intakes and toxicity values would not change during the exposure duration. This assumption requires that human activities and contaminant concentrations remain the same over the exposure duration, and that toxicity values would not be updated.

APPENDIX B

WILDLIFE FOUND IN THE BUFFALO RIVER AOC

The following tables provide a list of wildlife species found in the Buffalo River AOC.

TABLE B-1. ENDANGERED, THREATENED, AND SPECIAL CONCERN SPECIES OF NEW YORK STATE OBSERVED NEAR THE BUFFALO RIVER AOC

Common Name	Scientific Name	Status
Least Bittern	*Ixobrychus exilis*	Special Concern
Osprey	*Pandion haliaetus*	Threatened
Northern Harrier	*Circus cyaneus*	Threatened
Cooper's Hawk	*Accipiter cooperii*	Special Concern
Red-shouldered Hawk	*Buteo lineatus*	Threatened
Peregrine Falcon*	*Falco peregrinus*	Endangered
Common Tern	*Sterna hirundo*	Threatened
Black Tern	*Chlidonias niger*	Special Concern
Short-eared Owl	*Asio flammeus*	Special Concern
Common Nighthawk	*Chlordeiles minor*	Special Concern
Eastern Bluebird	*Sialia sialis*	Special Concern

* Also Federally Endangered

TABLE B-2. MAMMMALS, REPTILES, AND AMPHIBIANS RECORDED AT TIMES BEACH CONFINED DISPOSAL SITE, BUFFALO, NEW YORK (ANDRLE, 1986 CITED IN MANN-KLAGER, 1993)

Common Name	Scientific Name
Muskrat	*Ondatra zibethicus*
Eastern Cottontail	*Sylvilagus floridanus*
Raccoon	*Procyon lotor*
White-footed Mouse	*Peromyscus leucopus*
Meadow Vole	*Microtus pennsylvanicus*
Red Fox	*Vulpes fulva*
Weasel	*Mustela* sp.
Eastern Garter Snake	*Thamnophis sirtalis*
American Toad	*Bufo americanus*
Bullfrog	*Rana catesbeiana*

TABLE B-3. MAMMALS OBSERVED AT TIFFT NATURE PRESERVE, BUFFALO, NEW YORK (LANDSITTEL, 1990 CITED IN MANN-KLAGER, 1993).

Common Name	Scientific Name
Opossum	*Didelphis marsupialis*
Masked shrew	*Sorex cinereus*
Shorttail shrew	*Blarina brevicauda*
Longtail shrew	*Sorex dispar*
Starnose mole	*Condylura cristata*
Big brown bat	*Eptesicus fuscus*
Raccoon	*Procyon lotor*
Shorttail weasel	*Mustela erminea*
Longtail weasel	*Mustela frenata*
Mink	*Mustela vison*
Striped skunk	*Mephitis mephitis*
Coyote	*Canis latrans*
Red fox	*Vulpis fulva*
Gray fox	*Urocyon cinereoargenteus*
Woodchuck	*Marmota monax*
Eastern chipmunk	*Tamias striatus*
Eastern gray squirrel	*Sciurus carolinensis*
Beaver	*Castor canadensis*
White-footed mouse	*Peromyscus leucopus*
Deer mouse	*Peromyscus maniculatus*
Meadow vole	*Microtus pennsylvanicus*
Muskrat	*Ondatra zibethica*
Norway rat	*Rattus norvegicus*
House mouse	*Mus musculus*
Meadow jumping mouse	*Zapus hudsonius*
Woodland jumping mouse	*Mapaeozapus insignis*
Eastern cottontail	*Sylvilagus floridanus*
Whitetail deer	*Odocoileus virginianus*

APPENDIX C

HUMAN HEALTH TOXICITY ASSESSMENT INFORMATION

C.1 TOXICITY ASSESSMENT

The toxicity assessment step is an integral part of the human health baseline risk assessment. This step includes four tasks: (1) gather qualitative and quantitative toxicity information for substances being evaluated, (2) identify exposure periods for which toxicity values are necessary, (3) determine toxicity values [i.e., reference doses (RfDs)] for noncarcinogenic effects, and (4) determine toxicity values (i.e., slope factors) for carcinogenic effects (USEPA, 1989a). The EPA has performed the toxicity assessment step for a limited number of chemicals and these assessments have undergone extensive peer review. Therefore, the toxicity assessment step of this study involves primarily a compilation of available toxicity data.

