UPPER COLUMBIA RIVER

Baseline Ecological Risk Assessment Work Plan

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CONTENTS

LIST OF APPENDICES .... **vii**
LIST OF FIGURES .... **ix**
LIST OF MAPS .... **xi**
LIST OF TABLES .... **xiii**
ACRONYMS AND ABBREVIATIONS .... **xv**
UNITS OF MEASURE .... **xxi**

1 INTRODUCTION **1-1**
- 1.1 OBJECTIVES 1-2
- 1.2 WORK PLAN PROCESS AND ORGANIZATION 1-3

2 SITE DESCRIPTION AND SETTING **2-1**
- 2.1 HYDROLOGY 2-2
- 2.2 CLIMATE AND METEOROLOGY 2-5
- 2.3 HABITATS AND BIOLOGICAL COMMUNITIES 2-6
  - 2.3.1 Aquatic Habitat 2-6
  - 2.3.2 Aquatic Communities 2-7
  - 2.3.3 Terrestrial Habitats 2-8
  - 2.3.4 Special Status Species 2-9
- 2.4 SUMMARY OF SCREENING-LEVEL ECOLOGICAL RISK ASSESSMENT 2-9

3 EXISTING DATA **3-1**
- 3.1 SUMMARY OF EXISTING DATA 3-1
  - 3.1.1 Surface Water Quality 3-1
  - 3.1.2 Sediment Chemistry 3-14
  - 3.1.3 Sediment Toxicity 3-41
  - 3.1.4 Benthic Macroinvertebrate Tissue Chemistry 3-49
  - 3.1.5 Fish Tissue Chemistry 3-53
  - 3.1.6 White Sturgeon 3-75
  - 3.1.7 Soil Chemistry 3-82
  - 3.1.8 Air Quality 3-85
  - 3.1.9 Groundwater 3-90
- 3.2 DATA EVALUATIONS 3-90
  - 3.2.1 Surface Water Quality 3-91
  - 3.2.2 Sediment Chemistry 3-93
  - 3.2.3 Sediment Toxicity 3-95
  - 3.2.4 Fish Tissue Chemistry 3-98
  - 3.2.5 Soil Chemistry 3-99
  - 3.2.6 Groundwater 3-102
  - 3.2.7 Background Soil and Sediment Chemistry 3-103

4 PROBLEM FORMULATION 4-1
4.1 RISK DRIVERS 4-2
4.2 CONCEPTUAL SITE MODEL REFINEMENT 4-2
4.2.1 Conceptual Site Models 4-3
4.3 ASSESSMENT ENDPOINTS 4-5
4.4 RECEPTOR GROUPS OF CONCERN 4-6
4.4.1 Aquatic Receptor Groups 4-7
4.4.2 Terrestrial Receptor Groups 4-10
4.4.3 Special Status Species 4-11
4.5 CHEMICALS OF POTENTIAL CONCERN 4-11
4.5.1 Sediments 4-12
4.5.2 Surface Water 4-18
4.5.3 Soils 4-18
4.5.4 Biota 4-18
5 LINES OF EVIDENCE AND MEASUREMENT ENDPOINTS 5-1
5.1 LINES OF EVIDENCE 5-1
5.1.1 Aquatic Receptors 5-2
5.1.2 Terrestrial Receptors 5-6
5.2 MEASURES OF EXPOSURE AND EFFECTS 5-8
5.2.1 Measures of Exposure 5-8
5.2.2 Measures of Effect 5-13
5.3 UNCERTAINTIES 5-16
6 DATA GAPS AND STUDIES – PHASE II 6-1
6.1 SURFACE WATER QUALITY 6-1
6.2 FISH TISSUE CHEMICAL CONCENTRATIONS 6-3
6.3 SEDIMENTS 6-6
6.3.1 Spatial Refinements 6-6
6.3.2 COPC Refinements 6-6
6.3.3 Sediment Toxicity Refinements 6-7
6.3.4 Sediment and Porewater Studies 6-7
6.4 BEACH (SHALLOW WATER ZONE) SEDIMENTS 6-9
6.5 SOIL CHEMISTRY 6-9
6.6 WHITE STURGEON 6-10
6.6.1 Sturgeon Toxicity Tests (water) 6-11
6.6.2 Sturgeon Toxicity Tests (sediment) 6-11
7 ADDITIONAL PHASES 7-1
7.1 SEDIMENT STUDIES – PHASE III 7-1
7.1.1 Sediment-Water Interface Water – Phase III 7-2

7.1.2 Sediment Weathering – Phase III 7-2
7.2 AQUATIC RECEPTORS – PHASE III 7-2
7.2.1 Benthic Macroinvertebrates Tissue Chemistry – Phase III 7-2
7.2.2 Benthic Macroinvertebrate Abundance or Biomass (includes mussels) – Phase III 7-3
7.3 PLANKTON TISSUE CONCENTRATIONS – PHASE III 7-4
7.4 OTHER AQUATIC BIOASSAYS (INCLUDES MUSSELS) – PHASE IV 7-5
7.5 AQUATIC SPECIALTY STUDIES – PHASE IV 7-5
7.5.1 Bioavailability of COPCs from Fish and Macroinvertebrate Tissue – Phase IV 7-5
7.5.2 Dietary Toxicity to Fish – Phase IV 7-6
7.5.3 White Sturgeon – Phase IV 7-6
7.5.4 Emergent Insect Tissue Chemistry – Phase IV 7-7
7.6 TERRESTRIAL STUDIES – PHASE IVA 7-7
7.6.1 Soil Sampling and Bioavailability 7-8
7.6.2 Soil Background 7-8
7.6.3 Upland Lakes and Wetlands 7-8
7.7 TERRESTRIAL RECEPTOR STUDIES – PHASE IVB 7-9
7.7.1 Terrestrial Resource Study 7-9
7.7.2 Plant Tissue Chemistry 7-9
7.7.3 Plant Bioassay 7-9
7.7.4 Invertebrate Tissue Chemistry 7-9
7.7.5 Terrestrial Invertebrate Toxicity Studies 7-10
7.8 TRANSPORT AND FATE EVALUATIONS 7-10
7.9 ADDITIONAL STUDIES 7-10
8 ANALYSIS AND RISK ESTIMATION PLAN 8-1
8.1 RISK CHARACTERIZATION 8-1
8.1.1 Comparison to Benchmarks 8-1
8.1.2 Bioassays 8-2
8.1.3 Spatial Analysis 8-2
8.1.4 Tiered Approach to Reduce Uncertainty 8-3
8.2 INTEGRATION OF LINES OF EVIDENCE 8-4
8.3 INTERPRETATION OF ECOLOGICAL SIGNIFICANCE 8-4
8.4 UNCERTAINTY 8-5
9 SCHEDULE 9-1
10 REFERENCES 10-1
11 GLOSSARY 11-1

LIST OF APPENDICES

Appendix A – Summary of Aquatic Resources in the Study Area

Appendix B – Terrestrial Resource Inventory

Appendix C – Summary of Surface Water Quality Data

Appendix D - Sediment Chemistry Characteristics in the UCR

Appendix E – Evaluation of Sediment Toxicity

Appendix F – Evaluation of Existing Fish Tissue Data

Appendix G – Summary of Upland Soil Characteristics in the Study Area

Appendix H - Background Sediment and Soil Chemistry

LIST OF FIGURES

Figure 1-1. Relationships among Major Tasks in the UCR RI/FS

Figure 1-2. Eight-step Ecological Risk Assessment Process for Superfund

Figure 1-3. EPA Ecological Risk Assessment Framework

Figure 3-1. Total Arsenic: Comparison of Surface Water Concentrations at Birchbank, Waneta, the U.S.-Canada Border, and Northport (2001-2005)

Figure 3-2. Total Cadmium: Comparison of Surface Water Concentrations at Birchbank, Waneta, the U.S.-Canada Border, and Northport (2001-2005)

Figure 3-3. Total Copper: Comparison of Surface Water Concentrations at Birchbank, Waneta, the U.S.-Canada Border, and Northport (2001-2005)

Figure 3-4. Total Lead: Comparison of Surface Water Concentrations at Birchbank, Waneta, the U.S.-Canada Border, and Northport (2001-2005)

Figure 3-5. Total Zinc: Comparison of Surface Water Concentrations at Birchbank, Waneta, the U.S.-Canada Border, and Northport (2001-2005)

Figure 3-6. Average Nitrate Concentrations as a Function of Time

Figure 3-7. Average Nitrate Concentrations as a Function of Well Location

Figure 3-8. Average Calcium Concentrations as a Function of Well Location

Figure 3-9. Average Sodium Concentrations as a Function of Well Location

Figure 3-10. Average Sulfate Concentrations as a Function of Well Location

Figure 3-11. Average Iron Concentrations as a Function of Well Location

Figure 3-12. Average Lead Concentrations as a Function of Well Location

Figure 3-13. Average Manganese Concentrations as a Function of Well Location

Figure 3-14. Average Zinc Concentrations as a Function of Well Location

Figure 4-1. Sitewide Conceptual Site Model

Figure 4-2. Conceptual Site Model for the Riverine Transitional Areas (Reaches 1 and 2)

Figure 4-3a. Conceptual Site Model for Marcus Flats Shallow water Area (Reach 3)

Figure 4-3b. Conceptual Site Model Marcus Flats Deep water Area (Reach 3)

Figure 4-4a. Conceptual Site Model for Lacustrine Shallow water Areas of the Site (Reaches 4, 5, and 6)

Figure 4-4b. Conceptual Site Model for Lacustrine Deep water Areas of the Site (Reaches 4, 5, and 6)

Figure 4-5. Conceptual Site Model for Aquatic-Dependent Plants and Wildlife

Figure 4-6. Conceptual Site Model for the Terrestrial Upland Habitat

Figure 4-7. Aquatic Macrophyte CSM

Figure 4-8. Benthic Macroinvertebrate CSM

Figure 4-9. Fish CSM

Figure 4-10. Early Life Stage Amphibian CSM

Figure 4-11. Aquatic Dependent Wildlife CSM

Figure 4-12. Terrestrial Plant, Invertebrate, Bird, Mammal, Amphibian and Reptile CSM

Figure 4-13. National water quality criteria final acute values (FAVs) for all species versus values for benthic species only

Figure 4-14. Relationship between the toxicity of 50 nonpolar organic chemicals and acute toxicity to an indicator fish (*Lebistes reticulates*). Organics included chlorotoluenes, alphiphatic chlorohydrocarbons, alcohols, glycol derivatives and similar compounds

Figure. 5-1. Lines of Evidence for Assessing Benthic Macroinvertebrates

Figure 5-2. Lines of Evidence for Assessing Benthic Fish

Figure 5-3. Lines of Evidence for Assessing Pelagic Fish

Figure 5-4. Lines of Evidence for Assessing Early Life Stage Amphibians

Figure 5-5. Lines of Evidence for Assessing Aquatic Dependent Wildlife

Figure 5-6. Lines of Evidence for Assessing Terrestrial Plants

Figure 5-7. Lines of Evidence for Assessing Foliar/Soil Invertebrates

Figure 5-8. Lines of Evidence for Assessing Terrestrial Birds, Mammals, Reptiles, and Amphibians

Figure 7-1. Study Sequencing

Figure 9-1. Projected Schedule for BERA Phase II

LIST OF MAPS

Map 2-1. Site Location and River Reach Delineations

Map 3-1. Spatial Distribution of Sediment Classes in Phase I Data

Map 3-2. Summary of PECQs Greater Than 1.0 for Sediment in the 2005 Sediment Toxicity Study

Map 3-3. Summary of Toxic Units Greater Than 1.0 for Porewater in the 2005 Sediment Toxicity Study

Map 3-4. Calculated Floodplains for Post-1973 Average Daily Flow and Pre-1973 Maximum Flow Conditions form the U.S.-Canada Border to Grand Coulee Dam

Map 3-5. National Park Service Groundwater Well Locations

Map 6-1. Phase II Surface Water Sampling Stations

Map 6-2. Proposed 2009 Fish Sample Collection Areas and 2010 Zooplankton Collection Areas

Map 6-3. Phase II Beach (Shallow water) Sediment Sampling Locations

LIST OF TABLES

Table 2-1. Threatened and Endangered Terrestrial Species Present in the Surrounding Area of the UCR

Table 2-2. Chemicals of Interest for the UCR SLERA

Table 2-3. Scientific Management Decision Point Summary for the UCR SLERA

Table 3-1. Summary of Sediment Studies within the UCR included in the BERA (1996 to Present)

Table 3-2. Summary of other Qualified Data Collected within the UCR for use in the BERA (1996 to Present)

Table 3-3. Surface Sediment Samples by Reach, Class, and Channel Location

Table 3-4. Number Detected, Number Nondetected for Standardized Site Groups

Table 3-5. Statistics of Detected Values by Standardized Site

Table 4-1. Anticipated technical memoranda and data summary reports (Phases II and III)

Table 4-2. Summary of Aquatic Receptors and Related Assessment Endpoints for the BERA

Table 4-3. Summary of Terrestrial Receptors and Related Assessment Endpoints for the BERA

Table 4-4. Fish Species Representing Fish Receptor Groups in the UCR

Table 4-5. Representative Terrestrial Receptor Species for the BERA

Table 5-1. Measures of Exposure and Effects for Aquatic Receptors

Table 5-2. Measures of Exposure and Effects for Terrestrial Receptors

Table 5-3. Summary of Data Availability

Table 5-4. Data Needs and Descriptions

Table 5-5. Weighted Average of Stomach Contents (as percent by dry mass) of Fish Collected by the LRFEP, 1999 to 2005

Table 5-6. Exposure Factors for Aquatic-Dependent Wildlife

Table 5-7. Food and Water Ingestion Rate Estimation Formulas

Table 5-8. Series 850—Ecological Effects Test Guidelines

Table 6-1. List of Studies Described in the BERA

ACRONYMS AND ABBREVIATIONS

AAS	atomic absorption spectrometry
ABS	absorption factors
AC	alternating current
AChE	acetylcholinesterase
ACR	acute to chronic ratios
amsl	above mean sea level
ARAR	applicable or relevant and appropriate requirement
ASIL	acceptable source impact level
AUF	area use factor
AVS	acid volatile sulfide
AWQC	ambient water quality criteria
B.C.	British Columbia
BERA	baseline ecological risk assessment
BLM	biotic ligand model
BWMP	Basic Water Monitoring Plan
CBB	critical body burden
CCME	Canadian Council of Ministers of the Environment
CDF	cumulative distribution function
CEC	cation exchange capacity
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act
CF	condition factor
ChE	cholinesterase
CLP	Contract Laboratory Program
COI	chemical of interest
COPC	chemical of potential concern
CRB	Columbia River Basin
CRIEMP	Columbia River Integrated Environmental Monitoring Program
CSM	conceptual site model
CSOIII	Combined Sewer Outfall III
CT	central tendency
CTFWD	Colville Tribes and Fish and Wildlife Department
CTLBB	critical target lipid body burden
CV	comparison value

CVAAS	cold vapor atomic absorption spectrometry
DDE	dichloro-diphenyl-dichloroethene
DDT	dichloro-diphenyl-tricholorethane
DNA	Deoxyribonucleic acid
DO	dissolved oxygen
DOC	dissolved organic carbon
DQO	data quality objective
dw	dry weight
EC	Environment Canada
EC_x	effect concentration at a certain percentile
Eco-SSL	EPA ecological soil screening level
Ecology	Washington State Department of Ecology
E & E	Ecology and Environment, Inc.
ELS	early life stage
EPA	U.S. Environmental Protection Agency
ERA	ecological risk assessment
EROD	ethoxyresorufin-*O*-deethylase
ESG	equilibrium partitioning sediment guidelines
ESI	expanded site inspection
FAV	final acute value
FCV	final chronic value
FDA	Food and Drug Administration
f_{oc}	fraction organic carbon
FOD	frequency of detection
FSCA	fish sample collection area
GC/MS	gas chromatography/mass spectrometry
GFAA	graphite furnace atomic absorption
GIS	geographic information system
GLI	Great Lakes Initiative
GPS	global positioning system
HC5	hazardous concentration that protects all but 5 percent of the species
HF	hydrofluoric acid
HHRA	human health risk assessment
HSI	hepatosomatic index

ICAES	inductively coupled atomic emission spectrometry
ICP	inductively coupled plasma
ICP-MS	inductively coupled plasma-mass spectrometry
IDWA	inverse distance weighted averaging
IRM	iterative regression model
K_{ow}	octanol-water partition coefficient
Lake Roosevelt	Franklin D. Roosevelt Lake
LOAEC	lowest-observed-adverse-effect concentration
LOAEL	lowest-observed-adverse-effect level
LOE	lines of evidence
LRFEP	Lake Roosevelt Fisheries Evaluation Program
LRNRA	Lake Roosevelt National Recreation Area
MS/MSD	matrix spike and matrix spike duplicate
NASQAN	National Stream Quality Accounting Network
NCBP	National Contaminant Biomonitoring Program
NOAEC	no-observed-adverse-effect concentration
NOAEL	no-observed-adverse-effect level
NPL	National Priorities List
NPS	National Park Service
NURE-HSSR	National Uranium Resource Evaluation-Hydrogeochemical and Stream Sediment Reconnaissance
OA	Outer Area
PAH	polycyclic aromatic hydrocarbon
PBDE	polybrominated diphenyl ether
PCB	polychlorinated biphenyl
PCDD	polychlorinated dibenzo-p-dioxin
PCDF	polychlorinated dibenzofuran
PEC	probable effect concentration
PECQ	probable effect concentration quotient
PEQ	probable effect quotient
PM_{10}	Particulate Matter, 10 microns (or less)
PNEC	probable no effect concentration
PSDDA	Puget Sound Dredged Disposal Analysis
PSEP	Puget Sound Estuary Program
QICAR	quantitative ion characteristics activity relationship

QAPP	quality assurance project plan
QA	quality assurance
QA/QC	quality assurance/quality control
QC	quality control
QSAR	quantitative structure activity relationship
RI/FS	remedial investigation and feasibility study
RM	river mile
RMAO	Risk Management-Based Action Objective
RME	reasonable maximum exposure
SAP	sampling and analysis plan
SARA	Canada's Species Risk Act
SD	spatially distributed
SEM	simultaneously extracted metals
Site	Upper Columbia River Site
SLERA	screening-level ecological risk assessment
SMD	scientific management decision
SMDP	scientific management decision point
SOW	Statement of Work
SPMD	semipermeable membrane devices
SQO	sediment quality objective
SQG	sediment quality guideline
SQS	sediment quality standards
SSD	species sensitivity distribution
SSI	splenosomatic index
START	Superfund Technical Assessment and Response Team
SVOC	semivolatile organic compound
SWIW	sediment-water interface water
TAL	target analyte list
TAI	Teck American Incorporated
TCDD	tetrachlorodibenzo-p-dioxin
TCDF	tetrachlorodibenzo-furan
TDS	total dissolved solids
TEC	threshold effect concentration
Teck	Teck American Incorporated
TEL	threshold effect level

TEQ	toxic equivalent
TKN	total Kjeldahl Nitrogen
TLM	target lipid model
TN	total nitrogen
TOC	total organic carbon
TP	total phosphorus
TRC	tissue residual criterion
TRO	tissue residue objective
TRV	toxicity reference value
TSA	trend surface analysis
TSP	total suspended particulates
TSS	total suspended solids
TU	toxic unit
UCL	upper confidence limit
UCR	Upper Columbia River
UCRWSRI	Upper Columbia River White Sturgeon Recovery Initiative
U.S.	United States
USBR	U.S. Bureau of Reclamation
USFWS	U.S. Fish and Wildlife Service
USGS	U.S. Geological Survey
UTL	upper tolerance level
VOC	volatile organic compound
WDFW	Washington State Department of Fish and Wildlife
WDOH	Washington State Department of Health
Weston	Roy F. Weston, Inc.
WOE	weight-of-evidence
WQC	water quality criteria
WQO	water quality objective
ww	wet weight

UNITS OF MEASURE

°C	degrees Celsius
°F	degrees Fahrenheit
μg/g	micrograms per gram
μg/kg	micrograms per kilogram
μg/L	micrograms per liter
μg/m^3	micrograms per cubic meter
μm	micrometer
μmol/g	micromoles per gram
cfs	cubic feet per second
cm	centimeter(s)
dph	days post hatch
ft	foot/feet
ha	hectare(s)
hr	hour
in.	inch(es)
kg	kilogram(s)
kg/day	kilograms per day
km	kilometer(s)
L	liter
m	meter(s)
m^2	square meter(s)
m/s	meters per second
mg/kg	milligrams per kilogram
mg/L	milligrams per liter
mg/m^2	milligrams per square meter
mg/m^3	milligrams per cubic meter
mi^2	square mile(s)
mi^3	cubic mile(s)
mL	milliliter(s)
m^3/s	cubic meters per second
mm	millimeter(s)
mph	miles per hour
ng/g	nanograms per gram

ng/kg	nanograms per kilogram
pg/g	picograms per gram
ppb	parts per billion
ppm	parts per million
ppt	parts per trillion
pCi/g	pico Curies per gram
RKM	river kilometer

1 INTRODUCTION

This document presents the approach and rationale for conducting the baseline ecological risk assessment (BERA) for the Upper Columbia River (UCR) Site[1] in support of the remedial investigation and feasibility study (RI/FS). As described in the RI/FS work plan (USEPA 2008a)[2], the primary objectives of the RI/FS are to investigate the nature and extent of contamination at the Site, to provide information to support baseline risk assessments for human health and the environment, and to develop and evaluate potential remedial alternatives for the Site. All work performed for the BERA will be subject to oversight by the U.S. Environmental Protection Agency (EPA) in accordance with the Settlement Agreement (USEPA 2006a).

A BERA is required to evaluate risk to ecological receptor groups that may be exposed to hazardous substances, pollutants and contaminants, including chemicals of potential concern (COPCs), in both the aquatic and upland portions of the Site, as identified in the Screening Level Ecological Risk Assessment (SLERA; TAI 2010). The BERA will be completed by Teck American Incorporated (Teck, formerly Teck Cominco American Incorporated), in parallel with the human health risk assessment (HHRA) being conducted by the EPA. The overall RI/FS process is depicted on Figure 1-1 including the respective responsibilities of Teck and EPA. The BERA will follow EPA's eight-step process for Superfund ecological risk assessments (ERAs) (USEPA 1997a) (Figure 1-2), and will be consistent with EPA's general framework for ERA (USEPA 1998) (Figure 1-3), as well as other relevant EPA risk assessment and related documents (e.g., USEPA 2007a). Results of the BERA will inform the remedial investigation of potential ecological risks posed by hazardous substances, pollutants, or contaminants at the Site, and the feasibility study for the identification and selection of remedial alternatives.

Teck has conducted an Upper Columbia River SLERA (TAI 2010[3]) that evaluated whether existing data were adequate to make a determination of potential ecological risks at the Site. The SLERA, which is summarized in Section 2.4, describes the COPCs and environmental media for which data 1) are adequate and indicate that the COPC poses no risk to the environment, 2) are inadequate for determining risk, or 3) indicate the potential for risk. The BERA will characterize risk to ecological receptor groups in aquatic and terrestrial upland habitats of the Site.

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[1] As defined within the Settlement Agreement of June 2, 2006, the Site consists of the areal extent of hazardous substances contamination within the United States (U.S.) in or adjacent to the Upper Columbia River, including the Franklin D. Roosevelt Lake, from the U.S.–Canadian border to the Grand Coulee Dam, and all suitable areas in proximity to the contamination necessary for implementation of response actions.

[2] EPA modified Teck's second draft RI/FS work plan and issued the modified work plan to Teck in December 2008 (USEPA 2008a).

[3] The BERA Work Plan is consistent with the final SLERA.

To date, a number of environmental investigations, not necessarily associated with the RI/FS have been conducted at the Site. The most comprehensive and systematic investigations however specifically designed and conducted for the RI/FS include those conducted by EPA; in which a comprehensive survey of concentrations of metals and organic compounds in sediments and in tissues of large fish was completed (USEPA 2006b,c,d; 2007b). In addition, EPA conducted toxicity tests with a subset of the sediment samples collected in 2005 (USEPA 2006b,d). These 2005 studies constitute Phase I of the RI/FS and are represented as such throughout the remainder of this document. The work described in this BERA work plan contributes to Phase II of the RI/FS. Although the BERA will assess risks to ecological receptor groups (plant and animals), the same data sets may also be used in the HHRA conducted by EPA (USEPA 2009a), and in other studies supporting the RI and FS.

1.1 OBJECTIVES

The BERA work plan provides the context and describes the approaches for performing a BERA at the Site. It is a critical component of the RI/FS and documents and communicates the process and methods that will be employed in performing the BERA. The BERA work plan has the following specific objectives:

- Review existing information on physical, chemical, and biological attributes of the Site to identify data useful for conducting the BERA and where there are data gaps that need to be filled.
- Refine the problem formulation and conceptual site models (CSMs) developed in the SLERA allowing EPA and Teck to identify and focus the risk assessment. This includes defining receptor groups and their exposure pathways.
- Define the lines of evidence (LOEs) that will be considered and evaluated to address risk to ecological receptors at the Site.
- Describe the investigations and types of data that will be generated to fill known gaps, including decision points and if/then statements about possible additional investigations.
- Describe the analysis plan and adaptive (i.e., tiered) approach for evaluating the various LOEs and conducting the ERA.

Preparation and review of the BERA work plan that is being performed subject to EPA oversight, will include a process of ongoing communication to attain transparency and ensure the technical quality.

1.2 WORK PLAN PROCESS AND ORGANIZATION

The BERA work plan provides descriptive, interpretive, and planning information. This document is organized into the following sections:

- **Section 2–Site Description and Setting.** This section provides a general characterization of physical and biological components of the Site, and summarizes the SLERA (i.e., the first level of refinement in the iterative ERA process).
- **Section 3–Existing Data.** This section presents and describes results for studies reporting on a wide range of environmental media (e.g., sediments, water, soil) and biota (e.g., fish and benthic invertebrates). The data are used to help characterize and illustrate the nature and extent of contamination at the Site.
- **Section 4–Problem Formulation.** This section further refines the preliminary problem formulation described within the SLERA helping to focus the ERA by identifying risk drivers, refining the CSM, identifying receptor groups and exposure pathways, and setting out a plan for how to further refine the list of COPCs.
- **Section 5–LOEs and Measurement Endpoints.** This section presents and discusses the LOEs to be employed and evaluated within the ERA and describes the exposure and effects measurement endpoints to be used.
- **Section 6–Data Gaps and Studies – Phase II.** This section, in consideration of problem formulation refinement (Section 4), summarizes gaps in the data needed to complete the measurement endpoints for environmental media and receptors.
- **Section 7–Additional Phases.** This section presents a sequence of studies that will be completed to fill data gaps identified within Section 6.
- **Section 8–Analysis and Risk Estimation Plan.** This section summarizes the tiered ERA approach to be employed for the Site, and identifies deterministic and probabilistic approaches that can and will be used for characterizing risk.
- **Section 9–Schedule.** This section presents the schedule for the list of studies discussed in Section 7.
- **Section 10–References.** This section contains references for documents cited in this work plan.
- **Section 11–Glossary.** This section describes the terms within this work plan and for the ERA.

This document presents a plan to complete the eight-step ERA process per EPA guidance (USEPA 1997a). Additionally, this work plan anticipates a process of information development and coordination with EPA as studies progress to fill data gaps that will take place through two types of documentation

- Informal outlines (e.g., technical memoranda) for specific media in which analysis of existing data, proposed risk assessment methodologies, or descriptions of data quality objectives (DQOs) and approaches for filling data gaps are presented.[4]
- Data summary and data gap evaluation reports summarize results from approved studies and provide the rationale for the necessity for additional studies or one or more LOEs in the BERA process.

Technical memoranda are anticipated to provide details (such as data analyses and proposed models) needed to support the BERA DQO process required for development of additional sampling and analysis plans (SAPs) and quality assurance project plans (QAPPs). Data summary and data gap evaluation reports[5] represent scientific management decision points (SMDPs) and are significant communication points between the risk assessors and risk/project managers; they document key technical decisions to further the BERA process (USEPA 1997a). This work plan specifies points at which SMDP documents are currently anticipated as the assessment moves through successive levels of refinement.

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[4] The use of informal outlines is typically not required nor completed for a RI/FS. However, due to a formal dispute between EPA and the participating parties this is an element required as part of the resolution.

[5] Data summary reports are required by the Settlement Agreement for Implementation of Remedial Investigation and Feasibility Study at the Upper Columbia River Site, between EPA and Teck dated June 2, 2006.

2 SITE DESCRIPTION AND SETTING

The Site is located wholly within the state of Washington and consists of the areal extent of hazardous substances contamination in or adjacent to the Upper Columbia River, including the Franklin D. Roosevelt Lake (Lake Roosevelt), from the Grand Coulee Dam to the U.S.-Canada border (Map 2-1), and all suitable areas in proximity to such contamination necessary for implementation of response actions. This encompasses a river reach extending approximately 150 miles downstream of the U.S.-Canada border.

The Site includes land and waters within the boundaries of the Colville Indian Reservation and the Spokane Indian Reservation, over which the Tribes have civil regulatory jurisdiction, as well as land and waters administered by the National Park Service and the Bureau of Reclamation within the U. S. Department of the Interior (USEPA 2006a). The construction of Grand Coulee Dam, a federal reclamation project, was completed in 1940 on a portion of the Columbia River that forms the southern boundary of the Colville Reservation (DOI 1977). A multi-purpose project, it provides flood control, irrigation, hydropower production, recreation, stream flows, and fish and wildlife benefits (USEPA 2003a). Located immediately behind the Grand Coulee Dam is Lake Roosevelt, a large reservoir extending approximately 133 river miles north of the dam at full pool[6] and bordered by over 600 miles of shoreline, approximately 312 miles of which are part of the Lake Roosevelt National Recreation Area (LRNRA) (NPS 2006a; USEPA 2003a).

The upland area surrounding the Site is generally thinly populated and consists of forests and farmland. Communities located along Highway 395 to the west of the UCR include Barstow and Boyds. Communities located to the east of the UCR, along Highway 25, include, from north to south, Northport, Evans, Marcus, Rice, Daisy, Gifford, Cedonia, Hunters, Fruitland, and Enterprise. Further south, the Colville Indian Reservation borders Lake Roosevelt to the north and west for approximately 93 river miles. This area includes several communities, the largest of which are Coulee Dam, Inchelium, and Keller (USEPA 2003a). The Spokane Indian Reservation borders Lake Roosevelt to the east for about 8 miles, just north of the Spokane Arm (confluence of the Spokane River with the Columbia River) (USEPA 2003a).

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[6] Technical reviewers for the UCR RI/FS have noted that at full pool (elevation 1,290 ft above mean sea level [amsl]), under extreme hydrological conditions the reservoir may have backwater effects up to and perhaps north of the border. At high water levels, river areas upstream of Northport, Washington, may also be influenced by factors other than water level in the reservoir. For instance, if the volume of water passing the border exceeds the river conveyance at morphological constrictions, water levels may increase regardless of downstream conditions.

2.1 HYDROLOGY

Hydrology of the UCR and Lake Roosevelt influences the kinds of available habitat for fish and other aquatic organisms, and affects exposure of plants and wildlife to Site media (e.g., surface water, sediments, and prey). Within this setting, the UCR and its tributaries drain an area of approximately 74,700 mi^2, of which approximately 20,000 mi^2 is within the U.S. Major tributaries that influence hydraulic conditions at the U.S.-Canada border are the Columbia and Pend Oreille rivers (USEPA 2008a). The Kootenay (also seen as Kootenai) River enters the Columbia River between the Hugh Keenleyside Dam and the Pend Oreille River. Principal tributaries that join the UCR within the study area are the Kettle, Colville, Spokane, and Sanpoil rivers (USEPA 2008a). Numerous smaller tributaries also join the UCR within the Site, including Deep, Onion, Sheep, Sherman, Hall, Ninemile, and Hawk creeks (USEPA 2008a).

Just upstream of the U.S.-Canada border, the Columbia and Pend Oreille rivers above the border supply the majority of the annual inflow (90 percent) to Lake Roosevelt, the remainder primarily supplied by the Spokane, Sanpoil, Kettle, and Colville rivers (Stober et al. 1981).

Flow regimes in the UCR have varied over time. Over the past century, three distinct flow regimes have existed, as described below (USEPA 2008a).

- Unregulated (before Grand Coulee Dam or upstream flow control). Before flow regulation began, UCR flows were governed by precipitation and runoff, particularly the amount of snowpack and snowmelt. During the unregulated era, the river was free-flowing and subject to large, periodic high-flow (flood) events (USEPA 2008a).
- Downstream control (after Grand Coulee Dam but before upstream flow control). During the period of downstream control, UCR flows were determined by unregulated upstream flow and water-level regulation at Grand Coulee Dam. Although periodic high-flow events still occurred, the extent of the Lake Roosevelt impoundment and backwater effects in upstream areas were controlled entirely by operations at Grand Coulee Dam (USEPA 2008a).
- Regulated (after Grand Coulee Dam and after upstream flow control). During the contemporary era of regulation (post-1972), river flows are controlled by the operation of upstream dams in addition to management operations at Grand Coulee Dam (USEPA 2007c). As a result of the combined effects of dam operations, the size and frequency of large flood events has been reduced (USEPA 2008a).

The normal operating range on the lake is between 1,290 feet above mean sea level (amsl; full pool) to 1,208 amsl, resulting in maximum seasonal lake level fluctuations in excess of 80 ft (USEPA 2003a). During draw down, acres of bed and bank sediments of Lake Roosevelt are exposed (USEPA 2003a). At full pool (1,290 ft amsl), the reservoir has a surface area of approximately 82,300 acres and extends upstream of Grand Coulee Dam approximately 133 miles to Onion Creek (U.S. Geological Survey [USGS] river mile [RM] 730[7]), approximately 15 river miles south (downstream) of the U.S.-Canada border (USEPA 2008a). However, it is worth noting that the channel is constricted (and conveyance reduced) through the Little Dalles (RM 728) (Map 2-1), and that water levels upstream of this point may rise during high-flow events (USCGS 1950).

The extent of water level increases is expected to be influenced by interactions between flow magnitude, reservoir pool level, and conveyance limitations through the Little Dalles. At typical low pool levels, with a water surface elevation of approximately 1,245 ft amsl (USEPA 2007d), the reservoir extent is reduced and ends near RM 704. Outflow from Lake Roosevelt occurs via discharge through Grand Coulee Dam to the Middle Columbia River or through pumped discharge to Banks Lake for irrigation storage (USBR 2006).

Hydrodynamic transport in the UCR is affected by upstream and tributary inflow rates, which are dependent on watershed hydrology, and the operation of numerous upstream dams, as well as Grand Coulee Dam at the downstream end of the UCR. Water may also enter or leave the UCR through groundwater seepage or drainage and may also be transported in the interfacial bed area via hyporheic (interstitial) flow. Once particulate and dissolved COPCs enter the UCR, they are redistributed via the hydrodynamic transport processes of advection and turbulent mixing; these processes result in the dilution and dispersion of these materials within the system (USEPA 2008a). Hydrodynamic processes also influence the dynamic coupling between sediment and overlying surface water. The frictional interaction between moving fluid and riverbed roughness induces boundary shear stresses and steep vertical gradients of turbulent mixing near the bottom that regulate particle deposition and resuspension (i.e., scour). Near-bottom turbulence also regulates concentration gradients near the riverbed, altering the diffusional exchange of dissolved COPCs between sediments and surface waters. Advective processes in the bed, such as interactions with groundwater and hyporheic flow, can also impact the exchange of dissolved COPCs between sediments and surface waters.

In the RI/FS work plan (USEPA 2008a), the overall Site was subdivided into six river reaches that correspond to relatively distinct physiographic units (Map 2-1). The delineation of the river reaches is generally consistent with those used in past USGS studies by Bortleson et al. (1994) and Cox et al. (2005) and is summarized below.

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[7] There is a discrepancy in river mile designations used by USGS and those used by USEPA in the 2005 Phase I sampling reports (USEPA 2006b, 2006c, and 2006d). USGS river miles increase from RM 680 to RM 682 over a less than 1 river mile segment when transitioning between the Inchelium and Rice USGS quadrants, whereas the USEPA river miles increases from RM 680 to RM 681 over the same segment. The USGS river mile designations are used herein unless otherwise noted.

- **Reach 1 (U.S.-Canada Border at RM 745 to RM 730).** This reach is consistent with the Northport Reach identified by USGS (Bortleson et al. 1994; Cox et al. 2005). It begins at the upstream boundary of the Site and extends approximately to Onion Creek. The upper 3 miles of this reach is relatively narrow and shallow with water depths ranging to approximately 14 ft. Water depths of 50 ft or more are found at the downstream end of this reach. Generally, this reach can be characterized as a swift river environment (i.e., riverine) that is typically unaffected by the reservoir.

- **Reach 2 (RM 730 to RM 711).** This reach is consistent with the Upper Reservoir Reach identified by USGS. It extends to the vicinity of the towns of Evans and Powell, Washington, and can be characterized as a narrow channel within the reservoir that has few shoreline embayments and irregularities (Map 2-1). Although this reach is inundated at high pool (i.e., approximately 70 percent of the time; USEPA 2006d), currents through the widened canyon remain swift at lower pool levels. The deepest part of the channel at the upstream end of the reach increases from 50 ft to more than 100 ft in the vicinity of Little Dalles. From there, it shallows to 60 to 70 ft until about RM 718, where it narrows and deepens again to 100 ft, varying between 70 and 90 ft through the remainder of the reach. At RM 736, the UCR makes a sharp bend east, with the deepest part of the channel hugging the southern bank. Below China Bend, the UCR becomes more sinuous as it proceeds through a series of three additional broad bends before the end of the reach.

- **Reach 3 (RM 711 to RM 699).** This reach is consistent with the upper portion of the Middle Reservoir Reach identified by USGS, and it consists primarily of Marcus Flats. This reach can be characterized as a depositional area for coarse-grained sediments in the historical river channel and for fine-grained sediments in many of the shallower areas. At RM 710 and again between RMs 706 and 707, the historical river channel makes two 90-degree bends while passing through a relatively broad floodplain in the area of Marcus Flats. (Map 2-1) To the north of the second bend is the confluence of Kettle River with the UCR. The Kettle River is the first significant tributary confluence downstream of the U.S.-Canada border, contributing approximately 3,000 cfs of mean annual flow (USGS 2006d). Between RM 704 and 703, the UCR channel descends through a steep, narrow reach. Downstream of the narrow reach the UCR continues along a relatively straight path until the confluence with the Colville River at RM 699. Reach 3 is almost continuously inundated by the Lake Roosevelt pool, with full-pool water depths during seasonally high flow of 50 ft or more over the historical floodplain and more than 100 ft deep along sections of the historical river channel.

- **Reach 4 (RM 699 to RM 640).** This reach is consistent with the lower portion of the Middle Reservoir Reach identified by USGS and extends from above the mouth of the Colville River to above the mouth of the Spokane River. It can be further subdivided into Reaches 4a and 4b, with the boundary occurring at RM

676 near the towns of Inchelium and Gifford, where the width of the overall reach narrows considerably. Both portions of this reach can be, characterized as a lacustrine environment with low-flow velocities (maximum flow rarely exceeds 2 to 3 ft/second [USEPA 2006d]). Through Reaches 4a and 4b, the reservoir is roughly 0.25 to 1.75 miles (0.4 to 2.8 km) wide. Water depths through this reach range from 100 to 300 ft (30 to 91 m), but can become quite shallow near the banks, reflecting the topography of the drowned river valley. The Colville River is a major tributary to Reach 4, with a mean annual flow of 1,000 cfs.

- **Reach 5 (RM 640 to RM 617).** This reach is consistent with the upper portion of the Lower Reservoir Reach identified by USGS, and it extends upstream of the mouth of the Sanpoil River. It can be characterized as a lacustrine environment with slow-moving water. Within Reach 5, the Spokane River joins the Columbia River at RM 639.
- **Reach 6** (**RM 617 to Grand Coulee Dam near RM 597).** This reach is consistent with the lower portion of the Lower Reservoir Reach identified by USGS, and extends to the downstream boundary of the Site. It can be characterized as a lacustrine environment with slow-moving water. Within Reach 6, the Sanpoil River joins the Columbia River at RM 616.

Reaches 5 and 6 collectively represent the Lower Reservoir. Near the Grand Coulee Dam, the reservoir is roughly 1 mile wide, with maximum water depths on the order of 300 ft (91 m) deep. In many locations, shear valley walls rise nearly 1,000 ft (300 m) above the original river floodplain. Landslides and erosion along the banks of the flooded valley that forms the shores of Lake Roosevelt have largely occurred within lacustrine portions of the Site (Jones et al. 1961; USEPA 2006d; Whetten et al. 1969).

2.2 CLIMATE AND METEOROLOGY

The UCR area lies in the rain shadow of the Cascade Mountains, and therefore average annual rainfall is low in comparison to the western portion of the state. The northern areas of the Site receive about 20 in. (48 cm) of precipitation a year [NPS 2006b]). Moving south, the climate becomes far more arid, with average annual precipitation at Grand Coulee Dam of approximately 10 in. (24 cm). This precipitation occurs mostly in the winter and spring, while summer months are generally hot and dry. Short-term extreme rain events can occur. Trends in the last 50 to 100 years show a general decrease in winter precipitation and increase in summer precipitation (Ferguson 1999; USEPA 2008a).

During the summer months, temperatures at the Site typically range from 75 °F to 100°F (24 °C to 38°C) in daytime, dropping to 50°F to 60°F (10°C to 16°C) at night (NPS 2006b,c). Fall and spring provide plenty of sunshine and cooler temperatures. During these transitional times, the temperatures vary between 50°F and 80°F (10°C and 27°C) during the daytime and 30°F and 50°F (-1°C and 10°C) at night (NPS 2006a; USEPA 2008a).

Winters can be extremely cold in this area, with cold winds sweeping across the flat terrain. Daytime temperatures are generally between 25°F and 40°F (-4°C and 4°C), and nighttime temperature ranges may be as low as 15°F to 20°F (-9°C to -7°C). Trends in the last 50 to 100 years indicate a slight increase in winter temperatures and slight decrease in summer temperatures (Ferguson 1999; USEPA 2008a).

As a transition-type climate zone, climate within the Site is characterized by the interactions of three distinct types of air masses (Ferguson 1999)

- Moist marine air from the west that moderates seasonal temperatures
- Continental air from the east and south that is dry and cold in winter and hot with convective precipitation and lightning in summer
- Dry arctic air from the north that brings cold air to the area in winter and helps cool the area in summer.

The timing and extent of influence of these competing air masses are controlled largely by synoptic weather patterns and complex local topographic features that vary across the Site. For instance, prolonged periods of drought occur when Pacific storms are deflected around the region, preventing the intrusion of moist marine air. At these times, dry continental conditions prevail (USEPA 2008a).

2.3 HABITATS AND BIOLOGICAL COMMUNITIES

The UCR Site provides a variety of aquatic and terrestrial habitats that support diverse communities of aquatic life, wildlife, and vegetation. Organisms that comprise these communities form potential groups of ecological receptors that may be at risk from exposure to chemical stressors at the Site, and therefore warrant consideration in the BERA. The Site's broad ecological groups are summarized below, with detailed discussions presented within Appendices A and B.

2.3.1 Aquatic Habitat

Aquatic habitats of the UCR include those found in the riverine portion of the Site (Reaches 1 and 2), and those found in the lacustrine portion of the Site (Reaches 3, 4, 5, and 6). Depending on annual hydrologic conditions and interactions between the river and reservoir, portions of Reaches 2 and 3 may at times of the year illustrate transitional aquatic habitats sharing features of both a riverine and lacustrine environment. Short-term variations in the reservoir pool level further influence the kinds of available aquatic habitats, primarily in Reaches 3 through 6 and at the aquatic-terrestrial interface or near-shore areas.

2.3.2 Aquatic Communities

Major components of aquatic communities reported at the Site include plankton (phytoplankton/zooplankton), periphyton, macrophytes, benthic macroinvertebrates (including mussels), fish, and amphibians. The general characteristics of each aquatic community are briefly described below. More detailed descriptions are provided in Appendix A.

Phytoplankton abundance (in terms of chlorophyll *a* concentration) in the UCR cycles annually, generally reaching a maximum in May, and corresponding to the early part of the period of spring runoff (Lee et al. 2006; Scofield et al. 2007). Copepods and daphnids each make up approximately half of the total zooplankton biomass found in samples collected from throughout the UCR (1998 through 2007) (Scofield et al. 2007). Although copepods are numerically more abundant, daphnids reportedly are selectively consumed by fish, potentially because they are larger and more visible (Scofield et al. 2007). Zooplankton abundance is closely tied to phytoplankton productivity in the UCR, as in other lakes and reservoirs (Wetzel 2001).

In the reservoir portion of the Site, annual mean chlorophyll *a* concentrations of periphyton, also known as benthic or attached algae (Wetzel 2001), ranged from 2.0 to 15.1 mg/m^2, below the 20 mg/m^2 oligotrophic/mesotrophic threshold established by Dodds et al. (1998, as cited in Scofield et al. 2004). Periphyton have recently been dominated numerically by diatoms (96 percent), followed by green algae (4.0 percent) and cyanobacteria (0.1 percent). Quantitative descriptions of periphyton in the riverine portion of the Site were not found. Macrophyte beds are generally sparse throughout the UCR, and tend to be limited to areas of tributary mouths and embayments (Broch and Loescher 1991; Moore 1991, 1993).

Several studies provide descriptions of benthic invertebrate communities within the lacustrine portion of the UCR. Chironomids and oligochaetes appear to be most abundant in areas of the reservoir where investigations have been conducted, with gastropods also abundant in deeper waters (Bortleson et al. 1994; Johnson 1991a; Griffith et al. 1992; Voeller 1993).

Installation of the Grand Coulee Dam changed the UCR from a primarily lotic system to a combination of lotic and lentic conditions. This once salmonid-dominated fish community became a system dominated by cyprinid (minnow), centrarchid (sunfish), and catostomid (sucker) species, with remnant populations of redband trout (*Oncorhynchus mykiss gairdnerii*) and kokanee (*O. nerka*) (i.e., non-anadromous sockeye salmon) (Scholz et al. 1986; USFWS 1949). Fish abundance is not uniform throughout the UCR, likely due to differences in habitat types among the reaches. The Lake Roosevelt Fisheries Evaluation Program (LRFEP) has monitored fish in the reservoir for a number of years. In addition to redband trout and kokanee, other fish species found in the UCR include walleye (*Sander vitreus*), smallmouth bass (*Micropterus dolmieui*), yellow perch (*Perca flavescens*), burbot (*Lota lota*), lake whitefish (*Coregonus* *clupeaformis*), mountain whitefish (*Prosopium williamsoni*), rainbow trout (*O. mykiss*), white sturgeon (*Acipenser transmontanus*) and largescale sucker (*Catastomus macrocheilus*).

Several species of amphibians, including two salamander species, two toad species, and six frog species have been reported to occur in the UCR area (Hebner et al. 2000; Creveling and Renfrow 1986; Quigley et al. 2001; Marcot et al. 2003; WDFW 2008). Incidental observations by Washington State Department of Fish and Wildlife (WDFW) (McLellan 2008) while sampling fish confirm that amphibians are present.

2.3.3 Terrestrial Habitats

The overall terrestrial vegetation in the northern areas can be described as mixed conifer forests, and in the drier southern section of the Site as shrub-steppe vegetation. The dominant terrestrial habitat type immediately adjacent to the northern portion of the UCR (Reaches 1 through 4) is a mix of ponderosa pine (*Pinus ponderosa*) and eastside white oak (*Quercus alba*) forests, wetlands and riparian forests, eastside grasslands, and urban or agricultural lands. A transition from ponderosa pine forest to shrub-steppe occurs between Keller Ferry and the upper end of the Spokane River Arm at Little Falls Dam. In the southern portion of the UCR (i.e., Reaches 5 and 6), the habitat type is dominated by shrub-steppe shrublands mixed with agriculture, with ponderosa pine and eastside white oak forests occurring with increasing elevation. Additional information on plant species and vegetative land cover documented within the areas surrounding the UCR is available from Wilson (2006) and NPS (2005a,b) and is summarized within Appendix B.

Site-specific information on the presence and distribution of terrestrial wildlife has been compiled by a number of resource agencies (Hebner et al. 2000; Creveling and Renfrow 1986; CTFWD 2006; McCaffrey et al. 2003; Rodhouse 2005). Hebner et al. (2000) listed the wildlife species in the area from Grand Coulee Dam to the U.S.-Canada border and the reservoir to the surrounding ridges, while Quigley et al. (2001) and Marcot et al. (2003) have reported species data for the entirety of the interior Columbia Basin area. Wildlife species also have been listed by the Colville Tribes Fish and Wildlife Department (CTFWD 2006) and other resource agencies (McCaffrey et al. 2003; Rodhouse 2005). Approximately 98 species of upland mammals and 250 species of birds are cited in these documents. In addition to the amphibians noted above, several species of reptiles are present in the UCR uplands, including the painted turtle, five lizard species, western skink, and eight snake species (Hebner et al. 2000; Creveling and Renfrow 1986; Quigley et al. 2001; Marcot et al. 2003; WDFW 2008). Several ubiquitous terrestrial invertebrates have been reported near the UCR, notably the thatch ant (*Formica obscuripes*), western black widow spider (*Latrodectus hesperus*), and western yellow jacket (*Vespula pensylvanica*) (Quigley et al. 2001; Marcot et al. 2003).

2.3.4 Special Status Species

Special status species (wildlife, vegetation, and aquatic species) are summarized herein. Fifteen wildlife species reported to occur within the UCR and the surrounding area are listed as state and/or federal threatened or endangered (Table 2-1). The silver-bordered fritillary (butterfly) (*Boloria selene atrocostalis*) is the only federally listed terrestrial invertebrate, and is a state candidate species.

Threatened plant species reported within the area include the little bluestem (*Schizachyrium scoparium* var. *scoparium*) and the Palouse milk-vetch (*Astragalus arrectus*) (WDNR 2006). An endangered plant species reported in the area is the Columbia crazyweed (*Oxytropis campestris* var. *columbiana*) (WDNR 2006). Additional plant species that are of concern to the state due to their statewide decline include the fuzzytongue penstemon (*Penstemon eriantherus* var. *whitedii*), the least bladdery milk-vetch (*Astragalus microcystis*), black snake-root (*Sanicula marilandica*) and the Nuttall's pussy-toes (*Antennaria parvifolia*) (WDNR 2006).

A number of aquatic invertebrate species are also reported within the area, including the California floater (*Anodonta californiensis*), which is a mussel species that is a candidate for listing by both federal and state resource agencies (WDFW 2008). Bull trout (*Salvelinus confluentus*) is uncommon in the Site but is listed on the federal threatened and state candidate species lists.

Although not listed as a federal or state threatened or endangered species, Columbia white sturgeon is considered a species of special interest due to its known population and recruitment failure within portions of the Columbia River (i.e., between the Hugh Keenleyside and Grand Coulee dams), and the Kootenay River; its significant tribal importance; its endangered listing under Canada's Species at Risk Act (SARA 2009); and its listing on the Washington State Priority Species List.

2.4 SUMMARY OF SCREENING-LEVEL ECOLOGICAL RISK ASSESSMENT

An Upper Columbia River SLERA was conducted (TAI 2010) following the procedures described in EPA's ERA Guidance for Superfund (USEPA 1997a). Development of the SLERA incorporated the approaches defined in EPA's key ERA guidance documents (USEPA 1997a, 1998, 2006d, 2007a). The primary purpose of the SLERA was to determine if there are adequate data to make a determination on the potential risks posed by chemicals in UCR environmental media to ecological receptors. If adequate data are available for certain chemicals of interest and environmental media, and risks are determined to be acceptable, then no further assessment would be warranted. If inadequate data are available, or there is potential for risk, then further evaluation would be required.

The SLERA represents the initial two steps of EPA's eight-step process for conducting ERAs (Figure 1-2; USEPA 1997a). Step 1 involved a screening-level problem formulation and ecological effects characterization, and Step 2 included a screening-level exposure estimate and risk calculation. Results of the SLERA will be further refined in Section 4 of this document (i.e., Step 3 of the ERA process).

Consistent with EPA guidance (USEPA 1997a), Step 1 of the SLERA included a sitewide CSM, (refer to Figure 4-1 in Section 4) that addressed the following elements:

- Environmental setting
- Contaminants known or suspected to exist in the UCR
- Contaminant transport and fate mechanisms
- Mechanisms of ecotoxicity associated with broad classes of contaminants and potential ecological receptor groups
- Potentially complete exposure pathways
- Preliminary assessment endpoints
- Screening-level ecotoxicity values.

Step 2 of the SLERA addressed the following:

- Determination of screening-level exposure estimates
- Calculation of screening-level risk estimates
- Risk characterization and evaluation of uncertainties.

Any potential risks to ecological receptor groups identified in the SLERA are not considered definitive and will be evaluated further in Section 4 of this Work Plan by refining assumptions and developing Site-specific data.

Human activities may become sources of chemical stressors through the release or migration of contaminated Site media (air, sediment/soil, wastewater, stormwater, groundwater, or spills) to the UCR. Both the RI/FS work plan (USEPA 2008a) and the SLERA (TAI 2010) describe the primary anthropogenic sources for chemical stressors to the UCR, from which the CSMs were derived. The EPA concluded that both the smelter in Trail, British Columbia and the former Le Roi Smelter in Northport, Washington were sources of contamination to the UCR Site; however, the Trail smelter was identified as the primary source of contamination (USEPA 2003a). This is because until the practice was discontinued in mid-1995, up to 145,000 tonnes of slag were discharged annually from the smelter to the Columbia River. The slag was carried downstream and settled out in slower moving, depositional portions of the river (G3 Consulting 2001). Other discharges from the Trail facility included liquid effluent discharged to the Columbia River through five outfalls, including one outfall from the Warfield Fertilizer Operation, three outfalls from the metallurgical plants, and one outfall from the slag launder system (Cominco 1997), and atmospheric emissions from the smelter stack.

The expanded site investigations and the 2005 Phase 1 remedial investigation documented sediment contamination along the Site from the U.S.-Canada border to the Grand Coulee Dam (USEPA 2003a, 2006d). The mines and mills along the tributaries to the UCR were not identified as current sources of contamination to the Site because, with the exception of the Spokane River, Phase 1 sediment samples collected near the mouths of selected UCR tributaries did not contain notably elevated concentrations of contaminants of interest (COIs; USEPA 2008a). The mines and mills in the UCR drainage basin may be investigated in the future if anomalous and significant contaminant concentrations (relative to risk) are found at confluences of tributaries with the UCR and a potential upstream source is suspected (USEPA 2008a). Additional potential sources (e.g., dry blowing sediment dust) will also be considered within this work plan and as necessary in the BERA.

As outlined and presented within the SLERA (TAI 2010) and RI/FS work plan (USEPA 2008a), a number of potential point and non-point chemical sources were reviewed to develop the list of contaminants of potential concern (COPCs). The SLERA evaluated existing data for several groups of COIs in environmental media (i.e., surface water, porewater, sediment, soil, and fish tissue), including metals/metalloids, nutrients, semivolatile organic compounds (SVOCs), polycyclic aromatic hydrocarbons (PAHs), pesticides, polychlorinated biphenyls (PCBs), polybrominated diphenyl ethers (PBDEs), polychlorinated dibenzo-p-dioxins (PCDDs), and polychlorinated dibenzofurans (PCDFs) (Table 2-2). Maximum site-wide measured concentrations were compared to conservative accepted toxicity benchmarks or thresholds. This analysis resulted in an initial refinement of COPCs by screening out COIs from further evaluation for specific media, and identifying those that required further evaluation and refinement. A summary of the screening analysis per chemical by media is detailed in Table 2-3.

As illustrated within Table 2-3, conservative scientific management decisions (SMDs) about each chemical in each medium can be made. Those decisions include one of the following:

1. **SMD No. 1.** There are adequate data to conclude that there are no unacceptable ecological risks for a chemical (medium and receptor specific), and therefore there is no need for further study or for remediation on the basis of ecological risk.
2. **SMD No. 2.** The information was not adequate to make a risk-based management decision, and will be further evaluated and refined within the ERA process.
3. **SMD No. 3.** The available information indicates a potential for adverse ecological effects, and a more thorough assessment will be conducted in the BERA.

As detailed within the SLERA (TAI 2010), a number of chemicals or groups of chemicals were identified as not posing or presenting an unacceptable risk to ecological receptors or the environment. As such and on a receptor and medium specific basis, do not require further evaluation or refinement within the ERA (e.g., PAHs and PCBs in sediments for benthic invertebrates). For those chemicals in which SMDs Nos. 2 or 3 were obtained, and as per EPA guidance (USEPA 1997a), this work plan further refines the problem formulation (refer to Section 4) and approaches that will be used to conduct the BERA.

3 EXISTING DATA

Many studies that may be pertinent to understanding site conditions and completing the RI/FS and BERA have been conducted at the Site. This section presents summaries of several of these studies conducted to date and provides the results of preliminary evaluations of distribution of COPCs in environmental media at the Site, including identification of patterns and trends in the data. It should be noted that many of the studies and monitoring programs described below were conducted for purposes unrelated to the UCR RI/FS and BERA. EPA has determined that the data from these studies are useful for the purpose of understanding the distribution of chemicals in environmental media in the UCR and helpful in identifying data gaps that will need to be filled by additional studies (see Section 6). As the BERA progresses, the quality of the existing data and suitability for inclusion in the BERA and for other purposes will be assessed according to procedures that will be reviewed and approved by the EPA. Preliminary analyses and statements included in this section may be augmented in subsequent documents with additional information included in these studies but not summarized here, and that the summaries presented here may be modified during more rigorous and thorough evaluations in the development of subsequent documents supporting the BERA.

3.1 SUMMARY OF EXISTING DATA

This section provides summary descriptions of studies conducted to date at the Site. The descriptions are organized by media and are listed from oldest to newest within each medium. A synopsis of the investigations covered in this section is provided in Table 3-1.

3.1.1 Surface Water Quality

The surface water quality studies summarized in this section primarily focus on samples collected downstream of the U.S.-Canada border and were performed to meet a variety of objectives and may not be representative of all Site surface water.

In addition to information in reports and publications, surface water quality data relevant to the Site are also available from several electronic databases. Data collected and reported by Environment Canada (EC), USGS, and the Washington State Department of Ecology (Ecology) are of particular relevance. Further evaluation of select data from these studies is provided in Appendix C.

Environment Canada. Data from EC are available for locations upstream of the Site and include Federal-Provincial water quality monitoring stations on the Columbia River at Birchbank and Waneta, B.C., as well as a station on the Pend Orielle River at Waneta.

The Waneta station on the Columbia River is located immediately upstream of the confluence with the Pend Oreille River. The Waneta station on the Pend Oreille River is located immediately upstream of the confluence with the Columbia River. At Birchbank, samples have been collected every two weeks over the period 1983-present. At the Columbia River Waneta station, weekly samples have been collected over the period 1979-present. At the Pend Oreille Waneta station, monthly samples have been collected over the period 1979-present. Analytes at all three locations included acid/base chemistry, major ions, carbon (organic carbon content), nutrients, physical parameters such as temperature, and metals/metalloids/non-metals. At these stations, data have been acquired through 2007, but the sampling programs are ongoing. Additional data will be acquired for the BERA as they become available. However, it should be noted that the list of analytes available at each station, as well as analytical methods, varied over the period of record.

Ecology. For more than 50 years, Ecology has conducted monthly water quality monitoring at hundreds of stream stations throughout the state. One of the monitoring stations sampled as part of this program is located northeast of Northport at the bridge crossing the UCR on State Highway 25 (Station 61A070 at RM 735). Data collection at this station began in 1952 and continues into the present. Over the period January 1995 to June 2007, unfiltered and filtered water samples were collected on a monthly basis for analysis of metals (bimonthly), nutrients, major ions, other water quality parameters. At the Northport station, data have been acquired through 2007. Additional data will be acquired for the BERA as they become available. Other Ecology stations of potential relevance include the Kettle River near Barstow (60A070), Colville River at Kettle Falls (59A070), Sanpoil River at Keller (52A070), and the Columbia River at Grand Coulee (53A070). Not all analytes are available at all stations. The period of record for each analyte differs by stations. Additional data from these stations will be acquired for the BERA as they become available.

USGS. Data from USGS are available for stations on the Columbia River at Northport (12400520), Colville River at Kettle Falls (12409000), Sanpoil River above Jack Creek at Keller (12434590), and the Spokane River at Long Lake (12433000). The period of record at these stations vary. At Northport, data are available for the period 1958-2007. Analytes of note include total recoverable metal measurements. Data for these stations were acquired for the period of record through 2007. Additional data will be acquired for the BERA as they become available.

> *Johnson, A., B. Yake, and D. Norton. 1989. An assessment of metals contamination in Lake Roosevelt. Segment No. 26-00-04. Washington State Department of Ecology, Olympia, WA. 84 pp.*

This study was conducted by Ecology to determine the extent and significance of metal contamination in Lake Roosevelt. In 1986, Ecology conducted surveys of metal concentrations in bottom sediments, water, and fish from Lake Roosevelt, the dammed portion of the upper Columbia River in the reach between the U.S.-Canada border and Grand Coulee Dam. The impetus for these surveys were reports of metals contamination in fish and other environmental samples, coupled with the presence of a potentially significant source—the Cominco Limited lead-zinc smelter and refinery in British Columbia. It consisted of taking water samples from the lake and its tributaries, collecting fish tissue samples, a sediment core, and performing bioassays. The specific study objectives were to 1) determine extent of metal contamination of lake sediments as a function of distance from the U.S.-Canada border, 2) obtain a sediment core and analyze the history of metal accumulation, 3) screen sediment samples for toxicity to aquatic organisms, 4) survey metal concentrations in water from the lake and tributaries and assess potential impacts on aquatic organisms, 5) determine the extent of metal accumulation in fish and the potential for adverse effects, and 6) determine if metal concentrations in sport fish were detrimental to safe consumption by humans at recreational fishing rates.

Methods. Two sets of grab water samples were collected in May and August of 1986– one during spring runoff and one during summer low flow. Samples were taken at 13 sampling sites– 2 in the Columbia River (one collected at Northport and the other collected below Grand Coulee Dam) and the remainder in tributaries including the Colville, Kettle, Spokane, and Sanpoil rivers. Surface samples were collected by hand and bottom samples were collected using a 10-L, Teflon-lined, Go-Flo bottle and preserved in storage containers with nitric acid. Samples for dissolved metals were field filtered under N_2 atmosphere and filtered through a 0.45-μm filter. Temperature and pH were measured in the field.

Metals were measured using atomic absorption spectrometry (AAS). Methods for metals in water are as follows: Total recoverable zinc (EPA 289.2), total recoverable copper (EPA 220.2), total recoverable lead (EPA 239.2), total recoverable arsenic (EPA 206.2), total recoverable cadmium (EPA 213.2), and total recoverable mercury (EPA 245.1).

Results. The authors stated "Metals concentrations in water samples from the lake and its tributaries were generally low and within EPA (water quality criteria) WQC. However, long-term monitoring by the U.S. Geological Survey shows metals concentrations in the Columbia River 12 miles below the Canadian border to be highly variable, sometimes exceeding EPA criteria. EC has also observed this phenomenon in the Columbia and attributed this to Cominco discharges." It was also noted that "USGS, however, reports generally higher levels of zinc, copper, and lead in the Columbia River than found during the present study... One factor that may have contributed to these disparate results is sampling frequency. The water quality data from the present study are based on a small number of samples." The sediment, toxicity, and fish tissue results are summarized below within separate sections describing each medium.

Bortleson, G.C., S.E. Cox, M.D. Munn, R.J. Schumaker, and E.K. Block. 1994. Sediment-quality assessment of Franklin D. Roosevelt Lake and the upstream reach of the Columbia River, Washington, 1992. U.S. Geological Survey, Open-File Report 94-315, 130pp. (Reprinted 2001 as U.S. Geological Survey Water Supply Paper 2496, 130 pp.)

This study was conducted by researchers from the USGS, in cooperation with the USEPA, to provide a comprehensive characterization of the distribution of metals and organic compounds in surface sediments throughout the UCR, and represents a more detailed characterization of surface sediments than Johnson et al. (1989, 1991a) in the Columbia River upstream of Lake Roosevelt to the U.S.-Canada border. Surface water, suspended sediment, and porewater samples were also collected as part of this investigation that was conducted in September and October of 1992. Additional description of this study is provided in the sediment toxicity study summaries (Section 3.1.3).

Serdar, D., B. Yake, and J. Cubbage. 1994. Contaminant trends in Lake Roosevelt. Publication No. 94-185. Washington State Department of Ecology, Olympia, WA.

During 1990, 1992, and 1993, Ecology monitored concentrations of dioxins, furans, and trace metals in suspended particles and selected fish tissues collected from upper Lake Roosevelt. The "primary objective of the monitoring was to provide information about temporal trends in concentrations of these pollutants".

Methods. Suspended sediment samples were collected in 1990, 1992, and 1993 from surface water downstream of Northport using two Sedisamp II continuous-flow centrifuges. Particles were analyzed for polychlorinated dibenzodioxins (PCDDs), polychlorinated dibenzofurans (PCDFs), metals, total organic carbon, and percent moisture. Water was pumped from an intake situated in the main current approximately 50 ft offshore with the depth of the intake adjusted periodically to obtain a depth-integrated sample. XAD™ resin columns were used to estimate concentrations of PCDDs and PCDFs in the dissolved phase; however, "limited success" was achieved measuring dissolved PCDDs/PCDFs due to equipment failure.

Analysis of 2,3,7,8-substituted PCDD and PCDF congeners was conducted using gas chromatography/mass spectrometry (GC/MS) method 8290. Samples for zinc, lead, copper, arsenic, and cadmium were digested using EPA method 3050 modified by the addition of 200 µL hydrofluoric acid (HF) to the HNO_3 acid mix. The analytical method was inductively coupled plasma (ICP) method 200.7. Mercury analysis used the EPA cold vapor method 245.1 modified for sediment. General water quality parameters were also collected during fish sampling in 1990, 1992, and 1993.

Results. The most toxic dioxin, 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) was not detected in suspended particles in the 1990, 1992, or 1993 samples. The concentration of a less potent furan, 2,3,7,8-tetrachlorodibenzo-furan (TCDF) in suspended particles samples, and associated toxicity as measured by toxic equivalent (TEQs), decreased over time. However, suspended particle samples from 1993 contained several less potent dioxin/furan species that had not been detected in previous years. The presence of these less potent dioxins/furans resulted in slightly increased TEQs) between 1992 and 1993. TEQs were approximately 10.8 pg/g in 1990 to 1.1 in 1992, and 3.3 in 1993.

Trace metals in suspended particles from 1992 and 1993, in order of decreasing concentration, were zinc (1,478 and 1,130 ppm), lead (554 and 498 ppm), copper (352 and 256), arsenic (442 and 24.8 ppm), cadmium (16.1 and 10.3 ppm), and mercury (13.7 and 2.46 ppm). Except for mercury, the concentrations of specific metals in 1993 were 10 to 44 percent lower than in 1992. Mercury in the 1992 river samples was elevated by a spill at Cominco. In 1993, the mercury levels were 80 percent lower than the 1992 concentrations.

Conclusions. The authors stated

- "There is an apparent shift in the pattern of PCDDs and PCDFs in suspended particles entering Lake Roosevelt. Several congeners with 5 to 8 chlorine atoms are present at greater concentrations than during 1990 and, possibly, 1992. The reason for this shift in congener pattern is not known."
- "During the 1993 investigation, approximately 25 percent of the 2,3,7,8-TCDF in the Columbia River at Northport appeared to be associated with suspended particles. The remaining 75 percent appeared to be in the dissolved phase."
- "Loading of zinc, lead, copper, cadmium, mercury, and to a lesser degree arsenic, from upriver sources continues to represent a significant source of contamination to Lake Roosevelt. The Cominco smelter in Trail, B.C. is almost certainly the major source of these metals. 1993 data provided the first opportunity to look for trends in particulate contamination. Concentrations of all metals in the 1993 particle sample were lower than those in 1992. This is probably due, at least in part, to the elimination of a major sewer outfall at Cominco in 1993."
- "Despite these decreases, metals concentrations in Columbia River particles remain very high."

Riedel, J.L. 1997. Lake Roosevelt National Recreation Area, Washington: Water resources scoping report. National Park Service Technical Report NPS/NRWRD/NRTR-97/107. 84pp.

This report summarizes the state of knowledge about hydrologic and water resource issues including water quality in Lake Roosevelt at the time of writing. The National Park Service has principal responsibility for management of the Lake Roosevelt National Recreation Area, but concurrent jurisdictional authorities exist for water management. The authors noted that this body of water was classified as AA (extraordinary) by the

Washington Department of Ecology, and that USEPA and other agencies had detected high levels of lead, zinc, cadmium and arsenic in fish tissue. The report states that the "primary point sources for the heavy metal pollution were the slag and sewer discharge at the COMINCO lead and zinc processing plant in Trail, British Columbia" and that "high concentrations of chlorinated organic compounds in Lake Roosevelt fish...was the CELGAR paper mill near Castelgar B.C." They also noted that "Improvements in these plants over the past 10 years have drastically reduced the amount of pollution dumped into the Columbia River. Dioxin or furan is no longer detectable in the effluent from the CELGAR plant, and COMINCO has decreased the amount of heavy-metal bearing slag dumped from approximately 400 tons/day to less than 5 tons/day, and is scheduled to eliminate discharge in mid-1997. Nonetheless, vast quantities of the previously released pollutants likely remain in sediments at the bottom of Lake Roosevelt, and these industries continue to discharge pollutants, presenting a long-term challenge to water resource management" Bacteriological pollution was also a continuing concern.

Results. The report identifies several threats to water quality in Lake Roosevelt in addition to the potential continuing concerns from the above-mentioned Cominco and Celgar plants. The report states "Developments near Fort Spokane threaten the groundwater at the site, which the National Park Service (NPS) relies on for visitor and administrative use." The report further states that "Land use in and near the unit are a concern to water quality and aquatic resources as non-point sources of pollution. The NPS currently has authorized 155 special use permits for private developments, including vacation cabins, docks, buoys, water delivery systems, and agricultural activities such as grazing." The authors also stated "Reservoir bank erosion threatens park resources and facilities at hundreds of locations along the 435 miles of shoreline managed by the NPS…Risk of landslides is a moderate concern for 30-50 ft drawdowns, and a major concern for drawdowns of more than 50 ft…Bureau of Reclamation management of reservoir levels has attempted to avoid these conditions for the past 20 years. Natural variation in runoff and competing demands for water and reservoir capacity make managing reservoir levels difficult." The report states "Water pollution and reservoir operation for flood control, irrigation, and enhancing anadromous fish runs on the lower Columbia River System, impact and limit ecological development in the reservoir. Low retention time of water in the reservoir due to large inflow is believed to limit plankton growth, which is the basis for the entire reservoir ecosystem. Drawdowns also negatively impact several non-native popular game fish species. Several introduced species, including Eurasian milfoil, are a growing ecological concern. Concentrations of heavy metal and organochlorine pollutants remain high in predator and bottom-feeding fish."

Conclusion. This report identifies water resource issues such as the need for emergency response planning and baseline resource inventories (wetlands) and recommends that the NPS maintain at least one full-time water resource management position at Lake Roosevelt and that they develop a Water Resource Management Plan.

MESL (MacDonald Environmental Sciences Ltd.). 1997. Lower Columbia River from Birchbank to the international border: Water quality assessment and recommended objectives. Technical report. Available at: http://www.env.gov.bc.ca/wat/wq/objectives/birchbank/Birchbanktech.pdf. Prepared for Environment Canada and British Columbia Ministry of Environment, Lands and Parks.

The purpose of this report was to evaluate and develop water quality objectives (WQOs) to support management of water quality and quantity in the trans-boundary reach of the Columbia River.

Methods. The water quality of the Columbia River from Birchbank to the U.S.-Canada border was assessed using data collected at six stations within the reach. These sampling stations included the Columbia River at Birchbank (located 9 km upstream of Trail, BC), the Columbia River at East Trail (located at the Old Trail Bridge), the Columbia River at West Trail (located at the Old Trail Bridge), the Columbia River at Waneta, Stoney (Topping) Creek, and Beaver Creek. While data were available between the period of 1968 and 1996, data from 1990-1996 were primarily used in order to minimize the bias due to changes in analytical methods. Data were compiled from EC's ENVIRODAT database and the Ministry of Environment, Land and Parks' SEAM database. At all sites on the Columbia River, physical variables including pH, water temperature, specific conductivity and hardness were measured. In addition, data on major ions, nutrients, metals and metalloids, and microbiological indicators were also collected at each of the sites. Resin acid concentrations at the Columbia River at Waneta site were measured to assess the transport and fate of these pollutants from the pulp mill at Castlegar, BC. Data collected from Stoney (Topping) Creek (1986-1994) and Beaver Creek (1968-1983) included physical parameters, nutrients, metals and microbiological indicators. Total dissolved gases in the Columbia River were assessed using data from the Columbia River from Keenleyside Dam to Birchbank (Butcher 1992).

Results. Assessing water quality in the Columbia River from Birchbank to the U.S.-Canada border showed that WQOs were generally met. However, pH ranged outside the acceptable range (6.5 to 8.5) on some occasions. Data collected from previous assessments of the Columbia River also showed that total dissolved gas concentrations commonly exceeded the WQC of 110 percent downstream of the Keenleyside Dam and at the Birchbank site (Butcher 1992). The WQOs for microbiological indicators (i.e., faecal coliforms, *E. coli*, and *Enterococcus* spp.) were exceeded on multiple occasions at both the West Trail site and at the Waneta site on the Columbia River, indicating that contamination from municipal and domestic releases of treated sewage is common. The levels of metals in water collected at the Birchbank site showed considerable variability over the period of record. However, the more recent data indicated that the maximum and 30-day average WQOs were met for all of the metals. At the East and West Trail sites (Old Trail Bridge), metals concentrations varied between the east and west river banks, indicating incomplete mixing in this area. The most recent data collected at the

West Trail site showed that cadmium, copper, lead, and zinc exceeded their respective 30-day average WQO. At the Waneta site, elevated levels of copper, mercury and zinc were observed between 1990 and 1996. However, data collected at the same site in 1997 showed that cadmium, lead, and zinc concentrations exceeded WQOs; the reduction in mercury at the site was attributed to the termination of phosphate-fertilizer production at Cominco, which utilized contaminated apatite.

In Stoney Creek, the data showed elevated concentrations of arsenic, cadmium, chromium, copper, mercury, lead and zinc compared to levels in the Columbia River. In Beaver Creek, elevated levels of ammonia, total phosphorus, and sulphate were also observed. Additionally, very high levels of faecal coliforms were measured in Beaver Creek, likely reflecting the municipal waste discharges to the system.

Conclusions. The author concluded that WQOs were developed for pH, total dissolved gases (i.e., gas supersaturation), microbiological indicators (i.e., faecal coliforms, *E. coli*, and *Enterococcus* spp.), ammonia, and various metals (i.e., arsenic, cadmium, chromium, copper, lead, thallium, and zinc). Generally, WQOs were set to address concerns over flow regulation (from dams and impoundments along the Columbia River and its tributaries), which affect total dissolved gas concentrations; impaired water quality due to the discharge of municipal and domestic waste, which have increased concentrations of microbiological indicators and ammonia; and industrial discharges (from Cominco's lead-zinc smelter and the Celgar pulp mill), which have contributed to elevated levels of various metals and other contaminants in the Columbia River. WQOs for the Columbia River were generally set to protect aquatic life with both maximum and 30 d average WQOs developed. As the Columbia River is used as a drinking water source for communities in the area, WQOs for microbiological indicators and arsenic serve to address potential unacceptable risks to human health. While some improvements in water quality were observed in the Columbia River in more recent studies, exceedances of cadmium, lead, and zinc were still present at the Waneta site. Recommendations included the development of toxic-units models or chemical mixture models to evaluate the potential synergistic, additive, or antagonistic effects of key contaminants in the river.

> *USEPA. 2003a. Upper Columbia River expanded site inspection report, Northeast Washington (Region 10, START-2). Prepared for U.S. Environmental Protection Agency, Region 10, Seattle, WA, by the Superfund Technical Assessment and Response Team. 84 pp.*

Pursuant to EPA Superfund Technical Assessment and Response Team (START)-2 Contract Nos. 68-S0-01-01 and 68-S0-01-02, Ecology and Environment, Inc. (E & E) and Roy F. Weston, Inc. (WESTON) conducted an expanded site inspection (ESI) of sediment contamination in the upper Columbia River and its tributaries from the U.S.-Canada border downstream to approximately RM 675 near Inchelium. The impetus for this investigation was EPA's evaluation of information and data from previous studies indicating that sediment sampling of the upper Columbia River was necessary to determine if releases of hazardous substances are occurring and if there is a potential for releases to affect human health and the environment. The main objective of the ESI was to collect sufficient data to determine the site's potential for inclusion on the National Priorities List (NPL) and establish priorities for additional action, if warranted. Note that limited surface water samples were collected as part of the ESI (as summarized here). However, the ESI report focuses almost exclusively on the sediment metal data.

Methods. Surface water samples were taken from the City of Grand Coulee municipal drinking water system distribution point and from Lake Roosevelt near the system intake. These samples were analyzed for target analyte list (TAL) metals, Pesticides/PCBs, volatile organic compounds (VOCs), and SVOCs. In addition, seven surface water samples were collected from tributaries to the lower Pend Oreille River and Upper Columbia River and analyzed for TAL metals only.

The analytical methods were as follows: TAL metals (Contract Laboratory Program [CLP] Statement of Work [SOW] ILM04.1); total organic carbon (TOC) (EPA SW-846 Method 9060 [modified]); and pesticides/PCBs, VOCs, and SVOCs (CLP SOW OLM 04.2).

Results. The two surface water samples collected from the City of Grand Coulee municipal drinking water system distribution point (CW001) and from Lake Roosevelt near the system intake (CW002) did not contain pesticide/PCBs or SVOCs at concentrations above detection limits. The analysis for VOCs revealed the presence of chloroform at 26 μg/L in water sample CW001.

Conclusions. The concentrations of chloroform and detected metals were not above existing federal drinking water standards. However, the authors expressed concern for human exposure from the consumption of lake or river water due to direct contamination by slag or sediments.

> *Johnson, A., K. Seiders, C. Deligeannis, K. Kinney, P. Sandvik, B. Era-Miller, and D. Alkire. 2006. PBDE flame retardants in Washington rivers and lakes: Concentrations in fish and water 2005-06. Publication No. 06-03-027. Washington State Department of Ecology, Olympia, WA.*

Ecology analyzed PBDE flame retardants in freshwater fish and water samples collected statewide during 2005-2006. This was done in response to increasing PBDE levels in the environment and concern about potential adverse human health effects from fish consumption. The goal of this study was to establish baseline conditions that could be used to evaluate the effectiveness of the Washington State PBDE Chemical Action Plan and other efforts to reduce PBDE inputs to the environment.

Methods. Semipermeable membrane devices (SPMDs) were deployed for approximately 28 days. Samples were collected throughout the region, one of which, collected in September, 2005, was from the Upper Columbia River. SPMDs were extracted using a proprietary method at Environmental Sampling Technologies. The extract was analyzed for PBDEs and PCBs using methods EPA 8270 and EPA 8082, respectively. Grab samples were analyzed for total suspended solids (TSS) (EPA 160.2) and TOC (EPA 415.1) at the beginning, middle, and end of the deployment period. Conductivity was measured with a field meter. The latitude and longitude of each sampling site was recorded from a global positioning system (GPS) receiver.

Results. Data were obtained on concentrations of PBDE-47, -49, -66, -71, -99, -100, -138, -153, -154, -183, -184, -190, and -209 in 16 water samples. Upper Columbia river samples found detected concentrations of two congeners (PBDE-47 and PBDE-99). Relatively higher PBDE levels were found in the Spokane River. The report also states that "instrument tuning, calibration, internal standards, and matrix spikes were within QC limits".

Conclusions. The authors stated "The PBDE congeners most frequently detected in fish were also detected in the SPMDs. These included PBDE -47, -49, -99, -100, -153, and -154. Except for PBDE-47 and -99, most congeners were detected near the reporting limit. PBDE-209 is too large a molecule to be taken up effectively by SPMDs and was not detected. The amount absorbed by an SPMD is proportional to the local water concentration. As in the fish samples, SPMDs deployed in the Spokane River, Columbia River, Lake Washington, and Yakima River had the largest residues. The Duwamish River, Potholes Reservoir, Queets River, and Lake Ozette had very low residues."

> *Paulson, A.J., R.J. Wagner, R.F. Sanzolone, and S.E. Cox. 2006. Concentrations of elements in sediments and selective fractions of sediments, and in natural waters in contact with sediments from Lake Roosevelt, Washington, September 2004. Open-File Report 2006-1350. U.S. Geological Survey. 84 pp.*
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> *Paulson, A.J. and S.E. Cox. 2007. Release of elements to natural water from sediments of Lake Roosevelt, Washington, USA. Environmental Toxicology and Chemistry 26:2550-2559.*

This study was conducted by researchers from the USGS to present data regarding concentrations of elements in sediments, selective fractions of sediments, and natural waters in contact with sediments from Lake Roosevelt. Samples were collected in September, 2004. The specific objectives of the study were to 1) evaluate the release of elements from bed sediments of Lake Roosevelt; and 2) determine concentrations of elements in filtered water after contact with sediments. Several types of water (surface water, porewater, water incubated with sediments, and water tumbled with sediment) were analyzed for elemental concentrations. It should be noted that the primary focus of this study was to assess element releases from sediments rather than to characterize surface water, but the limited water data is summarized below.

Methods. Samples were collected from seven sites spaced throughout the reservoir to capture the areal distribution of trace element concentrations in sediments of the reservoir. As noted by the authors, one additional site on the Sanpoil River was also sampled and assumed to be representative of background conditions. Information for sediment from a site at RM 743 (Black Sand Beach) was also presented. Water samples in various degrees of contact with the sediment were analyzed for 10 alkali and alkaline earth elements, 5 non-metals, 25 metals, and 16 lanthanide and actinide elements. As described by Paulson et al. (2006), filtered water samples included 10 samples from the reservoir water column at 8 sites, 32 samples of porewater, 55 samples from reservoir water overlying sediments in 8 cores from the site incubated in a field laboratory, and 24 water samples that were filtered after being tumbled with sediments from 8 sites. Three surface water samples were collected directly from the water column at two sites (LR-5, LR-7, and a replicate at LR-7); water samples from the other six sites were overlying water from draining or sealed sediment core samples. Quality assurance/quality control (QA/QC) analyses are reported.

Results. Of the 10 samples of reservoir water analyzed (including those obtained from sealed or draining sediment cores), beryllium and cesium were not detected in any samples and lithium was detected in just one sample. Of the five non-metals analyzed, only selenium was not detected in any reservoir nor porewater nor core incubation sample in contrast to aluminum and lead, which were detected in all porewater samples and copper in all incubation samples. Concentrations of 9 of the 25 metals analyzed (aluminum, antimony, cadmium, cobalt, copper, manganese, lead, nickel, and zinc) were greater than the reporting limit. Concentrations of lead and zinc in reservoir water from a sealed box core were higher than those collected from the surface of the water column or those from draining a box core. It was not known if these elevated concentrations for reservoir water were due to proximity to the sediments or contamination from the box corer. Only two of the lanthanide and actinide elements (erbium and uranium) were detected in reservoir water. Twenty-three percent of all elements analyzed in reservoir water were detected. Within the group of elements, 71 percent of the alkali and alkaline earth element analyses, 38 percent of the non-metal analyses, 15 percent of the metal analyses, and 1 percent of the lanthanide and actinide element analyses were detected. Bismuth, chromium, silver, and zirconium were notable in not being detected in any water samples. However, in general, the number of detected elements increased with the extent of mixture and contact of water with sediment.

Conclusions. These authors state "The percentage of detected elements of each type of water sample generally reflects the extent of contact with sediment phases. Only 23 percent of the analyses of reservoir water, which had little contact with the sediments, were detected. The percentage of porewater concentrations that were greater than the reporting limit (47 percent) was only slightly less than that of the higher energy tumbling experiment (48 percent). The percentage of analyses of water overlying the incubation cores with concentrations greater than the reporting limit (29 percent) was intermediate between the reservoir water and the leachate from the higher energy tumbling experiments."

> *Scofield, B. and D. Pavlik-Kunkel. 2007. Trace metal concentrations in surface water of Lake Roosevelt. Supplemental Report, January 1998–March 2000. Prepared for U.S. Department of Energy, Bonneville Power Administration, Division of Fish and Wildlife, Portland, OR; Spokane Tribe of Indians, Department of Natural Resources, Lake Roosevelt Fisheries Evaluation Program, Wellpinit, WA.*

This study was conducted by researchers from the Spokane Tribe of Indians Department of Natural Resources to further understand trace metal concentrations in surface water and supplement annual fisheries/limnological reports for Lake Roosevelt. Arsenic, cadmium, copper, lead, mercury, and zinc were focused on because past work identified these trace metals as the most concerning. The Spokane Tribe co-manages the Lake Roosevelt fishery and is concerned with how these contaminants may be affecting the biota as well as providing a pathway for human exposure. The specific objectives of the study were to 1) report observed element concentrations in Lake Roosevelt surface water and any distinguishable spatial or temporal trends; 2) relate contaminants of concern with environmental variables; and 3) compare trace metal concentrations to Spokane Tribal and Washington State water quality standards. Although data usability concerns exist because concentrations of many analytes were below reporting limits, this study represents the only known large-scale survey of surface water conditions in the Columbia River between the U.S.-Canada border and Grand Coulee Dam.

Methods. Samples were collected as part of a series of 38 synoptic surveys conducted at 2 to 3 week intervals over the period January 1998 through March 2000. Samples were collected at 11 stations located throughout the Site (from Evan's Landing at RM 710 to the Sanpoil River at RM 616). During the 1998 surveys, surface water samples were collected using a Van Dorn bottle at two depths, the mid photic zone and 1 m below the photic zone. The photic zone depth was defined as the depth at which 1 percent of ambient irradiance penetrates the water column and was determined using an irradiameter. During 1999 to 2000, surface water was collected from the photic zone using a depth-integrated sampler. The authors noted that lead weights attached to the samplers had the potential to contaminate samples. Trace metal (element) concentrations were determined as total recoverable concentrations. Arsenic, cadmium, copper, zinc and most other trace elements were analyzed by inductively coupled atomic emission spectrometry (ICAES, method 200.7; NEMI 2005a). Arsenic and lead were analyzed by graphite furnace atomic absorption (GFAA, method 200.9; NEMI 2005b). Mercury was analyzed by cold vapor atomic absorption spectrometry (CVAAS, method 245.1; NEMI 2005c). Hardness was estimated from calcium and magnesium concentrations. A full list of analytical methods was tabulated by the authors. QA/QC analyses are reported.