Once a "verified" toxicity value is agreed upon by the EPA's toxicologists, it is entered into the EPA's Integrated Risk Information System (IRIS) data base; these values are updated as necessary. IRIS is the primary source of toxicity information used in baseline risk assessments. The Health Effects Assessment Summary Tables (HEAST) are the second most current source of toxicity information and include both verified and interim RfD and slope factor values. Interim values are used for chemicals that have not yet been approved by the EPA. Specific EPA workgroups, such as the Carcinogen Risk Assessment Verification Endeavor (CRAVE) and RfD Workgroups, are another source of interim toxicity values. If toxicity values are not available in the aforementioned sources, then interim values from other reports may be used.

This appendix includes brief descriptions of the most important toxicity values used to evaluate noncarcinogenic and carcinogenic effects; these subsections were summarized from the EPA guidance document: "Risk Assessment Guidance for Superfund. Volume 1. Human Health Evaluation Manual (Part A)" (USEPA, 1989a).

C.1.1 Noncarcinogenic Chronic Toxicity

The RfD is the toxicity value used most often in evaluating noncarcinogenic effects. RfDs are based on the assumption that thresholds exist for certain toxic effects (e.g., cellular necrosis) but may not exist for other toxic effects (e.g., carcinogenicity). The RfD is defined as an estimate of the daily exposure to the human population that is likely to be without an appreciable risk of deleterious effects during either a portion of the lifetime (i.e., subchronic RfD or "RfD_s") or during the lifetime (i.e., chronic RfD or "RfD"). This toxicity value has an uncertainty range of about an order of magnitude and

includes exposures to sensitive subgroups in the population. For each chemical, the RfD is calculated from the following equation:

$$RfD = \frac{NOAEL \text{ or } LOAEL}{UF \times MF}$$

where:

NOAEL = No-Observed-Adverse-Effect-Level
LOAEL = Lowest-Observed-Adverse-Effect-Level
MF = Modifying Factor
U = Uncertainty Factor

The NOAEL and LOAEL are derived from dose-response experiments. The NOAEL represents the highest exposure level tested at which no adverse effects occurred (including the critical toxic effect), whereas the LOAEL represents the lowest exposure level at which significant adverse effects occurred. Uncertainty factors usually consist of multiples of ten, with each factor representing a specific area of uncertainty included in the extrapolation from available data. An uncertainty factor of ten is usually used to account for variation in the general population so that sensitive subpopulations are protected. An additional ten-fold factor is usually applied for each of the following extrapolations: from long-term animal studies to humans, from a LOAEL to a NOAEL, and when subchronic studies are used to derive a chronic RfD. A modifying factor (MF), ranging from >0 to 10, is included as a qualitative assessment of additional uncertainties; the default value for the MF is one.

For Aroclor 1254, an oral RFD of 2×10^{-5} mg/kg/day has been approved for the IRIS data base. An uncertainty factor of 300 and a modifying factor of 1 were assigned to this RfD value. This was based on clinical and immunologic studies with monkeys.

C.1.2 Carcinogenicity

Human carcinogenic risks are usually evaluated for a chemical by using its slope factor (formerly designated as a cancer potency factor) and corresponding weight-of-evidence classification. These variables were listed in Table 5.2 for the Buffalo River chemicals. Slope factors are estimated through the use of mathematical extrapolation models, most commonly the linearized multistage model, for estimating the largest possible linear slope (within 95% confidence limits), at low extrapolated doses, that is consistent with the data. The slope factor is characterized as an upper-bound estimate so that the true risk to humans, while not identifiable, is not likely to exceed the upper-bound estimate.

The weight of evidence classification for a particular chemical is determined by the EPA's Human Health Assessment Group (HHAG). Chemicals are placed into one of five groups according to the weight of evidence from epidemiological studies and animal studies. These groups are designated by the letters A, B, C, D, and E which represent the level of carcinogenicity to humans (see Table 5.1). Quantitative

carcinogenic risk assessments are performed for chemicals in Groups A and B, and on a case-by-case basis for chemicals in Group C.

C.2 UNCERTAINTIES

A number of uncertainties are involved with using toxicity values for estimating noncarcinogenic and carcinogenic risks. Some of these qualitative uncertainties are listed below:

- Using dose-response information from healthy animal or human populations to predict effects that may occur in the general population, including susceptible subpopulations (e.g., elderly, children),
- Using dose-response information from animal studies to predict effects that may occur in human populations,
- Using NOAELs derived from short-term animal studies to predict effects that may occur in humans during long-term exposures,
- Using dose-response information from effects observed at high doses to predict the adverse health effects that may occur following exposure of humans to low levels of the chemical in the environment, and
- Using a toxicity value derived from exposure to a particular chemical mixture (e.g., Aroclor 1260) to represent the level of carcinogenic toxicity for other similar chemical mixtures (e.g., Aroclor 1242, 1248, and 1254).