Results. Spatially, mean euphotic depth during the study period increased while moving from the upper reservoir to the lower reservoir, from 11.2 m at Evan's Landing to 12.7 m at Spring Canyon. Seasonally, the average euphotic depth across all locations was least during spring runoff (6.8 m in April) and highest in fall (15.6 m in November). Turbidity measurements showed similar trends. Total recoverable arsenic was measured 15 times out of 608 samples. No spatial or temporal patterns were readily discernable in arsenic concentrations but measurable arsenic was most frequently measured at Porcupine Bay. Arsenic concentrations were always below chronic and acute toxicity criteria. Total recoverable cadmium was detected 8 times in 608 samples. No spatial or temporal patterns were readily discernable in cadmium concentrations. Cadmium concentrations exceeded acute and chronic criteria in 85 and 100 percent of samples respectively, although the reporting limit was greater than both the acute and chronic criteria and most of the exceedances were based on estimated concentrations. Total recoverable copper was measured 14 times out of 520 samples. No spatial or temporal patterns were readily discernable in copper concentrations. Copper concentrations exceeded chronic and acute toxicity criteria in 1.0 to 1.2 percent and 1.5 to 2.1 percent of samples, respectively. Total recoverable lead was measured 402 times out of 608 samples. Variability in measurable lead concentrations was high, possibly due to the use of lead fishing weights to hold water samplers at a given depth. As a result, lead results are questionable. Measurable lead concentrations were highest and most often measured at upper reservoir locations. Measured lead concentrations were also highest from photic zone depths (5.8 m) compared to aphotic depths (12.4 m). Lead concentrations typically exceeded WQC. Acute and chronic exceedances in lead criteria were 0.5 and 56 percent, respectively. Total recoverable mercury was greater than reporting limit in only 1 of 544 samples. No spatial or temporal patterns were readily discernable in mercury concentrations. Acute and chronic exceedances of mercury criteria were 0 and 100 percent, respectively. However, most exceedances were based on concentrations below the reporting limit. Total recoverable zinc exceeded reporting limits in 92 of 608 samples. Zinc concentrations were 3.5 times greater at Porcupine Bay compared to main stem reservoir locations. Exceedances of acute and chronic criteria for zinc were 0.3 and 0.8 percent, respectively.

Conclusions. The authors reported that trends for TSS were difficult to assess and that only 144 of 608 total samples had concentrations greater than the minimum reporting limit of approximately 2 mg/L. Total recoverable concentrations of arsenic, cadmium, copper, and mercury were frequently below minimum reporting levels. The least detectable elements were nickel and selenium (1 percent detection frequencies) and antimony, beryllium, chromium, cobalt, and silver (with no detectable concentrations). Measurable percentages of contaminants ranged from 0.2 to 2.7 percent. High reporting limits, few detected concentrations, and variable concentrations among samples made identifying spatial, temporal, and environmental relationships difficult. Similarly, comparisons to Tribal water quality standards are also difficult to assess due to the high reporting limits. The authors also noted that water quality in Porcupine Bay (on the Spokane Arm of the study area) had measurably different water quality, with typically larger trace element concentrations (particularly for zinc).

Lake Roosevelt Fisheries Evaluation Program Annual Reports (LRFEP). - Multiple References (cited below)

The LRFEP monitors various water quality parameters as part of an ongoing program to monitor limnological and biological resources in the lentic portion of the UCR from Marcus Flats (RM 710) to the vicinity of the Grand Coulee Dam (RM 599). Data are available from 1998 to 2005 in annual reports (Shields et al. 2002 [1998 annual report], McLellan et al. 1999 [1999 annual report], Lee et al. 2003 [2000 annual report], Scofield et al. 2004 [2001 annual report], Fields et al. 2005 [2002 annual report],, Pavlik-Kunkel et al. 2005 [2003 annual report], Lee et al. 2006 [2004 annual report], and Scofield et al. 2007 [2005 annual report]).

From 1999 through 2001, ten to fifteen fixed stations from Marcus Flats to the Grand Coulee Dam were sampled for water quality data each month. After January 2002, sampling was scaled back to one sample in January and monthly samples between May and October, and sampling locations were reduced to five stations (Gifford, Seven Bays, Keller Ferry, Spring Canyon in the UCR, and Porcupine Bay in the Spokane Arm). Additional Hydrolab data were also collected from four locations below Little Falls Dam in the upper Spokane River.

In all years for which data have been reported, water quality parameters collected from Lake Roosevelt have included temperature, dissolved oxygen, conductivity, turbidity, pH, redox potential, and total dissolved gas (degree of saturation) recorded in pelagic and near shore zones using a multi-sensor probe (Hydrolab Surveyor 4). Observations with the Hydrolab data were collected from water at the surface to a depth of 33 m, at 3-m intervals. Additional measurements were taken from 40 m to 90 m at 10 m intervals at Keller Ferry and Spring Canyon to characterize the water quality of deeper portions of the lower reservoir. Secchi disk depths were also recorded at all stations. Discrete water samples were analyzed for nutrients including NO_3 as N, NO_2 as N, NH_3 as N, total Kjeldahl nitrogen (TKN), total nitrogen (TN) TKN+nitrite+nitrate, total phosphorous (TP), ortho-phosphorous, alkalinity, TSS and turbidity.

3.1.2 Sediment Chemistry

Summaries of sediment chemistry data in reports and publications are summarized by reporting authors or organization in chronological order. These summaries primarily represent studies conducted on samples collected downstream of the U.S.-Canada border. Studies were performed to meet a variety of objectives and may not be representative of all Site sediments. Further evaluation of select data from these studies is provided in Appendix D.

> *Hopkins, B.S., D.K. Clark, M. Schlender, and M. Stinson. 1985. Basic water monitoring program, fish tissue and sediment sampling for 1984. Publication No. 85-7. Washington State Department of Ecology, Olympia, WA.*

This document is a result of Ecology's Basic Water Monitoring Plan (BWMP) that was initiated in 1978 and provides site data from 1984. Sediment samples were collected from fish tissue sampling locations and are described further in the fish tissue study summaries (Section 3.1.5).

> *Johnson, A., B. Yake, and D. Norton. 1989. An assessment of metals contamination in Lake Roosevelt. Segment No. 26-00-04. Washington State Department of Ecology, Olympia, WA. 84 pp.*
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> *Johnson, A., D. Norton, B. Yake and S. Twiss. 1990. Transboundary Metal Pollution of the Columbia River (Franklin D. Roosevelt Lake). Bull.Environ. Contam. Toxicol. 45:703-710.*

This study was conducted by researchers from Ecology to provide an initial characterization of the longitudinal distribution of metals in surface sediments throughout the UCR, and it represents the first large-scale study of metals in surface sediments in this part of the Columbia River.

Methods. A series of sediment grabs were collected in Lake Roosevelt between August 4 and 8, 1986, in a longitudinal transect from the U.S.-Canada border to Grand Coulee Dam. Surface sediments (top 2 to 4 cm) were sampled at 12 stations in the UCR from RM 743 near the U.S.-Canada border to RM 605 near the Grand Coulee Dam. Surface sediments were also collected at the mouths of the four major tributaries to the UCR (i.e., the Kettle, Colville, Spokane, and Sanpoil rivers). The Ministry of Environment provided a sediment sample from Lower Arrow Lake (the Columbia River reservoir above Trail) which served as a reference for Lake Roosevelt sediments. Surface sediments at most stations were sampled using a van Veen grab sampler, but an Emery pipe dredge was required to sample coarse sediments near the border. The Ministry of Environment used an Ekman grab in Lower Arrow Lake. The five stations located at RM 724 and above were located in back eddies and embayments in order to obtain fine grained material rather than the typical coarse sediment found in the main channel. Below RM 724, stations were sampled near mid-channel to a maximum water depth of 135 ft. A single core sample was also collected at RM 693 near French Rocks using a 5-cm gravity corer in September, 1986. In addition to metals concentrations, concentrations of cesium-137 were measured in the core samples to assign a time horizon to each sample. QA/QC analyses are reported.

Results. A distinct longitudinal gradient of grain-size distribution in the UCR was identified, with stations located at and above Marcus Flats being characterized predominantly by coarse-grained sediment (i.e., sand) and downstream areas being characterized predominantly by finer grained sediment (i.e., silt and clay). For the six stations sampled above RM 700, sand accounted for more than 96 percent of the sediment at all but one station. For fine-grained sediments in downstream areas, silt exhibited a peak concentration of 80 percent at RM 692 near French Rocks, and then declined steadily to a value of 46 percent at RM 604 near the Grand Coulee Dam. By contrast, clay exhibited gradually increasing concentrations from 15 percent at RM 692 to 34 percent at RM 604. The authors found that four patterns were evident, inferring different transport mechanisms and/or source materials.

- "Lake Roosevelt sediments were elevated in concentrations of all metals except aluminum. Peak concentrations of zinc, copper, and mercury (26,840, 4,870, and 2.7 μg/g, dry, respectively) were two orders of magnitude higher in the lake than in its tributaries or in Lower Arrow Lake. Similarly, iron (326,200 μg/g), manganese (5,900 μg/g), lead (550 μg/g), arsenic (32.2 μg/g), and cadmium (5.7 μg/g) concentrations were elevated by one order of magnitude. "
- The highest concentrations of iron, manganese, zinc, copper, and arsenic occurred at the border. Concentrations decreased rapidly from the border downstream to Kettle Falls, below which they remained relatively constant. The high concentrations in the upper reaches of the lake were almost certainly due to slag particles originating from Cominco. Slag was discharged from Cominco in the form of a coarse-grained sand; an average of 460 metric tonnes per day was reported to have been discharged to the Columbia River in 1977 (Ministry of Environment 1979). A matrix of correlation coefficients calculated using the data for Lake Roosevelt sediments showed significant intercorrelations among these five metals and the sand content of the sediments.
- Concentration gradients for cadmium and mercury varied inversely with the above metals (iron, manganese, zinc, copper, and arsenic) in that concentrations increased with distance downstream of the border and reached maximum levels below Kettle Falls. There appeared to be evidence of a trend toward decreasing mercury concentrations from Kettle Falls to Grand Coulee Dam. While the distribution of these metals could be construed as indicating a local source, it is more likely the result of increasing silt and clay content of the sediments. Cadmium, mercury, silt, and clay were significantly inter-correlated.
- Lead concentrations in the sediments followed a pattern generally similar to mercury, but with a broader region of high concentrations extending from China Bend to Gifford. The reason for the observed distribution of lead in the sediments is not clear. Lead was not correlated with other metals, grain size, or organic carbon.

- Metals derived from weathering of rocks are generally transported in rivers bound to aluminosilicates. Aluminum concentrations should therefore explain most of the natural variation in concentrations of other metals. Aluminum concentrations in Lake Roosevelt sediments, however, were poorly correlated with other physical/chemical variables; concentration gradients were unlike those of other metals. Metals ratios in tributary and lower Arrow Lake sediments were also very high relative to Lake Roosevelt sediments. This suggests artificial enrichment of other metals in the lake.
- Cesium-137 "...gives two dates 1) 1963, the peak of nuclear weapons testing and 2) 1954, the first major testing. Cesium-137 activity peaked at 10 to 15 cm, and could not be detected below 20 cm. On this basis, the increments 10 to 15 cm and 15 to 20 cm were considered to contain material laid down in 1963 and 1954, respectively."

Conclusions. The authors stated "Sediments in the upper reaches of the lake (i.e., above Kettle Falls) were found to contain high concentrations of iron, manganese, zinc, copper, and arsenic relative to tributary sediments. This was attributed to the presence of slag which is discharged from Cominco in the form of coarse-grained sand. Elevated concentrations of lead, cadmium, and mercury occurred farther downstream in association with finer grained materials. Analysis of a single core of the lake bottom indicated the level of sediment contamination had not changed appreciably since the 1950s." The authors speculated that the high concentrations of metals in sediment at and above Marcus Flats were largely the result of the presence of granulated slag in the coarse-grained fraction of the sediments. They characterized these sediments as primarily consisting of "*brownish black sand*." The authors also analyzed the cesium activity of sediments collected in Lake Roosevelt and the Upper Columbia River. This analysis of benthic lake cores indicated that sediment contamination had not changed appreciably since the 1950s.

With respect to the major tributaries, the authors stated "Metals concentrations in sediment from Lake Roosevelt tributaries were within a factor of two of those in Lower Arrow Lake, except for the Spokane River Arm which was relatively high in all metals, especially manganese, zinc, lead, arsenic, and cadmium. However, compared to Lake Roosevelt sediments above the Spokane confluence (i.e., Castle Rock and Hunters stations), the Spokane confluence sediments were elevated only for zinc (1,540 μg/g vs. 954 and 610 μg/g)."

> *Johnson, A.D., D. Serdar, and D. Norton. 1991a. Spatial trends in TCDD/TCDF concentrations in sediment and bottom fish collected in Lake Roosevelt (Columbia River). Publication No. 91-29. Washington State Department of Ecology, Olympia, WA.*

Johnson, A. 1991b. Results of screen for EPA xenobiotics in sediment and bottom fish from Lake Roosevelt (Columbia River). Publication No. 91-e24. Washington State Department of Ecology. Olympia, WA.

As part of Ecology's investigation of contaminants in Lake Roosevelt, a series of sediment and bottom fish samples were collected in June 1990 from Lake Roosevelt and vicinity for analysis of PCDDs and PCDFs. Sediment and bottom fish samples were also analyzed for 44 additional organic compounds. The impetus for this survey was the need to better understand the spatial distribution of these compounds as a result of their discharge by the Celgar bleached kraft pulp mill in Castlegar, B.C. (approximately 30 river miles above the U.S.-Canada border). Johnson et al. (1991a) provides original data for 13 PCDF's and 5 PCDDs while Johnson (1991b) gives the results of 44 additional organic analytes.

Methods. Six sites were sampled in the UCR between the U.S.-Canada border and Grand Coulee Dam, and one site each in the Spokane River behind Long Lake Dam and Rufus Woods Lake (the Columbia River reservoir below Lake Roosevelt formed by Chief Joseph Dam). One sediment sample was collected at each sample location. Each sediment sample represented a composite of three grabs taken with a stainless-steel van Veen grab sampler. Only the top 2-cm surface layer was sampled. The 2,3,7,8-substituted PCDDs and PCDFs, except octachlorodibenzo-*p*-dioxin and octachlorodibenzofuran, were analyzed at the ppt (pg/g) level. The remaining 44 bioaccumulative organic compounds were analyzed at the ppb (ng/g) level. QA/QC data are reported.

Results. Detection of target PCDDs and PCDFs in the sediments was limited almost exclusively to 2,3,7,8-TCDF, and in one instance, 2,3,7,8-TCDD. A low concentration of 2,3,7,8-TCDD was detected in the sediments at one site only, off French Point Rocks (RM 692), just below Kettle Falls (RM 700). Concentrations of 2,3,7,8-TCDF in the UCR occurred at much higher concentrations than at Rufus Woods Lake or the Spokane River at Long Lake, and they decreased downstream from Kettle Falls. TCDF was not detected in sediments upstream of Kettle Falls (i.e., Marcus Island, China Bend, and Northport). The authors stated "One possible explanation for the lower level of sediment contamination in the upper Lake Roosevelt is the velocity of the Columbia River which results in the upper reaches being an erosional rather than depositional environment. This is reflected in the coarse grain size and low TOC content of the sediments - samples obtained from Northport, China Bend, and Marcus Island." Sediments collected above Kettle Falls were of coarse sand with low TOC content whereas much finer grained material predominated downstream of that area.

For the remaining 44 organic compounds, detection was almost exclusively limited to PCBs and dichloro-diphenyl-dichloroethene (DDE) (a metabolite of DDT). PCBs and DDE were below detection limits (ranging from 2.5 ng/g for DDE and from 1.25 to 6.25

ng/g dry weight for PCB congeners) in the three most upstream samples collected from Northport to Marcus Flats. Detected concentrations of PCBs were found at French Point (3.7 ng/g) and corresponded with increased TOC (2.4 percent). Higher concentrations of PCBs were found at Grand Coulee (25.2 ng/g) and the Spokane River at Long Lake (20.8 ng/g). DDE was below detection limits in all sediments, except for trace amounts (2.1 ng/g) near Grand Coulee Dam.

Conclusions. The authors concluded that their survey demonstrated long distance transport of 2,3,7,8-TCDD and 2,3,7,8-TCDF through the UCR. Sediment samples from the riverine portion of the UCR (near the US-Canadian border) did not contain detectable concentrations of 2,3,7,8-TCDD or 2,3,7,8-TCDF. Significant deposition first occurs near Kettle Falls where 2,3,7,8-TCDD and 2,3,7,8-TCDF were detected at the highest concentrations in sediment samples. Dioxin (2,3,7,8-TCDD) was not detected in any other sediment sample. There was attenuation of 2,3,7,8-TCDF as a function of distance downstream of Kettle Falls. The highest levels of PCBs were found in the Long Lake section of the Spokane River and in the UCR downstream of the Spokane River near Grand Coulee Dam. Because of the environmental persistence of PCBs, the authors recommended a more thorough evaluation of these compounds in the Spokane drainage. However, the authors found that the Spokane River was not a significant source of PCDD/PCDF to the lake.

> *Norecol Environmental Consultants Ltd. 1993. A 1992 biological reconnaissance and sediment sampling in the Columbia River between the Hugh Keenleyside Dam and the International Boundary. Prepared for CRIEMP, Castlegar, B.C., Norecol Environmental Consultants Ltd. 302 pp.*

This study was conducted for the Columbia River Integrated Environmental Monitoring Program (CRIEMP) to evaluate the environmental quality of the Columbia River between the Hugh Keenleyside Dam and the U.S.-Canada border.

Methods. This study evaluated the structure of aquatic communities (i.e., periphyton, macrophytes, and benthic macroinvertebrates), sediment chemistry, toxicity, and bioaccumulation by macrophytes and benthic macroinvertebrates. Sampling was conducted in September 1 through 2, 1992. The study specifically sampled upstream and downstream of the Celgar and Cominco discharges and tested for both organic pollutants and metals. Sediment samples were collected from "three additional sites where sample splits for the U.S. Geological Survey." QA/QC data are also reported.

Results and Conclusions. Metal concentrations in sediment and biota (particularly arsenic, cadmium, copper, lead, mercury, and zinc) were elevated at stations located downstream from Trail. The authors stated that "the highest mercury concentration occurred at Waneta (IV-3A), where the sediment had the highest proportion of silt-clay particles (see Table 5-2). Mercury tends to be associated with sediment fines." Sediment lead, zinc and copper were one to two orders of magnitude higher immediately downstream from Trail as compared to all other sites. The authors concluded that the elevated metal concentrations in sediment at those locations were causally-related to elevated metal concentrations in the tissues of selected macrophytes and benthic macroinvertebrates (i.e., mussels and caddisflies). Concentration of metals in sediment samples collected downstream of Cominco (Reach IV) were higher than those detected in sediment samples collected in upstream reaches or at control sites. These elevated concentrations of metals apparently persist downstream into U.S. reaches of the Columbia River. Similarly, concentrations of selected organic compounds (i.e., chlorinated phenolic compounds, dioxins, and furans) were found in sediments and biota tissues below the pulp mill near Castlegar. However, some of these compounds were also found in tissues collected upstream of the pulp mill. "The highest 2,3,7,8 TCDF encountered anywhere but the Celgar site was 61 pg/g at Waneta."

The authors recommended that environmental monitoring be continued in this portion of the Columbia River.

> *Bortleson, G.C., S.E. Cox, M.D. Munn, R.J. Schumaker, and E.K. Block. 1994. Sediment-quality assessment of Franklin D. Roosevelt Lake and the upstream reach of the Columbia River, Washington, 1992. U.S. Geological Survey, Open-File Report 94-315, 130pp. (Reprint 2001 as U.S. Geological Survey Water Supply Paper 2496, 130 pp.)*

This study was conducted by researchers from the USGS, in cooperation with the EPA, to provide a comprehensive characterization of the distribution of metals and organic compounds in surface sediments throughout the UCR, and represents a more detailed characterization of surface sediments than Johnson et al. (1989 and 1991a) in the Columbia River upstream of Lake Roosevelt to the U.S.-Canada border. Bed sediments, suspended sediments, and water were sampled as part of this study.

Methods. Sampling was conducted in September and October of 1992. Surface sediments (top 1 to 2 cm) were sampled at 41 stations in the UCR from the U.S.-Canada border to the Grand Coulee Dam. Surface sediments were also collected in the mouths of the four major tributaries to the UCR (i.e., the Kettle, Colville, Spokane, and Sanpoil rivers), in the mouths of three minor tributaries (Sherman, Hall, and Hawk creeks at RMs 700, 675, and 634 respectively), at four beaches (Big Sheep Creek, Colville River, Bradbury, and Keller Ferry at RMs 736, 699, 694, and 615, respectively), and in the river bank at three locations (Ninemile Creek, Seven Bays, and Sanpoil River, at RMs 648, 636, and 616, respectively). Above the U.S.-Canadian border, the authors sampled surface sediments in the Kootenay and Pend Oreille rivers (which are tributaries to the UCR) and Lower Arrow Lake, B.C. Surface sediments were sampled using a stainless steel van Veen grab sampler. At most locations in the UCR, stations were sampled along a transect perpendicular to the river that included a station in the historical river channel as well as stations on either bank. Bed sediment sieved to less than 2 mm (coarse sand and smaller) was analyzed for trace elements, including but not limited to key elements of interest (arsenic, cadmium, copper, lead, mercury, and zinc). Trace-organic compounds selected for analysis were those which had not been analyzed in Lake Roosevelt bed sediment, including the determination of many methylene-chloride extractable compounds that are usually associated with urban and industrial activities. Bed sediments from three sites were also analyzed for trace-organic compounds related to pulp-mill discharge.

Results and Conclusions (inorganics). The longitudinal patterns of metals in UCR sediments were generally similar to those identified previously by Johnson et al. (1989), with the highest concentrations of some metals (e.g., copper and zinc) associated with the sandy sediments found at and above Marcus Flats. Median concentrations of copper and zinc were 2,600 mg/kg and 13,000 mg/kg, respectively, in the Northport reach. Whereas median concentrations for copper and zinc were 300 mg/kg and 2,000 mg/kg in the upper reaches of Lake Roosevelt. Most other trace elements (antimony, arsenic, barium, chromium, iron, and manganese) followed a similar pattern with higher concentrations near the U.S.-Canada border and lower concentrations downstream. The authors found that further down the reservoir, beyond French Point Rocks, the concentrations of these eight elements were fairly uniform. The authors stated "The elevated concentrations of copper and zinc are explained by the presence of slag particles that contained concentrations of these trace elements as large as 6,000 mg/kg and 25,000 mg/kg, respectively.

Sandy bed sediment from three samples in the Northport reach were analyzed for slag particles by microscopy. The authors found that the slag particles, consisting of dark glassy needles and angular grains, were progressively finer at sites downstream. The slag particles were easily recognized as chemically and morphologically distinct from natural sediment. Major elements present in slag particles determined from X-ray spectrum were calcium, iron, and silicon with minor and varying amounts that included aluminum, chromium, copper, manganese, potassium, sodium, sulfur, titanium, and zinc. In addition to the common calcium-iron silicate inclusions (lesser minerals) within the glass matrix, other inclusions included copper-zinc-iron sulfides, iron-titanium oxides, iron oxides, and aluminum-manganese-iron-zinc oxide phase. The authors concluded that concentrations of copper and zinc are therefore due in part to inclusions found in the slag.

Total cadmium, lead, and mercury in sediments followed a different pattern or spatial variability, being associated primarily with fine-grained sediments in areas farther downstream. The authors speculated that the elevated concentrations of cadmium, lead, and mercury found in downstream areas may be associated with liquid effluent discharged to the river above the U.S.-Canada boundary. Trace element concentrations of copper, lead, and zinc exceeded Canadian sediment guidelines for severe effects at 18 of 41 sites clustered near Northport and the upper reaches of Lake Roosevelt.

With respect to the river banks, the authors stated "The banks along Lake Roosevelt are also a source of sediment to the reservoir because of the active sliding and slumping of sediments, but even though bank sediments may be a natural source of arsenic, they are probably not a significant source for enriched concentrations or other trace elements."

With respect to the beaches, the authors concluded that trace-element concentrations from three public beaches were small compared to those in reservoir bed sediment.

With respect to the various tributaries, the authors stated "The major and minor tributaries to the reservoir are also sources of riverine sediment containing trace elements." The authors further state "Total concentrations of arsenic, cadmium, copper, lead, mercury, and zinc in bed sediment of the Colville, Kettle, and Sanpoil Rivers were considerably smaller than those of Lake Roosevelt sediment;" and, "Elevated zinc concentrations in the Spokane River are a potential source of zinc to the lower reach of the reservoir."

The authors concluded "Overall, the evidence indicates that sediment carried by the Columbia River is the primary source of the widespread trace-element-enriched bed sediment of Lake Roosevelt and the Northport reach of the Columbia River."

Results and Conclusions (organics). For organic compounds, the UCR and its major tributaries generally did not contain detectable concentrations of methylene chloride-extractable compounds. In fact, 52 of the methylene chloride-extractable compounds analyzed were not detected or were below the minimum reporting level. However, 15 compounds were detected, but at concentrations below minimum reporting levels. Of these, 11 were PAHs and two were phenolic compounds. PAHs were detected less frequently in the lower part of the UCR than in the upper part. The authors suggested that the widespread presence of PAHs may have been due to atmospheric deposition from incomplete combustion of fossil fuels (Sanders et al. 1993).

Although chlorinated phenols and wood pulp-related compounds of veratroles, anisoles, and vanillins were not detected in sediment at Marcus Flats and French Rocks, a few chlorinated guaiacols and catechols were found during the study at those locations. In addition, 8 of 15 unchlorinated resin and fatty acids (i.e., components of pulp mill effluent), were found at both Marcus Flats and French Rocks. Extractable organic halides, a gross indicator of chlorinated organic compounds, were not found above their minimum reporting levels of 10 mg/kg dry weight (dw) at either sample site.

The authors concluded that sediments collected from the UCR and its tributaries generally were not contaminated with methylene chloride-extractable compounds or common wood pulp-related compounds. However, a few of the non-dioxin and non-furan compounds were found at small concentrations as far downstream from the U.S.-Canada boundary as French Rocks (RM 692). The authors noted that the concentrations of selected PAHs in UCR sediment were several orders of magnitude less than sediment quality values developed by EPA for freshwater ecosystems. They also concluded that even though the extractable organic halides analytical method does not measure individual chlorinated organic compounds, the data suggested that total concentrations of chlorinated compounds in UCR sediments were low.

> *Tielens, J.T. 1994. Bioavailability of trace metals in Franklin D. Roosevelt Lake, Washington, sediments. Environmental Science and Regional Planning, Washington State University. Masters Thesis. December 1994.*

This is a Washington State University graduate study evaluating metal concentrations in surface sediments, porewater, and benthic macroinvertebrates in the UCR. Additional information is provided in the benthic invertebrate study summaries (Section 3.1.4).

> *MESL (MacDonald Environmental Sciences Ltd.). 1997. Lower Columbia River from Birchbank to the international border: Water quality assessment and recommended objectives. Technical report. Available at: http://www.env.gov.bc.ca/wat/wq/objectives/birchbank/Birchbanktech.pdf. Prepared for Environment Canada and British Columbia Ministry of Environment, Lands and Parks.*

The purpose of this report was to develop sediment quality objectives (SQOs) for the Columbia River from Birchbank to the U.S.-Canada border, to support assessment of sediment quality conditions.

Methods. Provisional SQOs were developed for the Columbia River between Birchbank and the U.S.-Canada border using a three step process. First, priority variables were identified using information on existing and potential contaminant sources in the study area. Second, preliminary SQOs for those priority variables were selected from the Canadian sediment quality guidelines (SQGs) of the Canadian Council of Ministers of the Environment (CCME) when available. When CCME SQGs were not available, the preliminary SQOs were selected from the British Columbia approved and working sediment quality criteria. Third, background concentrations of priority variables were calculated as the average of measurements from Birchbank and vicinity. The higher of the two values, either the average background concentration or the preliminary SQO, was selected as the provisional SQO for each variable in the Columbia River.

In addition to developing SQOs, data from a number of surveys on sediment chemistry and toxicity conducted in the Columbia River between Birchbank and the U.S.-Canada border between 1976 and 1997 were compiled and evaluated to assess the status and spatial trends of ambient sediment quality conditions. Data collected by BCMOE in 1976 measured arsenic, cadmium, lead, copper, mercury and zinc in bed sediments upstream and downstream of the Cominco lead-zinc smelter. Under the CRIEMP, EC collected data in 1992 on concentrations of barium, cadmium, chromium, copper, iron, lead, manganese, mercury, molybdenum, and zinc at four sites– Birchbank, Ryan Creek, Beaver Creek, and Waneta. EC also collected data on the concentrations of arsenic, cadmium, copper, lead, mercury, and zinc in suspended and bed sediments from one sampling station near Waneta in 1990/91. Fisheries and Oceans Canada (Nener 1992) evaluated the composition and toxicity of slag from Cominco's lead-zinc smelter in 1992. Organic contaminants (i.e., abietic acid, dehydroabietic acid, isopimaric acid, palustric acid, arachidic acid, and behenic acid) were evaluated at Birchbank, Waneta and Ryan Creek sties under the CRIEMP in 1993. Organic contaminant concentrations were also evaluated by Hatfield Consultants Ltd. (1997) immediately downstream of the pulp mill, and at Genelle. Godin and Hagen (1992) conducted a suite of aqueous-phase and solid-phase toxicity tests in 1992 on Cominco's slag from three sampling locations, Genelle Island, Beaver Creek, and the back eddy pool at Genelle, using rainbow trout (*Oncorhynchus mykiss*), amphipods (*Hyalella azteca*), midges (*Chironomus dilutus*), cladocerans (*Daphnia magna*), and algae (*Selenastrum capricornutum*). Solid-phase toxicity tests with *Daphnia magna* and *Hyalella azteca* were also conducted with sediment samples collected in the vicinity of Beaver Creek, Birchbank, Ryan Creek vicinity, and Waneta under the CRIEMP in 1992.

Results. Provisional SQOs were developed for seven metals– arsenic (5.87 mg/kg dw), cadmium (0.6 mg/kg dw), chromium (36.4 mg/kg dw), copper (35.1 mg/kg dw), lead (33.4 mg/kg dw), total mercury (0.16 mg/kg dw), and zinc (120 mg/kg dw). All provisional SQOs for metals were selected directly from the CCME threshold effect level (TEL), as none of the background concentrations for any of these seven metals was greater than the CCME TEL. Insufficient data existed at the time of the report to develop an objective for thallium. The CCME interim SQG for protection of aquatic life of 0.25 ng TEQ/kg DW was selected as the provisional guideline for total PCDDs and total PCDFs. Insufficient data were available at the time of the report to establish provisional SQOs for resin acids or fatty acids, both suspected contaminants originating from pulp mill sources. It was recommended that the SQOs be compared to the mean concentration of at least three replicate surficial (i.e., top 5 cm) sediment samples.

Not enough data existed to evaluate temporal variability in sediment quality conditions in the Columbia River from Birchbank to the U.S.-Canada border. However, there were clear spatial patterns of sediment quality conditions in the study area. Elevated concentrations of arsenic, cadmium, copper, lead, mercury, and zinc were typically observed in bedded sediments down-stream of the lead-zinc smelter, as compared to upstream areas. Similar trends in metal concentrations were also observed for suspended sediments. Barium, chromium, iron, manganese, molybdenum also exhibited similar upstream/downstream trends in bedded sediments, although the magnitude of difference between upstream and downstream was smaller for these metals. Metal concentrations were also higher in sediments on the west side of the river, compared to the east side, although this trend diminished further downstream from the smelter. Total concentrations of metals also generally decreased with distance downstream from the Trail facility. Metal concentrations in sediments generally exceeded the provisional SQOs downstream of the lead-zinc smelter, while they were generally lower than provisional objectives upstream of the smelter. Elevated concentrations of cadmium, chromium, copper, and zinc were attributed to slag disposal into the river system. The highest organic contaminant concentrations were observed in 1992, downstream of the pulp mill.

Conclusions. The authors identified cadmium, copper, lead, mercury, and zinc as chemicals of greatest concern in sediments. However, it was recommended that future sediment quality monitoring include metals and organic contaminants. They also indicated that future sediment quality monitoring should identify additional depositional areas for sediment sampling sites.

> *Era, B. and D. Serdar. 2001. Reassessment of toxicity of Lake Roosevelt sediments. Publication No. 01-03-043. Environmental Assessment Program, Washington State Department of Ecology, Olympia, WA. 54 pp.*

This study was conducted by researchers from Ecology. Prior studies conducted by Ecology and the USGS have included bioassay tests on Lake Roosevelt and upper Columbia River sediments (Johnson 1991a; Bortleson et al. 1994). Those studies reported significant toxicity at seven main stem Columbia River locations and near the mouths of the Kettle and Sanpoil rivers, which are tributaries to the Columbia. Based upon the toxicity results, these sites were included on the federal Clean Water Act Section 303(d) list of impaired waterbodies for 1998. Due to questions about the relevance of these decade-old data, the Water Quality Program at Ecology's Eastern Regional Office requested a reassessment of sediment toxicity in Lake Roosevelt and the upstream reach of the Columbia River. Ecology's Environmental Assessment Program conducted the reassessment by analyzing sediment metals and toxicity at the nine sites previously found to have sediment toxicity, and at a reference site. Objectives of the survey were to obtain current information on sediment metals chemistry and toxicity, and to determine if the sites should remain on the 303(d) list (Era and Serdar 2001).

Methods. Sampling was conducted in May of 2001. Surface sediments (top 10 cm) were sampled at seven stations in the UCR, with three stations located in the upper portion of the river (RM 745 to 738) near the U.S.-Canada border (i.e., Boundary, Auxiliary Gage, and Goodeve Creek), and at four stations located in the lower part of the river (RM 645 to 596; i.e., Castle Rock, Whitestone Creek, Swawilla Basin, and Grand Coulee). Surface sediments were also collected in the mouths of the Kettle and Sanpoil rivers, as well as at an upstream reference area in Lower Arrow Lake, B.C. Main stem Columbia River sediments were collected using a van Veen grab sampler. The Kettle and Sanpoil River samples were collected by wading into the water and scooping sediments up with a stainless steel spoon. With the exception of the Kettle River and Sanpoil River sites, each sediment sample was a composite of three grabs. Compounds analyzed for included arsenic, cadmium, copper, lead, mercury and zinc as well as 3 toxicity tests of bulk sediment and porewater.

Results. In general, the longitudinal patterns of metal concentrations found in the UCR were similar to those identified previously by Johnson et al. (1989) and Bortleson et al. (1994), with the highest concentrations of zinc and copper associated with the coarse-grained sediments found in the upper UCR. Another pattern observed was elevated mercury concentrations in the lower part of Lake Roosevelt. Lower Lake Roosevelt has historically contained high percentages of clay sediments that have correlated with elevated levels of mercury. In the study, mercury also correlated with the clay sediments of lower Lake Roosevelt. Lead was relatively high throughout the study area.

Conclusions. The authors found no strong longitudinal patterns for lead and cadmium. The authors noted that much of the coarse-grained sediments at the three stations located in the upper parts of the UCR consisted of a visibly dark sandy mixture indicating the presence of slag from the Cominco smelter. Elevated levels of copper and zinc were found at the upper Columbia River sites but low in Lake Roosevelt, indicating that slag material may still be present in the upper Columbia River. Elevated concentrations of metals were not found in the Kettle and Sanpoil rivers.

> *Fischnaller S., P. Anderson, and D. Norton. 2003. Mercury in Edible Fish Tissue and Sediment from Selected Lakes and Rivers of Washington State. Washington State Department of Ecology. Publication No. 03-03-026.*

During 2001 and 2002, Ecology conducted a screening survey for mercury concentrations in fish tissue and sediments from selected lakes and rivers across Washington State. These data are discussed further in the fish tissue study summaries (Section 3.1.5).

> *Majewski, M.S., S.C. Kahle, J.C. Ebbert, and E.G. Josberger. 2003. Concentrations and distributions of slag-related trace elements and mercury in fine-grained beach and bed sediments of Lake Roosevelt, Washington, April-May 2001. Water Resources Investigations Report 03-4170. U.S. Geological Survey, Reston, VA.*

This study was conducted by researchers from USGS to determine the concentrations and distribution of trace elements in the fine-grained fraction of exposed beach, bed, and bank sediments (i.e., <63 μm) along the Columbia River from the U.S.-Canada border to the Grand Coulee Dam. Previous studies conducted in Lake Roosevelt focused on contamination levels in fish, water, and bed sediment. There was, however, a growing concern over the potential threat of airborne contaminants to human health. In response to these concerns, the USGS conducted an assessment of trace-element concentrations in the relatively shallow fine-grained sediment along the shore of Lake Roosevelt that is exposed annually during periods of reservoir drawdown. The principal contaminants of interest in this study were trace elements associated with slag discharge (arsenic, cadmium, copper, lead, and zinc) and smelting effluent discharge (mercury).

Methods. Sampling was conducted in April and May of 2001 during the spring drawdown of the lake. Surface sediments (top 2 to 3 cm) were collected using two different, but complementary, sampling designs—targeted and spatially distributed (SD). Samples of exposed fine-grained beach and bed sediments were collected at 24 SD and targeted sampling areas (18 SD depositional areas and 6 targeted areas) in the UCR from RM 731 near Northport, Washington to RM 601 near the Grand Coulee Dam. Sediments were also collected at an upstream reference area in Lower Arrow Lake, the same upstream reference area sampled by Johnson et al. (1989), Bortleson et al. (1994), and Era and Serdar (2001). In addition, a riverine slag sample was taken at about RM 747 near Waneta, British Columbia, about 8 miles downstream from Trail, British Columbia. Surface sediments were sampled using plastic spoons. At the targeted sites, three composite samples were collected from the five transects to determine if there was a gradient in concentrations from the shoreline to the 1,290-ft contour. This was done to assess the spatial variation in concentrations of slag associated trace elements at the sampling areas. The three samples were composed of a composite of the five sampling sites nearest the water's edge; a composite of the five sampling sites midway between the water and the 1,290-ft contour; and, a composite of the five sampling sites nearest the 1,290-ft contour. At the SD sampling areas, samples from each of the 15 sites were composited into one sample. Reference samples collected at the upper bluff area of French Rocks boat ramp, the upper bluff at road cut on Hwy 25 about 6 miles north of Cedonia, and the upper bluff area across from Seven Bays were subsurface composites. That is, an exposed bank above the high water line was selected, and at least the top 15 cm of material was removed before the sample was taken. Reference samples collected along Lower Arrow Lake upstream from Trail, B.C. were surface composites and included the upper 2 to 3 cm of beach sediment. The riverine-slag sample was collected using the same procedures as used at the targeted and SD sites. The analytical methods included freeze-drying the samples, then sieving to obtain a size fraction of < 63 um. Each sample was then digested using a four-acid digestion process and analyzed using ICP-mass spectrometry (MS) or Flame-AAS for trace elements. Cold Vapor-AAS was used to analyze for mercury. Although air-quality sampling commonly analyzes the size fraction less than 10 um Particulate Matter, 10 microns (or less) (PM_{10}), the size fraction less than 63 um was used as a screening method for this investigation.

Results. The authors found that the mean concentrations of the slag-related trace elements at the targeted areas (the averaged concentration of the three composite samples) exhibited a gradient of decreasing concentrations downstream, from the U.S.-Canada border to the Grand Coulee Dam. The trace-element concentrations at the SD areas varied, but the slag-related trace elements showed generally pronounced decreases in concentration from near the U.S.-Canada Border to Grand Coulee Dam. Concentrations of all trace elements analyzed for varied greatly between RM 699 (south side of confluence of Colville River and Lake Roosevelt) and RM 675 (west bank of Lake Roosevelt at Inchelium). The large variations in this area were attributed to the addition of natural sediments from streams and creeks in the area draining into the lake, as well as from erosive slumping of bank deposits. The subsequent downstream areas showed a gradual decrease in concentration following the general gradient trend. Trace-element concentrations showed a gradient from water to shore at some of the targeted areas. Of the slag-related trace elements, concentrations of cadmium, copper, lead, and zinc decreased from the water line to the bank at Swawilla Cove, across from Seven Bays, and at Rogers Bar, at the lower part of the reservoir. At the Barnaby Creek area, the gradients for concentrations for lead and mercury were reversed. Areas at Marcus Flats and Evans, at the upper reach of the lake, showed no gradient trends for any analyzed trace elements.

Although this pattern was inconsistent with the patterns identified for mercury and cadmium in one or more of the previous studies described above, the authors speculated that it may have been related to the near-shore targeted sediment samples, whereas the previous studies evaluated sediments at greater water depths.

Conclusions. The authors stated "This study concluded that trace elements associated with slag and metallurgical waste are present in fine-grained fraction (less than 63 μm) of bed sediments along the length of Lake Roosevelt, and as such, could be components of the airborne dust resulting from exposure, drying, and wind mobilization of the sediments exposed during the annual drawdowns of the reservoir. Trace-element concentrations in the surficial bed varied, but the major components in slag-arsenic, cadmium, copper, lead, and zinc-showed generally pronounced gradients of decreasing concentrations from near the U.S.-Canada Border to the Grand Coulee Dam. The results of this study provide base-line information needed to plan and conduct air monitoring of trace elements in wind-blown dust along Lake Roosevelt." Cadmium, lead, zinc, and mercury exceeded the Canada PEL guidelines at most of the sampling areas upstream of RM 675 (west bank of Lake Roosevelt at Inchelium). Lead and zinc exceeded the PEL values at approximately 50 percent of the downstream sites, as well.

> *USEPA. 2002a. Upper Columbia River/Lake Roosevelt expanded site inspection – Northeast Washington, Sediment Investigation Statistical Analysis. Prepared for U.S. Environmental Protection Agency, Region 10, Seattle, WA, by the Superfund Technical Assessment and Response Team.*
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> *USEPA. 2003a. Upper Columbia River expanded site inspection report, Northeast Washington (Region 10, START-2). Prepared for U.S. Environmental Protection Agency, Region 10, Seattle, WA, by the Superfund Technical Assessment and Response Team. 84pp.*

This study was conducted by EPA to provide information for determining whether the UCR site should be included on the National Priorities List, and to establish priorities for additional action, if warranted. Concentrations of metals and organic compounds were evaluated in surface sediments.

Methods. Sampling was conducted in May and June of 2001. The samples included 53 surface sediment samples from the Columbia River/Lake Roosevelt, including 51 from depth intervals ranging from 0 to 8 in,, and two from depth intervals from 18 to 24 in.; 115 surface sediment samples from tributaries of the Columbia River/Lake Roosevelt; 6 soil samples from the Columbia River watershed; and 5 surface sediment samples from tributaries of the Pend Oreille River. Surface sediments (top 2 to 20 cm) were sampled at stations in the upper and middle portions of the UCR from the U.S.-Canada border to RM 675 near Inchelium. Surface sediments were sampled using several kinds of stainless steel equipment, including a petite Ponar grab sampler, hand augers, and spoons. All sediment and soil samples were analyzed for TAL metals. Selected samples were analyzed for pesticide/PCB (Pest/PCB), TOC, or SVOC analysis. The data were compared to a background, reference sample collected at Lower Arrow Lake in Canada. The background sediment sample used by EPA was collected by Ecology in May 2001. The Canadian Department of Foreign Affairs and International Trade refused the request from the U.S. Government for EPA to collect a sample from Lower Arrow Lake. For this reason, existing data for the sediment sample collected by Ecology was used as the background sample for the ESI.

Results. For metals, an evaluation of the analytical data relative to background sediment showed elevated concentrations of arsenic, cadmium, lead, mercury, and zinc in the majority of samples, ranging from 48 percent of all samples for mercury to 97 percent of all samples for copper. The area of elevated concentrations included the entire study area. The authors also found that concentrations of zinc, copper, and lead tended to decrease with increasing distance downstream. The highest concentrations of copper and zinc were found near the U.S.-Canada border with elevated concentrations approximately two orders of magnitude over other elevated concentrations located further downstream. Concentrations of cadmium and mercury tended to be highest toward the downstream portion of the study area. Sediments at a number of sites consisted of a dark glassy sandy mixture visually characterized by field personnel as slag.

For organic compounds, only three organic compounds were detected in the 49 sediment samples (i.e., 6 percent) collected between RM 675 and the U.S.-Canada border. These are methoxychlor, Aroclor 1254, and Aroclor 1260. Methoxychlor was measured at 52 μg/kg dw at a near shore site by Marcus Flats, but was below its detection limit at all other UCR sites. None of the other chlorinated pesticides were detected at any UCR sampling station. Aroclor 1254 and Aroclor 1260 were detected at low concentrations (38 and 17 μg/kg dw, respectively) in a single sample collected at RM 688, but were undetected at all other UCR sites.