APPENDIX D

TOXICITY PROFILES

The following toxicity profiles were largely derived from those given in U.S. EPA (1993b) and the EPA's IRIS data base. Although other organic chemicals (i.e., dieldrin, chlordane, DDD, DDE) were found to contribute to baseline human health (Crane, 1993a) and wildlife (Mann-Klager, 1993) risks, these chemicals were not included in the comparative risk assessments due to time and cost constraints.

D.1 POLYCHLORINATED BIPHENYLS (PCBs)

PCBs are base/neutral compounds that are formed by direct chlorination of the biphenyl ring. There are 209 different PCB compounds, termed congeners, based on the possible chlorine substitution patterns ($C_{12}H_{10-N}Cl_N$, where N = 1-10). PCBs were manufactured by Monsanto under the trade name Aroclor. Aroclors contain a mixture of congeners, and are named with numbers which indicate the weight percent of chlorine in each mixture (e.g., Aroclor 1242 represents 42% chlorination of the biphenyl ring).

PCBs have been widely used in industrial systems. Important properties of PCBs for industrial applications include thermal stability, fire and oxidation resistance, and solubility in organic compounds (Hodges, 1977). PCBs were used as insulating fluids in electrical transformers and capacitors, as plasticizers, as lubricants, as fluids in vacuum pumps and compressors, and as heat transfer and hydraulic fluids (Hodges, 1977; Nimmo, 1985). Although use of PCBs as a dielectric fluid in transformers and capacitors was generally considered a closed-system application, the uses of PCBs, especially during the 1960s, were broadly expanded to many open systems where losses to the environment were likely. Heat transfer systems, hydraulic fluids in die cast machines, and uses in speciality inks are examples of more open-ended applications that resulted in serious contamination in fish near industrial discharge points (Hesse, 1976).

Although PCBs were once used extensively by industry, their production and use in the United States was banned by the EPA in July 1979 (Miller, 1979). Prior to 1979, the disposal of PCBs and PCB-containing equipment was not subject to Federal regulation. Prior to regulation, of the approximately 1.25 billion pounds purchased by U.S. industry, 750 million pounds (60 percent) were still in use in capacitors and transformers, 55 million pounds (4 percent) had been destroyed by incineration or degraded in the environment, and over 450 million pounds (36 percent) were either in landfills or dumps or were available to biota via air, water, soil, and sediments (Durfee et al., 1976).

PCBs are extremely persistent in the environment and are bioaccumulated throughout the food chain (Eisler, 1986; Worthing, 1991). There is evidence that PCB health risks increase with increased chlorination because more highly chlorinated PCBs are retained more efficiently in fatty tissues (IRIS,

1994). However, individual PCB congeners have widely varying potencies for producing a variety of adverse biological effects including hepatoxicity, developmental toxicity, immunotoxicity, neurotoxicity, and carcinogenicity. The non-ortho-substituted coplaner PCB congeners, and some of the mono-ortho-substituted congeners, have been shown to exhibit "dioxin-like" effects (Golub et al., 1991; Kimbrough and Jensen, 1989; McConnell, 1980; Poland and Knutson, 1982; Safe, 1985, 1990; Tilson et al., 1990; USEPA 1993c). The neurotoxic effects of PCBs appear to be associated with some degree of ortho-chlorine substitution. There is increasing evidence that many of the toxic effects of PCBs result from alterations in hormonal function. However, because PCBs can act directly as hormonal agonists or antagonists, PCB mixtures may have complex interactive effects in biological systems (Korach et al., 1988; Safe et al., 1991; Shain et al., 1991; USEPA, 1993c). Because of the lack of sufficient toxicological data, the EPA has not developed quantitative estimates of health risk for specific congeners.

A recent summary of the National Contaminant Biomonitoring Program data from 1976 through 1984 indicated a significant downward trend in total PCBs in fish, although PCB residues in fish tissue remained widespread (Schmitt et al., 1990). Total PCBs were detected at 91 percent of 374 sites surveyed in the National Study of Chemical Residues in Fish (USEPA, 1992a, 1992b). Currently, PCB contamination in fish and shellfish has resulted in the issuance of consumption advisories in 31 states (RTI, 1993).

PCBs may be analyzed quantitatively as Aroclor equivalents or as individual congeners. Historically, Aroclor analysis has been performed by most laboratories. This procedure can, however, result in significant error in determining total PCB concentrations (Schwartz et al., 1987) and in assessing the toxicologic significance of PCBs, because it is based on the assumption that the distribution of PCB congeners in environmental samples and parent Aroclors is similar.

The distribution of PCB congeners in Aroclors may be altered considerably by physical, chemical, and biological processes after release into the environment, particularly when the process of biomagnification is involved (Norstrom, 1988; Oliver and Niimi, 1988; Smith et al., 1990). Recent aquatic environmental studies indicate that many of the most potent, dioxin-like PCB congeners are preferentially accumulated in higher organisms (Bryan et al., 1987; Kubiak et al., 1989; Oliver and Niimi, 1988). This preferential accumulation probably results in a significant increase in the total toxic potency of PCB residues as they move up the food chain. Consequently, the congener-specific analysis of PCBs is required for more accurate determination of total PCB concentrations and for more rigorous assessment of the toxicological effects of PCBs.

D.2 POLYCYCLIC AROMATIC HYDROCARBONS (PAHs)

PAHs are base/neutral organic compounds that have a fused ring structure of two or more benzene rings. PAHs with two to five benzene rings are generally of greatest concern for environmental and human health effects. These include benzo[a]pyrene and benzo[a]anthracene. Benzo[a]pyrene, one

of the most widely occurring and potent PAHs, and several other PAHs have been classified by the EPA as probable human carcinogens (B2) (IRIS, 1992). Although benzo[a]pyrene is one of the most toxicologically potent PAHs, it may represent only a small fraction of the total PAH concentration in fish or shellfish tissue. Evidence for the carcinogenicity of PAHs in humans come primarily from epidemiologic studies that have shown an increased mortality due to lung cancer in humans exposed to PAH-containing coke oven emissions, roof-tar emissions, and cigarette smoke (U.S. DHHS, 1990). However, it is not possible to conclude from this information that benzo[a]pyrene is the responsible agent.

PAHs are ubiquitous in the environment and usually occur as complex mixtures with other toxic chemicals. They are components of crude and refined petroleum products and of coal. They are also produced by the incomplete combustion of organic materials. Many domestic and industrial activities involve pyrosynthesis of PAHs, which may be released into the environment in airborne particulates or in solid (ash) or liquid by-products of the pyrolytic process. Domestic activities that produce PAHs include cigarette smoking, home heating with wood or fossil fuels, waste incineration, broiling and smoking foods, and use of internal combustion engines. Industrial activities that produce PAHs include coal coking; production of carbon blacks, creosote, and coal tar; petroleum refining; synfuel production from coal; and use of Soderberg electrodes in aluminum smelters and ferrosilicum and iron works (Neff, 1985). Historic coal gasification sites have also been identified as significant sources of PAH contamination (J. Hesse, Michigan Department of Public Health, personal communication, March, 1991).

Major sources of PAHs found in marine and fresh waters include biosynthesis (restricted to anoxic sediments), spillage and seepage of fossil fuels, discharge of domestic and industrial wastes, atmospheric deposition, and runoff (Neff, 1985). Urban stormwater runoff contains PAHs from leaching of asphalt roads, wearing of tires, deposition from automobile exhaust, and oiling of roadsides and unpaved roadways with crankcase oil (MacKenzie and Hunter, 1979). Solid PAH-containing residues from activated sludge treatment facilities have been disposed of in landfills or in the ocean (ocean dumping was banned in 1989). Although liquid domestic sewage contains <1 ug/L total PAH, the total PAH content of industrial sewage is 5 to 15 ug/L (Borneff and Kunte, 1965) and that of sewage sludge is 1 to 30 mg/kg (Grimmer et al., 1978; Nicholls et al., 1979).

PAHs can accumulate in aquatic organisms from water, sediments, and food. BCFs of PAHs in fish and crustaceans have frequently been reported to be in the range of 100 to 2,000 (Eisler, 1987). In general, bioconcentration was greater for the higher molecular weight PAHs than for the lower molecular weight PAHs. Biotransformation by the mixed function oxidase system in the fish liver can result in the formation of carcinogenic and mutagenic intermediates, and exposure to PAHs has been linked to the development of tumors in fish (Eisler, 1987).

Sediment-associated PAHs can be accumulated by bottom-dwelling invertebrates and fish (Eisler, 1987). For example, Great Lakes sediments containing elevated levels of PAHs were reported by Eadie

et al. (1983) to be the source of the body burdens of the compounds in bottom-dwelling invertebrates. NAS (1991) reported that PAH contamination in bivalves has been found in all areas of the United States. Bivalves are good bioaccumulators of some PAHs because they do not metabolize these compounds as rapidly as fish. Varanasi et al. (1985) ranked benzo[a]pyrene metabolisms by aquatic organisms as follows: fish > shrimp > amphipod crustaceans > clams. Half-lives for elimination of PAHs in fish ranged from less than 2 days to 9 days (Niimi, 1987).