Conclusions. The report states Analytical data show that widespread contamination is present in lake and river sediments throughout the upper Columbia River between Inchelium, Washington, and the U.S.-Canada border. Potential sources of contamination to the upper Columbia River include industries such as mining, milling, smelting, pulp, and others that have discharged hazardous substances into the river. Results of the EPA site investigation indicate that the Cominco smelter in Trail, B.C., is a primary source of contamination to the upper Columbia River. The report also states "Further detailed investigation of the upper Columbia River under CERCLA is recommended, including consideration of the site for proposal to the NPL, based on an evaluation of hazardous substances found in sediment samples collected from the upper Columbia River and based on a review of prior studies conducted documenting elevated levels of metals, dioxins, and furans in sediment samples at numerous locations from the U.S.-Canada border to Lake Roosevelt."

> *USGS. 2004. Geochemistry of sediments in the U.S. from the NURE-HSSR database. http://tin.er.usgs.gov/nure/sediment/. U.S. Geological Survey. Accessed September 2007.*

The National Uranium Resource Evaluation Hydrogeochemical and Stream Sediment Reconnaissance (NURE-HSSR) program, a nationwide survey of the elemental composition of soils and sediments, was conducted to assess the location of potential deposits of uranium and other strategic minerals in the U.S. Sampling and analysis was conducted by four national laboratories during the 1970s and 1980s. In the Pacific Northwest, sampling and analysis was conducted primarily by the Savannah River Laboratory, with some samples collected and processed by the Lawrence Livermore Laboratory. NURE-HSSR data were obtained from USGS (2004).

During the NURE-HSSR program, samples were collected near the UCR study area. The location of each sample was described, and those descriptions included an indication of whether or not each sampling location was potentially influenced by municipal, agricultural, industrial, mining, or other (unspecified) anthropogenic activity. Documentation of the field and laboratory methods followed during the NURE-HSSR program was reviewed to assess the usability of these data. Overall, the QA measures implemented during the NURE-HSSR field and laboratory programs followed the standards of good laboratory practice.

> *Cox, S.E., P.R. Bell, J.S. Lowther, and P.C. Van Metre. 2005. Vertical distribution of trace element concentrations and occurrence of metallurgical slag particles in accumulated bed sediments of Lake Roosevelt, Washington, September 2002. Scientific Investigations Report 2004 5090. U.S. Geological Survey, Reston, VA. 70 pp.*

This study was conducted by researchers from USGS to evaluate the vertical distributions of trace elements within the accumulated bed sediments and porewater of

Lake Roosevelt; to evaluate sediment accumulation rates; and, to evaluate sediment from selected core intervals for the occurrence of metallurgical slag.

Methods. Sampling was conducted in September 2002. Sediments were sampled at five stations from RM 705 to 624 and at one station in the Spokane River Arm of Lake Roosevelt. According to the authors, sites of continuously accumulating sediments were not found upstream from RM 705, so no cores were collected in the uppermost portion of the UCR. The five cores in the downstream section of the UCR were collected near the original river channel where the accumulation of sediment was thought to be thickest and least likely to be disturbed by fluctuations in water level and river flow. The core at RM 705 (the most upstream station) was located away from the historical river channel toward the left bank on a submerged terrace, because fine-grained sediments were not found in the channel. The core in the Spokane River was collected 8 miles upstream of the confluence of the Spokane River Arm and the confluence of the pre-reservoir Columbia River channel. . All cores were located in areas thought to be minimally affected by recent large landslides along the shoreline, which could potentially confound the vertical patterns of metals concentrations. Each core was collected using a 6.5 cm diameter gravity corer. Core depths ranged from 38 to 164 cm and sectioning occurred in intervals of 2 to 5 cm, depending on the core. In addition to metals, concentrations of cesium-137 were measured in the core samples to assign a time horizon to each core. Time horizons were assigned to each core based on the stratigraphic position of significant changes in the concentration profile of cesium-137. Additional time-stratigraphic markers used in assignment of time horizons within the sediment core included 1) the upper sediment surface, assigned the date of sample collection and 2) for cores that penetrated pre-reservoir sediments, the lower surface of the accumulated lake bed sediments was assigned a time coincident with the initial filling of the reservoir. Cesium-137 concentrations were measured by gamma counting and reported as pCi/g. Samples from selected intervals of the cores used to determine trace-element profiles were examined for slag-like particles. Intervals were determined based on elevated concentrations of trace elements identified in the analysis of slag samples, which included zinc, copper, antimony, and silver.

Results. The analysis of cesium-137 data indicated that two sediment cores contained sedimentary records extending to the initial reservoir filling in 1941 and the remaining four cores contained at least 60 percent of the sedimentary record. Elevated concentrations of arsenic, cadmium, copper, lead, mercury, and zinc occurred throughout much of the accumulated sediments as compared to reference locations. Concentrations varied greatly within the sediment core profiles (often covering a range of 5 to 10 fold), and concentrations typically were highest below the surface sediments in the lower half of each core profile, with generally decreasing concentrations from the 1964 horizon to the surface of the core. With the exception of copper, the trace-element profiles of the five cores collected along the pre-reservoir Columbia River channel typically showed trends of decreasing concentrations in sediments deposited after the

1964 time horizon. Trace-element concentration in sediments from the Spokane Arm of the reservoir showed distinct differences compared to the similarities observed in cores from along the pre-reservoir Columbia River. The authors stated that these differences were likely due to the greater influences of sediment inflow from the Coeur d'Alene basin to that portion of the reservoir.

The authors estimated minimum sediment accumulation rates for each station that ranged from 0.7 cm/year at RM 624, 692, and 705 (i.e., in the upper and lower portions of the UCR) to 2.8 cm/year at RM 668 in the middle portion of the UCR. The minimum sediment accumulation rate at RM 643 in the middle portion of the UCR was 1.3 cm/yr, and the minimum rate in the Spokane Arm was 1.7 cm/year. All of the cores from the UCR showed some evidence of disturbance from landslides in their deeper horizons, based on the concentration profiles of both metals and cesium-137. However, three cores (i.e., at RMs 705, 692, and 624) showed no evidence of potential disturbance from landslides since the 1964 time horizon.

The authors further stated "Particles of slag, which have physical and chemical characteristics of slag discharged to the Columbia River by a lead-zinc smelter upstream of the reservoir at Trail, British Columbia, were found in sediments of Lake Roosevelt. Slag particles are more common in the upstream reaches of the reservoir. The chemical composition of the interior matrix of slag collected from Lake Roosevelt closely approximated the reported elemental concentrations of fresh smelter slag, although evidence of slag weathering was observed. Exfoliation flakes were observed on the surface of weathered slag particles isolated from the core sediments. The concentrations of zinc on the exposed surface of slag grains were smaller than concentrations on interior surfaces. Weathering rinds also were observed in the cross section of weathered slag grains, indicating that the glassy slag material was undergoing hydration and chemical weathering."

Conclusions. With respect to the vertical distributions of metal concentrations in the sediment cores collected in the UCR, the authors concluded that the trace-element profiles reflect changes in historical discharges of trace elements to the Columbia River by an upstream smelter.

While iron concentrations have remained relatively unchanged since 1964, concentrations of the other metals generally exhibited declines in one or more of the three cores. Concentrations of zinc, lead, and cadmium exhibited relatively steady declines in all three cores since 1964 (although lead concentrations increased slightly in the upper 8 cm of the core from RM 692). Concentrations of these three metals at the core surfaces in 2002 were only 14 to 53 percent of the values associated with 1964. However, cadmium, lead, and zinc were in exceedance of Colville Tribe cleanup guidelines in all samples. In contrast to the temporal patterns observed for zinc, lead, and cadmium, concentrations of copper and mercury exhibited a relatively consistent decline since 1964 (only at RM 692) where the surface concentrations in 2002 were only 35 to 39 percent of the concentrations associated with 1964. For cores collected at RM 705 and 624, copper and mercury only exhibited declining concentrations in the top 3 to 6 cm, if at all. The metals co-vary in concentrations together over time and show clear evidence of periodic dilution of sediment concentrations by landslide inputs. However, the metals in the cores reflect the changes in effluent loading to the river from the Trail smelter over the last 30 years.

Discharges of metals in liquid effluent appear to have been the primary cause of the pattern of sediment metals deposition observed in the river/reservoir system. The authors further stated "Slag, while common in the uppermost reaches of the reservoir above RM 720, may be present only in minor amounts in sediments of the lower and mid reaches of the reservoir. Trace elements in sediments in the lower and mid reaches of the reservoir are thus largely due to the liquid effluent discharged to the Columbia River, which can easily be transported the length of the reservoir. Relative concentrations of trace elements in the cores from those reaches of the reservoir resemble relative loading of trace elements from the liquid effluent, further supporting that conclusion."

> *USEPA. 2006d. Phase I sediment sampling data evaluation, Upper Columbia River site, CERCLA RI/FS. Draft final. Prepared by CH2M HILL and Ecology and Environment Inc. under Contract No. 68-S7-04-01. U.S. Environmental Protection Agency, Region 10, Seattle, WA. August 2006.*

In 2005, EPA conducted a comprehensive survey of the concentrations of metals and organic compounds in surface sediments of the UCR in 2005, as Phase I of the UCR RI/FS. Stations were distributed from the U.S.-Canada border to the Grand Coulee Dam and were located at multiple water depths, from the dewatered banks to the deepest part of the channel.

Methods. Sediment sampling was conducted in April and May during the period of low pool in the UCR, to facilitate sample collection and depth determinations. Four kinds of sediment stations were sampled– transect stations, tributary mouths, reference areas, and beaches. Surface sediments (0 to 15 cm) were collected using a van Veen grab sampler or stainless steel hand tools. Sediments were analyzed for a variety of metals and organic compounds.

Results (inorganics). For metals, the discussion of the results of the 2005 Phase I sediment study presented below focuses only on the 12 metals identified as COIs by EPA (USEPA 2006d). The 12 COIs were antimony, arsenic, cadmium, chromium, copper, iron, lead, manganese, mercury, nickel, uranium, and zinc. In addition, the spatial patterns of the metals arsenic, cadmium, copper, lead, mercury, and zinc are discussed in greater detail than the patterns for the remaining six metals because they are considered representative of the three kinds of longitudinal distributions identified in the UCR.

In general, the longitudinal patterns of sediment grain size distribution and TOC were consistent with findings of the various historical sediment surveys in the UCR, with percent sand being highest at and above Marcus Flats and finer grained sediment being more prevalent at downstream locations. Based on this pattern, the authors suggested that Marcus Flats represents the transition point from predominantly riverine to reservoir conditions in the UCR. In addition, based on visual field observations of sediment characteristics made during the Phase I sampling, the authors concluded that black, coarse-grained granulated slag was largely confined to Marcus Flats and upstream areas. Fine-grained sediments in mid-channel samples are < 20 percent upstream and > 70 percent downstream of Marcus Flats. Clay and colloids are only found in the lower reservoir sediments. Side banks are mostly composed of sand. The distribution of total organic carbon content in the sediment follows the general trend of fine sediments.

The longitudinal distributions of iron, zinc, manganese, and copper exhibited a pattern that was generally consistent with the distribution of percent sand, with the highest concentrations found above Marcus Flats and lower and relatively consistent concentrations found in most downstream areas. However, several peaks in the concentration gradients were found on the left and right banks in downstream areas. Above Marcus Flats, two sets of elevated concentrations were apparent, with the highest values occurring upstream from EPA RM 725. The differences between the two sets of concentrations were particularly large for the left and right banks, compared to the mid-channel stations. This pattern suggests that sediments in the main channel are subjected to more uniform transport processes than are sediments on the banks, and that the banks include depositional areas where sediment and associated metals can accumulate.

The longitudinal distributions of cadmium and mercury (and to a lesser extent, lead and nickel) at mid-channel stations exhibited a pattern that was generally consistent with the distribution of fine-grained sediments, with the highest concentrations found below Marcus Flats and relatively low concentrations found above that location. For both metals, there were two separate areas of elevated concentrations, one at mid-channel in the middle portion of the UCR between EPA RM 690 and USGS RM 665 and one in the lower portion of the UCR below USGS RM 645. The longitudinal distributions of cadmium and mercury at the stations on the left and right banks of the UCR were generally opposite the distributions found at the mid-channel stations, with the highest concentrations found above Marcus Flats and relatively low concentrations found at most downstream areas, although several peak concentrations were found at several downstream locations, particularly near the confluence with the Spokane River.

Longitudinal distribution of lead at the mid-channel stations showed no clear upstream/downstream relationship. By contrast, the longitudinal distribution of lead on the left and right banks of the UCR exhibited a distribution similar to copper and zinc, with highest concentrations found above Marcus Flats and relatively low concentrations found in downstream areas, with the exception of peak concentrations at several reservoir stations. The longer distance transport of lead may have been due to a relatively greater association with organic carbon and suspended sediments (compared to other metals).

The cross-sectional concentrations of grain size parameters, TOC, and metals were evaluated by the authors at the six transects. The grain-size patterns observed for the uppermost station location near Marcus Flats showed that sediments in the mid-channel consisted of primarily sand-sized particles, with relatively little fine-grained material present. By contrast, bank stations contained relatively high percentages of fine-grained material, particularly silt. Sediments at the side-channel stations showed intermediate values, depending on which side of the channel they were located on and how far from mid-channel they were.

Conclusions (inorganics). These patterns suggested that the historical river channel in this part of the river was largely a high-energy environment, whereas the shallower areas represented depositional environments in many locations. Below Marcus Flats, the cross-sectional patterns indicated that the opposite depositional patterns generally occurred, with the historical river channel representing a depositional environment for fine-grained material, while coarser-grained particles identified in the banks. Coarse-grained sediments along banks within the middle and lower portions of the reservoir may have been the result of landslides or bank erosion, or from the original pre-reservoir soils of the areas flooded after construction of the Grand Coulee Dam. The authors suggested that the coarse-grained sediments in many of the bank areas in the lower portions of the UCR may have been partly the result of soil washing and redistribution of fine-grained material to deeper areas, as the result of the combined effects of the seasonal rising and falling water levels in the reservoir, wave action, exposure to rainfall, and localized sheet runoff.

In general, metal concentrations in the historical river channel at the cross-sectional transects corresponded closely to the patterns observed for the grain-size parameters. Near Marcus Flats, zinc and copper were elevated in association with the coarse-grained material found in the historical river channel. Lead exhibited a similar pattern. By contrast, cadmium and mercury were elevated only in the shallower side-channel and bank areas on the left side of the river. Below Marcus Flats, all five metals were elevated primarily in the historical river channel. These patterns suggest that one or more metals may be found at elevated concentrations in most areas at or above Marcus Flats, but that below that area, elevated metal concentrations are confined largely to the historical river channel. None of the tributary mouths appear to show evidence the tributaries are contributing to metals loading to mid-channel sediments, with the possible exception of nickel from the Colville and Kettle rivers.

Sediment core results showed the highest concentrations of arsenic, cadmium, lead, and mercury at USGS RM 676 at the 3- to 5-ft and 5- to 7-ft core intervals (middle reservoir reach, near Gifford). These metals are either associated with slag (arsenic) or with the aqueous Cominco effluent (other metals). It is assumed the other metals are preferentially adsorbed to fine organic clays or colloids.

Results (organics). For organic compounds, the authors evaluated SVOCs, pesticides, PCBs, PAHs, and dioxins and furans and, based on screening evaluations, identified the following organic compounds as COPCs: 2,4′-DDE, 2,4′-DDT, 4,4′-DDD, 4,4′-DDE, 4,4′-DDT, Aldrin, Aroclor 1016, Aroclor 1260, dioxins/furans, and PAHs (benzo[a]anthracene, benzo[a]pyrene, benzo[b]fluoranthene, benzo[k]fluoranthene, chrysene, dibenz[a,h]anthracene, and indeno[1,2,3-cd]pyrene). The following discussion focuses on the organic compounds identified as COIs, with the exception of all dioxins/furans. Of the latter, only 2,3,7,8-TCDD and 2,3,7,8-TCDF are discussed because they were some of the most frequently detected dioxins/furans in UCR sediments and both were detected in the Johnson et al. (1991a) study.

Conclusions (organics). No clear pattern between the bank and mid-channel stations was apparent for the concentrations of pesticide COIs, which were generally undetected. In addition, no clear upstream/downstream increase or decrease in concentrations was found. In instances where a lone data point was an outlier on a dry weight basis, that same point no longer appeared to be an outlier when TOC-normalized. Because non-polar organic compounds partition to TOC, such a pattern was expected.

Aroclor 1016 and Aroclor 1260 were both detected only once at EPA RM 687 on the left bank. Again, these data points no longer appeared to be outliers when they were TOC-normalized.

Like the pesticides, the PAHs showed no clear pattern between the bank and mid-channel stations. Similarly, no clear upstream/downstream increase or decrease in concentrations was observed, and the detected concentrations were relatively low, with most concentrations below 10 μg/kg dw. As noted previously for the other organic COIs, apparent outlier data points on a dry weight basis were no longer outliers once they were TOC-normalized. Sediment core data revealed that organic contaminants were consistently of greatest concentrations in the top 1-ft of the cores.

Fate and Transport of COIs. In addition to the general patterns of deposition as mentioned above, the authors concluded that Grand Coulee dam operations, reductions of known upstream sources of sediment COIs, slag distribution, sediment accretion and burial, benthic flux, and TOC sorption all played a significant role in the distribution patterns of sediment COIs in the river/reservoir system.

> *Paulson, A.J., R.J. Wagner, R.F. Sanzolone, and S.E. Cox. 2006. Concentrations of elements in sediments and selective fractions of sediments, and in natural waters in contact with sediments from Lake Roosevelt, Washington, September 2004. Open-File Report 2006-1350. U.S. Geological Survey. 84 pp.*
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> *Paulson, A.J. and S.E. Cox. 2007. Release of elements to natural water from sediments of Lake Roosevelt, Washington, USA. Environmental Toxicology and Chemistry 26:2550-2559.*

This study was conducted by researchers from the USGS to describe the release of metals from sediments of Lake Roosevelt measured by determining concentrations of these elements in natural waters in contact with sediments under three conditions varying in time and degree of physical mixing. The three types of natural waters were ambient interstitial water in the sediments, gently stirred water overlying cores incubated for one month, and supernatants of sediment slurries that were tumbled end-over-end for 43 days. Porewater samples simulated equilibrium between aqueous and sediment phases; a gently stirred incubation cores simulated lacustrine conditions, and end-over-end tumbling simulated turbulent riverine conditions.

Methods. Sampling was conducted in 2004. Sediments were collected at eight stations between approximately RM 601 and RM 736, to represent the areal distribution of trace element concentrations between the border and Grand Coulee Dam, and one station in the Sanpoil River to represent background conditions. Sampling stations were selected to target locations unaffected by reservoir fluctuations, although one station was likely above the range of maximum drawdown elevations, and locations that had been permanently sub-aqueous for the previous 2 years. Areas of landslides where sediment chemistry may be affected by bank material were purposely avoided.

Samples were collected using a (13.5 x 13.5 x 12 cm) deep box corer, following EPA protocols (USEPA 2001a). Water depths at the stations ranged from 12 to 26 m. Subsamples collected at each site included one 4.4 cm diameter core (up to 10 cm long) for analysis of elemental vertical distribution in sediments. A total of 28 sediment samples (1 to 4 from each site) were analyzed for total concentrations of 10 alkali and alkaline earth elements, 2 non-metals, and 20 metals, not including mercury. The samples consisted of discrete surface (0 to 1 cm) samples and composite near-surface samples (ranging from 3 to 10 cm) from four locations that were analyzed to provide vertical profile information; replicate 0 to 2 cm interval samples from four sites, which had been previously centrifuged for porewater analysis, to provide data on horizontal concentration variations; composite samples from nine sites representing material from 4 to 10 cm below surface; and samples of 0 to 2 cm below surface from two sites that had previously undergone a tumbling experiment. A sample of unsorted beach sand collected at RM 743 as part of a previous study (Cox et al. 2005) was also included in the data set for this study.

Results. Spatial trends in the total concentrations of arsenic, copper, cadmium, lead, and zinc in sediments were found to be similar to those of other studies. Concentrations of copper, lead, and zinc were highest in the river reach sediments and generally decreased downstream, while cadmium concentrations, relatively lower in the river reach sediments, generally increased downstream.

Aqueous concentrations of cadmium, copper, lead, and zinc in contact with Lake Roosevelt sediments generally were higher than the concentrations obtained from waters in contact with reference sediments. Porewater and water tumbled with sediments had the highest metals concentrations. Cadmium concentrations were found to be related to fine-grained sediment content, and inversely related to granulated slag content.

A dried aliquot from the composite sample from each site was sequentially and selectively extracted to estimate associations with sediment mineral phases. Aluminum, cadmium, cobalt, copper, iron, lead, manganese, vanadium, and zinc were the metals detected in the extracts and residual sediment from all sites. In contrast, silver was not detected in the extracts from the first three extract fractions and was detected only in the residual sediment from three sites. Bismuth was not detected in any extract from fractions one and three, but was detected in five of eight extracts from fraction two and in the residual sediment from all eight sites. The percentage of concentrations of metals that were greater than the reporting limit were 65, 73, 69, and 84 percent for the first, second, and third extraction fractions and the residual sediment, respectively. Uranium was the only actinide element analyzed and was detected in all fraction extracts and residual sediment samples.

Conclusions. The authors stated "An index of potential toxicity, called a toxic unit, was calculated for each type of natural water by comparing the aqueous concentration of an element to the chronic water-quality criterion that was corrected for water hardness. The three measures generally provided consistent indications of potential toxicity of copper and cadmium for sites in which the aqueous concentrations exceeded the water-quality criteria. All three measures indicated potential toxicity by the release of copper from sediments at two sites in the riverine reach of Lake Roosevelt (near the U.S.-Canada border). Likewise, all three measures indicate potential toxicity by the release of cadmium from the sediments collected at one site (LR-3) in the middle reach of Lake Roosevelt. Results of two of the three measures indicate potential toxicity by the release of lead from the sediments in the river reach, with the differences among the results possibly caused by differences in experimental conditions of artifacts. Higher concentrations of hardness in water in contact with sediments in the upper reach of Lake Roosevelt, which likely is caused by the differences in the carbonate geology, resulted in higher chronic water-quality criteria for this reach, thereby lessening the potential toxicity of lead releases. Concentrations of zinc and arsenic did not exceed the chronic water-quality criteria in any environment. Given that the total concentrations of arsenic, cadmium, copper, lead, and zinc in Lake Roosevelt sediments determined in this and previous investigations are consistent with elemental transport from the liquid effluent and slag discharged from the smelter, the trends in the releases of elements reported here suggest that future studies should focus on the release of copper from the river reach sediments and the release of cadmium from the middle and lower reaches."

Two locations in the lower reach of the study area contained the highest cadmium concentrations, which the authors attributed to sorption from the aqueous phase to organic-rich particulates and subsequent downstream transport.

> *Dowling, B. 2007. Field reconnaissance and sediment sampling report - Upper Columbia River site, Washington. Toxics Cleanup Program, Washington State Department of Ecology, Olympia WA. 36 pp.*

This one-day study was conducted by Ecology to evaluate the general sediment depositional patterns and conditions in the upper portions of the UCR, including visual indications of slag material, and to evaluate metal concentrations in surface sediments collected from selected locations, including exposed river shorelines and island locations. The timing of the reconnaissance coincided with the normal seasonal drawdown of Lake Roosevelt, resulting in more exposed shoreline and islands.

Methods. Sampling was conducted at 10 stations on May 14, 2007. Stations were distributed from Deadman's Eddy (RM 737) to approximately Kettle Falls (RM 700). At each station, surface sediment (0 to 15 cm) was collected using a stainless steel spoon and composited in plastic bags from 5 individual sub-sites within a 10-ft radius. Each station included a visual observation of the variations in sediment deposition, grain size, and slag content. Sites were selected with an emphasis on exposed bank and beach terrace areas within the zone of reservoir drawdown.

In the laboratory, each sediment sample was sieved using a 2- mm mesh sieve, and the fine fraction was submitted for metals analysis.

Results. Chemistry results were compared to selected human health and ecological screening levels. Concentrations of cadmium and copper in exceedance of screening levels were found at 3 to 4 of the 10 stations and elevated concentrations of lead and zinc were found at 7 to 8 of the 10 stations. Black, glassy, angular sand-size particles were observed at 4 of the 10 stations.

Conclusions. The authors suggested the black, glassy, angular sand particles were slag and that the slag/chemistry relationships observed in this reconnaissance survey could be used to help structure future, comprehensive sampling efforts.

> *Besser, J.M., W.G. Brumbaugh, C.D. Ivey, C.G. Ingersoll, and P.W. Moran. 2008. Biological and chemical characterization of metal bioavailability in sediment from Lake Roosevelt, Columbia River, Washington, USA. Arch. Environ. Contam. Toxicol. 54:557-570.*

This study was conducted by researchers from the USGS to assess bioavailability and toxicity of copper, zinc, arsenic, cadmium, and lead in sediments from Lake Roosevelt. This study is one of the most recent studies of metal concentrations in UCR sediments. Although only seven stations were sampled, they were relatively evenly distributed from the Grand Coulee Dam to the vicinity of Northport, Washington and were located in most of the major reaches of the UCR. The purpose of the study was to better understand sediment metals bioavailability and mobility within the UCR system.

Methods. Sediment sampling was conducted in September of 2004. Seven stations were located in the UCR between approximately RMs 735 and 601. One additional station was sampled near the head of the Sanpoil Arm, the designated reference area. Surface sediments (i.e., top 4 to 10 cm) were collected using a stainless steel box corer (20 cm diameter). Sediments were homogenized using Teflon blades, placed in polyethylene containers, and refrigerated. Sediments were then stored in the dark at 4°C prior to toxicity testing. All toxicity testing was conducted within 14 days of sediment collection.

Results and Conclusions. Relationships among metal concentrations in sediment and porewater, metal bioaccumulation, and toxic effects suggest that that sorption and/or co-precipitation of metals to iron and manganese hydrous oxides is a dominant control on metal bioavailability in Lake Roosevelt sediments under oxidizing conditions. Concentrations of acid-volatile sulfide in Lake Roosevelt sediments were too low to provide strong controls on metal bioavailability, and selective sediment extractions indicated that metals in most Lake Roosevelt sediments were primarily associated with iron and manganese oxides. Elevated metal concentrations in porewaters of some Lake Roosevelt sediments suggested that metals released from iron and manganese oxides under anoxic conditions contributed to metal bioaccumulation and toxicity. Results of both chemical and biological assays indicate that metals in sediments from both riverine and reservoir habitats of Lake Roosevelt are available to benthic invertebrates. Bioavailability of metals associated with hydrous oxides might be increased under reducing conditions, which might have developed during storage of some Lake Roosevelt sediments.

The authors concluded that concentrations of most of the metals of concern (i.e., arsenic, copper, lead, and zinc) were highest at the two most upstream stations near Northport and China Bend. However, they also noted that the highest concentrations of cadmium and mercury were found at sediments located in the downstream reservoir portions of the site. Five of the seven sediment samples collected exceeded probable effect concentrations (PECs, MacDonald et al. 2000). Whereas, the relatively low metal concentrations found at two of the stations in the reservoir portion of the site may have been influenced by landslides of bank soils, a factor that has been noted in other studies. Despite collecting samples from water depths greater than 10 m that have been submerged for at least two years, the sediment acid volatile sulfides were insufficient to provide strong controls on metal bioavailability. However, elevated metal concentrations in porewaters indicated iron and manganese oxides released metals under anoxic sediment conditions, thus contributing (and largely, controlling) the observed bioavailability and toxicity.

3.1.3 Sediment Toxicity

Summaries of sediment toxicity data in reports and publications are summarized by reporting authors or organization in chronological order. These summaries primarily represent studies conducted on samples collected downstream of the U.S.-Canada border. Studies were performed to meet a variety of objectives that and may not be representative of all Site sediments. Further evaluation of select data from these studies is provided in Appendix E.

> *Johnson, A., B. Yake, and D. Norton. 1989. An assessment of metals contamination in Lake Roosevelt. Segment No. 26-00-04. Washington State Department of Ecology, Olympia, WA. 84 pp.*

This study is the earliest evaluation of sediment toxicity in the UCR. In response to reports of elevated metal concentrations in fish and other environmental samples from Lake Roosevelt, Ecology conducted a series of field surveys between May and September 1986 to determine the extent and significance of contamination. Metals—primarily zinc, copper, lead, arsenic, cadmium, and mercury—were analyzed in sediment and water samples from the lake and its tributaries, and in a variety of the lake's fish species. A single core of the lake sediments was analyzed to determine the history of metals contamination. Bioassays were also conducted with select sediment samples to evaluate their potential toxicity. Fish were also tested for bioaccumulation.

Methods. Five stations were sampled between August 4 and 18, 1986. Four stations were sampled in the UCR between RMs 635 and 738. An additional station was sampled in a reference area located in Lower Arrow Lake, B.C. Two sediment toxicity tests were conducted– the 10-day amphipod test using *Hyalella azteca* and the 48-hour daphnid test using *Daphnia pulex*. The endpoint evaluated in both tests was survival, and both tests were conducted using whole sediments. Surface sediments (i.e., top 2 cm) in the UCR and Lower Arrow Lake were collected using a van Veen grab and an Ekman grab sampler, respectively. Sediments were transferred to a stainless steel container and homogenized using a stainless steel spoon. The homogenization devices were washed with LiquiNox® detergent, 10 percent nitric acid, and deionized water between stations. Sediments for chemical analysis and toxicity testing were transferred to glass jars with

Teflon-lined lids and transported to the laboratory on ice. All toxicity testing was conducted within 14 days of sediment collection.

Results. For the daphnid test, low survival found in the negative control added uncertainty to the validity of the test results. However, toxicity tests with neither species yielded significant differences from controls.

Conclusions. The authors noted that interpretation of the toxicity data were confounded by the relatively poor reference-area survival found for both tests. Given these test performance issues, the authors concluded that the toxicity results should not be considered conclusive. They noted that none of the sediments appeared to be toxic based on the amphipod test, but that sediment from Seven Bays appeared to be toxic based on the daphnid test, despite the low survival found in the negative controls. They also concluded that neither toxicity test exhibited a pattern of response that was clearly related to metal concentrations or the physical characteristics of the sediments. The inconclusive toxicity tests but clear patterns in sediment and fish tissues led the authors to recommend a monitoring and testing program for the UCR.

> *Johnson, A. 1991a. Review of metals, bioassay, and macroinvertebrate data from Lake Roosevelt benthic samples collected in 1989. Publication No. 91-e23. Washington State Department of Ecology, Olympia, WA.*

This study was a follow-up study of the initial UCR sediment toxicity conducted in 1986 by Johnson et al. (1989). It employed one test (Microtox®) that was not used in the 1986 study, as well as the two of the tests that were used in 1986 (*Hyalella* and *Daphnia*).

Methods. Six stations were sampled between August 14 and 17, 1989. Four stations were sampled in the UCR between RMs 605 and 728, one station was sampled near the head of the Sanpoil Arm, and one station was sampled in the Spokane Arm at RM 8. No stations were sampled in a designated reference area. Three sediment toxicity tests were conducted– the 10-day amphipod test using *Hyalella azteca*, the 7-day daphnid test using *Daphnia magna*, and the 15-minute Microtox® test using the saltwater bacterium *Vibrio fischeri.* Survival was the endpoint evaluated in the former two tests, whereas bacterial luminescence was the endpoint evaluated in the later test. The amphipod test was conducted using whole sediments, the daphnid test was conducted using both whole sediments and sediment elutriates, and the Microtox® test was conducted using a sediment elutriate. As for the 1986 study, the daphnid test was conducted using only whole sediments with a 1:4 ratio of sediment to water, mixed for 30 minutes and settled overnight. Surface sediments (i.e., top 2 cm) were collected using a 0.1-m^2 van Veen grab sampler. Sediments were transferred to a stainless steel container and homogenized using a stainless steel spoon. The homogenization devices were washed with LiquiNox® detergent, 10 percent nitric acid, and deionized water between samples. Sediments for chemical analysis and toxicity testing were transferred to glass jars with Teflon-lined lids and transported to the laboratory on ice.

Results and Conclusions. The authors noted that because the *H. azteca* and daphnid tests showed significant reduction of survival from exposure to Castle Rock and Swawilla Basin sediment, and because of the limited number of stations sampled, the resulting data provide limited insight as to whether or not adverse biological effects are occurring in the UCR sampled sediments. They concluded that the toxicity tests showed some evidence of toxicity at several sites, but that there were few correlations with metal concentrations in sediments. They noted that none of the sediment samples appeared to be extremely toxic, and that where some toxicity was indicated, there were inconsistencies in the toxicity results for the various tests. They also concluded that the Microtox® test may be an indicator of cadmium toxicity. In addition, the sediment chemistry indicated levels of metal contamination that would be expected to be toxic to benthic organisms. Metal concentrations considered as "heavily polluted" included manganese, zinc, copper, lead, arsenic, cadmium, and mercury (at the highest concentrations). For porewater, EPA chronic criteria were exceeded for lead and mercury at all sites, with Little Dalles and French Point Rocks sites having the highest concentrations. The greater zinc and copper concentrations at the Little Dalles site (RM 728.1) was indicative of the presence of slag in the sample.

The two sites with the most significant sediment toxicity results (*Hyalella* and *Daphnia*) were Castle Rock (RM 644.8) and Swawilla Basin (RM 604.9). Uncertainty with these toxicity data include an underestimation of the concentrations of aluminum, iron, manganese, and zinc (and probably mercury) due to incomplete digestions, poor precision of mercury analyses, and sulfide measurements were too erratic to use. However, the absence of clear toxicity despite elevated metal concentrations is not changed by the potential underestimation of metal concentrations. These results suggest that metals from these UCR sediments are not bioavailable to organisms in bench-top bioassays.

> *Godin, B. and M. Hagen. 1992. Cominco sediment bioassays, sediment and water chemistry - October and November 1991. Environment Canada, Conservation and Protection, Environmental Protection, Pacific and Yukon Region. Regional Data Report DR 92-12. May 1992.*

This study was conducted on behalf of Teck to evaluate the toxicity of slag from the Trail smelter, as well as sediment from the Columbia River, both above and below Trail.

Methods. A sample of mid-tap furnace slag was collected for toxicity testing during the middle of a pulse of quenched slag discharge in October, 1991. Surface sediments from the Columbia River were sampled using plastic spoons at two locations upstream from Trail (i.e., Genelle Island and Genelle Back Eddy Pool) and at one location downstream from Trail (i.e., Beaver Creek) in November of 1991. Sediment toxicity was evaluated using a 10-day daphnid bioassay with *D. magna* as the test species following methods described in Nebeker et al. (1984). Seven-day old test organisms were exposed to 2 cm of sediment or slag in a 1-L chamber filled with 800 mL of culture water allowed to equilibrate with the sediment sample for one day. The test organisms were indirectly exposed to sediment through the overlying water under this dated test method. Five replicate chambers were evaluated for each station. Total and dissolved metal concentrations (except mercury) were measured in the overlying water at the end of testing, and total metal concentrations were measured in the test sediments. The slag sample and river sediment sample bioassays were completed separately.

Results. For the slag sample, all test organisms died by the second day of exposure and the 11 offspring that were produced during this period had also died. For the corresponding negative control sample, 100 percent survival of test organisms was found after the 10-day exposure period, with an average of 37 offspring per female. For the Columbia River samples, 100 percent survival and a mean offspring per female of 157 and 264 were reported for Genelle Island and Genelle Back Eddy Pool upstream stations, respectively. Sediment collected from the Beaver Creek station downstream of the Teck discharge had 0 percent survival and 15 offspring per female. For the negative control sample corresponding to the river samples, 100 percent survival and 248 offspring per replicate were found.

Total and dissolved metal concentrations in water overlying the slag sample were reported, with elevated levels of copper (0.478 mg/L total and 0.165 mg/L dissolved), and zinc (0.208 mg/L total and 0.071 mg/L dissolved). Acutely toxic Columbia River sediments from Beaver Creek also caused elevated levels of copper (0.042 mg/L total and 0.021 mg/L dissolved) and zinc (0.040 mg/L total and 0.017 mg/L dissolved) in overlying water samples. Total metals and particle sizes in Columbia River sediment samples are also reported.

Conclusions. The authors concluded that sediments from the upstream stations were non-toxic, whereas the slag and the sediment collected downstream of the Teck discharge were toxic to *D. magna*. They also concluded that the observed toxicity in the latter two samples could be attributed at least partly to elevated concentrations of copper and zinc in the overlying water of the test chambers. The authors reported that copper, lead, and zinc concentrations in whole sediments from Beaver Creek were 32 to 155 times greater than the concentrations found in whole sediment from the two upstream sites. They noted that because mercury was not measured in the overlying water of the toxicity test chambers, but was found to be elevated in Beaver Creek sediments, it was not possible to evaluate the potential contribution of mercury to the observed toxicity.

> *Bortleson, G.C., S.E. Cox, M.D. Munn, R.J. Schumaker, and E.K. Block. 1994. Sediment-quality assessment of Franklin D. Roosevelt Lake and the upstream reach of the Columbia River, Washington, 1992. U.S. Geological Survey, Open-File Report 94-*

315, 130pp. (Reprinted in 2001 as U.S. Geological Survey Water Supply Paper 2496, 130 pp.)

This study represents the largest historical study of sediment toxicity in the UCR. It included toxicity testing of sieved, bulk sediment and porewater using three species at up to 27 sites.

Methods. Sediment samples were collected in September and October of 1992 in the UCR from RM 596 to 745. Sediments were also collected in the Sanpoil and Spokane Arms, as well as the Kettle and Colville Rivers. Lower Arrow Lake, B.C., was designated as the reference area for the study and two stations were sampled in that water body. Additional stations were sampled in the Kootenay and Pend Oreille rivers. Three sediment toxicity tests were conducted– the 7-day amphipod test using *H. azteca* (endpoint = survival), the 7-day daphnid test using *Ceriodaphnia dubia* (endpoints = survival and reproduction), and the 15-minute Microtox® test using the saltwater bacterium *V. fischeri* (endpoint = luminescence). Within the UCR, the amphipod and daphnid tests were conducted at a subset of 14 stations, whereas the Microtox® test was conducted at all 27 stations. The amphipod and daphnid tests were conducted using whole sediments, whereas the Microtox® test was conducted using whole sediments and sediment porewater (obtained by centrifugation). Surface sediments (i.e., top 1.3 to 2.5 cm) were collected using a stainless steel van Veen grab sampler. Sediments were transferred to a glass bowl and homogenized using Teflon blades and a stainless steel spoon. Sediments were then wet sieved through a 2-mm nylon screen, before shipment to the bioassay laboratories. Sediments for toxicity testing using the *H. azteca* and *C. dubia* tests were transferred to high-density polyethylene containers, whereas sediments for testing using Microtox® were transferred to glass jars. All sediments were maintained at or below 4^{o}C prior to toxicity testing. All toxicity testing was conducted within 14 days of sediment collection.

Results and Conclusions. The authors reported that all three toxicity tests indicated toxicity responses significantly different from that of the negative controls or reference sediments sampled near and downstream of the U.S.-Canada border. They also noted that survival or reproduction of at least one, and in some cases all three, of the test species was adversely affected in the Northport reach of the UCR, from the U.S.-Canada border to Onion Creek (RM 730). In general, little toxicity was found at stations located immediately downstream from the Northport reach. However, the overall longitudinal pattern of sediment toxicity showed an increase in toxicity as stations approached the lower reservoir and were dominated by fine-grained sediments. For example, the Microtox® tests showed toxicity in one third of the sites in the mid and lower reaches of the lake and *Ceriodaphnia* at two of the lower lake sites.

This pattern of toxicity in the Northport reach, followed by no toxicity in the upper reservoir, and eventual toxicity in the lower reservoir seems linked to fine grained sediments but not the associated porewater. For example, the Microtox® results showed no toxicity with predominately sandy sediments or porewater, but did exhibit toxicity to fine-grained sediments. The authors concluded that metals were the likely cause of toxicity as exhibited in the Microtox® results.

> *Era, B. and D. Serdar. 2001. Reassessment of toxicity of Lake Roosevelt sediments. Publication No. 01-03-043. Environmental Assessment Program, Washington State Department of Ecology, Olympia, WA. 54 pp.*

This study of sediment toxicity in the UCR was conducted on sediments collected in 2001. The primary objective of this study was to reassess metal concentrations and toxicity of sediments through bioassay tests and to make recommendations on the continued listing or de-listing of the upper Columbia River and Lake Roosevelt sites on the 303(d) list.

Methods. Ten stations were sampled between May 7 and 9, 2001. Seven stations were sampled in the upper and lower portions of the UCR, with four stations located between RMs 596 and 645 and three stations located between RMs 738 and 745. One additional station was sampled near the head of the Sanpoil Arm, and a second additional station was sampled in the Kettle River. The designated reference station was located in Lower Arrow Lake, B.C. Three sediment toxicity tests were conducted– the 10-day amphipod test using *H. azteca* (endpoint = survival), the 20-day chironomid test using *Chironomus dilutus* (endpoints = survival and growth), the 15-minute Microtox® test using the saltwater bacterium *V. fischeri* (endpoint = bacterial luminescence). The amphipod and chironomid tests were conducted with whole sediments, whereas the Microtox® test was conducted with porewater. Surface sediments (i.e., top 10 cm) in the UCR were collected using a 0.1-m^2 stainless steel van Veen grab sampler. The sample for the Kettle River was collected to a depth of 5 cm. Surface sediments in the Sanpoil Arm and Kettle River were collected by wading into the water and scooping sediments with a stainless steel spoon. Sediments were transferred to a stainless steel container and homogenized using a stainless steel spoon. The homogenization devices were washed with LiquiNox® detergent, tap water, 10 percent nitric acid, and deionized water between stations. The equipment was then air-dried and wrapped in aluminum foil. Sediments for chemical analysis and toxicity testing were transferred to pre-cleaned glass jars with Teflon-lined lids and transported to the laboratory on ice. All toxicity testing was conducted within 14 days of sediment collection.

Results. The authors found that with the exception of the station located near Grand Coulee Dam, all nine stations exhibited a significant toxicity response for at least one of the three toxicity tests. The reference sediment showed low toxicity in all of the bioassay tests. Chironomid survival appeared to be a more sensitive endpoint than the growth endpoint due to six hits compared to two hits respectively. Samples with the greatest toxicity were from the Auxiliary Gage and Goodeve Creek (7 miles downstream from the border). The least toxic sediments were from Boundary, Kettle River, Whitestone Creek, Sanpoil River, and Grand Coulee sediment samples.

Conclusions. The authors stated "Metals concentrations and toxicity levels in the upper Columbia River and Lake Roosevelt sediments remain relatively high. All but the Grand Coulee site had at least one toxicity hit out of the suite of bioassay tests performed on the sediments. In the upper Columbia River reach, above the town of Northport and below the U.S.-Canada border, cadmium, copper, lead, and zinc concentrations were elevated. Based on these findings and the findings of previous studies, bioassay toxicity for the upper Columbia River and Lake Roosevelt may be attributed to metals contamination. Although the Kettle River and Sanpoil River sites showed toxicity to some of the bioassay tests, metal concentrations were very low. The cause of the apparent sediment toxicity for these sites is unknown and could possibly be attributed to other toxic parameters not tested for in the present study." The authors further state "Based on the existing policy for 303(d) listings (one toxicity hit per segment), eight of the nine sites (1 through 8) should be listed. Based on the newly proposed listing criteria (toxicity at three separate locations within a segment), none of the nine sites should be placed on the 2002 303(d) list. Regardless of the 303(d) listing status, there is sufficient toxicity at the majority of sites in the upper Columbia River and Lake Roosevelt to warrant further investigation."

> *Besser, J.M., W.G. Brumbaugh, C.D. Ivey, C.G. Ingersoll, and P.W. Moran. 2008. Biological and chemical characterization of metal bioavailability in sediment from Lake Roosevelt, Columbia River, Washington, USA. Arch. Environ. Contam. Toxicol. 54:557-570.*

This study is one of the two most recent studies of sediment toxicity in the UCR. Although only seven stations were sampled, they were relatively evenly distributed from the Grand Coulee Dam to the vicinity of Northport, Washington.