D.3 COPPER

Copper is commercially extracted from ores that typically contain several additional metals such as zinc, cadmium, and molybdenum. Once extracted, the primary uses of copper are for heating equipment, plumbing, and electrical equipment. Copper has also been widely used as an algicide and aquatic herbicide (USEPA, 1985a).

Copper is an essential dietary element for plants and animals. In animals, copper makes up an essential part of many enzymes and is important in hemoglobin formation (Rand and Petrocelli, 1985). Ingestion of large doses may cause liver or kidney damage.

In aquatic systems, copper's toxic effects tend to decrease with increasing water hardness (USEPA, 1985a). Copper has a low potential to be bioaccumulated in freshwater organisms. Little information exists relating copper contaminated soils to increased body burdens in mammals.

D.4 LEAD

Lead is derived primarily from the mining and processing of limestone and dolomite deposits, which are often sources of lead, zinc, and copper (May and McKinney, 1981). It is also found as a minor component of coal. Historically, lead has had a number of industrial uses, including use in paints, in solder used in plumbing and food cans, and as a gasoline additive. As recently as the mid-1980s, the primary source of lead in the environment was the combustion of gasoline; however, use of lead in U.S. gasoline has fallen sharply in recent years. At present, lead is used primarily in batteries, electric cable coverings, some exterior paints, ammunition, and sound barriers. Currently, the major points of entry of lead into the environment are from mining and smelting operations, from fly ash resulting from coal combustion, and from the leachates of landfills (May and McKinney, 1981).

In aquatic environments, water-borne lead is the most toxic form, with uptake modified by water hardness (USEPA, 1985b). As water hardness, increases, precipitation also increases, making lead less available and reducing toxic effects. Lead has been shown to bioaccumulate, with the organic forms, such as tetraethyl lead, appearing to have the greatest potential for bioaccumulation in fish tissues. Lead is primarily accumulated in the epidermis and intestine of fish, whereas very little is accumulated in muscle. High concentrations of lead have been found in marine bivalves and finfish from both estuarine and marine waters (NOAA, 1987, 1989). Although water-borne lead is concentrated by

biota, it has not been shown to transfer through the food chain (USEPA, 1979). Lead concentrations tend to decrease with increasing trophic level in both detritus-based and grazing aquatic food chains (Wong et al., 1978). Lead concentrations in freshwater fish declined significantly from a geometric mean concentration of 0.28 ppm in 1976 to 0.11 ppm in 1984. This trend has been attributed primarily to reductions in lead content of U.S. gasoline (Schmitt and Brumbaugh, 1990).

In terrestrial environments, lead has been shown to cause toxicity in animals, including waterfowl. The skeleton is the main long-term storage site for lead, with bone concentrations reflecting total chronic exposure levels. The highest lead concentrations occur in bone, followed by the kidneys. The lowest lead values are found in the liver, brain, and muscle tissues. Species and individuals respond differently to lead exposure, with effects of organolead compounds more pronounced than inorganic lead compounds. Younger developmental stages are more sensitive to lead than older life stages, and effects are exacerbated by diets deficient in minerals, fats, and proteins.

Lead is particularly toxic to children and fetuses. Subtle neurobehavioral effects (e.g., fine motor dysfunction, impaired concept formation, and altered behavior profile) occur in children exposed to lead at concentrations that do not result in clinical encephalopathy (ATSDR, 1988). A great deal of information on the health effects of lead has been obtained through decades of medical observation and scientific research. This information has been assessed in the development of air and water quality criteria by the Agency's Office of Health and Environmental Assessment (OHEA) in support of regulatory decisionmaking by the Office of Air Quality Planning and Standards and by the Office of Drinking Water (ODW). By comparison to most other environmental toxicants, the degree of uncertainty about the health effects of lead is quite low. It appears that some of these effects, particularly changes in the levels of certain blood enzymes and in aspects of children's neurobehavioral development, may occur at blood lead levels so low as to be essentially without a threshold. The Agency's Reference Dose Work Group discussed inorganic lead (and lead compounds) in 1985 and considered it inappropriate to develop an RfD for inorganic lead (IRIS, 1994). Lead and its inorganic compounds have been classified as probable human carcinogens (B2) by EPA (IRIS, 1992). However, at this time, a quantitative estimate of carcinogenic risk from oral exposure is not available (IRIS, 1994).