Methods. Sediment sampling was conducted in September of 2004. Seven stations were located in the UCR between approximately RMs 735 and 601. One additional station was sampled near the head of the Sanpoil Arm, the designated reference area. Two chronic sediment toxicity tests were conducted– the 28-day amphipod test using *H. azteca* (endpoints = survival and growth), and the 12-day chironomid test using *C. dilutus* (endpoints = survival and growth). Surface sediments (i.e., top 4 to 10 cm) were collected using a stainless steel box corer (20 cm diameter). Sediments were homogenized using Teflon blades, placed in polyethylene containers, and refrigerated. Sediments were then stored in the dark at 4°C prior to toxicity testing. All toxicity testing was conducted within 14 days of sediment collection.

Results and Conclusions. The authors concluded that the observed toxic responses in the amphipod and chironomid tests were generally consistent with predictions of metal toxicity based on empirical and equilibrium partitioning-based SQGs. Five of the seven sediment samples collected exceeded PECs (MacDonald et al. 2000). Midge growth, but not survival was significantly affected by exposure to UCR sediments, while neither *Hyalella* growth nor survival (corrected for suspected predation by indigenous organisms) differed significantly from the reference. Midge growth and amphipod survival were significantly correlated with porewater and sediment metal concentrations, and that the observed effects likely reflect concentrations of multiple metals. They also concluded that the bioavailability of metals in sediments from the UCR to benthic macroinvertebrates is highly variable across the site. Chronic toxic effects on amphipods (*H. azteca*; reduced survival) and midge larvae (*C. dilutus*; reduced growth) in whole-sediment exposures were generally consistent with predictions of metal toxicity based on empirical and equilibrium partitioning-based SQGs.

> *USEPA. 2006d. Phase I sediment sampling data evaluation, Upper Columbia River site, CERCLA RI/FS. Draft final. Prepared by CH2M HILL and Ecology and Environment Inc. under Contract No. 68-S7-04-01. U.S. Environmental Protection Agency, Region 10, Seattle, WA. August 2006.*

This data set represents the most recent and most extensive evaluation of sediment toxicity in the UCR.

Methods. Sediment sampling was conducted in April and May of 2005. Fifty stations were sampled for sediment toxicity testing between RMs 603 and 745. Six additional stations were sampled in six tributaries to the UCR between RMs 685 and 732, to represent reference areas. Three sediment toxicity tests were conducted– the 28-day amphipod test using *H. azteca* (endpoints = survival and growth), the 10-day chironomid test using *C. dilutus* (endpoints = survival and growth), and the 7-day daphnid test using *C. dubia* (endpoints = survival and reproduction). All three toxicity tests were conducted using whole sediments. Surface sediments were collected using a van Veen grab sampler at most stations below RM 725, and generally the top 8 to 16 cm of the sediment column was sampled. Above RM 725, at several stations below that location, and at the six reference areas located in tributaries to the UCR, surface sediments were collected using hand tools, and generally the top 4 to 8 cm of the sediment column was sampled. Following collection, sediments were transferred to an aluminum-lined stainless steel bowl and homogenized using disposable hand tools. Any obvious abnormalities (e.g., wood/shell fragments, large organisms) and coarser-grained sediment (e.g., pebbles, gravel) were removed by hand or using the disposable hand tools prior to homogenization. Sediment subsamples were then distributed to pre-labeled sample containers and stored in a cooler at 4°C. All toxicity testing was conducted within 14 days of sediment collection, which is considered an appropriate sediment holding time for sediment toxicity testing (ASTM 2000; USEPA 2000).

Results. A final report was not yet been prepared by the authors; however, the data are presented in Appendix E.

3.1.4 Benthic Macroinvertebrate Tissue Chemistry

Summaries of benthic macroinvertebrate data in reports and publications are summarized by reporting authors or organization in chronological order. These summaries primarily represent studies conducted on samples collected downstream of the U.S.-Canada border. Studies were performed to meet a variety of objectives and may not be representative of all Site conditions.

> *Tielens, J.T. 1994. Bioavailability of trace metals in Franklin D. Roosevelt Lake, Washington, sediments. Environmental Science and Regional Planning, Washington State University. Masters Thesis. December 1994.*

This study was conducted by a graduate student at Washington State University to evaluate metal concentrations in surface sediments, porewater, and benthic macroinvertebrates in the UCR. Stations were located so that comparisons could be made with the earlier study conducted by Johnson et al. (1989).

Methods. Sampling was conducted in August of 1994. Surface sediments were sampled at four stations in the UCR from the U.S.-Canadian border to the Grand Coulee Dam (i.e., China Bend, Gifford, Seven Bays, and Swawilla Basin). Surface sediments were also collected in the Kettle River, which was designated as a reference area by the author. Triplicate sediment samples were collected at each station at water depths of 11 to 12 m using a Ponar grab sampler. Benthic macroinvertebrates were collected in the field using a 0.6-mm sieve. Porewater was collected in the laboratory using centrifugation (7,000 revolutions per minute [rpm] for 1 hour), followed by filtration through a 0.45-µm filter.

Results. The abundance of benthic macroinvertebrates at all four UCR stations were lower than the value found for the Kettle River reference area. In addition, diversity was low at all four UCR stations, with only three major taxa found (i.e., isopods, oligochaetes, and chironomids), and chironomids accounting for over 80 percent of total abundance at each station. Tissue concentrations (un-depurated) of zinc and copper were elevated in benthic macroinvertebrates at all four UCR stations, compared to the Kettle River reference area. Porewater concentrations of copper and zinc were also higher than the reference values at all four UCR stations.

Conclusions. The authors suggested that the source of metals may differ between the upper and lower two stations, with the Spokane River being a potential source of metals for the lower two stations. The author concluded that benthic macroinvertebrates represent a major pathway by which metals can be transferred to fish in the UCR, and recommended that other food sources (e.g., zooplankton) be evaluated in the future.

MESL (MacDonald Environmental Sciences Ltd.). 1997. Lower Columbia River from Birchbank to the international border: Water quality assessment and recommended objectives. Technical report. Available at: http://www.env.gov.bc.ca/wat/wq/objectives/birchbank/Birchbanktech.pdf. Prepared for Environment Canada and British Columbia Ministry of Environment, Lands and Parks.

This study was undertaken to evaluate the levels of contaminants in aquatic organisms of the Columbia River and to develop tissue residue objectives (TROs) that may be used in management decisions. TROs were developed to protect aquatic life, wildlife, and human health in waterbodies that are affected by anthropogenic activities. The TROs were set to protect the most sensitive use (i.e., aquatic life, wildlife, or human health) in the ecosystem. Data collected under the CRIEMP and other historical studies were used to assess contaminant residues in the tissues of algae and aquatic macrophytes, benthic invertebrates, and fish in the portion of the Columbia River from Birchbank to the U.S.-Canada border.

Methods. Data on the concentrations of contaminants in the tissues of algae and aquatic macrophytes, benthic macroinvertebrates, and fish in the Columbia River were compiled from previous studies. Samples of *Ulothrix* sp. and *Enteromorpha* sp., available from a 1978 survey, and pondweed (*Potamogeton perfoliatus*) collected during the 1992 CRIEMP were used to assess levels of contaminants in algal and aquatic macrophyte tissues. In 1977, contaminant-residue levels in benthic macroinvertebrates were assessed by analyzing metals concentrations in the stomach contents of largescale suckers captured in the Columbia River, as well as in mayflies, stoneflies, and caddisflies collected at three locations on the River (i.e., at Birchbank and two locations downstream of the Cominco smelter). In 1992 invertebrate samples were collected, under the CRIEMP, to assess the levels of metals and organic contaminants in emergent caddisflies and freshwater mussels in the Columbia River at Waneta and further upstream near the Celgar Pulp Company facility.

The TROs were developed for the Columbia River using a three-step process. First, bioaccumulative contaminants were identified based on the existing and potential sources of contamination in the lower Columbia River basin. Next, tissue residue guidelines were compiled from various sources for the protection of aquatic life, wildlife, and human health. Finally, the tissue residue guideline for the most sensitive receptor group was selected as the TRO for each substance. In some cases, TROs were developed using the CCME protocols.

Results. Tissue residue concentrations in Columbia River benthic macroinvertebrates were indirectly measured by analyzing the concentrations of cadmium, lead, and zinc in the stomach contents of largescale suckers. These metal concentrations were higher downstream of the Cominco smelter than in fish gut contents from Lower Arrow Lake. The levels of cadmium, lead, and zinc in mayflies and stoneflies collected at the

Birchbank site were similar to the levels found in the largescale suckers; furthermore, levels of the same metals in caddisflies collected downstream of the smelter were elevated compared to the levels observed near Birchbank. More recent studies showed that levels of cadmium, lead, and zinc in aquatic invertebrate tissues had declined significantly since the previous surveys were conducted. However, levels of cadmium, lead, and zinc and other metals in freshwater mussels remained elevated near Waneta, BC and levels of chromium in mussel tissue collected near the Celgar Pulp Company site were significantly higher than those at other sites. Levels of most organic contaminants in macroinvertebrate samples were below detection limits, except for PCDDs and PCDFs, which were detected at both sites in 1992. Additionally, several chlorophenols were also detected in caddisfly tissue.

The TROs for the Columbia River from Birchbank to the U.S.-Canada border were determined for various metals (i.e., arsenic, cadmium, chromium, copper, lead, mercury, thallium and zinc) and for the organic contaminants, PCDD and PCDF. Of these TROs, *de novo* provisional objectives for the most sensitive receptor group were developed for arsenic, chromium, and lead. Insufficient information was available to determine TROs for copper, thallium, and zinc.

Conclusions. The synthesis of contaminant residues in tissues of aquatic organisms from the Columbia River showed that elevated levels of various metals and organic contaminants were present in algae and aquatic macrophytes, benthic macroinvertebrates, and fish, with the highest concentrations observed downstream of industrial activities and discharges. Improvements in water quality were observed between 1976 and 1996, resulting in declining levels of contaminants in the tissues of benthic macroinvertebrates and fish of the Columbia River. The TROs were developed for various metals (arsenic, cadmium, chromium, lead, and mercury) for the Columbia River. The authors recommended the development of tissue residue guidelines for copper, thallium and zinc to address the lack of relevant information for these variables.

> *Besser, J.M., W.G. Brumbaugh, C.D. Ivey, C.G. Ingersoll, and P.W. Moran. 2008. Biological and chemical characterization of metal bioavailability in sediment from Lake Roosevelt, Columbia River, Washington, USA. Arch. Environ. Contam. Toxicol. 54:557-570.*

This is the only study of laboratory based bioaccumulation using UCR sediments. Evaluations of sediment chemistry (whole sediments and porewater) and sediment toxicity were concomitantly conducted for the effects of copper, zinc, arsenic, cadmium, and lead. Although only seven stations were sampled, they were relatively evenly distributed from the Grand Coulee Dam to the vicinity of Northport, Washington.

Methods. Sediment sampling was conducted in September of 2004. Seven stations were located in the UCR between approximately RM 735 and 601. One additional station was sampled near the head of the Sanpoil Arm, the designated reference area. Surface sediments (i.e., top 4 to 10 cm) in the UCR were collected using a stainless steel box corer (20 cm diameter). Sediments were homogenized using Teflon blades, placed in polyethylene containers, and refrigerated during shipment to the testing laboratory. Bioaccumulation evaluations were conducted using the 28-day test based on the oligochaete *Lumbriculus variegatus*. Test chambers were 6-L glass aquaria containing 1 L of sediment, with four replicates per sediment sample. Oligochaetes were not fed during the test. At test termination, the oligochaetes were depurated for 6 hours and analyzed for five metals (i.e., arsenic, cadmium, copper, lead, and zinc). Toxicity tests were conducted on the same sediment samples, using standard amphipod and midge tests for survival and growth.

Results. Bioaccumulation of the five metals analyzed (copper, zinc, arsenic, cadmium, and lead) were greater when exposed to UCR sediments than control sediment. They also found that bioaccumulation of all five metals differed among sites, with oligochaetes in three or more UCR sediment samples having tissue concentrations of cadmium copper, lead, and zinc significantly greater than those accumulated from the reference sediment. Tissue concentrations of cadmium, copper, and lead showed the greatest increases, with values that exceeded the reference concentrations by maximum factors of 47 (site LR3), 20 (LR7), and 22 (LR3), respectively. Zinc did not exhibit elevations comparable to those observed for the three previous metals (significantly elevated only at sites LR1, LR3, and LR7), and the authors suggested that variations in zinc bioavailability among stations were minimal, or that the oligochaetes were able to regulate internal zinc concentrations. Limited mercury data (single samples from seven of the eight sites) also indicated a wide range of tissue concentrations from 0.03 to 0.57μg/g. Tissue concentrations of arsenic did not differ significantly from the reference area value at any UCR station. Tests of association between ambient chemistry and bioaccumulation revealed significant positive associations between oligochaete uptake and sediment or porewater concentrations of copper, arsenic, and lead. The only significant correlation between amphipod and midge toxicity test endpoints and bioaccumulation in worms was for copper bioaccumulation and amphipod survival.

The highest concentrations of total metals in sediments used in these tests were recorded from the riverine Northport site, LR7 (primarily for lead and zinc), whereas the most elevated porewater concentrations were from further down-reservoir at LR2 and LR3 (lower and mid-lake, respectively).

Conclusions. The authors concluded that the observed bioaccumulation in the UCR reflected a general enrichment of background metal concentrations. They also found that oligochaetes accumulated the greatest concentrations of copper from the most upstream riverine sediments, and the greatest concentrations of cadmium, and lead from the lower reservoir portion of the site.

3.1.5 Fish Tissue Chemistry

Summaries of fish tissue data in reports and publications are summarized by reporting authors or organization in chronological order. These summaries primarily represent studies conducted on samples collected downstream of the U.S.-Canada border. Studies were performed to meet a variety of objectives and may not be representative of all Site conditions. Further evaluation of select data from these studies is provided in Appendix F.

> *Hopkins, B.S., D.K. Clark, M. Schlender, and M. Stinson. 1985. Basic water monitoring program, fish tissue and sediment sampling for 1984. Publication No. 85-7. Washington State Department of Ecology, Olympia, WA.*

This document is a result of Ecology's BWMP that was initiated in 1978. Fish tissues were analyzed to obtain information on the incidence and distribution of metals and synthetic organic compounds in the aquatic environment. The data collected were used to identify potential problem areas requiring further investigation. This document reports data from the 1984 field season, and BWMP data from 1978 to 1983 for reference. The 1984 BWMP effort was the first year that stream sediments were sampled at each station where fish were collected.

Methods. Twelve stations were selected for sampling based on the 1983 BWMP results. Two sites were not sampled however, due to field conditions. Sampling locations included Wenatchee River at Wenatchee; Lake Chelan at outlet; Okanogan River near Malott; Columbia River at Northport; Palouse River at Hooper; Walla Walla River below Warm Springs; Yakima River below Kiona; Yakima River at Birchfield Drain; Skagit River near Mount Vernon; and the Green/Duwamish River.

At each station fish species were collected representing two trophic levels; the same species were collected at each station when possible to provide comparability. The primary collection method used to obtain fish samples throughout this program was electrofishing; however, hook-and-line was used whenever the sample area was not conducive to this sampling method. Six species were collected overall–bridgelip sucker, longnose sucker, mountain sucker, mountain whitefish, northern pikeminnow, and largemouth bass. Sediment was also collected from the stream channel at each station. Three tissue types were isolated from each composite (liver, gill, fillet with skin) and analyzed for pesticides and metals.

Conclusions. The authors concluded that for samples collected at Northport, Washington (in the UCR) fillet tissue contained an average lead level of over 90 percent of the unofficial Food and Drug Administration (FDA) guidelines for other food types. Cadmium, copper, and zinc concentrations were also considered to be elevated, both in fish tissue and sediment. The authors observed that "the highest level of pollutants [arsenic, cadmium, copper, mercury, lead, and zinc in sediments] was found at Northport on the Columbia River."

> *Johnson, A. and W. Yake. 1989. Survey of mercury and dioxin in Lake Roosevelt sport fish in 1989. Preliminary results for mercury. Publication No. 89-e29. Washington State Department of Ecology, Olympia, WA.*

The purpose of this survey was to address concerns raised by the Colville Tribes and the Lake Roosevelt Water Quality Council. A previous mercury analysis was performed in 1988 showing uniformly low concentrations of mercury, but because of elevated concentrations in the lake's bottom sediments and discharge of mercury from the Cominco lead-zinc smelter and refinery, another survey was considered justified. Muscle tissue samples were collected off Marcus Island (sturgeon), at the Colville River mouth (walleye), and at the mouth of Hawk Creek (walleye) to coincide with popular sport fishing areas. Mercury concentrations, expressed in wet weight, ranged from 0.05 to 0.24 μg/g (mean of 0.155 μg/g) in walleye and 0.02 to 0.10 μg/g (mean of 0.05 μg/g) in white sturgeon. Except for a few small walleye, all the fish were of legal sport fishing size. Walleye were collected by electrofishing. The sturgeon samples were muscle tissues from the severed heads from sturgeon caught by anglers in the upper lake, as arranged for collection by the Department of Wildlife (e.g., not standard fillet samples). The authors noted that the sturgeon tissue samples did not include material exposed to the open cut in the field.

Conclusions. The authors concluded 1) None of the samples exceeded the FDA action level at that time of 1.0 μg/g of mercury for commercially marketed fish nor the Canadian criterion of 0.5 μg/g of mercury; 2) Results were consistent with the 1986 survey and posed no threat to human health; and 3) Mercury concentrations in other sport fish species in Lake Roosevelt were expected to be equal or lower.

> *Johnson, A., B. Yake, and D. Norton. 1989. An assessment of metals contamination in Lake Roosevelt. Segment No. 26-00-04. Washington State Department of Ecology, Olympia, WA. 84 pp.*

The study was conducted by researchers from Ecology in response to reports of elevated metal concentrations in fish and other environmental media. Whole fish (largescale suckers) and fillets from walleye, rainbow trout and other sport fish were analyzed.

Methods. Fish were collected September 23 through 26 at three locations in Lake Roosevelt and the UCR. The three stations were RM 732 below Northport; RM 680 above Gifford; and RM 635 at Seven Bays. The primary target species were: walleye and largescale sucker. Lake whitefish, yellow perch, and rainbow trout were also collected at each station. Fish were collected using gill nets except for rainbow trout which were collected using hook and line. Composites were composed of four to five individuals of each species.

Muscle tissues were analyzed from walleye and the other sport fish while suckers were analyzed as whole body. The muscle tissues were skinless fillets. The analyte list consisted of aluminum, arsenic, cadmium, copper, iron, lead, manganese, mercury, and zinc.

Results. Zinc, copper and lead in whole body largescale suckers had strong trends of decreasing concentrations from the Northport downstream to Seven Bays. Lead showed the greatest decline with concentrations 15 times higher at the Northport station (6.09 µg/g) than at Seven Bays (0.39 µg/g). Zinc and copper concentrations were higher by factors of about 2 and 7, respectively, from the Northport station to Seven Bays. Cadmium concentrations were highest at Gifford but ranged throughout the site (0.26 to 0.38 µg/g). It is not known if the zinc, copper, lead, and cadmium concentrations in whole body largescale suckers were due to accumulations in tissues or reflected the presence of contaminants in gut contents. The relative abundance of the metals in the suckers also approximately corresponded to the relative abundance in sediment samples collected. The authors calculated that the amount of sediment in the gut of the suckers need only to have been between 1 and 0.1 percent of the total weight of the fish to account for the zinc, copper and lead concentrations measured in the tissues.

Metal concentrations in sport fish muscle were lower than in whole suckers, except for mercury. Zinc, lead, arsenic, and cadmium concentrations were similar among species while walleye had higher mercury and lower cadmium concentrations. There were no significant differences among stations for walleye. The authors stated that only mercury appeared to accumulate significantly in the muscle tissue of Lake Roosevelt sport fish. Mercury tissue concentrations found within this study were similar to concentrations reported by the B.C. Ministry of the Environment in fish collected near the Cominco smelter. In addition, some muscle tissue from largescale suckers collected at the Gifford station was analyzed for mercury by the Ministry of the Environment and the results indicated that suckers, and perhaps other benthic fish species in Lake Roosevelt, may have been accumulating higher levels of mercury in muscle tissue than expected.

Conclusions. Whole largescale suckers from the upper part of the reservoir had significantly higher concentrations of metals (cadmium, copper, lead, and zinc) than samples from the lower reservoir. Lead and cadmium concentrations in fish tissue were among the highest in the nation, as indicated by USFWS National Contaminant Biomonitoring Program (NCBP). The significance of these metals data were difficult to determine due to the lack of fish tissue based criteria. Metal concentrations in Lake Roosevelt sport fish fillets were within guidelines set by the FDA and other countries. The authors stated that they did not find evidence that levels of fish tissues at the time of the study posed a serious threat to human health. However, due to the level of contamination in some whole fish from some parts of the system, and the narrow margin between natural background and potentially toxic concentrations of mercury in sport fish muscle tissue, it was recommended that Ecology establish a program to periodically monitor the edible tissue of Lake Roosevelt sport fish for mercury.

> *Johnson, A. 1990. Results of screen for dioxin and related compounds in Lake Roosevelt sport fish. Technical Memorandum. Washington State Department of Ecology. Olympia, WA.*

This study was conducted by the Ecology's Water Quality Program in response to concerns raised in an EC report which found elevated concentrations of dioxins and furans in lake whitefish below the Celgar Pulp Mill. The extent of contamination of dioxins and furans in sport fish was evaluated.

Methods. Both walleye and white sturgeon were collected in July of 1989. Two sturgeon were caught in the vicinity of Marcus Island. The heads and some of the anterior muscle tissue, not including skin or tissue exposed in the field, were used in the analysis. The ages of the two sturgeon were estimated to be 12 to 13 years. Two walleye were collected downstream of the confluence with the Colville River using electrofishing. The ages of the walleye were determined to be at least five years old. Walleye tissue samples consisted of skinless fillets.

Results. Dioxins were only detected in one sturgeon and one walleye sample while furans were detected in all samples. Dioxin and furan concentrations varied greatly within species for both the walleye and sturgeon; however, comparable concentrations were found from one sturgeon duplicate sample (2.2 to 2.6 ppt ww TCDD and 221 to 321 ppt ww TCDF) and one walleye sample (4.0 ppt ww TCDD and 326 ppt ww TCDF).

Conclusions. Reasons for the within-species differences were not known, but the large and variable species ranges were raised as a possible contributing factor. The concentrations of dioxins and furans reported from the fish collected as a part of this study were compared to the results of the EC report. It was found that dioxin and furan concentrations were much less than those of fish collected just below the Celgar Pulp Mill; however, different species were collected in these studies. The Washington State Department of Health determined it could not make any conclusions about these study data due to the small sample size.

> *Johnson, A. 1991b. Results of screen for EPA xenobiotics in sediment and bottom fish from Lake Roosevelt (Columbia River). Publication No. 91-e24. Washington State Department of Ecology. Olympia, WA.*
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> *Johnson, A., D. Serdar, and D. Norton. 1991a. Spatial trends in TCDD/TCDF concentrations in sediment and bottom fish collected in Lake Roosevelt (Columbia River). Publication No. 91-29. Washington State Department of Ecology, Olympia, WA.*

In June of 1990, Ecology collected sediment and fish tissue samples from Lake Roosevelt for analysis of PCDDs and PCDFs and 44 other compounds taken from EPA's National Bioaccumulation Study. The authors' objective was to evaluate the transport and distribution of these chemicals throughout the lake.

Methods. Largescale sucker were collected from six locations in Lake Roosevelt, including Northport (RM 735), China Bend (RM 724), Marcus Flats (RM 709), French Point Rocks (RM 692), Hunters (RM 661), and the Grand Coulee Dam (RM 601). The fish were collected by electro-shocking. Whole-fish composites consisting of five fish each were analyzed for 2,3,7,8- substituted dioxin and furan compounds, and percent lipids.

Results. TCDD and TCDF were the only congeners detected in fish samples. TCDD was detected in all Lake Roosevelt fish samples, but not in fish from Rufus Woods Lake or the Spokane River. Concentrations of 2,3,7,8-TCDD in whole sucker ranged from 0.9 to 2.6 pg/g ww, with the highest concentrations occurring in fish collected from Marcus Flats (RM 709). Fish from all sites had detectable concentrations of TCDF. Concentrations of 2,3,7,8-TCDF in whole sucker composites ranged from 17 pg/g ww to 48 pg/g ww, again with the maximum at Marcus Flats. Samples from Rufus Woods Lake and the Spokane River had the lowest TCDF concentrations. PCBs and DDE were also detected in fish tissues collected as a part of the study. Total PCBs ranged from 35.8 to 61.1 ng/g ww with highest concentration found at the China Bend station. DDE concentrations ranged from 18.9 to 41.8 ng/g ww with the highest concentration found at the Grand Coulee station. In general, TCDD and TCDF were detected more often in fish than in sediments.

Conclusions. The authors stated "Results of this survey showed long distance transport of TCDD and TCDF through Lake Roosevelt, with the latter compound still detectable at slightly elevated levels in whole fish samples collected over 200 river miles from the presumed source (the Celgar Pulp mill). The Spokane River did not appear to be an important source of either TCDD or TCDF to Lake Roosevelt. Significant deposition of these contaminants to the bottom sediments of Lake Roosevelt appeared to first occurred in the region of Kettle Falls, about 53 miles downstream of the border. This was the only site where TCDD was detected in the sediments. The distribution of TCDD and TCDF in whole fish samples from Lake Roosevelt resembled the pattern in the sediments. A consistent ratio in TCDF concentrations was observed between fish and sediments throughout the study area, averaging 0.07 on a lipid/TOC normalized basis (their Bioavailability Index). This ratio may have predictive value for SQGs in Lake Roosevelt or impacts to fisheries from process changes at Celgar." The authors also compared the PCB and DDE fish tissue concentrations to the results of the USFWS and EPA national fish monitoring programs and have concluded that the results from the study were moderate to very low in comparison.

Johnson, A., D. Serdar, and S. Magoon. 1991b. Polychlorinated dioxins and furans in Lake Roosevelt (Columbia River) sport fish, 1990. Publication No. 91-4. Washington State Department of Ecology, Olympia, WA.

The objective of this study was to estimate the mean concentrations of TCDD and TCDF in muscle tissue of major sport fish in Lake Roosevelt. The impetus for this survey was the detection, by EC and the British Columbia Ministry of Environment, of contaminated lake whitefish and mountain whitefish below the Celgar bleached kraft pulp mill in Castlegar, B.C., about 30 river miles upstream of Lake Roosevelt.

Methods. Muscle tissue samples of 12 walleye, 12 rainbow trout, 12 lake whitefish, 4 white sturgeon, 2 kokanee, and 2 burbot from two areas of Lake Roosevelt were collected. Each sample was a composite of 5 fish (4 fish were used in burbot composites). The two parts of the reservoir sampled included "upper" Lake Roosevelt from Northport to Kettle Falls (RMs 735 to 700), and "lower" Lake Roosevelt (RMs 637 to 600). Lake Rufus, downstream of Lake Roosevelt, was also sampled.

TCDD and TCDF compounds were analyzed by EPA Method 8290 (isotope-dilution, high resolution GC/MS). Lipid content was also analyzed.

Results. TCDD was detected in all samples of kokanee, lake whitefish, and white sturgeon, and in the majority of rainbow trout samples. TCDF was detected in all species, and were generally higher than TCDD concentrations. Mean 2,3,7,8-TCDD TEQ concentrations ranged from 0.3 pg/g in burbot in the upper reservoir to 17 pg/g in white sturgeon, also in the upper reservoir.

Conclusions. Concentrations of dioxins and furans in fish from Lake Roosevelt were compared to those provided by several national data sets. The authors concluded that TCDF in lake whitefish and white sturgeon was elevated relative to local and national data. TEQ concentrations in lake whitefish and white sturgeon from Lake Roosevelt were the highest that had been reported in the Columbia River at the time the report was published, and "ranked in the top 10% of a USEPA nationwide study". In general, differences in contamination among fish species was linked to lipid content and location. White sturgeon had the highest lipids and the highest TCDD and TCDF content. Rainbow trout were significantly elevated in TCDF in upper Lake Roosevelt as compared to the lower lake. The authors recommended wastewater treatment at the Celgar Pulp Company mill, establishment of a fish tissue monitoring program, and evaluation of adverse biological effects.

Johnson, A. and D. Serdar. 1991. Metals concentrations in Lake Roosevelt (Columbia River) largescale suckers. Memorandum to Carl Nuechterlein June, 21, 1991. Publication 91-e26. Washington State Department of Ecology, Olympia, WA.

Largescale suckers had been collected from Lake Roosevelt and Lower Arrow Lake, British Columbia (upstream of Trail) in September 1989 and kept frozen as part of a B.C. Ministry of Environment study. The fish were given to Ecology in 1990 to be analyzed for metals. Each metals was analyzed from a different tissue type. The muscle tissue from suckers collected at Lake Roosevelt was analyzed for mercury, bone tissue for lead, and liver tissue for cadmium; muscle tissue from suckers collected at Lower Arrow Lake was analyzed for mercury and bone tissue was analyzed for lead.

Results. Results from the Lake Roosevelt specimen for lead, mercury, and cadmium were 36.9, 1.59, and 10 mg/kg dw, respectively. Results from the Lower Arrow Lake specimen for lead and mercury were 0.35 and 1.17 mg/kg dw, respectively. The dw results converted to ww, assuming 70 percent moisture, are as follows. Results from the Lake Roosevelt specimen for lead, mercury, and cadmium were 11.1, 0.48, and 3 mg/kg ww, respectively. Results from the Lower Arrow Lake specimen for lead and mercury were 0.11 and 0.35 mg/kg ww, respectively.

Conclusions. The authors concluded that lead in bone samples from the Lake Roosevelt fish were particularly noteworthy in how much higher in concentration they were than the upstream samples; they were two orders of magnitude higher than samples from Lower Arrow Lake. In contrast, muscle tissue samples from Lake Roosevelt had only slightly higher mercury concentrations than from Lower Arrow Lake. The authors noted that the suckers were probably not a good indicator of mercury contamination because they are not predators. No inter-lake comparison was possible for cadmium because of the lack of samples from Lower Arrow Lake; however, the authors noted that the Lake Roosevelt results appeared elevated in comparison to other published literature data. No details were provided on collection methods.

> *Serdar, D., B. Yake, and J. Cubbage. 1994. Contaminant trends in Lake Roosevelt. Publication No. 94-185. Washington State Department of Ecology, Olympia, WA.*

The purpose of this study was to evaluate changes in pollutant loads to Lake Roosevelt over time in lake whitefish and largescale suckers; and thereby document the effects of pollution controls being implemented by Canadian industries related to contaminants from Celgar and Cominco.

Methods. The fish were collected from the UCR (from the U.S.-Canada border south to Kettle Falls) in 1992 and 1993. Largescale suckers were analyzed as whole fish and lake whitefish as muscle tissue and eggs. Chemical analytes included: dioxins, furans, cadmium, copper, lead, mercury, and zinc. Results were compared to prior studies and national averages.

Conclusions. The authors stated:

- "2,3,7,8-TCDF in Columbia River suspended particles and 2,3,7,8-TCDD and 2,3,7,8-TCDF in lake whitefish muscle have decreased significantly since 1990. These reductions appear to be largely due to modifications at the Celgar pulp mill in British Columbia."
- "The degree of metals contamination of Lake Roosevelt water and sediment is reflected in uptake by bottom-feeding fish species (largescale suckers). Average lead and cadmium concentrations in suckers from Northport are higher than any fish analyzed during a nationwide survey of metals in fish. This underscores the degree of contamination in this reach of the Columbia River."
- "Comparison of 1993 whole fish data with Ecology's 1986 data suggests no improvement over these seven years and should allow these data to serve as a representative "baseline" for judging future trends in metals contamination in the upper Columbia.

Munn, M.D., S.E. Cox, and C.J. Dean. 1995. Concentrations of mercury and other trace elements in walleye, smallmouth bass, and rainbow trout in Franklin D. Roosevelt Lake and the Upper Columbia River, Washington 1994. U.S. Geological Survey Open-File Report 95-195. U.S. Geological Survey, Tacoma, WA. 35 pp.

The purpose of this study was to determine the concentrations of mercury and other trace elements in sport fish in the Columbia River. Prior studies identified concerns about bioaccumulation of trace elements in sport fish in the Columbia River posing a risk to human and environmental health. The primary objectives of the study were to 1) determine the concentrations of total mercury, arsenic, cadmium, copper, lead, manganese, selenium, and zinc in fillets of walleye, smallmouth bass, and both native and net-pen rainbow trout; and, 2) determine the liver tissue concentrations of cadmium, copper, lead, and zinc in the same species as a point of comparison for future studies. Walleye, smallmouth bass, and rainbow trout were chosen for this study because of historically high concentrations of mercury (walleye) and popularity as sport fish (smallmouth bass and rainbow trout).

Methods. Composites of fish muscle tissue were collected in 1994 from three areas:

- Upper reach—Columbia River and Lake Roosevelt near Kettle Falls
- Mid reach—Lake Roosevelt and lower Spokane River
- Lower reach —Sanpoil River embayment.

Four size classes of walleye were collected: 10 to 13 in., 13 to 16 in., 16 to 19 in., and 19 to 22 in. A total of 34 walleye composites were collected, each composite consisting of 8 individual fillets from fish of the same size class. Individual fillets were also analyzed from the 13 to 16 in. size class. Smallmouth bass were sampled the same as walleye, but with only a single size class of 8 to 12 in. Rainbow trout were not sorted into size classes, but were analyzed as individuals. Fillet samples included the belly flap, and had the skin removed.

Results. Mercury concentrations ranged from 0.11 to 0.44 mg/kg, with the lowest concentrations reported from the 10 to 13 in. size class, and the highest in the concentrations in the 19 to 22 in. size class. Concentrations of mercury in smallmouth bass ranged from 0.16 to 0.62 mg/kg, native rainbow trout from 0.16 to 0.24 mg/kg, and net-pen rainbow trout from 0.11 to 0.16 mg/kg. All results were reported in wet weight.

Concentrations of other trace elements in walleye, smallmouth bass, and rainbow trout fillets included:

- Arsenic – below detection to 0.14 (est.) mg/kg
- Cadmium – below detection
- Copper – 0.27 to 0.68 mg/kg
- Lead – below detection (< 0.05) to 0.1 mg/kg (highest in rainbow trout)
- Manganese – 0.09 to 0.54 mg/kg
- Selenium – below detection to 0.39 mg/kg (highest in walleye)
- Zinc – 3.7 to 6.1 mg/kg (11 samples outside lab control limits) (highest in smallmouth bass).

Cadmium, copper, lead, and zinc in liver tissue included:

- Cadmium – 0.9 to 15.7 µg/g (highest in walleye and native rainbow trout)
- Copper – 10.3 to 140 µg/g (highest in native rainbow trout)
- Lead – < 0.03 to 10.9 µg/g (similar among species [< 2] except for the single high outlier for large walleye)
- Zinc – 64.6 to 622 µg/g (highest for large walleye).

Conclusions. The authors concluded that walleye fillets had a higher concentration of total mercury in larger fish; although, concentrations of trace elements generally were low. Lead and zinc followed the same pattern in showing highest concentrations in the livers of the largest walleye (but not for fillets). Concentrations of zinc may have been overestimated because 11 of 16 samples were noted by the lab as having spike sample recoveries associated with them outside of the lab control limits. The authors provided no discussion or general conclusions related to the fishery or environment other than a data summary.

> *MESL (MacDonald Environmental Sciences Ltd.). 1997. Lower Columbia River from Birchbank to the international border: Water quality assessment and recommended objectives. Technical report. Available at: http://www.env.gov.bc.ca/wat/wq/objectives/birchbank/Birchbanktech.pdf. Prepared for Environment Canada and British Columbia Ministry of Environment, Lands and Parks.*

This study was undertaken to evaluate the levels of contaminants in aquatic organisms of the Columbia River and to develop TROs that may be used in management decisions. TROs were developed to protect aquatic life, wildlife, and human health in waterbodies that are affected by anthropogenic activities. The TROs were set to protect the most sensitive use (i.e., aquatic life, wildlife, or human health) in the ecosystem. Data collected under the CRIEMP and other historical studies were used to assess contaminant residues in the tissues of algae and aquatic macrophytes, benthic invertebrates, and fish in the portion of the Columbia River from Birchbank to the U.S.-Canada border.

Methods. Data on the concentrations of contaminants in the tissues of algae and aquatic macrophytes, benthic macroinvertebrates, and fish in the Columbia River were compiled from previous studies. Beginning in 1976, tissue samples were collected from five fish species, including largescale sucker (*Catostomus macrocheilus*), mountain whitefish (*Prosopium willliamsoni*), rainbow trout (*Oncorhynchus mykiss*), peamouth chub (*Mylocheilus caurinus*), and kokanee (*Oncorhynchus nerka*). Further data were collected under other studies including the USEPA National Dioxins Study (in 1987) and a five-year study initiated by the Department of Fisheries and Oceans (in 1992) to provide data on organics contamination and fish health, respectively.

The TROs were developed for the Columbia River using a three-step process. First, bioaccumulative contaminants were identified based on the existing and potential sources of contamination in the lower Columbia River basin. Next, tissue residue guidelines were compiled from various sources for the protection of aquatic life, wildlife, and human health. Finally, the tissue residue guideline for the most sensitive receptor group was selected as the TRO for each substance. In some cases, TROs were developed using the CCME protocols.

Results. The results of contaminant residue monitoring in Columbia River fish showed that levels of arsenic and cadmium in fish tissue were below TROs. Copper levels in largescale sucker, rainbow trout, mountain whitefish, and peamouth chub/northern pikeminnow were 0.62 to 1.6 mg/kg ww, 0.59 to 1.2 mg/kg ww, 0.45 to 1.1 mg/kg ww, and 1 to 1.75 mg/kg ww, respectively. At the time, TROs had not been developed for these metals. Levels of both lead and mercury in fish tissues collected in the Columbia River exceeded TROs in most samples, with the highest levels of lead and mercury found in largescale suckers and northern pikeminnow, respectively. Samples of northern pikeminnow had levels of mercury (mean = 0.56 mg/kg ww) exceeding aquatic life, wildlife, and human health guidelines. In 1988, fish tissue samples were collected for the analysis of organic contaminants, primarily dioxins and furans. Results of the analyses showed detectable concentrations of both dioxins (specifically, 2,3,7,8-T_4CDD) and furans (primarily 2,3,7,8-T_4CDF). More recent analyses showed that levels of these contaminants in fishes of the Columbia River had declined from the earlier levels observed. The Department of Fisheries and Oceans completed a study in 1992 to assess contaminant residue concentrations on fish health. Stress-related abnormalities observed, including altered histology of the liver, kidney and pyloric caeca, were attributed to poor water quality in the Columbia River. Follow-up studies showed significant improvements in fish health following improvements in the water quality of the Columbia River.

The TROs for the Columbia River from Birchbank to the U.S.-Canada border were determined for various metals (i.e., arsenic, cadmium, chromium, copper, lead, mercury, thallium and zinc) and for the organic contaminants, PCDD and PCDF. Of these TROs, *de novo* provisional objectives for the most sensitive receptor group were developed for arsenic, chromium, and lead. Insufficient information was available to determine TROs for copper, thallium, and zinc.

Conclusions. The synthesis of contaminant residues in tissues of aquatic organisms from the Columbia River showed that elevated levels of various metals and organic contaminants were present in algae and aquatic macrophytes, benthic macroinvertebrates, and fish, with the highest concentrations observed downstream of industrial activities and discharges. Improvements in water quality were observed between 1976 and 1996, resulting in declining levels of contaminants in the tissues of benthic macroinvertebrates and fish of the Columbia River. The TROs were developed for various metals (arsenic, cadmium, chromium, lead, and mercury) for the Columbia River. The authors recommended the development of tissue residue guidelines for copper, thallium and zinc to address the lack of relevant information for these variables.

> *EVS. 1998. Assessment of dioxins, furans, and PCBs in fish tissue from Lake Roosevelt, Washington, 1994. Final Report. December. EVS Environmental Consultants, Inc., Seattle, WA.*

In 1994, EPA initiated a study to measure concentrations of dioxins, furans, and PCBs in fillet tissue of kokanee, rainbow trout (wild and hatchery-raised), smallmouth bass, walleye, lake whitefish, and white sturgeon. The primary objective of the study was to collect information to evaluate the potential human health risks associated with these organochlorines, and therefore targeted the first four of these fish species (because they were the most common in creels at the time preceding this study). Additionally, white sturgeon were included because of their longevity, feeding characteristics, and lipid content and lake whitefish were included because of the availability of historical data on this species preceding this study. Other objectives were to compare tissue concentrations between different geographic areas, between size classes, between composites and individual, and for comparison with historical data for whitefish. Samples were primarily for fillet with skin, but there were whitefish samples without skin, and the white sturgeon samples included only muscle tissue. All samples were within fixed size categories. Both composites (of eight fish each) and individual fish fillets were analyzed. Fish were collected from four areas spanning the UCR (at Northport, at the mouth of the Colville River, in the Seven Bays area, and near the Grand Coulee Dam), and in the Sanpoil Arm using gill nets, electrofishing, and angling.

Results and Conclusions. Dioxins and furans were detected in all of the fish species evaluated, and 2,3,7,8-TCDF was the most commonly detected congener among the dioxins/furans found in UCR fish. More dioxin and furan congeners were detected in hatchery rainbow than in wild rainbow. 2,3,7,8-TCDD TEQ were also calculated, and used in comparisons. The highest TEQ concentrations were measured in white sturgeon and lake whitefish. Statistical comparisons showed no significant differences in TEQ concentrations among fish of the same species from the three geographic areas of the lake.

The authors also reported PCB concentrations in tissues from kokanee, lake whitefish, rainbow trout, smallmouth bass, walleye, and white sturgeon. PCB concentrations in wild rainbow trout fillets were higher in the upper reach of the UCR, near Northport (mean total PCB concentration = 88 µg/kg ww) than in hatchery rainbow in parts of the lower reservoir (mean total PCB concentration = 22 µg/kg ww). The authors concluded that mean concentrations of 2,3,7,8-TCDF in lake whitefish declined either 7-fold (on a wet-weight basis) or 34-fold (when normalized for lipid content) from 1990 to 1994. These differences were highly significant ($p \leq 0.01$; Spearman's rank correlation coefficient). There were substantial declines in tissue concentrations of TCDF and TCDD in kokanee, rainbow trout, walleye, and white sturgeon since 1990 in the study area.

> *Munn, M.D. 2000. Contaminant trends in sport fish from Lake Roosevelt and the Upper Columbia River, Washington, 1994-1998. Report 00-4024. U.S. Geological Survey, Water Resources Division, Tacoma, WA.*

The objective of this study was to collect and analyze fish tissue data to compare to and follow-up on prior fish tissue studies in the Columbia River area. Studies in the 1980s were the first to report that concentrations of certain contaminants in fish tissue from the Columbia River that posed a risk to human health. This study was to determine if the concentrations of mercury, PCBs, dioxins, and furans had changed in fish tissue from previous work (specifically EVS [1998] and USGS [1995]). Species collected, locations of sampling, and chemical analysis were chosen based on the past studies to allow comparisons across time periods.

Methods. The sampling locations were:

- Upper reach—Northport south to Kettle Falls
- Lower reach —Spokane River west to Grand Coulee Dam.

Muscle tissue samples from walleye, wild and net-pen rainbow trout, and mountain whitefish were collected using electrofishing and gill nets. Total length (cm) and total weight (g) were recorded for each fish. Individual fillet samples were analyzed using standard procedures; samples included the belly flap in most cases. For mercury samples, skin was removed on the individual walleye fillets. The skin was left on the muscle tissue samples for all other chemicals and species analyzed.

Rainbow trout and mountain whitefish were analyzed for dioxins and furans (EPA Method 1613B for 17 dioxin and furan congeners). Rainbow trout were analyzed for PCB Aroclors (EPA Method 8082) and a small subset of samples analyzed for 13 individual dioxin-like PCB congeners (EPA Method 1668). All chemical concentrations were reported as wet weights. Standard quality assurance and quality control procedures were used for all laboratory analysis and resulted in all data meeting quality criteria.

Results and Conclusions. The authors concluded that concentrations of contaminants in fish that were identified as a potential threat to human health had either not changed since the 1994 studies, or had decreased, based on non-parametric statistical comparisons. More specifically:

- Total mercury concentrations in walleye decreased by about 50 percent from 1994 to 1998, although concentrations in walleye remained greater than in other species.
- Dioxins and furans, as indicated by 2,3,7,8-TCDF, decreased significantly in rainbow trout fillets from 1994 to 1998. However, there was no apparent change in the average 2,3,7,8-TCDF concentrations in mountain whitefish. Average concentrations of 2,3,7,8-TCDF were higher in mountain whitefish than in rainbow trout.
- Rainbow trout from the upper reach had a higher TEQ concentration than rainbow trout from the lower reach, with trout from the upper reach having a higher percentage of the toxicity from dioxin-like PCBs than dioxin and furan compounds.
- PCB concentrations in rainbow trout (both wild and pen) remained elevated and not significantly changed, although the results were highly variable.
- Decreases in some contaminants could have been a function of reductions in industrial loadings to the Columbia River and/or changes in reservoir management practices (as might affect mercury methylation).

- The toxic equivalent concentrations (TEC) of the organics showed a spatial pattern, with higher TEC for trout in the Northport region as compared to more downstream sites.

> *USEPA. 2002b. Columbia River Basin Fish Contaminant Survey; 1996 – 1998. EPA 910/R-02-006. July 2002.*

This study was designed to estimate risks to Columbia River Intertribal Fish Commission's member tribes and to quantify differences in contamination among fish species and study sites.

Methods. The study area was confined to the Columbia Basin below Grand Coulee dam to the north, the Clearwater River to the east, just below Bonneville Dam to the west and the Willamette River to the south, six rivers and creeks and the mainstream Columbia. Fish were collected in 1996 and 1998 using electrofishing or various types of netting. Cancer and non-cancer human health risk models were developed to evaluate risk to consumers as compared to the results.

Results. A total of 281 samples of fish and fish eggs were collected from the Columbia River Basin. The fish species included five anadromous species (Pacific lamprey, smelt, coho salmon, fall and spring chinook salmon, and steelhead) and six resident species (largescale sucker, bridgelip sucker, mountain whitefish, rainbow trout, white sturgeon, walleye). Four types of samples were collected: whole-body with scales, fillet with skin and scales, fillet without skin (white sturgeon only), and eggs. The type of tissue tested (whole body, fillet, egg) varied with species and sample location.

The fish tissues were analyzed for metals, pesticides, PCBs, dioxins, furans, and other organics. DDE, PCBs, zinc, and aluminum were detected in the highest concentrations in most of the tissues, with resident fish having higher concentrations than anadromous species for organics but no such distinct differences for metals. The Hanford Reach of the Columbia tended to have the highest organic compounds in fish tissues. PCBs and mercury were the major contributors to human risk, along with DDT compounds for some resident fish. Lead concentrations did not exceed levels of concern for human consumption guidelines. For metals, largescale sucker had the highest concentrations of aluminum, barium, manganese, mercury, and vanadium. White sturgeon had the highest concentrations of beryllium, chromium, cobalt, and selenium. Most metals were higher in whole body than fillet concentrations. The authors concluded that the greatest risks from fish consumption was from persistent bioaccumulative compounds (DDTs, PCBs, dioxins, furans) as well as some naturally occurring metals (arsenic, mercury).

> *Fischnaller S., P. Anderson, and D. Norton. 2003. Mercury in Edible Fish Tissue and Sediment from Selected Lakes and Rivers of Washington State. Washington State Department of Ecology. Publication No. 03-03-026.*

During 2001 and 2002, Ecology conducted a screening survey for mercury concentrations in fish tissue and sediments from selected lakes and rivers across Washington State.

Methods. Mercury was analyzed from 185 fish and sediment collections from 20 sites across Washington collected in 2001 and 2002. Several sites were considered part of the Columbia River Basin, but the Columbia River itself was not sampled. Largemouth bass and smallmouth bass were the target fish species.

Results. Tissue mercury concentrations varied widely by location and among individual fish within water bodies. The highest concentrations were in western Washington. Mercury concentrations varied positively with fish age, weight, and length. Twenty-three percent of the fish tissue mercury concentrations exceeded the EPA fish tissue residual criterion (TRC) of 300 μg mercury/kg ww.

> *Hinck, J.E., C.J. Schmitt, T.M. Bartish, N.D. Denslow, V.S. Blazer, and P.J. Anderson. 2004. Biomonitoring of Environmental Status and Trends (BEST) Program: Environmental contaminants and their effects on fish in the Columbia River Basin. Scientific Investigations Report 2004 – 5154. U.S. Geological Survey, Washington, D.C.*
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> *Hinck, J.E., C.J. Schmitt, V.S. Blazer, N.D. Denslow, T.M. Bartish, P.J. Anderson, J.J. Coyle, G.M. Dethloff, and D.E. Tillitt. 2006. Environmental contaminants and biomarker responses in fish from the Columbia River and its tributaries: Spatial and temporal trends. Sci. Tot. Environ. 366:549–578.*

The primary objective of this study was to document and assess spatial and temporal trends in the concentrations of environmental contaminants and their effects on fish throughout the Columbia River Basin. Secondary objectives were to compare results from the Columbia River Basin to other U.S. river systems, to further define benchmarks for the quantification of long-term trends, and interpretation of biomarker results.

Methods. Fish were collected at sixteen sites in the Columbia River Basin (CRB). Eight of the 16 sites were located on the Columbia River, two were on the Willamette River in western Oregon, three were on the Snake River in Idaho and Washington, and one site each were on the Yakima River in Washington, Salmon River in Idaho, and Flathead River in Montana. Ten sites were NCBP stations where contaminants in fish were monitored from the late 1960s to the mid-1980s. These sites were selected to ensure spatial and temporal continuity with historical data and to facilitate trend analysis. Five stations were National Stream Quality Accounting Network (NASQAN) sites. Upper Columbia sites were Northport and Grand Coulee, WA. Most fish were collected between early September and November 1997. Carp and largemouth bass were the preferred taxa at all sites due to prevalence, distribution, and extant contaminant and biological endpoint data. Hinck et al. (2004) sampled largescale sucker, walleye and rainbow trout in the UCR as whole body, single-gender composites of 2 to 10 individuals per composite. Data for metals, pesticides and PCBs in these UCR fish were generated.

A suite of chemical and biological methods was used to characterize the exposure of fish to chemicals including reproductive biomarkers, measures of cytochrome P450 enzyme induction and concentrations of chemicals in whole fish. Measures of potential effects of chemical exposures included fish health assessments; measures of fish health included 1) gross abnormalities; 2) condition factor (CF); hepatosomatic index (HSI); splenosomatic index (SSI); 3) histopathology; and 4) several measures of reproductive condition.

Conclusions. The authors provided the following conclusions about chemical concentrations in fish of the Columbia Basin:

- Overall fish from the middle Columbia River and the lower Columbia River had higher concentrations of organochlorine contaminants than fish from the UCR
- Where historical data were available, concentrations of PCBs declined in fish at all sites. This was not the case for p,p′-DDE, which remained consistent from 1967 to 1997.
- Except for mercury, selenium, and lead, concentrations of metals were relatively low and stable or declining relative to historical levels at most sites.
- Concentrations of PCBs and TEQs were low in most samples, but ethoxyresorufin-o-deethylase (EROD) rates in bass, carp, and largescale sucker exceeded threshold levels reported in 1995.
- Concentrations of mercury in the Columbia River Basin accumulated more in bass than in carp and largescale sucker, as reported in other studies.
- Carp and sucker had greater concentrations of cadmium, copper, chromium, and nickel compared to bass, and concentrations of zinc in carp were consistently five times higher than in other species.
- Concentrations of pesticides were similar among bass, carp, and sucker.
- Pesticide concentrations were greatest in fish from lower Columbia River Basin sites and elemental concentrations were greatest in fish from upper Columbia River Basin sites and these patterns reflected land uses.
- Lead and mercury concentrations in fish from the Columbia River at Northport and Grand Coulee, Washington, exceeded fish and wildlife toxicity thresholds (>0.4 and >0.1 μg/g, respectively). Lead concentrations in fish tissues from Northport were the highest found in the study.
- Mercury concentrations in fish were elevated throughout the basin but were greatest (>0.4 μg/g) in predatory fish from the Salmon River at Riggins, Idaho,

the Yakima River at Granger, Washington, and the Columbia River at Warrendale, Oregon.

- Other organochlorine pesticides did not exceed toxicity thresholds in fish or were detected infrequently.
- Total polychlorinated biphenyls (PCBs >0.11 μg/g) and 2,3,7,8-TCDD equivalents (>5 pg/g) exceeded wildlife guidelines in fish from the middle and lower Columbia River Basin. Concentrations of PCBs also exceeded criteria protective of wildlife at Northport.
- Temporal trend analysis indicated decreasing or stable concentrations of lead, selenium, mercury, p,p′-DDE, and PCBs at most sites where historical data were available. Lead and zinc showed decreases at the Northport station from the 1970s to the 1980s.
- A total of 74 percent of all fish sampled throughout the Columbia River Basin had some type of external anomaly, and 50 percent or more of fish had external anomalies at any given station. Many largescale sucker from the Columbia River at Northport and Grand Coulee, Washington had external lesions and enlarged spleens, both signs of compromised immune systems. Northport largescale suckers had the second highest spleenosomatic index in the entire Columbia River Basin. The majority of external and internal lesions observed were the result of inflammatory responses to parasitic or bacterial infections.

The authors also concluded that results from this study and other investigations indicated that continued monitoring in the Columbia River Basin was warranted to identify consistently degraded sites and those with emerging problems.

> *Johnson, A., K. Seiders, C. Deligeannis, K. Kinney, P. Sandvik, B. Era-Miller, and D. Alkire. 2006. PBDE Flame retardants in Washington rivers and lakes: Concentrations in fish and water 2005-06. Publication No. 06-03-027. Washington State Department of Ecology, Olympia, WA.*

Ecology analyzed PBDE flame retardants in freshwater fish and water samples collected statewide during 2005-2006. This was done in response to increasing PBDE levels in the environment and concern about potential adverse human health effects from fish consumption. The goal of this study was to establish baseline conditions that could be used to evaluate the effectiveness of the Washington State PBDE Chemical Action Plan and other efforts to reduce PBDE inputs to the environment. Freshwater fish and water samples were collected statewide to identify spatial, temporal and between species patterns in the environmental distribution of PBDEs.

Methods. Freshwater fish fillets were collected from 20 waterbodies statewide between June and November, 2005. The study focused primarily on waterbodies that drained large areas and are a significant fisheries resource. The 20 waterbodies were composed of ten rivers and ten lakes spread evenly across the state with primary focus on the

Columbia River system, due to its size and importance, but also due to reported increases in PBDE concentrations in fish in the Upper Columbia River.

Three different species of game fish were attempted to be collected from each site with four to five individual fish for each composite. The Upper Columbia River sample station was located in the Marcus Flats reach with largescale sucker, rainbow trout, walleye and lake whitefish collected and analyzed. Fish were collected by electro-shocking, gill net, dip net, or hook and line. Data on length, weight, age, and sex of each fish were recorded. The latitude and longitude of the sampling sites were recorded from a GPS receiver or taken from USGS quad maps. The fish tissue samples were analyzed following Manchester Laboratory's standard operating procedure for PBDEs. The samples were extracted with methylene chloride/acetone by EPA SW-846 Method 3540 then solvent exchanged to iso-octane. The extracts were analyzed by capillary gas chromatography with mass spectrometry detection (GC/MS) following EPA SW-846 Method 8270.

Results. Data were obtained on concentrations of PBDE-47, -49, -66, -71, -99, -100, -138, -153, -154, -183, -184, -190, and -209 in approximately 120 fish fillet samples and 23 whole fish samples, representing 32 waterbodies. Fish collected from the Upper Columbia River had the second highest PBDE concentrations among the 20 stations with on fish tissue collected from the Long Lake portion of the Spokane River being higher. Among the four species of fish collected, lake whitefish had the highest concentrations of total PBDEs (18 µg/kg, wet weight) while rainbow trout had the lowest concentrations of total PBDEs (0.92 µg/kg, wet weight). The most frequently detected congener was PBDE-47, which was detected in all four fish species collected from the Upper Columbia River. Congener PBDE-47 also contributed the most to the total concentration of PBDEs in fish tissue.

Conclusions. The authors stated: "The recent data collected on fillets from Washington freshwater fish indicate that total PBDE concentrations are less than 10 µg/kg (ppb) in most rivers and lakes. Certain fish species from several large water bodies – Palouse River, Columbia River, Lake Washington, Snohomish River, Cowlitz River, and Snake River – have total PBDE concentrations in the 10 to 200 µg/kg range. Fish in watersheds with minimal human disturbance (e.g., the Queets River and Lake Ozette) have PBDE concentrations at or below the limit of detection (0.2 to 0.5 µg/kg in the present study). Rivers clearly have higher PBDE levels than lakes. High PBDE levels are found throughout the Spokane River, exceeding 1,000 µg/kg in some cases." Total PBDE concentrations found in fish tissues from the Upper Columbia River were within the range reported from an EPA national study of 500 lakes nationwide. The authors noted, however, that at the time of this study there were no regulatory, human health, or natural resource agencies in the United States that had established levels of PBDEs that may adversely affect human health and the environment.

> *USEPA. 2007b. Phase I fish tissue sampling data evaluation report, Upper Columbia River site CERCLA RI/FS. Prepared by CH2M HILL. U.S. Environmental Protection Agency, Region 10, Seattle, WA. October 30, 2007.*

EPA collected and analyzed fish tissues from six locations throughout the Site in 2005, and issued a summary of the results (USEPA 2007b). Tissue samples from six species (burbot, largescale sucker, rainbow trout, lake whitefish, mountain whitefish, and walleye) consisting of both fillet and whole body samples, primarily as composites of five fish each, were collected and analyzed for a selected list of chemicals. Fish also were observed for the presence of external lesions.

Methods. Fish were collected in October, 2005, by a variety of methods (gill nets, line fishing, burbot traps) from six Fish Sample Collection Areas (FSCAs) within the Site. Each FSCA was in a separate reach of the river and represented an area sufficiently large to catch the required number of fish. The size range of fish collected in this study approximately bracketed a mean size determined from UCR creel census data and/or reports of mean size from scientific collections. The study included chemical analysis of whole body fish from six fish species (walleye, rainbow trout, lake whitefish, mountain whitefish, largescale sucker, and burbot) and both whole body and fillet tissue from two species (walleye and rainbow trout). For fish from which fillets were analyzed, the offal (the remainder of the fish after removing fillets) was also analyzed to facilitate estimation of whole body concentrations. Five fish from the same species were composited in each sample, and four to five samples were collected within each FSCA. Tissues were analyzed for 23 metals, total mercury, PCB, Aroclors, and PCDDs, and PCDFs. One composite sample of each species from each collection area was analyzed for PCB congeners and approximately 10 percent of all samples were analyzed for inorganic arsenic and organic arsenic species. The occurrence and types of external lesions observed on fish were recorded prior to processing fish for chemical analysis. Tissue anomalies recorded included lesions, deformities, abnormalities, fin erosion, and visible external parasites. Examination of fish for external lesions followed the protocol described by Smith et al. (2002).

Results. For most metals, the results for all samples analyzed were greater than the detection limit and greater than the human health or ecological comparison values (CVs). The following metals and organics were classified as preliminary contaminants of interest based on tissue chemistry results: aluminum, arsenic, cadmium, chromium, copper, iron, lead, nickel, selenium, uranium, zinc, mercury, total PCBs, and 2,3,7,8-TCDF.

Silver and beryllium were reported as non-detected for all samples of each species. Antimony was detected in two of the four composite samples of largescale suckers at the most upstream collection area (FSCA 1). Thallium was detected only in walleye fillets from the collection area nearest Grand Coulee Dam (FSCA 6), while concentrations were lower than the detection limit in many other tissue/areas. Uranium and vanadium results were mostly reported as non-detected in some fishes and/or tissue types, and variation was limited in samples where these compounds were detected. Lead concentrations in whole-body samples of largescale suckers were more than 10 times greater than that of all other species of fish sampled in each collection area. Largescale suckers also had the greatest concentrations of cadmium, chromium, cobalt, manganese, and nickel at every site. Burbot had total arsenic concentrations two to three times greater than other species throughout the study area. For walleye and rainbow trout, metal concentrations in fillets were lower than in whole body samples from the same location. The exception was for mercury, for which concentrations in fillets were generally greater than concentrations in whole bodies.

TCDD was detected in whole body tissues at a frequency of nine percent. Other dioxins/furans were detected between 0 and 73 percent in UCR fish tissues. Aroclors 1254 and 1260 were summed and were detected in all species and tissues. All other Aroclors (except Aroclor 1016) were never detected. Most congeners were detected at a frequency greater than 10 percent. Whole-body (ww) TCDF concentrations were highest in lake whitefish, followed by burbot and largescale suckers. Lipid-normalized TCDF concentrations were highest in burbot compared to other species. Fillet concentrations of TCDF were higher in rainbow trout than walleye, and hatchery and wild rainbow trout concentrations were comparable. Walleye fillet tissues had higher TCDF concentrations per lipid content than rainbow trout fillets. Aroclor 1254/1260 wet-weight concentrations in whole body tissues were highest in largescale suckers. Burbot, walleye, and largescale suckers had higher Aroclor concentrations per lipid content than other fish species. Rainbow trout fillet tissues contained higher concentrations of Aroclors than walleye, and wild rainbow trout concentrations were higher than hatchery concentrations. Walleye fillets had higher Aroclor concentrations per lipid content than rainbow trout. Spatial variation of metal concentrations among collection areas in the UCR was common. For largescale suckers, the species with the most spatial variation in concentrations, most metal concentrations were greater at upstream sites, with some exceptions for mercury, selenium, and arsenic. Copper, lead, and zinc in largescale sucker whole body samples (including gut contents) declined with distance downstream from the U.S.-Canada border. Largescale suckers, walleye, burbot, and to some degree rainbow trout consistently showed the highest spatial variability in metals, and the authors hypothesized that some of that may have been due to sediment and slag particles in the gut. For walleye, there was an increase in fillet concentrations, moving downstream, for mercury, arsenic, and lead, with no apparent pattern for other metals. Trout showed no such spatial trends.

Spatial differences within species and among sites for organic chemicals did not indicate a consistent trend. Differences in concentrations of organics among FSCAs were variable and did not constitute a significant declining or increasing trend when comparing upstream versus downstream collection areas.

EPA recorded results of external examinations of individual fish for all fish that were used in the composite samples, plus a random selection of additional fish that were available; selection of fish was not dependent upon whether or not external anomalies were apparent. Lesions were counted individually, but in many cases more than one lesion occurred on a single fish. The percent of all fish examined in each FSCA that had external anomalies was highest in FSCA 5 at 81 percent. When the percent of anomalies was considered by species, the maximum for each species was also in FSCA 5, with the exception of lake whitefish. For all species combined, the average number of lesions per fish (within species) generally increased moving downstream.

In comparing to past USGS and Ecology data, these study results indicated that mercury had been declining in walleye and rainbow trout (both wild and hatchery); 2,3,7,8 TCDF continued to decline in lake whitefish; metals were unchanged in walleye and rainbow trout fillets; and, lead in largescale sucker from Northport was lower in the current study (although significantly higher than at any other location).

Conclusions. The authors suggested that their results supported the preliminary CSM and the assumption that UCR fish are exposed via surface water (i.e., surface water and suspended particulates), sediment, and diet. These results indicated that exposure varies with species and location within the reservoir. The authors also recommended that additional data should be collected to support the evaluation of human health and ecological risk including: additional sample locations, additional target species (white sturgeon, in particular), expanded fish sizes, sampling individual fish, an expanded analyte list (including PCBs and arsenic speciation), further investigation of the potential effects of gut contents on largescale sucker whole body measurements, and measurements of temporal trends in fish tissue concentrations.

> *Washington State Department of Ecology, Measuring Mercury Trends in Freshwater Fish in Washington State. 2005-2007 (Furl 2007[2006 data]; Furl et al. 2007[2005 data]; Furl and Meredith 2008[2007 data]).*
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> > *Furl, C. 2007. Measuring Mercury Trends in Freshwater Fish in Washington State: 2006 Sampling Results. Washington State Department of Ecology. Publication No. 07-03-043.*
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> > *Furl, C. and C. Meredith 2008. Measuring Mercury Trends in Freshwater Fish in Washington State: 2007 Sampling Results. Washington State Department of Ecology. Publication No. 08-03-027.*
> >
> > *Furl, C., K. Seiders, D. Alkire, and C. Deligeannis. 2007. Measuring Mercury Trends in Freshwater Fish in Washington State: 2005 Sampling Results. Washington State Department of Ecology. Publication No. 07-03-007.*

The objective of this study was to monitor mercury levels in edible tissue from freshwater fish at six sites per year for five years (30 sampling events total) to characterize temporal and spatial trends in fish tissue mercury levels.

Methods. In 2005, largemouth and smallmouth bass were collected from six sites in Washington. In 2006, 85 individual fish and 30 fish composite samples, representing 10 species, were analyzed from seven lakes in Washington. In 2007, 60 individual largemouth bass and 32 composite samples representing 8 species were analyzed from six locations. Three lakes sampled in 2001-2002 were compared to 2007 results for a trend analysis over 5 years. The other data were compared among fish, between species, among sites, and among the larger, 3 year data set (Furl and Meredith, 2008).

Results. Higher mercury concentrations were found in western Washington bass as compared to those from other areas. There was a slight downward trend in concentration in one of the three lakes evaluated for long term trends. Seventy-three percent of individuals and 28 percent of composites exceeded the EPA fish TRC of 300 μg mercury/kg ww.

Teck Cominco Ecological Risk Assessment (EcoRA) and Related Studies (Antcliffe et al. 1997a, 1997b; Lewis 2000; Teck Cominco 2001; Golder 2003; Golder 2007).

> *Antcliffe, B.L., D. Kieser, J.A.J. Thompson, W.L. Lockhart, D.A. Metner and J.R. Roome. 1997a. Monitoring of mountain whitefish, Prosopium williamsoni, from the Columbia River system near Castlegar, British Columbia: Fish health assessment and contaminants in 1994. Canadian Technical Report of Fisheries and Aquatic Sciences. 2142; 101.*
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> *Antcliffe, B.L., D. Kieser, G. Lawrence, W.L. Lockhart, D.A. Metner and J.A.J. Thompson. 1997b. Monitoring of mountain whitefish, Prosopium williamsoni, from the Columbia River system near Castlegar, British Columbia: Fish health assessment and contaminants in 1996. Canadian Technical Report of Fisheries and Aquatic Sciences. 2184.*
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> *Lewis, B. 2000. Memorandum to Bonnie Antcliffe, Department of Fisheries and Oceans, regarding Mountain Whitefish Health Assessment Survey in March 1999. November 3, 2000.*
>
> *Golder Associates. 2003. Aquatic Problem Formulation. Prepared for Teck Cominco.*
>
> *Golder Associates. 2007. Teck Cominco Aquatic Ecological Risk Assessment 2004 Fish Health Study. Prepared for Teck Cominco.*
>
> *Teck Cominco Ltd. 2001. Assessment of Columbia River Receiving Waters – Final Report. G3 Consulting Ltd. December, 2001.*

Teck Cominco and the Department of Fisheries and Oceans (DFO) have conducted fish tissue collection as part of monitoring efforts for smelter improvements and an ERA. These studies include separate reports on the Problem Formulation portion of the EcoRA, a fish health study, and the results of the tissue chemistry monitoring efforts.

Methods. Fish were collected from a 56-km area encompassing the Columbia River basin upstream of the U.S.-Canada border. Whole body and muscle tissue samples were analyzed.

Results. Mountain whitefish were collected by the DFO in 1994, 1995, 1996 and 1999. In 1995 and 1999, the DFO also collected rainbow trout and walleye. In 2001, sampling consisted of large-scale sucker, mountain whitefish, rainbow trout and walleye. In 2004, mountain whitefish and prickly sculpin were collected from sites located upstream and downstream of the smelter effluent outfalls.

Most fish tissue concentrations showed decreasing concentrations between 1994 and 1999, with some individual mountain whitefish showing criteria exceedances for lead and mercury (although average concentrations of all metals were below criteria).

The study of fish health on mountain whitefish and prickly sculpin revealed equivocal results, with whitefish showing higher metals bioaccumulation above the smelter versus downstream, whereas the sculpin showed elevated bioaccumulation at downstream sites. Neither fish species exhibited excessive abnormalities or links to upstream/downstream locations. Overall indicators of health of mountain whitefish showed improvement over the period of the mid to late 1990s.

In terms of the EcoRA, potentially unacceptable risk was identified to fish in the main stem UCR from cadmium, copper, lead, and zinc. In addition to these potentially unacceptable risks, arsenic, iron, mercury, selenium, and silver are potentially unacceptable risks to fish in the UCR tributary streams. Bioaccumulated tissue concentrations of cadmium and copper may pose risks to fish, as well.

3.1.6 White Sturgeon

Summaries of white sturgeon data in reports and publications are summarized by reporting authors or organization in chronological order. Studies were performed to meet a variety of objectives and may not be representative of all Site conditions. Further evaluation of select data from these studies is provided in Appendix F.

> *Lapirova T.B., V.R. Mikryakova, A.S. Mavrin, and G.A. Vinogradova. 2000. Effect of sublethal concentrations of mercury, cadmium, and copper salts on the lysozyme content in fry of the Lena River Sturgeon Acipenser baeri. J. Evol. Biochem. Physiol. 36:47-51.*

This study investigated the sublethal effects of metals on juvenile sturgeon over a 30 to 60-day period.

Methods. The effects of copper, cadmium, and mercury on 2-month-old Siberian sturgeon (*Acipenser baeri*) were evaluated by examining changes in lysozyme content of individual body tissues (spleen, liver, and heart).

Results and Conclusions. The authors used one tenth of measured 96 hr. LC50s as the dosages used for the sublethal experiments (0.03 mg/L for cadmium; 0.015 mg/L for copper; and 0.003 mg/L for mercury) for a 30- to 60-day study. The study revealed an enzymatic response to these low levels of metals, with significant differences lasting up to 30 days, and with spleen and mercury showing the greatest effects, but all three metal/organ combinations showing responses. The authors noted that both increases and decreases of lysozyme could be expected with differing stresses; the responses measured in this study were nonspecific. The authors showed the greatest number of significant responses from exposure to mercury and copper as compared to cadmium, which is contrary to observations made for other fish species (e.g., Besser et al. 2007). The authors also noted that there are a number of uncertainties that limit the use of data from this study. The applicability of the results to other sturgeon species is uncertain because different sturgeon species can differ substantially in their sensitivity to metals such as copper (Dwyer et al. 2005). In addition, because this study tested 2-month-old juveniles, there are uncertainties in the applicability of these data to the sensitivity of earlier life-stages (embryo and fry) to the metals.

> *Kruse, G.O. 2000. The effects of contaminants on reproduction, embryo development and related physiological processes in Kootenai River white sturgeon, Acipenser transmontanus richardson. Masters Thesis. University of Idaho – Moscow.*

"This study used biomarkers to evaluate the effects environmental levels of organochlorine, organophosphate, organonitrate, and carbamate pesticides, PCBs, and metals in the aquatic system on Kootenai River white sturgeon, *Acipenser transmontanus*." The three objectives of this study were 1) determine if bioaccumulated contaminants affect reproductive processes in adult sturgeon; 2) determine contaminant uptake in developing embryos exposed to contaminants in Kootenai River sediment, suspended solids and water; and 3) assess contaminant residue bioaccumulation and its effects on physiological processes in juvenile sturgeon.

Methods. To study the first objective fifteen grams of ovarian tissue samples were collected from 34 adult female sturgeon from river kilometers (RKM) 215 to 120 on the Kootenai River. The fish are reported to have ranged in ages 18 to 52 years old. Surface water samples and river-bottom substrate samples were collected at eight sites between river kilometers 205 and 244.5. Water samples were drawn within 1.5 meters of the surface using a LaMotte sampler while a Ponar dredge was used to collect sediment. In addition, eggs were collected from spawning female sturgeon. Both the ovarian tissue and eggs were tested for the above listed contaminants. Plasma steroids were also collected from 46 sturgeon (19 mature males, 22 mature females and 5 immature and non-reproductive sturgeon of unknown sex).

To study the second objective fertilized white sturgeon eggs were incubated in three groups consisting of exposure to surface sediment, suspended solids and unfiltered river water collected from the Kootenai River. The surface sediment and suspended solids were collected from sturgeon spawning areas in the Kootenai River (RKM 229-230) using a Ponar dredge and a 80-μm mesh plankton net respectively.

To study the third objective 25 4-year-old hatchery-reared juvenile white sturgeon were collected between rkm 203.5 and 234.4 on the Kootenai River. Blood was drawn from each of the juvenile white sturgeon in addition to an additional 10 blood samples drawn from wild adult sturgeon and one duplicate set from a wild juvenile. The liver and brain tissues from the hatchery-raised juvenile white sturgeon were collected and the remaining carcass analyzed as a whole-body tissue.

Results and Conclusions. The author concluded that "Results from chemical residue analysis indicated that copper, zinc, iron, and the PCB Aroclor 1260 were at levels that could adversely affect sturgeon reproduction as well as other aquatic organisms and overall system productivity. Plasma steroid concentrations in Kootenai River sturgeon were comparable to those reported for other species of sturgeon. However, the significant negative correlations between testosterone production and bioaccumulated Aroclor 1260 (Spearman; $r = -0.729$, $r = -0.820$), total organochlorine compounds (Spearman; $r = -0.753$) and zinc (Spearman; $r = -0.652$) suggest that males may experience decreased sperm production if they have bioaccumulated these contaminants at levels similar to those found in females. The significant positive correlation between the female hormone estradiol and DDT (Spearman; $r = 0.893$) also suggests potential feminization of male sturgeon that bioaccumulate DDT levels similar to those found in females. Zinc residues in sturgeon ovarian tissue were significantly (Mann-Whitney U test; $P < 0.05$) higher than in samples taken between 1989 and 1991. River bottom sediments were found to be a significant source of metal and PCB exposure for incubating white sturgeon embryos. Environmental levels of copper and PCB Aroclor 1260 in the rearing media were associated with increased mortality (Spearman; $r = 0.568$) and decreased incubation time of sturgeon embryos. Results from liver histology, cholinesterase and deoxyribonucleic acid (DNA) analyses in juveniles indicated that although juvenile sturgeon were experiencing low-level contaminant exposure, physiological functions were not limited or altered. The biomarkers used in this study indicate effects and are not a measure of cause. Therefore, it was concluded that embryonic, juvenile and adult life stages are potentially experiencing sublethal effects from contaminants in the Kootenai River. The embryonic life stage appears to be the most susceptible to the effects of these contaminants."

> *Kruse, G.O. and D.L. Scarneccia. 2002a. Assessment of bioaccumulated metal and organochlorine compounds in relation to physiological biomarkers in Kootenai River white sturgeon. J. Appl. Ichthyol. 18: 430–438.*

This field study was conducted in the Kootenay River, B.C., to assess the potential effect of environmental mixtures of metals and organochlorine chemicals on white sturgeon.

Methods. Concentrations of selected chemicals were measured in different tissues (ovary, testis, juvenile whole body) of wild caught white sturgeon and related to a series of biological endpoints including plasma sex steroid production, plasma vitellogenin concentrations, egg size/production, DNA chromosomal variability, liver histology, and acetylcholinesterase (AChE) activities.

Results. The authors found correlations between some individual contaminants measured in tissues and selected biological endpoints such as plasma sex steroid concentrations, chromosomal DNA content, and AChE activities. However, most of these correlations were strongly influenced by individual outliers. Clear trends (before or after removal of outliers) were the positive relationship between egg numbers produced and selenium concentrations and the negative correlation between blood butyryl ChE activity and lead concentrations. Other significant relationships included negative effects on ChE activities as related to tissue concentrations of chromium and aluminum. Plasma steroid biomarkers showed negative effects from zinc, DDE, DDT, and Aroclor 1260. No pathologies were observed in juvenile liver histology as a function of exposure to selected chemicals. The authors concluded that existing levels of metals and organochlorine compounds may potentially impact reproduction, enzyme expression, and physiological integrity of Kootenai River white sturgeon.

> *Kruse, G.O. and D.L. Scarneccia. 2002b. Contaminant uptake and survival of white sturgeon. American Fisheries Society Symposium 28: 151-160.*

This study investigated the potential effects of various de-adhesion treatments commonly used during rearing of white sturgeon eggs on contaminant uptake and survival of embryos in Kootenay River water. The study was part of a larger concern that the exposure of eggs or sperm to contaminants from sediments, water column, or suspended sediment during fertilization might have an impact on recovery efforts.

Methods. Embryos were exposed to different matrices (unfiltered river water, filtered river water, river bottom sediment, suspended river solids, and Fullers Earth) and analyzed for selected metals, organochlorine pesticides, and PCBs. The exposure duration was from fertilization of eggs through 13 days post fertilization.

Results and Conclusions. Significant differences in embryo mortality were found among treatment groups, and the authors concluded that two contaminants in the rearing medium (i.e., copper and Aroclor 1260) could have been high enough in concentration to contribute to the decrease in survival of embryos. In addition, significantly higher concentrations of arsenic, cadmium, copper, iron, magnesium, lead, and PCBs were discovered in embryos exposed to river bottom sediments as opposed to exposure to suspended solids or Fuller's Earth. The authors concluded that river-bottom sediments may be a more significant source of uptake of metals and organochlorines than water and suspended sediments. However, the authors cautioned that although mortality rates in relation to contaminant exposure were not excessive, they may be an additional stress on viable reproduction for this already endangered population. Further studies would be necessary to be able to establish relationships between contaminant concentrations and survival of embryos.

> *Bruno, J. 2004. Effects of two industrial effluents on juvenile white sturgeon (Acipenser transmontanus). Prepared for the Sturgeon Contaminants Working Group.*

Studies were conducted to assess the toxicity of two effluents on early life stage (ELS) of white sturgeon over 50 days in support of the UCR White Sturgeon Recovery Initiative (UCRWSRI). The larger question was "…what role might pollution play in the lack of successful recruitment of Columbia River white sturgeon?" In addition to sturgeon, the toxicity tests included early life stage rainbow trout and bacterial bioluminescence for short term tests.

Methods. The studies were conducted during summer/fall of 2002. The matrices tested were Combined Sewer Outfall III (CSOIII) effluent from Teck's Trail smelter, and effluent from the secondary foam tank of Celgar Pulp Company Ltd in Castlegar. The duration of the long term study was from 11 to 14-days post hatch (dph) through 61 to 64 dph. The chemistries of each effluent were measured as well. Note that Teck was in the process of a summer maintenance shutdown; therefore samples were collected over an abbreviated period (4 days).

Results. Short-term test results indicated the CSOIII effluent was not toxic to photo-luminescent bacteria but had some toxicity to rainbow trout fry. In contrast, no toxicity was exhibited by white sturgeon or rainbow trout to the Celgar Pulp mill effluent.

For the longer-term sturgeon tests, one hundred percent mortality was observed in the 50 and 100 percent CSOIII treatment groups after 17 and 5 days of exposure, respectively. No increased mortality relative to the controls occurred at 1 percent CSOIII concentration or in any of the two pulp mill exposure groups (1 and 100 percent effluent concentration). Control mortalities were 38.4 ± 28.4 percent (mean ± SD). No fish mortality occurred from the Celgar effluent.

The Teck Cominco effluent tested during these studies showed values ranging from 128 to 273 μg/L zinc, 12.5 to 54 μg/L lead, 1 to 6 μg/L copper, and 2.28 to 3.35 μg/L cadmium. The zinc, lead, and cadmium values were above the Canadian WQC for water. The authors concluded that the Teck Cominco effluent was lethal to juvenile sturgeon at high concentrations (50% and 100%), but not at low dilutions (1%), and that "long-term exposure to the effluent sample may present potential problems to the developing fish". Sturgeon may be more sensitive than rainbow trout to the Teck Cominco effluent, but the authors suggested this could also be due to the choice of younger life stages for the sturgeon.

There were a number of uncertainties associated with this study. The limited exposure period did not include evaluation of potential effects on possibly more sensitive life stages (e.g., eggs, embryos) and immediate post hatch effects. In addition, mortalities in the controls were highly variable (average = 40 percent; coefficient of variation = 74 percent), which adds uncertainty to the overall results.

> *Dwyer, F.J., F.L. Mayer, L.C. Sappington, D.R. Buckler, C.M. Bridges, I.E. Greer, D.K. Hardesty, C.E. Henke, C.G. Ingersoll, J.L. Kunz, D.W. Whites, T. Augspurger, D.R. Mount, K. Hattala, and G.N. Neuderfer. 2005. Assessing contaminant sensitivity of endangered and threatened aquatic species: Part I. Acute toxicity of five chemicals. Arch. Environ. Contam. Toxicol. 48:143-154.*

This study was a literature review of the acute toxicity of selected chemicals to aquatic species (96 hr. LC 50s).

Methods. A review of the sensitivity of 20 threatened aquatic vertebrate species to carbaryl, copper, 4-nonylphenol, pentachlorophenol, and permethrin was conducted based on acute toxicity data. Among the different species assessed, three were sturgeon: Atlantic sturgeon (*Acipenser oxyrhynchus*), shortnose sturgeon (*Acipenser brevirostrum*), and shovelnose sturgeon (*Scaphirhynchus platorynchus*).

Results. Based on the analysis conducted by the authors, it appeared that sturgeon are among the more sensitive species when exposed to copper, as well as for all the organic chemicals tested. The study also revealed relatively large differences in the sensitivity among different sturgeon species to copper, with ranks ranging from 1.5 to 10.5 out of 20 species (1 = most sensitive species; 20 = least sensitive species). However, when comparing across chemicals and species, one of the sturgeon species, the Atlantic sturgeon, was most sensitive out of the 20 species tested. In general, after reviewing these results and other literature, the authors concluded that sturgeon should be considered as a sensitive species when evaluating effects of toxicity and when comparing to surrogate species results (e.g., rainbow trout tests).

> *Kruse and Webb. 2006. Upper Columbia River White Sturgeon Contaminant and Deformity Evaluation and Summary. Prepared For the Upper Columbia River White Sturgeon Recovery Team Contaminants Sub-Committee*

Kruse and Webb (2006) collected white sturgeon in an area downstream of the Teck Cominco facility and north of the U.S.-Canada border to Hugh Keenleyside Dam in 2002 and performed a variety of toxicity tests and laboratory assays of contaminant effects on sturgeon physiology.

Methods. Tissue chemistry samples were collected from whole body adult and juvenile sturgeon, egg and sperm samples, and organs from a reproductive female for the analyses of metals, PCBs, PDBEs, organochlorine pesticides, dioxins, and furans. Toxicity and other tests with Teck Cominco effluent were performed on young stages of sturgeon.

Results. The Teck Cominco effluent was found to be lethal to larval sturgeon at the 100% and 50% levels. Genetic effects testing indicated that the effluent caused growth regulation and gene expression problems for the sturgeon. Greater amounts of external lesions and deformities were found in UCR sturgeon versus Kootenai populations. An increasing trend of fish with deformities was seen from 2002 to 2004 (39 and 55 percent, respectively). During this time frame hatchery-reared sturgeon had a high incidence of deformities but decreased between the years 2002 and 2004 (42.7 and 12.5 percent incidence of deformity, respectively). These rates of hatchery-reared sturgeon deformities are considered excessive. Recaptures of hatchery-reared sturgeon had high rates of pectoral fin deformity with the lowest rate (43.9 percent) of fish with deformed pectoral fins being captured in 2004. The authors concluded that the long lives of sturgeon may put them at greater risk than other species for the effects of these contaminants. Metals of interest for future study (on the basis of elevated levels in the sturgeon tissues) were cadmium, chromium, copper, lead, mercury, nickel, tin, and zinc.

> *USFWS and USGS. 2008. Summary of Kootenai River white sturgeon studies. information sheet. U.S. Fish and Wildlife Service, Upper Columbia Fish and Wildlife Office, Spokane, WA, USA, and U.S. Geological Survey (2007/2008).*

In 2007, the U.S. Fish and Wildlife Service (USFWS) and the USGS initiated a study to evaluate the acute toxicity of selected chemicals to white sturgeon and as compared to a standard "sensitive" test species, rainbow trout.

Methods. The acute toxicity of copper, chlorine, and three herbicides were evaluated for two different life stages (30 and 160 days post swim-up) of white sturgeon from the Columbia River and the Kootenai River. No detailed methods were provided.

Results. The study indicated that earlier life stages (30 days post swim-up) of white sturgeon were more sensitive to exposure to copper than older (160 days post swim-up) individuals by a factor of approximately 50 (LC50: 4.9 vs. 249 µg/L in 30 vs. 160 day post swim-up fish, respectively). The mean acute LC50 value of 4.9 µg/L, for Columbia River sturgeon is less than half of the EPA ambient water quality criteria (AWQC) of 12 µg/L adjusted for hardness. However, the LC50 is generally confirmed by the value of 3.1

µg/L copper LC50 for the Kootenai River sturgeon. Also, the results of the herbicide and copper studies show the increased sensitivity of sturgeon from either river as compared to rainbow trout. The authors did not provide any detailed information on study design or general conclusions.

3.1.7 Soil Chemistry

Summaries of soil chemistry data in reports and publications are summarized by reporting authors or organization in chronological order. Studies were performed to meet a variety of objectives and may not be representative of all Site soils. Further evaluation of select data from these studies is provided in Appendices G and H.

> *Holmgren, G.G.S., M.W. Meyer, R.L. Chaney, and R.B. Daniels. 1993. Cadmium, lead, zinc, copper, and nickel in agricultural soils in the United States of America. J. of Environ. Qual. 22:335-348. 16 pp.*

Surface soil was collected from 3,045 agricultural areas in the U.S., analyzing the samples for cadmium, copper, lead, nickel, and zinc. The purpose of the study was to characterize background soil chemistry in agricultural areas; sampling sites were selected to represent normal agricultural practices, uninfluenced by any known nonagricultural sources of cadmium and lead. Three samples were located immediately to the south of the UCR Site.

Methods. Soil samples were not sieved, and were prepared for metals analysis by strong acid digestion. Lead was analyzed by anodic stripping voltmeter. Cadmium was analyzed using GFAA spectrophotometry with deuterium lamp background correction. Copper, nickel, and zinc were analyzed via a multi-element direct current plasma spectrograph. QA analysis using standard reference materials and spiked sediments demonstrated complete recovery of the target metals.

Results. The authors listed the geometric means (in mg/kg dw) of trace metals in Washington State agricultural soils as follows: cadmium at 0.184; zinc at 66.0; copper at 26.7; nickel at 26.4; and lead at 8.5.

> *Ecology. 1994. Natural background soil metals concentrations in Washington State. Publication No. 94-115. Washington State Department of Ecology, Toxics Cleanup Program. 275 pp. October 1994.*

This extensive report (publication No 94-115, October 1994) covers many aspects of the soils of Washington State. The primary objective was to define a range of values that represent the natural concentration of metals in surficial soils throughout Washington State. Ecology (1994) provides data on soil chemistry for the Spokane Basin that are useful for comparison to Site soils. The Spokane Basin group was a combination of data from Spokane, Lincoln, and Pend Oreille Counties.

Methods. Sampling was conducted by USGS Water Resources Division personnel between June 1987 and January 1993. Five surface samples were collected with a stainless steel soil auger (2-ft depth) within 1-acre plots throughout Washington State, and 5 to 13 vertical profile samples were taken from hand-dug test pits within each area. Equipment was decontaminated between samples. Each sample consisted of approximately 20 L of soil. All samples were sieved through a 19 mm screen. Within the Spokane Basin, 79 soil samples were collected from 22 locations at depths of 24 in. to 36 in. Chemical analysis was performed at Manchester Laboratory. Samples were dried and again sieved using a 2 mm sieve (larger particles were excluded from chemical analyses) and acid digested (EPA Method 3050) prior to analysis by ICP (EPA Method 6010), manual cold-vapor technique (for mercury; EPA Method 7471), and GFAA methods (for arsenic and selenium; EPA Methods 7060 and 7740). A total of 27 samples are available for the Spokane Basin, including 22 locations, and 5 field duplicates.

> *USEPA. 2002a. Upper Columbia River/Lake Roosevelt expanded site inspection – Northeast Washington, Sediment Investigation Statistical Analysis. Prepared for U.S. Environmental Protection Agency, Region 10, Seattle, WA, by the Superfund Technical Assessment and Response Team.*
>
> *USEPA. 2003a. Upper Columbia River expanded site inspection report, Northeast Washington (Region 10, START-2). Prepared for U.S. Environmental Protection Agency, Region 10, Seattle, WA, by the Superfund Technical Assessment and Response Team. 84pp.*

This study was conducted by EPA to provide information for determining whether the UCR site should be included on the National Priorities List, and to establish priorities for additional action, if warranted. Concentrations of metals and organic compounds were evaluated in soils and are described further in the sediment study summaries (Section 3.1.2).

> *USGS. 2004. Geochemistry of sediments in the U.S. from the NURE-HSSR database. http://tin.er.usgs.gov/nure/sediment/. U.S. Geological Survey. Accessed September 2007.*

The National Uranium Resource Evaluation Hydrogeochemical and Stream Sediment Reconnaissance (NURE-HSSR) program, a nationwide survey of the elemental composition of soils and sediments, was conducted to assess the location of potential deposits of uranium and other strategic minerals in the U.S. Sampling and analysis was conducted by four national laboratories during the 1970s and 1980s. In the Pacific Northwest, sampling and analysis was conducted primarily by the Savannah River Laboratory, with some samples collected and processed by the Lawrence Livermore Laboratory. NURE-HSSR data were obtained from USGS (2004).

During the NURE-HSSR program, samples were collected near the study area. The location of each sample was described, and those descriptions included an indication of whether or not each sampling location was potentially influenced by municipal, agricultural, industrial, mining, or other (unspecified) anthropogenic activity. Documentation of the field and laboratory methods followed during the NURE-HSSR program was reviewed to assess the usability of these data. Overall, the QA measures implemented during the NURE-HSSR field and laboratory programs followed the standards of good laboratory practice.

> *Weston, R.F. 2005. Le Roi Smelter removal action report, Northport, Stevens County, Washington. Prepared for U.S. Environmental Protection Agency. Weston Solutions, Inc., Seattle, WA.*

A Comprehensive Environmental Response, Compensation, and Liability Act removal action was conducted in 2004 at the Le Roi smelter in Northport, Washington, under the direction of EPA Region 10. During Phase 2 sampling in May 2004, samples were collected on public lands distant from the former smelter site ("Outer Area" or "OA" samples). The OA samples were analyzed for arsenic, cadmium, copper, and lead.

Methods. Sample analyses were carried out following the EPA Statement of Work 5.3, and a QA review of the laboratory results was conducted as part of the remedial program. The analyses met acceptability standards for precision, accuracy, representativeness, comparability, and completeness. The OA samples were spatially within or beyond the band of samples around Northport that were designated for no action. A total of 17 OA samples were collected, but location coordinates were available for only 11 of these, and only data for those 11 OA samples were loaded into the project database and used for the characterization of background soil. One of the OA samples was found to have an arsenic concentration above the removal action level (Weston 2005), and that sample was not included among those used for this analysis.

> *Sanei, H., F. Goodarzi, and S. Hilts. 2007. Site-specific natural background concentrations of metals in topsoil from the Trail region, British Columbia, Canada. Geochemistry: Exploration, Environment, Analysis 7:41-48. 8 pp.*

This paper discusses the establishment of background levels of the trace metals arsenic, cadmium, copper, mercury, lead, and zinc in topsoil surrounding the Teck-Cominco zinc-lead smelter in Trail, B.C. The authors mention natural metals enrichment of soils resulting from mineralized bedrock, forest fires, volcanic ash, as well as historical anthropogenic operations leading to high metal concentrations in area soils. This study focused especially on finding diverse soil types that matched those near the smelter, yet were far enough away to minimize suspected smelter and other anthropogenic contamination.

Methods. Thirty-seven samples of the top 0 to 15 cm of soil were collected within a quadrant encompassing 49° 00′ to 49° 23′ N and 117° 25′ to 118° 00 W. Sampling was conducted with a stainless steel coring device which was cleaned between samples;

un-decomposed organic matter was removed from each core before storage and labeling. Soil samples were sieved through a 2 mm mesh in the laboratory and microwave digested with acid. Metal concentrations were determined with ICP-MS and CVAAS (for mercury). Analyses of blanks and laboratory standards indicated that analytical precision and accuracy was better than 10 percent.

Results. The paper presented background metal concentrations in soils by several statistical methods, including medians plus or minus two median absolute deviations, as well as various mean and percentile derived values. The 95th percentile results for background soils were as follows (units are mg/kg): arsenic, 19.7; cadmium, 1.67; copper, 51.5; mercury, 0.07; lead 37.9; and zinc, 168.

3.1.8 Air Quality

Summaries of air quality data in reports and publications are summarized by reporting authors or organization in chronological order. Studies were performed to meet a variety of objectives and may not be representative of all Site conditions.

> *Ecology. 1998a. Northport, Washington Air Quality Study: Phase III. Publication No. 98-210. Washington State Department of Ecology, Olympia, WA. 31 pp. October 1998.*
>
> *Ecology. 1998b. Northport, Washington Air Quality Study: Phase IV. Publication No. 98-211. Washington State Department of Ecology, Olympia, WA. December 1998.*
>
> *WDOH. 1994. Air Monitoring Data and Evaluation of Health Concerns in Areas of Northeast Tri-County; Summary of Activities.*

Ecology, in cooperation with the Washington State Department of Health (WDOH), conducted an air quality study between December 15, 1992, and December 31, 1998. The purpose of the air quality study was to 1) determine the possibility of cross-border transport of pollutants, 2) identify a potential contributing source(s), and 3) measure pollutant concentrations. The study consisted of four phases:

- Phase I, December 15, 1992, through February 13, 1993 (WDOH 1994)
- Phase II, August 1993 through October 30, 1993 (WDOH 1994)
- Phase III, November 3, 1993, through August 6, 1994 (Ecology 1998a)
- Phase IV, September 5, 1997, through December 31, 1998 (Ecology 1998b).

Methods. Phase I air sampling (WDOH 1994) included five sampling sites, four in the Northport area and one in Kettle Falls. At all sites, samplers monitored concentrations of total suspended particulates (TSP), lead, and arsenic. In addition, at one of the Northport area sites, a sampler measuring particulate matter with a diameter less than 10 µm (PM_{10}) was also used.

Phase II included seven monitoring distributed in a north-south pattern from Trail, B.C. to Kettle Falls as well as areas suspected of high metal concentrations in the Northport and Kettle Falls area. Phase II included analysis of seven metals likely to come from a smelting operation, a local power plant, and mining operations: antimony, arsenic, cadmium, copper, lead, manganese, and zinc. At each of the seven monitoring sites, PM_{10} was monitored using an instrument certified by EPA for use in sampling PM_{10} on an one-in-six-day schedule consistent with national monitoring programs. One of the two Northport sites had two additional samplers installed. One sampler measured TSP and the other measured PM_{10} using a sequential instrument that is not certified by EPA, but does have the capability to sample on two sets of filters, for six days in a row. Wind speed, wind direction, and temperature data were also collected at this site for a limited period (August 10, 1993, through September 27, 1993). Air quality modeling was performed to simulate the metal pollutant concentrations throughout the area. Meteorological data from local weather stations were used in conjunction with smelter production rates obtained from Cominco, Ltd. and metal concentrations measured at the seven monitoring sites on September 28, 1993.

Phase III air monitoring of PM_{10}, lead, arsenic, zinc, and cadmium at the Paparich Farm site at Northport was conducted from November 3, 1993, through August 6, 1994 (Ecology 1998a). Ecology reported the need for additional analysis to evaluate the risk to populations exposed to these metals. Meteorological data were collected at Northport elementary school, and sulfur dioxide concentrations were collected at Northport elementary school and the Northport airport.

Phase IV monitoring was initiated following the upgrade at the Trail facility in B.C., which came on line in the spring of 1997. After the implementation of the new process, Ecology (1998b) further required that three long-term monitoring sites for air quality be operated in the Northport area to track the changes of air quality from the upgraded Trail facility and to aid in future air quality modeling.

Results. Results of Phase I indicated no violation of the state or federal lead standards. WDOH (1994) reported that concentrations of lead and arsenic measured during the study were some of the highest recorded anywhere in the state during that time.

As reported by WDOH (1994), September 28, 1993, was the worst air quality day of the Phase II study period. Phase II results indicated no violation of the 150 $\mu g/m^3$ state particulate standard in effect at the time of sampling. In addition, the maximum concentration of lead measured during the Phase II study at each site was lower than the federal and state standard for lead. Ecology's acceptable source impact levels (ASILs) for antimony, copper, manganese, and zinc (based on 24-hr averages) were not exceeded by maximum measured concentrations of these metals. Maximum concentrations of arsenic and cadmium measured during the 42-day study period did exceed corresponding ASIL values based on annual average concentrations. However, Ecology

(1998a) summarized Phase I and II results by stating "Although no state or federal particulate standards were exceeded during these two phases, metal concentrations of lead, arsenic and cadmium were found to be well above expected normal rural background levels."

Ecology's (1998a) detailed analysis of the monitoring samples collected during Phase III of the study showed that: the quarterly average lead concentration at the monitoring site (0.14 $\mu g/m^3$) was significantly less than the quarterly standard (1.5 $\mu g/m^3$). The maximum monitored 24-hr sulfur dioxide concentration was 0.064 ppm at Northport elementary school. This value is less that the National Ambient Air Quality and Washington State standards of 0.14 ppm and 0.10 ppm, respectively. The annual average arsenic (0.03 $\mu g/m^3$) and cadmium (0.01 $\mu g/m^3$) concentrations were higher than Ecology's ASILs (0.00023 $\mu g/m^3$ and 0.00056 $\mu g/m^3$, respectively) with both detected and non-detected concentrations of arsenic and cadmium exceeding their ASILs [8]. The highest observed value of PM_{10} (104 $\mu g/m^3$) was approximately two-thirds of the National Ambient Air Quality Standard (150 $\mu g/m^3$) for a 24-hour period.

Phase IV monitoring results showed that the maximum observed 24-hr averages at Northport for particulate matter, arsenic, cadmium, lead, and zinc were 121, 0.04, 0.02, 0.42, and 0.89 $\mu g/m^3$, respectively (Ecology 1998b). "Cadmium and arsenic concentrations were found to be either at or below their respective detection limits at both sites. Lead and zinc were above their detection limits in most of the samples."

Conclusions. WDOH (1994) noted that the difference in averaging times for the ASILs and the study could be misleading given that average concentrations over the entire study period would tend to be much lower than maximum values and still greater than data measured and averaged over an entire year. Specifically, WDOH (1994) stated: "There is a strong likelihood that, had Phase II lasted a whole year, the resulting yearly arsenic concentrations would be much lower still, perhaps much closer to the ASIL values."

Based on the Phase III air sampling results for PM_{10} and metals in particulate phase, Ecology (1998a) found that for the period of August 13, 1993, through August 6, 1994, "arsenic and cadmium were almost always deposited with lead in a similar pattern, i.e., where lead concentrations are high, arsenic and cadmium concentrations are also high and vice-versa". Therefore, Ecology (1998a) concluded that "lead, arsenic and cadmium

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[8] Comparison of arsenic and cadmium concentrations to the ASIL values is problematic because the arsenic and cadmium detection limits in this study exceeded ASILs, and background concentrations typically exceed these values even in rural areas. Reported mean levels of arsenic in ambient air in the United States range from < 0.001 to 0.003 $\mu g/m^3$ in rural areas and from 0.02 to 0.03 $\mu g/m^3$ in urban areas (ASTM 2004). Similarly, Agency for Toxic Substances and Disease Registry (1999) reports that during the 1980s and 1990s, mean cadmium levels ranged from < 0.001 $\mu g/m^3$ in remote areas to 0.003–0.05 $\mu g/m^3$ in urban areas in the United States.

were deposited in the Northport area in ratios that stayed consistent between the metals, which indicated a common source, i.e., Cominco Ltd." Additional conclusions were:

1. Lead values in Northport were below the state and federal standards. Particulate values were below the state standard for both TSP and PM_{10}.
2. The arsenic annual average concentration exceeded its ASIL, value by a factor of at least 87. The cadmium annual average concentration exceeded its ASIL, value by a factor of 18.
3. A correlation coefficient of 0.995 shows that PM_{10} can be used in place of TSP for lead sampling in Northport.
4. Sulfur dioxide levels in Northport were below both the NAAQS and Washington State standards throughout the Cominco, Ltd. and Ecology Phase III monitoring periods.

Analysis of available data indicates a poor correlation between lead and sulfur dioxide in the Northport area.

Ecology (1998b) indicated that a final report "including analysis and presentation of all the data" for the Phase IV study would be released but later confirmed that no further analysis was done.

> *USGS. 2005. Occurrence and distribution of trace elements in air along Lake Roosevelt. Summary and data available from http://wa.water.usgs.gov/projects/roosevelt/summary.htm.*

In collaboration with the Lake Roosevelt Water Quality Council, the Confederated Tribes of the Colville Reservation, U.S. Bureau of Reclamation (USBR), National Park Service, and USGS conducted an air quality study as part of ongoing investigations within Lake Roosevelt ("Occurrence and Distribution of Trace Elements in Air along Lake Roosevelt between 2002 and 2006") (Majewski and Kahle 2003, 2005; USGS 2006b). According to the project design by USGS, the purpose of the study was to:

- Determine the occurrence, concentration, distribution, and seasonal variability of select trace elements on airborne dust particles at several locations along Lake Roosevelt.
- Compare the composition and concentration of airborne trace elements in the ambient atmosphere to that of high wind events occurring during the winter/spring and fall reservoir drawdown periods.
- Determine, to the extent possible, what percent of the measured concentration of airborne trace elements originated from exposed beach, bed, and bank sediments.

The objective of this study was to compare the occurrence, composition, and concentration of trace elements measured in airborne dust samples collected before, during, and after the drawdown of the reservoir to the results of a previous study that sampled exposed bed sediments along the entire length of Lake Roosevelt.

Methods. Based on the proximity to Lake Roosevelt, the availability of alternating current (AC) power, being reasonably free from local source contamination, and the prospects that the site would be reasonably secure, three monitoring stations were set up along the UCR at Kettle Falls (located on the west bank at RM 703.), Marcus Flats (east bank near RM 708), Inchelium (west bank near RM 675), and Seven Bays (east bank near RM 636). The monitoring station at Kettle Falls was operated only in 2002, and it was moved to Marcus Flats in 2003 and thereafter.

The sampling period for 2002 to 2005 started in January and ended in September. For 2006, sampling occurred from March to June. Sampling frequency for 2002 to 2003 was once every sixth day. For 2004 to 2005, the sampling began on January 4 with a frequency of once every twelfth day (1 in 12) and continued through February. The sampling frequency was increased to one day out of every six during March, and increased again to one in three in April through May 25. For the rest of May and June, the sampling frequency decreased again to one day out of every six. Sampling ended on June 28. The second sampling period began on August 26 with a sampling frequency of one day out of every six, and ended on September 19. For 2006, samples were collected every seventh day during the sampling period.

In addition, samples were collected during high-wind events for 2003 to 2004. Where a high-wind event was operationally defined as those days in which the wind speed exceeded 5.14 m/s (or 11.5 mph) for four or more consecutive hours.

Sampling equipment at each site consisted of a high-volume air sampler for particulate matter with an aerodynamic mean diameter of 10 µm or less (PM_{10}) with a fully instrumented meteorological station that recorded hourly averages of air temperature, relative humidity, wind speed and direction, barometric pressure, and rainfall.

Using the PM_{10} sample collected on the quartz-fiber filter for each of the sampling days and the corresponding blank sample, PM_{10} mass was determined and PM_{10} air concentration was calculated. Chemical analysis was performed for arsenic, lead, cadmium, copper, zinc, and more than 20 other elements (such as lithium, beryllium, phosphorous, scandium, titanium, chromium, manganese, iron, cobalt, nickel, gallium, rubidium, strontium, yttrium, niobium, molybdenum, gold, antimony, cesium, barium, lanthanum, cerium, bismuth, thorium, and uranium) using ICP-MS. The mean result of the blank analyses was subtracted from each field sample result to obtain the final sample mass per analyte. These values were then divided by the total volume of air sampled to obtain the concentration in $\mu g/m^3$.

Particulate phase mercury was sampled from the air monitoring stations at Seven Bays, Inchelium, and Marcus Flats following a different protocol than that for arsenic, cadmium, copper, lead, and zinc. The air samples for mercury were collected from April through September for 2004 and 2005, respectively, and no samples were collected for 2002, 2003, and 2006. Mercury concentration in air for each of the sampling days was determined in the same way as that for other metals.

Results. Results, as summarized in the RI/FS Work Plan (USEPA 2008a), did not show any spatial or temporal trend across the three monitoring stations for 2002 to 2004. All the mean concentrations were within the range of 11 to 22.8 $\mu g/m^3$. Concentrations of arsenic, cadmium, and lead showed a gradual decreasing trend from Marcus Flats to Seven Bays (north to south) within the three monitoring years. Copper and zinc consistently had the highest mean air concentrations for all sites across all sampling periods, with lead being the next highest.

Conclusions. The authors concluded that for PM_{10}, 24-hour average concentrations for all the monitoring stations did not exceed the short-term standard (150 $\mu g/m^3$) and the annual average concentrations did not exceed the long-term standard (50 $\mu g/m^3$). Air concentrations of slag-related trace elements were low, and the study did not single out any trace element as being of concern. It should be noted that a formal report has not been prepared by the authors using the above-mentioned air quality data, but rather has been summarized in a poster presented at the 5th Symposium on the Hydrogeology of Washington State (April 12 to 14, 2005).

3.1.9 Groundwater

The NPS provided analytical data from their routine monitoring of wells located within the LRNRA. These data were received from the NPS in 2008 as *.pdf files on the data reporting forms from the analytical laboratories. They included data on concentrations (in unfiltered samples) of the TAL metals; selected organics such as bromodichloromethane, ethylbenzene, xylenes, iron, fluoride, corrosivity, odor, pH, nitrate, calcium, sodium, and sulfate. Not all analytes were measured at all wells or at all time periods for each well. There were no reports associated with these data.

3.2 DATA EVALUATIONS

This section summarizes the results of evaluations conducted using selected existing site-related data for each medium in order to derive more in-depth understanding of the patterns and trends in COPC distributions at the Site. The analyses presented in these sections are not a summary of the studies in Section 3.1; rather, they are evaluations of existing data. These analyses also refer to additional data from off-site to establish context see Appendix H. Consistent with the HHRA, the data evaluated are only from

1996 and later. Additional details about the evaluations, as well as additional evaluations and further clarification of which data were used for each medium are within Appendices C through H.

As noted earlier, many of the studies and monitoring programs that contributed data to the evaluations presented here were conducted for purposes unrelated to the UCR RI/FS and BERA. Consequently, the sample collection procedures, target analyte lists, analytical methods, and detection limits employed during these studies may not meet the current standards of practice and/or the data quality requirements necessary for use in the ecological risk assessment for the Upper Columbia. However, for purposes of this BERA Work Plan, the data presented are assumed to be of adequate quality to provide an understanding about the distribution of chemicals in the UCR and sufficient to identify data gaps and support general site descriptions. As the BERA progresses, the quality of the existing data and suitability for inclusion in the BERA will be assessed according to procedures that will be reviewed and approved by the EPA. In addition, the analyses and statistical tests presented in this section, while appropriate for the purposes of the BERA Work Plan, may not be appropriate for use in the baseline risk assessment calculations and should not be considered definitive.

3.2.1 Surface Water Quality

As noted in Section 3.1.1, surface water quality data (with emphasis on COPCs) collected to date within the UCR have been spatially limited, and may be of limited value informing risk-based decisions. Notwithstanding these limitations, preliminary analyses of existing site and off-site data can be made to guide future surface water sampling programs. The following provides a brief summary of this analysis for a few metal COPCs using data from the EC, USGS, and Ecology databases. Additional discussion of select surface water quality evaluations are presented in Appendix C.

Metals

Total metals have been monitored at several locations in the Columbia River and Pend Oreille River, a major tributary to the UCR just north of the U.S.-Canada border. Stations along the main stem of the Columbia River in Canada include the Birchbank station (Federal ID BC08NE0005/Provincial ID 200003), and the Waneta station (Federal ID BC08NE0001/Provincial ID 200021). Additional data are available from the Pend Oreille River at Waneta B.C. (Federal ID BC08NE0029/Provincial ID 200021) and further upstream along the Pend Oreille at a station referred to as "International Boundary" (Federal ID BC08NE0020/Provincial ID E237493). In addition to the above-mentioned off-site surface water quality monitoring stations, and as noted within Section 3.1.1, water quality monitoring data have been collected in the UCR at Northport, Washington (USGS Station 12400520; Ecology Station 61A070 at RM 735.1).

A qualitative comparison (i.e., box plots) of data collected from 2001 through 2005 at the above-listed monitoring stations was completed for a limited number of COPCs (arsenic, cadmium, copper, lead and zinc), see Figures 3-1 through 3-5. It should be noted that the box plots are based only on detected metal concentrations so that differences in detection limits do not influence comparisons between stations. However, differences in detection limits skew the results and make graphical comparisons difficult in some cases. The following general observations are noted:

- **Arsenic.** Total arsenic concentrations in the Columbia River at Birchbank and Waneta are comparable; while concentrations in the Pend Oreille River were higher. At Northport, total arsenic concentrations appear intermediate between those measured in the Columbia upstream of the border and those measured in the Pend Oreille River.
- **Cadmium.** Total cadmium concentrations in the Columbia River at Waneta were slightly higher than those measured at Birchbank, the Pend Oreille River, or Northport in 2003, 2004, and 2005, but not 2001. Interpretation is complicated by difference among stations in detection limits and frequency of non-detects.
- **Copper.** Total copper concentrations appear to have been slightly higher in the Columbia River at Waneta than at Birchbank and concentrations in the Pend Oreille River were higher than those measured in the Columbia at Waneta. Copper concentrations at Northport are intermediate between those measured in the Columbia upstream of the border and those measured in the Pend Oreille River. However, there is large overlap among concentrations at these stations.
- **Lead.** Total lead concentrations appear to have been similar among the B.C. sites and Northport, although concentrations may have been slightly higher in the Pend Oreille River at Waneta.
- **Zinc.** Overall, total zinc concentrations appear to have been higher in the Columbia River at Waneta than upstream at Birchbank, and total zinc concentrations in the Pend Oreille River appear intermediate between those measured in the Columbia River. At Northport, total zinc was infrequently detected above a detection limit of 5 µg/L, and samples at this location included the highest individual values in the dataset (2003/2004).

Additional data was reported within Scofield and Pavlik-Kunkel (2007), as identified in Section 3.1.1, Due to elevated analytical detection limits and suspected lead contamination (i.e., use of lead weights during sampling) some of the COPC data are of limited value for the BERA. Notwithstanding this limitation, data are still useful for limited analyses of the detected concentrations and for non-COPCs (e.g., sodium).

3.2.2 Sediment Chemistry

As indicated within Section 3.1.2, the greatest volume of data collected to date within the UCR has been measures of bulk sediment chemistries and physical characteristics (e.g., grain size). These data have consistently identified longitudinal trends where elevated concentrations for elements such as antimony, arsenic, chromium, copper, iron, manganese, and zinc are highest in the portion of the river upstream from Marcus Flats (above RM 700), where coarse-grained sediments predominate. Concentrations of these five metals tend to display a step decline at Marcus Flats and are relatively lower in the reservoir portion of the Site. Lead also tends to be associated with coarse-grained sediments above Marcus Flats, but differs in that elevated concentrations persist through the reservoir portions of the site. Cadmium, mercury, and nickel appear to be more closely associated with finer grained sediments and concentrations are generally higher in the reservoir portion of the site than in the riverine reaches above RM 734. .

Exploratory analyses were carried out to evaluate whether subsets of sediments have distinct characteristics that can provide a basis for better understanding the distribution of COPCs in Site sediments, thereby helping to refine our understanding of the Site and focus future evaluations. As detailed within Appendix D, these analyses were carried out using statistical and graphical tools to examine potential relationships among chemical concentrations and physical characteristics. It is important to recognize that these *a priori* analyses may be useful for describing site sediments for the purposes of future data collection and for a preliminary understanding of site conditions, but they do not necessarily inform risk management decisions, nor do they preclude alternative analyses that can or will be conducted on future data (e.g., synoptic sediment and toxicity data to develop concentration-response relationships). One result from these analyses (i.e., an evaluation of characteristic element ratios), was the identification of three classes of sediment with distinguishing characteristics. Distinguishing characteristics were based on the relative abundance of zinc and vanadium using an iterative regression model (IRM) procedure. Because no unambiguous method to separate these subsets using the IRM method exists, a conservative cutoff value was chosen for this distinction at the zinc sediment screening level. Specifically, samples with a zinc concentration greater than 121 mg/kg or a zinc/vanadium ratio greater than 10 were identified separate classes, and the remaining samples were considered a third class:

- Class I—Slag influenced
- Class II—Not strongly influenced by slag
- Class III—Other, not corresponding to either class I or class II.

It is important to note that the descriptions used above are not intended to be full or complete characterizations; but rather are intended to refine our understanding of the Site and focus further evaluations. For instance, class I sediments represent samples that appear to be influenced by granulated slag; but should not be interpreted to mean they *are* granulated slag. Similarly, class II sediments represent samples that appear to have little or no influence of slag, but are not necessarily free of slag. Class III sediment are also not necessarily free of slag, but they simply do not correspond with zinc to vanadium ratios similar to those in classes I or II.

In addition to chemical differences, the spatial distribution of these classes was evaluated to determine if this sediment classification contributes to a better understanding of the distribution of COPCs within the UCR.

Evaluation of spatial distributions was carried out both for Phase I sediments (which were used to develop the classification tool), and also for surface sediment samples to which the classification criterion could be applied (i.e., those data sets having detected concentrations of both zinc and vanadium). Using this criterion, the following data sets were evaluated: Phase I data, EPA START data, and data collected by Grosbois et al. (2001), Cox et al. (2005), and Paulson et al. (2006). Map 3-1 illustrates the spatial distribution of sediment classes among Phase I samples, and Table 3-2 summarizes the spatial distribution, by reach, for the above-mentioned studies[9].

Based on this analysis, samples in class I contain a relatively high fraction of coarse particle sizes and are exclusively located above RM 701 (Marcus Flats); while samples in classes II and III have a greater percentage of fines and are predominantly located downstream of Marcus Flats. Additional spatial distributions of individual COPCs along the length of the UCR were evaluated in terms of variation between river reaches (Appendix D, Figures 29 through 126). Three distinctive patterns were observed from this analysis and can be summarized as follows:

- **Pattern 1—Maximum concentrations that are highest upstream and decrease downstream.** This pattern is exhibited by most of the common metals and metalloids. Specifically, this pattern is best shown by antimony, arsenic, barium, calcium, cobalt, copper, iron, magnesium, manganese, silver, uranium, and zinc. The identities of the metals that are highest in the upstream reaches indicate that this pattern is a result of the presence of class I sediments.
- **Pattern 2—Maximum concentrations that are lowest upstream and increase downstream.** This pattern is exhibited by all of the semi-volatile organic compounds, pesticides, and dioxins and furans, and most of the exotic metals but also nickel.
- **Pattern 3—Non-systematic variation.** A few chemicals show little longitudinal variation, or variations that do not represent a monotonic trend of either increasing or decreasing concentrations with reach. These chemicals are

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[9] Note: no sediment samples were collected where gravel content exceeded 35 percent; in areas where samples could not be collected, sediment classes cannot be assigned.

aluminum, beryllium, bismuth, cadmium, lead, mercury, selenium, and vanadium.

Although sediment classes were defined on the basis of COPCs, these classes may also be distinguished by sediment particle size. Class I contains the highest fraction of sand-sized particles, class III contains the highest fraction of clay- and silt-sized particles, and class II contains an intermediate distribution of particle sizes, with the greatest fraction in the sand-sized category.

In summary, the analysis of the existing sediment database results in a number of observations that may help guide development of the ecological risk assessment approach and future BERA sampling plans:

- Site sediments may be categorized into three classes based on the zinc/vanadium ratios. One of these classes (class I) appears to potentially reflect the influence of smelter slag, one (class II) may not be strongly influence by slag, and the third (class III) is unlike either of the other two.
- The distinction between the three sediment classes can also effectively be a distinction between sediment grain sizes—class III is predominantly fine (clay- and silt-sized) sediment, and class I is predominantly coarse (sand-sized) sediment.
- Class I sediments appear to be confined to areas upstream and including Marcus Flats.

Despite the significant amount of sediment data collected to date and identified by the SLERA (also refer to Section 2.4 of this work plan), there remain uncertainties associated with some COPC sediment concentrations (e.g., PBDEs, SVOCs, gold, indium etc.).

Subsurface sediment sampling within the UCR was conducted by Cox et al. (2005) and Phase I RI/FS studies (USEPA 2006d). Although the data set for subsurface samples was smaller than that for surface sediments, with the exception of Reaches 1 and 2 where coarse substrates did not permit the collection of cores), subsurface sediment samples were collected within representative reaches of the reservoir (Reaches 3 to 6).

3.2.3 Sediment Toxicity

As identified within Section 3.1.3, several historical studies have evaluated sediment toxicity in the UCR. More recent information has been collected at seven stations in 2004 by the USGS and at 50 stations in 2005 (i.e., Phase I). Sediment toxicity data will be an important source of information with respect to evaluating potentially unacceptable risks of COPCs in sediment and porewater to benthic macroinvertebrates. A detailed summary and evaluation of the historical (1986 to 2001) and recent (2004 to 2005) sediment toxicity data is presented within Appendix E of this work plan. The following is intended to provide a brief summary.

Evaluations of longitudinal patterns in Phase I toxicity data showed that the greatest number of stations with responses less than a benchmark value of 80 percent of negative control[10] was within Reach 1. No apparent site-wide longitudinal gradient in sediment toxicity was found. Comparisons between historical (1986 to 2001) and recent (2004 to 2005) sediment toxicity data illustrate indicate variable patterns among studies with some reporting elevated toxicity to multiple test species while others reporting toxicity was not different from reference samples. Based on data collected to date, toxicity has routinely been observed in sediments located within and upstream of Marcus Flats, and in fine grained sediments of the lower reservoir.

In addition to basic sediment toxicity test results that encompass a range of receptor classes and toxicity endpoints (e.g., survival, growth, and reproduction), several additional measures of exposure of aquatic organisms can be considered in an overall weight of evidence approach to evaluating risks from UCR sediments. These measurements include 1) total metal concentrations (bulk chemistry); 2) benthic invertebrate bioaccumulation tests to evaluate the bioavailability of contaminants; 3) sequential extracts of bulk sediments, including acid volatile sulfide (AVS) and simultaneously extracted metals (SEM); and 4) porewater concentrations. These different types of assessment methods can be used in a weight-of-evidence (WOE) approach to assess the potential for adverse effects on sediment dwelling organisms (e.g., benthic invertebrates).

3.2.3.1 Bulk Chemistry

The comparison of metal concentrations to benchmarks (e.g., TECs) is considered to be a conservative analysis because TECs do not explicitly account for the bioavailability of COPCs. Comparisons to PEC are less conservative, but sediments with varying degrees of bioavailable metals still tend to show effects at these concentrations. Applying this less conservative approach to Phase I data identified that sediment concentrations exceeded a probable effect concentration quotient (PECQ) for one or more metals at 24 of the 50 stations sampled. All but two of those stations were located in Reaches 1 through 3, while no exceedances were found below RM 687 (Map 3-2). Upon further evaluation, copper, lead, and zinc were the only elements with sediment concentrations that exceeded PECQs at multiple stations. In general, the highest PECQs were found for zinc and copper, for which the maximum values were 31 and 13, respectively. PECQs for arsenic, chromium, mercury, and nickel were not exceeded at any station, and the PECQ for cadmium was exceeded at only one station. Similarly, the longitudinal

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[10] Review of the Phase I sediment toxicity tests, identified that reference areas contained somewhat dissimilar TOC from conditions within the Site. For the purpose of this BERA Work Plan an evaluation of Phase I data was based on a comparison of responses relative to the negative controls. Alternative methods may also be used to evaluate these data.

distribution of PECQs showed a gradient within Reaches 1 to 4a, and then a relatively flat distribution within the lacustrine CSM see Appendix E.

Besser et al. (2008) also used PECs to evaluate the sum of PECQs for copper, cadmium, lead, zinc, and arsenic (the notation used therein was probable effect quotient [PEQ] rather than PECQ). Interestingly, PECQs for sediment samples from Besser et al. (2008) were as high as 85 at one of the stations (LR7 at RM 735; Reach 1). In this particular case, the PECQ for individual metal concentrations of copper, lead, and zinc were considerably higher than their respective PECs (18.7-, 8.5- and 56.0-fold, respectively).

3.2.3.2 Sequential Extracts (SEM-AVS) and TOC

Measurements of AVS and SEM involve a relatively simple, single-step extraction procedure. The results can be used as a basis for assessing when levels of selected cationic metals in sediments are not expected to cause adverse effects on benthic organisms (USEPA 2005). Under reducing conditions AVS will tightly bind cationic metals so they are not bioavailable and will not exert toxicity. Metal toxicity is not typically observed when the concentration of AVS exceeds the concentration of SEM. The following section, presents a summary of analyses conducted using SEM, AVS, and related parameters to evaluate the potential bioavailability of the SEM in sediments. Detailed discussions are found in Appendix E.

The longitudinal distribution of Phase I ∑SEM showed concentrations exceeding 5.0 μmol/g at all but two of the 26 stations in Reaches 1 to 3, and concentrations less than that value at all but three of the 24 stations in Reaches 4 to 6 (lacustrine CSM). With respect to the relative contributions of individual SEM to ∑SEM, zinc accounted for the greatest contribution at most stations, but cadmium, nickel, and lead combined to account for relatively large contributions between RMs 678 and 658. Copper accounted for a relatively constant contribution at most stations. The longitudinal distribution of AVS was generally similar to that for ∑SEM, with all but eight values being greater than 0.5 μmol/g in Reaches 1 through 3; all measured values were less than 0.1 μmol/g downstream of Marcus Flats. The longitudinal distribution of TOC did not show a pattern consistent with that found for ∑SEM and AVS. Concentrations of TOC generally increased from Reach 1 to Reach 3 (i.e., Marcus Flats, where all four values were 0.8 percent or greater); and varied markedly (ranging from 0.04 to 2.2 percent) in reaches downstream of Marcus Flats.

Concentrations of (∑SEM-AVS) measured during Phase I were compared with generic thresholds (i.e., 1.7 and 120 μmol/g) developed by EPA (USEPA 2005) to prioritize sediments of concern. For this analysis, ∑SEM was used instead of (∑SEM-AVS) for 16 of the 50 toxicity stations for which measured AVS concentrations were less than 0.1 μmol/g (i.e., the limit of the applicability of the SEM and AVS methodology), and for 12 of the 50 stations for which no AVS data were reported. Results of the analyses indicated that, in general, SEM exceed AVS by an order of magnitude at most of the 2005 stations located downstream of Marcus Flats. However, with respect to ∑SEM-AVS sediment toxicity is unlikely at these same stations. The highest cluster of ∑SEM-AVS results linked to biomass responses on toxicity tests were in Reach 1.

3.2.3.3 Porewater

Concentrations of dissolved metals in porewater collected in 2004 (Paulson et al. 2006, and Besser et al. 2008) and 2005 (USEPA 2006d) were measured to assist in the interpretation of the sediment toxicity results. The purpose was to evaluate which sediments may be toxic to benthic macroinvertebrates based on porewater concentrations.

The potential toxicity of porewater metal concentrations (i.e., arsenic, cadmium, copper, chromium, nickel, lead, mercury, selenium, and zinc) for Phase I sediment samples was evaluated by comparisons to EPA chronic AWQC (USEPA 2006e). For the metals that have hardness-dependent criteria (all of the above except arsenic and selenium), the AWQC was adjusted to the hardness of porewater found at each station using EPA methods (USEPA 2006e). Comparisons to AWQC were conducted using the toxic unit (TU) approach, where TUs were computed as the ratios of the metals concentrations to their respective AWQC. Using this approach, no TUs greater than 1.0 were found for arsenic and mercury, and only a single station exhibited a TU greater than 1.0 for chromium and nickel. Spatial distributions of TUs greater than 1.0 are illustrated in Map 3-3. In general and as illustrated, the results indicate that most or all TUs greater than 1.0 for all metals, except cadmium and lead, were found in Reaches 1 to 3. Based on similar work completed by Paulson et al. (2006), neither arsenic nor zinc concentrations in porewater yielded TUs >1; and only one of seven stations had a copper TU >1 (i.e., 1.6 for sample LR7 [RM 735] located in Reach 1). TUs >1 were reported for cadmium at LR3 [RM 665] (2.8) and LR1 [RM 602] (1.5); for lead at LR7 (1.6) and at three other reservoir sites (ranging from ~1.5 to 3.0).

Comparison of dissolved metals concentrations in porewater with hardness-based WQC suggest that sediment toxicity reported at some locations in the UCR, particularly at upstream riverine stations, may be associated with elevated exposure levels of some metals in porewater. However, other factors affecting bioavailability will be considered as part of an overall weight of evidence to determine risk.

3.2.4 Fish Tissue Chemistry

Evaluation of the pre-2005 (historical) and Phase I fish tissue data identified several spatial and interspecific patterns that may be useful for the BERA. A detailed discussion of this evaluation including the assumptions and statistical tests performed are provided in Appendix F. The following is a summary of the observations:

- Pre-2005 data were primarily for fillet tissue and for species and sizes considered edible by people. A few studies provided data for whole largescale sucker, published in 1997. As such, the data may not be sufficiently systematic, current, or representative of fish and tissues eaten by piscivorous fish and wildlife to have substantial value for the BERA.
- Temporal and interspecific contaminant trends for data collected pre-2005 can be summarized as follows:
  - Concentrations of copper, lead, and zinc in whole largescale sucker declined significantly ($p \leq 0.05$) with increasing distance downstream from Northport. Similar decreasing trends were not observed for arsenic, cadmium, selenium, and mercury. Mercury may show the opposite trend.
  - Concentrations of mercury were higher in walleye and largescale sucker than in other species. Mercury in walleye tissues have declined as a function of time.
- Phase I data may provide the most systematic and robust data set and is useful for the BERA. Phase I data added the following information on contaminant trends:
  - Phase I data identify the highest concentrations of most metals occurred in largescale sucker, burbot and walleye. Consistent with historical trends, the highest mercury concentrations were in walleye and largescale sucker; relatively high concentrations were also detected in burbot, which were not sampled historically.
  - As in the historical data, concentrations of copper, lead, and zinc declined with distance downstream from the U.S.-Canada border, but only for largescale sucker in Phase I data.
- Simple, qualitative comparisons between pre-2005 and Phase I data sets suggest declining tissue concentrations of copper, mercury, lead, and 2,3,7,8-TCDF from the mid- to late-1990s to 2005; while data for arsenic and cadmium are equivocal, due to high detection limits.

3.2.5 Soil Chemistry

As identified within the SLERA, soil contamination may result from three distinct processes. The first is from atmospheric point source emissions (e.g., smelter stacks), the second from hydrologic transport and deposition (i.e., floodplains), and the third from exposed sediments subject to wind erosion. The following provides a summary of interpretations of the existing data for each source/release process, with detailed descriptions and calculations presented in Appendix G.

3.2.5.1 Stack Emissions

The extent of the smelter aerial deposition from stacks was evaluated through an analysis of existing soil data and was designed to help address the following: 1) is there evidence that smelter stack aerial emissions have resulted in elevated concentrations of metals in soils, and 2) what is the spatial extent of any such effects within the UCR study area? Major elements of the approach used to evaluate the spatial extent of aerial deposition of metals from smelter stacks included 1) inclusion of all known surface soil data that are either known or likely to be influenced by aerial deposition (based on proximity) and all surface soil data in the U.S. near the study area that is not known to be influenced by other sources (as defined in the documentation accompanying the NURE database of soil concentrations), 2) use of standard quantitative methods to estimate by interpolation the concentrations of metals in surface soil between sampling locations, and 3) comparison of the measured and interpolated concentrations in soil to screening concentrations for ecological effects. This analysis is detailed in Appendix G.

Based on the above approach, the spatial patterns of cadmium and lead concentrations indicated elevated concentrations were contiguous from Trail to Northport along the UCR valley. Lead concentrations were elevated within the area from the U.S.-Canada border to just south of Northport. Soil lead may reflect emissions from the Le Roi smelter mixed with lead from the Trail smelter within this area, but available data are not sufficient to allow these the two sources to be distinguished. Unbounded elevated concentrations of lead were present to the west of Northport, and there are insufficient data in this region to establish the complete extent of elevated soil lead concentrations.

Zinc and mercury concentrations were elevated in Canadian soils near the Columbia River up to the U.S.-Canada border. Neither metal had available data from soil samples on the U.S. side of the border, and the presence and spatial extent of elevated concentrations in U.S. soil cannot be definitively determined. Based on the overall spatial distribution of zinc and mercury in Canadian soils adjacent to the border, elevated concentrations may also exist in U.S. soil.

Similar analyses were conducted for arsenic and copper and it appears that soil concentrations of these analytes were elevated relative to conservative soil benchmarks in localized areas. Nevertheless, there remains a great deal of uncertainty associated with the nature and extent of contamination within upland soils associated with stack emissions.

3.2.5.2 Relict Floodplains

The extent of potential soil contamination resulting from water-borne contaminant transport and deposition onto relict floodplains was evaluated using a number of geographic information system (GIS) and hydraulic analysis tools. These included:

ArcGIS 9.2 (ESRI 2005), HEC-GeoRAS 4.0 (USACE 2005), and HEC-RAS 4.0 (USACE 2006). As part of this evaluation, the geostatistical method outlined within the modified RI/FS work plan (USEPA 2008a) was refined using site-specific data as indicated in Appendix G.

A primary goal of this analysis was to identify relict floodplain areas along the UCR that may be areas of historical soil contamination. For this analysis, a relict floodplain is defined as an area that may have been subjected to flooding under past flow conditions but that is not expected to be flooded under present flow and pool level management controls. Three flow conditions were relevant to the analysis 1) the average daily post-1973[11] flow at low pool; 2) the average daily post-1973 flow at high pool; and, 3) the maximum reported pre-1973 flow. The water surface extent for the average daily post-1973 flow at low pool defines the minimum area that was expected to be under water at all times (i.e., areas that are always flooded). In continuously flooded areas, sample collection activities would typically need to be conducted from a boat (i.e., sediment sampling). The water surface extent for the average daily post-1973 flow at high pool defines the area that was expected to be periodically flooded (i.e., areas that are sometimes flooded) during the year. In periodically flooded areas, sample collection activities could be performed from a boat when wet or by ground crews when dry. The maximum reported pre-1973 flow was 15,577 m^3/s (550,100 cfs) occurring on June 12, 1948 and as such, defines the largest area where flooding may have occurred. The area between high pool floodplain and the maximum pre-1973 floodplain represents the relict floodplain. Sample collection activities in these relict floodplains could be performed by ground crews (i.e., soil sampling) to verify whether or not flooding resulted in elevated COPC concentrations in soils.

A map showing calculated floodplain extents for daily average flow conditions as well as the historical (relict) floodplain maximum extents from the U.S.-Canada border to Grand Coulee Dam is presented in Map 3-4. These results indicate that in most locations the pre-1973 maximum floodplain was no larger than the present-day floodplain extent at high pool. However, in a few locations (e.g., the area around the Deep Creek near Northport) the relict floodplain is larger than the present-day floodplain. Based on this analysis, relict floodplain areas exist upstream of Little Dalles (approximately RM 729).

3.2.5.3 Wind Blown Sediments

Sediments along the banks of Lake Roosevelt become exposed and dry out during the draw-down period (January through July). This allows smaller particles to be suspended by the wind and blown onto the shores. Over sufficiently long time-frames, some of the sediment-associated COPCs could build up in soils to levels that may pose risk to terrestrial ecosystems. To evaluate whether this has occurred, locations with the

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[11] 1973 is the year that the flood control dams on the Columbia River in Canada were completed.

highest probability of enrichment by wind-blown sediments will be selected for soil sampling. The following section describes the process for the most probable locations along the reservoir in which soil contamination may result due to wind-blown sediments.

It is assumed that areas where the wind blows with most frequency and highest velocity in a direction that takes particles towards shore have a high probability of suspending and blowing sediments. This will be enhanced in areas with the highest amount of small particles, as lower threshold velocities are required for picking up and moving smaller sized particles. Soil enrichment will be highest in areas where there are elevated concentrations of COPCs that have physical/chemical properties that result in binding to soil and very slow leaching processes. Therefore, where these three data sets intersect (i.e., maximum winds, highest percent fines, and highest concentrations of critical COPCs) will be the area(s) of greatest probability of exposure. The process for evaluation of each of these three factors is described in detail within Appendix G. It should be noted that this analysis was conducted for areas located between RM 597 (Grand Coulee Dam) to RM 707 (above Marcus Flats), as this is the area affected by the annual drawdown and has exposed sediments with significant percent fine particles.

Based on this analysis it was identified that Marcus Flats and Seven Bays on the left bank (east side) of the river represent the reasonable worst-case scenario for enrichment of soils by COPCs in wind-blown sediments.

3.2.6 Groundwater

Groundwater data available from within the UCR Site were limited to the NPS database of routine monitoring of wells located within the LRNRA, see Map 3-5. These data were reviewed qualitatively prior to an assessment of the data quality.

The database was examined geographically, temporally, and by reservoir elevation to determine if these parameters were associated with concentrations of measured analytes. For all analytes that were sampled for all years (1978 to 2007), the number of samples detected and non-detected were summarized by standardized site (Table 3-3). For the majority of analytes, the count of total samples analyzed was less than, or equal to, three, with most samples below detection limits. Sample sizes and analytes varied among sites, and no site was sampled for all chemical analytes.

The mean, minimum, maximum, and count of detected analytes at each standardized site are presented in Table 3-4. If a standardized site had only one detected value for a given analyte, its minimum and maximum were not presented in Table 3-4 (non-detected values were excluded). Analytes having the highest number of detected values included nitrate, calcium, sodium, and sulfate.

The average (with standard errors) for detected values of nitrate, calcium, sodium, sulfate, iron, lead, manganese, and zinc are presented in Figures 3-6 through 3-13, respectively (non-detected values were excluded). The standardized sites are presented in order from most upstream (North Gorge) to most downstream (Spring Canyon), going left to right along the x-axis with the depth of screen below the reservoir at full pool noted in feet behind each standardized site name, if available. Data from sites that are not located on the Columbia River (i.e., Kettle River, Porcupine Bay, and Detillion) were excluded from the figures, as were data from sites where the elevation at well screen was above the reservoir elevation (i.e., Fort Spokane No. 2 and Fort Spokane Spring).

No clear upstream/downstream trend was evident, although individual stations showed elevated concentrations (e.g., Camp Na-Bor-Lee). Similarly, no clear trends in the concentrations of nitrate, calcium, sodium, sulfate, iron, lead, manganese, and zinc as function of depth of screen below reservoir were evident. To examine temporal trends, the average concentrations of detected values of nitrate at all sites were plotted as a function of time (Figure 3-14). Data from sites not located on the Columbia River (i.e., Kettle River, Porcupine Bay, and Detillion) were excluded from this analysis. Nitrate was selected for this analysis since it has more detected values than any other analyte measured. Data for other analytes were too limited to allow for a thorough investigation of temporal trends in metals or other analytes. Interestingly, nitrate concentrations at most sites have increased from 1991 to 2004, and limited data since 2004 suggest that nitrate levels have declined again.

Based on the preliminary CSM (SLERA; TAI 2010), groundwater does not pose a direct risk to ecological receptors unless or until it daylights to surface water bodies. Therefore, sampling for the purposes of the RI/FS will be considered after data from the sediment (including porewater), soil, and surface water programs have been obtained, analyzed, and evaluated. Following an integrated review and evaluation of the aforementioned data, should uncertainties remain for the risk assessment, a groundwater investigation may be necessary.

3.2.7 Background Soil and Sediment Chemistry

Characteristics of sediment and soils in the UCR may be strongly influenced by local geological conditions. COIs developed for the Site in the RI/FS Work Plan included a number of exotic metals that have been measured at least once in the study area but that have no known relationship to any specific sources. Sediment and soil data collected by investigators in the Columbia Basin watershed but outside the river itself also include many of these exotic metals.

The purpose of the evaluation detailed in Appendix H, was to identify and describe available sediment and soil data to characterize potential background conditions; which, if required during the ERA process, and per EPA guidance (USEPA 1989, 1997a), may facilitate comparisons of individual chemical measurements from within the Site to regional background and will help inform risk-based management decisions. The distribution of concentrations for each COPC in the background data set was characterized by an upper tolerance limit (UTL), as recommended by EPA (USEPA 1992, 2002c, 2006f) (see Table 1 in Appendix H).

4 PROBLEM FORMULATION

This section presents a refinement of elements of the problem formulation initially provided in the SLERA (TAI 2010, including the Site wide CSM, ecological receptor groups and endpoints. It also describes the process that will be used to complete problem formulation when appropriate data are available for the refinement of COIs and identification of COPCs for each receptor group. The objective of refining the problem formulation is to focus the ecological risk analyses on chemicals most likely to pose risks to one or more receptor groups, and more clearly define the areas where such risks may be occurring. One goal of a BERA within the RI/FS context is to establish acceptable risk thresholds for ecological receptors in different media (sediment, soil, water), acknowledging that ranges of acceptable risk may be media/receptor/habitat specific. What is presented here are the core aspects of the work; another document (the Problem Formulation Plan) is forthcoming that will provide the necessary detail.

The RI/FS work plan presented a Site wide CSM for the UCR (Figure 4-1). The SLERA described the environmental setting, chemicals known or suspected to be present at the Site, physical fate and transport mechanisms, general receptor groups and pathways, and preliminary assessment endpoints. To refine the problem formulation, the Site wide CSM developed previously has been revised to reflect key physical and ecological differences among areas of the Site, and to describe more specifically exposure media and pathways for each receptor group. Assessment endpoints, which have not changed from those presented in the SLERA, are reviewed. As appropriate, major receptor groups are refined to reflect feeding guilds, and surrogate species for each receptor group are identified. For example, birds are divided into aquatic-associated birds and terrestrial birds (each a receptor group); both categories are further refined by feeding guild (e.g., piscivorous birds, herbivorous birds, insectivores); species in each of these categories are selected as surrogates for the purposes of the BERA, such that each species' life history provides the basis for exposure models. These refinements will result in a more Site-specific risk analysis.

The SLERA as summarized within Section 2 of this work plan presented a preliminary list of COPCs. This section describes the approach for refining the COPC list, specifically for sediments, but in general terms for all media. Because the data sets for water and soil chemistry are relatively small, COPCs within these media cannot be refined until after the next Phase of data collection. Refinement of the COPC list for fish and wildlife will incorporate multiple-media exposures. Development of benchmarks for COPCs for which none currently exist, and the results of a subsequent refinement of the COPCs employing more realistic exposure estimates than previously used in the

SLERA, will be detailed and submitted under separate cover as a technical memorandum (see Table 4-1 for a complete listing of potential technical memoranda and data summary reports to be developed during the BERA process). Chemicals without benchmarks, or chemicals with benchmarks that are not appropriate, will be addressed in the uncertainty section of the BERA. Site-specific toxicity data (bioassays) will be used to resolve uncertainty, where needed.

4.1 RISK DRIVERS

Based on information collected to date (refer to Section 3), primary risk drivers at the Site stem from slag discharges, liquid effluent, and stack emissions from the Trail smelter. Based on the analysis presented within the SLERA, this results in the inclusion of major metals (e.g., lead, copper, and zinc), metalloids (e.g., arsenic) and some less common elements (e.g., indium[12]). Additional risk drivers that emerged from the Phase I analyses include selected organochlorines such as dioxins/furans, PCBs, DDT, and methoxychlor.

4.2 CONCEPTUAL SITE MODEL REFINEMENT

A CSM is a simplified representation of relationships among the components of a site that result in complete exposure pathways linking stressors, such as COPCs, to the assessment endpoints (USEPA 1998). The CSM provides a framework within which the complex suite of chemical, physical, and biological processes and interactions at a site can be viewed in a systematic and organized manner. A CSM typically considers the sources of COPCs, the physical transport and chemical reaction pathways that control concentrations of COPCs over time and space, and exposure routes to selected receptor groups. It is a tool for focusing the risk assessment on the most important stressor-receptor group links, and thereby guides development of the analysis plan.

A Site wide CSM for ecological receptor groups and people was presented in the UCR RI/FS work plan (USEPA 2008a) (Figure 4-1). The Site wide CSM presented in the RI/FS work plan was intended to be systematic, broadly (rather than specifically) representative, and conservative (i.e., more inclusive than exclusive). The primary focus was on ecological receptor groups because a CSM specific to human health is being developed by EPA (USEPA 2009a).

As part of the BERA planning process, the Site wide CSM has been refined (where data and information permit) to reflect an improved understanding of the aquatic and terrestrial habitats occurring within the physiographic units of the UCR (refer to Section

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[12] As outlined within Sections 2 and 3 of this work plan, indium has been identified as a COPC. At this time however, no site-specific information has been collected to assess the potential risks.

2 of this work plan), as well as COPCs, fate/transport mechanisms, and receptor groups present in different parts of the Site. The refined CSMs described in this section describe exposure areas for the receptor groups and will enable more effective interpretation of data in the BERA. The CSMs recognize the different physical characteristics within various parts of the UCR that govern exposure pathways, and explicitly create a conceptual framework for ecological risk analysis.

4.2.1 Conceptual Site Models

Recognition of differences in the structural habitat elements, such as hydrology, water level fluctuations, tributary influences, and terrestrial vegetation zones in the CSM provides a framework for a more accurate representation of the distributions of receptor groups throughout the Site. CSM refinement improves characterization of exposures of receptor groups across the large area of the UCR, and identifies differences in receptor groups among locations.

This section introduces five aquatic and two terrestrial CSM. The aquatic CSMs correspond to the major hydrologic divisions of the Site, namely riverine, transitional, and lacustrine, as well as the unique physical characteristics of Marcus Flats within the transitional zone. The plant and wildlife CSMs correspond to two unique habitats 1) the aquatic-dependent CSM describes exposures and receptor groups inhabiting the shoreline (i.e., littoral and semi-aquatic habitats and upland areas occurring within the Site, and 2) the terrestrial upland CSM describing exposures to media and receptors relevant to only upland portions of the Site.

4.2.1.1 Aquatic CSMs

To refine the CSM for the aquatic portions of the UCR, variations were considered in the key physical variables that affect the development of exposure pathways to aquatic receptor groups. These include the channel morphology and hydrology of the Site, along with depth of light penetration, which together control the distribution of habitats (including sediment types) and receptor groups. In light of these considerations, the Site wide CSM has been refined to the following five aquatic CSMs, illustrated in Figures 4-2 to 4-4 (In each figure, only those receptor groups that may occur in that location are shown).:

- **Riverine CSM.** The riverine CSM consists of Reaches 1 and 2, and represents the portion of the Site most representative of a river. It is further characterized by a relatively narrow channel, periodic high-flow conditions, and sediments with relatively large particle sizes, including cobble in many areas. Ecological receptor groups found in this area are characteristic of the kinds of organisms found in riverine systems. This CSM is illustrated in Figure 4-2.

- **Transitional CSMs (shallow and deep).** The transitional CSMs (Marcus Flats) consists of Reach 3, and represent an environment that is riverine in character under low pool conditions, and lacustrine in character under high pool conditions. It is subject to variations in water level such that upland areas adjacent to the aquatic habitats may or may not be flooded at various times throughout the year. In addition to experiencing unique annual hydrologic conditions, Marcus Flats is also unique in that it is a large depositional area of granulated slag (refer to Section 3). Sediments in the historical river channel are generally sand-sized particles and consist of granulated slag, whereas sediments in many of the shallower areas are fine-grained. This CSM is subdivided into two CSM units: shallow (≤80 ft below full pool) and deep water (>80 ft below full pool). The COPC refinement will evaluate sediment concentrations by CSM unit and depth category to determine if COPCs can be further refined spatially. The shallow and deep water zones are illustrated in Figure 4-3a and Figure 4-3b, respectively.
- **Lacustrine CSMs (shallow and deep).** These CSMs consist of Reaches 4 through 6, and represent the lacustrine (reservoir) portions of the Site. The channel is relatively deep in many areas, creating a large deep water zone where sediments are continuously submerged; during drawdown, shallow water zone sediments are exposed, influencing the kinds of ecological receptor groups that can use those areas. As with Marcus Flats, the lacustrine CSM also is subdivided into shallow and deep water CSMs, see Figures 4-4a and 4-4b, respectively.

4.2.1.2 Terrestrial CSMs

Terrestrial areas at the Site include two distinct types, representing different exposure pathways 1) the terrestrial-aquatic interface environment and 2) the truly upland environment. The terrestrial-aquatic interface includes distinct conditions in both terrestrial and aquatic environments that results in a number of complex ecological exposure pathways. Semi-aquatic areas (e.g., shorelines) are accessed by aquatic-dependent birds and wildlife as well as fish and invertebrates and provide direct exposure pathways to contaminated sediments. True upland areas of the Site are identified as terrestrial habitats occurring at elevations above the shoreline and adjacent semi-aquatic areas. Receptor groups in true upland areas will not be directly dependent upon the UCR for their food or water (i.e., exposures to sediment, UCR surface water or aquatic biota are not complete pathways). Terrestrial CSMs defined for the BERA are as follows:

- **Aquatic-Dependent CSM.** Shoreline or semi-aquatic areas of the UCR where ecological receptors are exposed to shallow water zone sediments and true upland areas of the Site. A CSM depicting the relevant ecological receptor groups, exposure media, and exposure pathways is presented in Figure 4-5.

- **Terrestrial Upland CSM.** Figure 4-6 depicts the exposure media, exposure pathways, and ecological receptor groups relevant to upland habitats of the UCR. Specifically, the upland CSM differs from the Site wide CSM in the removal of certain secondary release/transport mechanisms, tertiary sources, tertiary release/uptake mechanisms, exposure media, receptor groups, and exposure pathways that are applicable only to aquatic and semi-aquatic habitats.

4.3 ASSESSMENT ENDPOINTS

An assessment endpoint is "an explicit expression of the environmental value to be protected, operationally defined as an ecological entity and its attributes" (USEPA 2003b). Clearly defined assessment endpoints help tie the BERA to risk management concerns. For practical reasons, it often is useful to identify assessment endpoints that can be measured with well-developed test methods and field assessment techniques (Suter and Barnthouse 1993). Clarity in assessment endpoints is essential to their role in refining the direction of the risk assessment and in communicating the emphasis of studies for the BERA.

Assessment endpoints for the BERA are focused on those ecological values that are relevant to maintaining the structure and function of the UCR biological communities (aquatic and terrestrial) and that should be protected under applicable or relevant and appropriate requirements (ARARs). The criteria used to develop specific assessment endpoints for the UCR are 1) ecological relevance, 2) endpoint sensitivity (susceptibility[13]), and 3) ability to actually measure the endpoint (Suter and Barnthouse 1993; USEPA 1998, 2003b). Ecological relevance of an assessment endpoint can be evaluated by considering the importance of the endpoint to higher levels of biological or ecological organization. For example, a measure of survival of individuals may be clearly tied to the long-term viability of a population, and therefore provides an effective assessment endpoint. An unambiguous operational definition anchors the statement of value in one or more measurable attributes. Finally, the ability to measure an assessment endpoint creates a framework for a specific evaluation that all parties can understand and which has a concrete result that can be considered. Assessment endpoints listed in Tables 4-2 and 4-3 were derived to conform to these guidelines.

The SLERA provided basic assessment endpoints for all receptor groups, given as the survival, growth, and reproduction of the subject receptor group. These assessment endpoints were not changed for the purposes of the BERA because, paired with refinements to the list of receptor groups, these generic assessment endpoints are consistent with the guidelines discussed above, and therefore provide both practical and specific direction required for the BERA. However, other than for some special status

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[13] Susceptibility pertains to the sensitivity of a particular endpoint to the stressor, relative to its potential exposure.

species, risk managers at the Site ultimately are concerned with the potential for COPCs to affect levels of organization higher than the individual organism, such as populations and communities. This is recognized in the preliminary ecological Risk Management-based Action Objectives (RMAOs) articulated in Appendix A of the RI/FS work plan (USEPA 2008a). In that discussion, the preliminary ecological RMAOs for the Site refers to decreasing unacceptable population- and community-level risks associated with the presence of COPCs at the Site.

The terms "population" and "community" in this context refer to groups of organisms that could occupy the UCR at any given time, or could occupy a smaller area (USEPA 2003b). For example, the BERA will recognize the potential for different fish communities in the riverine and lacustrine portions of the Site (and therefore different exposure regimes), but will not interpret risks to a riverine piscivorous fish in terms of the riverine piscivorous fish community across all river systems connected to the Site.

Assessment endpoints of "survival, growth and reproduction" will be interpreted in the context of these resource and risk management goals, consistent with the RI/FS work plan. Employment of population and community concepts in the risk characterization is discussed in Section 8 of this work plan.

4.4 RECEPTOR GROUPS OF CONCERN

The BERA will address several broad categories of organisms potentially exposed to COPCs within the Site that are classified into receptors groups. From these, surrogate plant and animal species are selected that are susceptible to COPC exposure and representative of the broader group.

In selecting receptor groups of concern for the Site, the following criteria were considered (USEPA 1998):

- The receptor group is or could be present at the Site
- The receptor group is important to the structure or function of the UCR ecosystem
- The receptor group is directly related to the assessment endpoints for the Site
- The receptor group includes representatives of one or more of the UCR feeding guilds
- The receptor group is known to be either sensitive and/or potentially highly exposed to COPCs.

The exposure route, or manner in which a chemical enters an organism (e.g., ingestion, inhalation, or absorption), provides the basis for selecting and organizing receptor groups. The use of a broad, taxa-rich category of organisms as a receptor group (e.g., benthic macroinvertebrates) allows the BERA to address the dominant exposure route without requiring that all individual species within a group be assessed. Assessment of all species would not provide practical information for management decisions. This approach applies to macrophytes, benthic macroinvertebrates, fish feeding guilds, and amphibian early life stages in the aquatic environment and to terrestrial plants, soil invertebrates, and wildlife feeding guilds in the terrestrial environment.

This section identifies the receptor groups and their surrogate species for the aquatic and terrestrial environments that meet the criteria discussed above and will therefore be addressed in the BERA.

4.4.1 Aquatic Receptor Groups

Aquatic ecological receptor groups can be exposed to COPCs through contact with and ingestion of surface water, near-bottom water, sediment, and sediment porewater; through ingestion of tissues of other organisms; and through combinations of environmental media via a specific exposure route. Some receptor groups will be exposed primarily through only one environmental medium; while others will be exposed through more than one. The selected aquatic receptor groups provide a complete representation of the general categories of aquatic plant and animal life found in the UCR (refer to Section 2), represent all of the exposure pathways in the aquatic environment, and include the following:

- Aquatic macrophytes
- Benthic macroinvertebrates (including mussels)
- Benthic and pelagic fish (including white sturgeon)
- Amphibian early life stages.

4.4.1.1 Aquatic Macrophytes

Aquatic macrophytes are vascular plants that are rooted in aquatic sediments and can be submerged or emergent. Aquatic macrophytes assimilate COPCs (e.g., Cu^{2+}) through their roots via the porewater and through surfaces of stem and leaf cells via the water column[14] (Jackson et al. 1993; Jackson 1998) (Figure 4-7). Exposures via these media will be considered individually, and will be evaluated in a way that allows direct comparison to available toxicity data. For example, if a COPC has a benchmark for aquatic macrophytes that is expressed as a concentration in surface water, the exposure estimate (from either surface water or porewater) will be presented as aqueous concentrations. Factors affecting exposure of aquatic macrophytes include substrate types, hydrology, and light penetration of the water column. Therefore, aquatic

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[14] Concentrations in bulk sediment will be considered an exposure metric for aquatic macrophytes if an appropriate benchmark is found, and a benchmark based on water or porewater is not found.

macrophyte exposure estimates based on surface water or porewater will include data only for those areas that are suitable for macrophyte growth.

4.4.1.2 Benthic Macroinvertebrates

Benthic macroinvertebrates (including mussels) will be exposed to COPCs via one or more water compartments: surface water, sediment-water interface or porewater via two exposure routes: via respiration and surface membranes—direct contact) and via diet (e.g., particulate COPCs, prey) (Borgmann et al. 2007; Morrison et al. 1996) (Figure 4-8). Particulate COPCs are those associated with living and dead organic matter (i.e., phytoplankton, seston, detritus). Filter feeders (e.g., mussels, sphaeriid clams, larvae of many insects) will be exposed to dissolved and particulate COPCs in water at the sediment-water interface and in the water column (surface waters). Grazers and scrapers (e.g., snails, mayfly larvae) live on the surface of the sediment and rocks, respire the water, and consume aufwuchs (the film of periphyton, microbes and invertebrate larvae on rock surfaces). Infaunal macroinvertebrates—mainly oligochaetes and chironomids—are largely detritivores that live and 'breathe' porewater and feed on particulate COPCs in the sediment detritus (e.g., total organic matter). For the purposes of this work plan, "benthic macroinvertebrate" will be used to generically refer to all types of invertebrates, including those that live on either soft or hard bottom substrates.

4.4.1.3 Fish

The fish species inhabiting the UCR are grouped into species that are mainly bottom-dwelling (which includes white sturgeon) and those that are pelagic. Species that principally live on or are associated with the bottom include burbot, sculpins, suckers, and white sturgeon (refer to Section 2 of this work plan). They also include the fry and juvenile stages of all fish species, which transiently live on, in and near the bottom for protection from predators. As the latter grow, juveniles of the pelagic species (e.g., percids, minnows, centrarchids, salmonids) increasingly forage in the water column (see Section 2). As shown in Figure 4-9, both bottom-dwelling and pelagic species are exposed to bioavailable COPCs in surface water and at the sediment-water interface during their life cycles. For brief periods—when they are larvae and fry—all of the fish species are expected to be exposed transiently (acutely) for hours to days to COPCs in porewater in shallower areas within the shallow water zone. Thus, all fish species are expected to be exposed for different durations to bioavailable COPCs in these three water compartments. Different life stages of different species also will be exposed via several dietary pathways to COPCs in their foods, whether it be the aufwuchs[15] and benthic macroinvertebrates consumed by suckers, the benthic macroinvertebrates

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[15] Aufwuchs refers to the microorganisms, plants, and animals that are associated with hard substrates, such as rocks, in aquatic environments.

consumed by sculpins, or the zooplankton and fish consumed by many pelagic species (e.g., Black et al. 2003; Lee et al. 2006) (Figure 4-9).

Surrogate species have been selected to represent each fish feeding guild (Table 4-4); species selected as surrogates are those known to be relatively abundant in the UCR and those for which detailed diet information is available (refer to Section 2). Species not selected but which are represented by each surrogate fish species are identified in Table 4-4.

4.4.1.4 Amphibians (Early Life Stages)

Ten amphibian species are known to occur in the region, including eight frogs or toads and two salamanders (refer to Section 2). Exposures to early life stages (eggs and larvae) are addressed here (Figure 4-10); while exposures to adults are discussed in sections on terrestrial receptor groups (see below). Amphibian eggs have layers of semipermeable membranes enclosing the ova (Duellman and Trueb 1994), and therefore may be exposed to COPCs in water through contact (diffusion and adsorption). Eggs may also contain COPCs as a result of maternal transfer, which also occurs in fish and other vertebrates (e.g., Barron 2003). Larvae may be exposed by respiration of water or by ingestion of the aufwuchs (periphyton, detritus) and benthic macroinvertebrates. Therefore, surface water, water at the sediment-water interface, and sediment porewater of shallow water zone sediments will be considered the primary exposure media for amphibian early life stages. Concentrations in prey will not be a primary LOE for assessment of exposure to amphibian early life stages, as available toxicity information for this receptor group generally is expressed as water concentrations.

4.4.1.5 Aquatic Dependent Wildlife

Aquatic-dependent wildlife includes those species exposed to COPCs through diets composed of aquatic organisms (fish, invertebrates, and plants) and to a lesser extent in their drinking water. All the exposure pathways to be evaluated are shown in Figure 4-11. Omnivorous aquatic-dependent wildlife (e.g., painted turtle; raccoon) are exposed via exposure to total COPCs in their food (plants, benthos, juvenile fish) and in the surface water whereas piscivorous wildlife (e.g., osprey, mink) are exposed only to COPC residues in their fish prey and in surface water. Herbivorous wildlife (e.g., dabbling ducks, muskrat) are exposed only to COPC residues in aquatic macrophytes and in the surface water. Exposures of invertivorous wildlife—e.g., sandpipers, bats and adult amphibians—are primarily through ingestion of benthic macroinvertebrates and surface water. All wildlife, with the possible exception of some piscivores (e.g., kingfisher, osprey), also incidentally ingest shallow water sediments.

4.4.2 Terrestrial Receptor Groups

As detailed within Section 2, there are approximately 98 mammalian species, 250 avian species, 10 amphibian species, and 15 reptilian species that may be present at the Site. Where appropriate, the same surrogate species for each receptor group (defined by feeding guild) were identified for the two types of terrestrial ecosystems present in the UCR as represented by CSMs (aquatic-dependent and terrestrial, see Section 4.2.1.2). This selection was made to increase efficiency and consistency in the risk evaluations and to allow for the comparative evaluation of exposure and risk across terrestrial habitat areas. The six main groups of terrestrial receptors are:

- Terrestrial plants
- Terrestrial (soil and foliar) invertebrates
- Amphibians (adults)
- Reptiles
- Birds
- Mammals.

A CSM for all upland receptors is shown in Figure 4-12. Terrestrial plants are exposed to COPCs primarily through uptake from the soils via their roots. Some lipophilic or volatile organic chemicals may be taken up through the leaf stomata, but this generally is considered a minor exposure route and there is little information about toxic responses. Therefore, soils will be considered the primary exposure medium for plants, and leaf uptake will not be used in the exposure assessment.

Soil invertebrates are important contributors to decomposition and carbon cycling. The invertebrate community is a multilayered food web, with some species feeding on dead and decaying plant material (detritivores), and other carnivorous species feeding on these primary consumers and detritivores. Some species, such as springtails (e.g., *Folsomia* spp.), have hard, chitonous exoskeletons and receive most of their exposure to chemicals through their diet (Suter 2007). Others, such as earthworms (e.g., *Lumbricus* spp.) or potworms (*Enchytraeus* spp.), are softbodied and may take up some chemicals through the dermis (Suter 2007). However, nearly all toxicity studies have been conducted based on bulk soil concentrations, so the route of exposure to all soil invertebrates is unknown, and exposure will be assessed as concentrations in bulk soil.

Foliar invertebrates are exposed to the COPCs through their diet; thus, they are dependent on uptake of the chemical from the soil into plants and subsequent translocation to above-ground parts. Some aerial invertebrates (e.g., bees, flies) are nectar feeders, but because little data are available on the transference of chemicals into nectar, all exposures to aerial invertebrates will be considered to occur via plant uptake. This will remain an uncertainty in the final risk analysis.

In upland (wholly terrestrial) areas, the exposure pathway to wildlife includes uptake of COPCs from soils by plants or soil invertebrates, and then transfer of the chemical through the food web (although some COPCs, such as many metals, biodilute in the food chain [USEPA 2007a]) and ingestion by the receptor. Incidental soil ingestion that occurs during foraging, or as a result of grooming, may also contribute significant COPC exposures (Barron and Wharton 2005). A minor amount of the chemical may be ingested from wet or dry deposition on plants (including splash-back from contaminated soils). Exposure from drinking water not consumed directly from UCR surface waters is expected to be acceptable. Inhalation and dermal exposures may be minor routes of uptake for most or all COPCs at the Site. However, methods for calculating exposure to wildlife via these routes are not well developed (USEPA 1993), and it is likely that fur and feathers significantly minimize direct dermal contact; therefore, inhalation and dermal exposures will not be used as LOEs for estimates of exposure to terrestrial receptors.

Herbivorous birds and herbivorous mammals (e.g., voles) are exposed only to COPCs associated with the plants they are consuming (Figure 4-12), whereas invertivorous birds (e.g., robin,) and mammals (e.g., shrew) consume soil incidentally while preying upon soil invertebrates. Omnivorous birds (e.g., crows, jays) and mammals (skunk) consume foliar invertebrates, plants, and herbivorous and invertivorous birds and mammals and incidentally ingest soil. The carnivores (birds, mammals, and some amphibians and reptiles), being at the top of the food chain, consume animals as they are available.

Surrogate receptors, and those species which they represent, are summarized by class (bird, mammal, plant, amphibian, and reptiles); CSM (aquatic-dependent and upland); and feeding guild (wildlife only) in Table 4-5.

4.4.3 Special Status Species

As noted within Section 2 of this work plan, there are several special status species (i.e., those on endangered and threatened lists at the federal and state level) that are known to inhabit portions of the UCR. The surrogate, aquatic and terrestrial species that have been selected for the BERA provide adequate representation of exposures to special status species (e.g., Dwyer et al. 2005; Milam et al. 2005), so exposures to these special status species will not be specifically modeled[16].

4.5 CHEMICALS OF POTENTIAL CONCERN

The list of COIs was defined initially in the RI/FS work plan (USEPA 2008a). The SLERA (TAI 2010) identified COIs that could be eliminated from further consideration within broad media categories, and COPCs that required further evaluation. The SLERA

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[16] The exception to this is the white sturgeon, for which site specific toxicity tests may be conducted.

comprised Steps 1 and 2 of the ERA process (USEPA 1997a). Due to the conservative nature of the assumptions used in the SLERA, it is possible that some COPCs, that may pose acceptable risks were screened through to the BERA. Step 3.2 of the ecological risk assessment process allows for review of conservative assumptions used in the SLERA and new information (e.g., Site data or toxicity information) (USEPA 1997a, 2001b).

Refinement of COPCs will be presented to EPA in technical memoranda (see Number 2 in Table 4-1). As part of the refinement, benchmarks will be presented in a technical memorandum (or technical memoranda) for EPA review and approval. These COPC refinement memoranda will be used to synthesize and evaluate existing data to focus the BERA on COPCs for which a) there is adequate information to conclude that ecological risks are acceptable, b) there is not enough information (uncertainty) to make a decision, or c) the information indicates the potential for unacceptable ecological risk. Data used in these analyses will be those that qualified for the baseline data set (see Section 3.3). The procedures for conducting initial COPC refinements by media are discussed in the following sections. It is important to note that these procedures represent initial tools and methods for COPC refinement and are not intended to be exhaustive. Additional refinements of the COPC lists in each media will differ by receptor group; this is described in more detail in Section 5 (Lines of Evidence). As additional site-specific information is collected, further refinement steps will be considered, evaluated and documented within technical memoranda and data summary / data gap evaluation reports (see Table 4-1).

4.5.1 Sediments

The SLERA (TAI 2010) used a simplified screening process wherein maximum concentrations of COPCs in bulk sediments across the Site were compared to conservative screening benchmarks (i.e., threshold effect concentrations). Screening benchmarks were identified from readily available literature or governmental sources. Results of the screening identified COPCs with sediment concentrations above benchmarks and identified numerous COPCs without readily available benchmarks. The SLERA did not account for spatial differences in sediment concentrations or representative receptors (i.e., distinct CSM units), nor did it incorporate results from on-Site toxicity bioassays or porewater chemistry. Therefore, additional data interpretation and chemical refinement will be done to determine which sediment COPCs and which locations should be prioritized for further study. The procedures for conducting additional refinements are detailed below and will be reported in a technical memorandum (see Number 4 in Table 4-1).

4.5.1.1 Exposure Concentrations

As previously mentioned, the SLERA utilized maximum Site-wide concentrations that did not account for differences among CSM units and water depth. In addition, and as described above, CSMs (e.g., Marcus Flats and the lacustrine region) can further be refined to distinguish shallow (≤80 ft below full pool) and deep water sediments (>80 ft below full pool). The COPC refinement will evaluate sediment concentrations by CSM unit and depth category to determine if COPCs can be further refined spatially.

Exposure metrics for each CSM unit will include analysis of the concentrations of individual samples, central tendencies of the distribution and the reasonable maximum exposure (RME) concentrations (e.g., 95 percent upper confidence level [UCL] of the mean). The 95 percent UCLs of the means will be estimated using appropriate statistical tests and follow EPA guidance (USEPA 2002c, 2009b,c). ProUCL (V4.00.04) is a software tool developed by EPA (USEPA 2009b,c) and will be used to estimate 95 percent UCLs of the means. Calculated 95 percent UCL mean concentrations will be compared (by location and depth) to sediment benchmarks to further refine the COPC list. Additionally, individual data points will be plotted by river mile to identify potential outliers or anomalies, and to provide a visual examination of the number of individual locations where measured concentrations are above estimated toxicity thresholds. See further detailed discussion in Section 8.1.

4.5.1.2 Sediment Benchmarks (Inorganics)

The SLERA identified COPCs with and without readily available sediment benchmarks for protection of benthic organisms (macroinvertebrates). For inorganics, the conservative screening benchmarks used in the SLERA were the TEC reported by MacDonald et al. (2000), and the SQSs reported by Ecology (2003). The TEC represents a concentration below which adverse effects are not expected to occur (MacDonald et al. 2000). The SQS is defined as a level below which adverse effects are not observed (Ecology 2003). Since these values are considered conservative for screening purposes and do not represent expected biological effect levels, less conservative and biologically relevant benchmarks will be considered in the refinement and documented in a technical memorandum (see Number 1 in Table 4-1). Benchmarks that will be used to evaluate COPCs will include the PECs, defined as the concentration above which adverse effects are expected to occur more often than not (MacDonald et al. 2000). The probabilities of observing adverse effects have been examined by Ingersoll et al. (2000, 2001) and Long et al. (2006) for different endpoints, groups of COPCs and PEC calculation methods.

Sediment toxicity benchmarks used in the SLERA were restricted to governmental standards and those that did not depend on calculated or modeled values. Consequently, the sediment benchmarks used for the inorganic COPCs were restricted to those proposed by MacDonald et al. (2000). The next iteration of COPC refinement will rely upon additional methods. Literature providing sediment-based toxicity data is unavailable for many of the trace elements on the COPC list (e.g., dysprosium, erbium, europium, gadolinium, praseodymium). However, water-based toxicity data are available for most of the inorganic COPCs from EPA's ECOTOX database and the general literature (e.g., Borgmann et al. 2005). These data can be used because toxicity data for water column species are known to protect benthic species (Ankley et al. 1996) and because data on porewater toxicity can be used to estimate overall sediment toxicity for most COPCs (Di Toro et al. 1991, 2001). Consequently, the available water based toxicity information can be evaluated to estimate secondary chronic values using the process developed for the Great Lakes Initiative (GLI) (USEPA 1995, 2008b; Suter and Tsao 1996; Appendix A to USEPA 2008b). Application of water based toxicity results to sediments is based on the premise that WQC or other benchmarks based on water exposures, when applied to the freely dissolved concentration in sediment porewater, will also be protective of sediment dwelling organisms. This premise is supported by direct comparisons between the Final Acute Values (FAVs) for aquatic and benthic organisms (Figure 4-13; Ankley et al. 1996; USEPA 2000). As shown, the WQC derived for aquatic organisms are very similar to those derived for benthic organisms for both freshwater and saltwater species. Therefore, in the COPC refinement for sediments, primary (USEPA 2006e) or secondary chronic water quality values (USEPA 2008b) will be compared to existing porewater and surface water data to determine which inorganic COPCs can be prioritized for further evaluation in the BERA.

When toxicity data are not available for some inorganic COPCs, such as is the case for thulium, quantitative structure activity relationships (QSARs) have been shown to be good predictors of toxicity for both inorganic and organic COPCs. The QSAR approach involves relating a known chemical or physical property of an organic molecule to its chemical reactivity or biological activity, such as the solubility product constant for inorganics (Biesinger and Christensen 1972; Khangarot and Ray 1989). Once a mathematical relationship has been established between the two, the QSAR can then be used to make quantitative predictions of reactivity or biological activity of a new organic molecule provided that its chemical or physical properties are known.

Quantitative ion characteristic activity relationships (QICARs) are the metal ion analogue to an organic chemical QSAR. In a QICAR, one or more ion characteristics of a metal are used as the known chemical or physical property of the metal ion to construct predictive relationships for metal ion reactivity or biological activity. A number of studies have employed this technique to evaluate ion characteristics and metal associated toxicity (McCloskey et al. 1996; Newman and McCloskey 1996; Newman et al. 1998; Tatara et al. 1998; Biesinger and Christensen 1972; Khangarot and Ray 1989). If QICARs or QSARs are employed, the methods and results of their application will be documented in the technical memorandum (see Number 1 in Table 4-1). If toxicity benchmarks are not identified through the published toxicological literature or through the use of QSAR/QISAR techniques, they will be documented (e.g., technical memorandum), and discussed in the uncertainty section of the BERA in consideration of site-specific data (e.g., standard sediment toxicity tests).

4.5.1.3 Sediment SEM-AVS

As noted within Section 3, SEM-AVS was evaluated in a subset of Phase I sediment samples (those used for toxicity testing). Following EPA guidelines (USEPA 2005), existing SEM-AVS results will be evaluated further along with bulk sediment chemistry, porewater and sediment toxicity results. A preliminary evaluation of the existing data is presented within Appendix E of this work plan. EPA (USEPA 2005) identified two sets of generic thresholds for AVS and SEM parameters that can be used to prioritize sediments of concern for further evaluation. Results of the evaluation will be documented in a technical memorandum (see Number 4 in Table 4-1).

4.5.1.4 Sediment Benchmarks (Organics)

For organic COPCs (as was the case for inorganics), benchmarks for the protection of benthic organisms used in the SLERA consisted of TECs, SQS, and Tier 2 equilibrium partitioning sediment guidelines (ESGs) for the protection of aquatic life (USEPA 2004a). Tier 2 ESGs were derived using the equilibrium partition coefficient method (Di Toro et al. 1991; Fuschmann 2003; USEPA 2004b, 2008b) and the chronic ambient WQC (USEPA 2006e). The ESGs were developed for nonionic organic chemicals and are dependent on the TOC. If PECs or Tier 2 ESGs are available for organic COPCs they will be used again in the chemical refinement. If Tier 2 ESGs were not available from EPA (USEPA 2004a, 2008b) they will be developed using the methodologies detailed in Appendix A of EPA (USEPA 2008b). This methodology is the same as used by the Great Lakes Initiative (USEPA 1995) to develop secondary (Tier 2) WQC (USEPA 1995; Suter and Tsao 1996). Thus, for organic COPCs without an existing chronic WQC (USEPA 2006e) or a secondary chronic WQC USEPA 2004a, 2008b), a Tier 2 criterion will be developed. The methods for identifying and evaluating toxicity literature and deriving secondary chronic criteria will be documented in the technical memorandum (see Number 4 in Table 4-1).

Toxicity data may be lacking for some of the organic chemicals on the COPC list, such that the Tier 2 ESGs cannot be developed using the general toxicological literature. In these cases, QSAR—which are widely employed by USEPA's Office of Pesticide Programs and Toxic Substances (USEPA 2009d[17])—will be used (Figure 4-14). The target lipid model (TLM) is a QSAR method that is applicable to compounds that exert a non-specific toxic mode of action (e.g., narcosis). It is well established that toxic effects are additive for this mode of action, such that once the chemicals are in an organism they all have the same toxic potency (McCarty et al. 1985; Di Toro et al. 2000; McGrath and Di Toro 2009). This is a common mode of action for many organic contaminants and

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[17] See EPA's ECOSAR program: http://www.epa.gov/oppt/newchems/tools/21ecosar.htm

additivity is a reasonable first approximation for risk assessment purposes (Meador 2006). This means that the risk due to narcotic chemicals is due to the dose-additive response of all of the organic chemicals having the same mode of action that are present in a sample (Konemann and Pieters 1996).

The TLM provides a scientifically defensible method for estimating effect levels of non-ionic organic compounds (e.g., Type I and II, or nonpolar and polar, narcotic compounds). The TLM is a critical body burden model, meaning that once a threshold chemical concentration is reached within an organism, then toxic effects will occur (McCarty et al. 1985, 1991). The TLM assumes that a fraction of the lipid content of an organism is the site of action so the critical body burden (CBB) is more specifically referred to as the critical target lipid body burden (CTLBB). The TLM was originally calibrated with effects data for over 250 chemicals and has a database of CBBs for more than 40 aquatic species (Di Toro et al. 2000). The model has been subsequently validated and extended to include algae (McGrath et al. 2004) and the organisms used in wastewater treatment (Redman et al. 2007). More recent efforts have focused on extrapolation of the TLM to polar chemicals such as phenols and anilines (Kipka and Di Toro 2009).

The acute effects benchmarks calculated by the TLM must be converted to chronic effect benchmarks for use in the evaluation of ESB$_{OC}$. The TLM has been adapted for estimating no effect concentrations using acute to chronic ratios (ACR—Stephan et al. 1985) and a statistical extrapolation that accounts explicitly for variation in the organism sensitivity and ACR (e.g., HC5—the hazardous concentration that protects all but 5 percent of the species) (Aldenberg and Slob 1993; Aldenberg and Jaworska 2000). The HC5 is similar to other chronic benchmarks—the final chronic value or FCV (Stephan et al. 1985) and the probable no effect concentration (PNEC)—since it is a statistical extrapolation based on a species sensitivity distribution (SSD) of more than 40 individual species that are included in the TLM database. It provides a conservatively rigorous method for establishing aquatic toxicity reference values (TRVs) (Newman et al. 2000; McGrath et al. 2004). The TLM-HC5 has been successfully applied to soil and sediments using the EqP approach (Di Toro and McGrath 2000; Redman et al. 2008). The TLM-HC5 can be in the same manner as the secondary chronic value (described above) to develop Tier 2 ESGs by the equations detailed by EPA (USEPA 2008b). If no benchmarks are available after evaluating the aforementioned techniques for organic COPCs, then the COPC will be discussed in the uncertainty analysis of the BERA.

4.5.1.5 Frequency of Detection

The frequency of detection (FOD) of COPCs in existing and new Site data will also be evaluated to refine the COPC list consistent with EPA guidance (USEPA 1989, 2001b). EPA guidance provides a list of factors to be considered during the review of data, "given adequate data quality," to determine if the FOD should be applied to COPC refinement (USEPA 2001b):

1. *Influence of random and/or biased sampling on the frequency and magnitude of detected values within the distribution of data*
2. *Spatial and temporal pattern of contaminants identified as low frequency [i.e., could detection patterns be indicative of a potential hotspot]*
3. *Comparison of detection limits with toxicity benchmarks.*

In general, if the data for an analyte are spatially representative of current conditions and meet data quality standards (including levels of detection below benchmark values), the FOD will be considered during the COPC refinement process. Clusters of samples with detectable levels, even if low FOD, will be evaluated further. Frequency of detection analyses will be provided in the technical memoranda for COPC refinement (see Number 2 in Table 4-1).

4.5.1.6 Sediment Porewater

Paulson et al. (2006) and Cox et al. (2005) evaluated porewater in sediments throughout the UCR. Porewater data from both studies are considered reliable estimates of metal concentrations in porewater because steps were made to minimize oxidation during sampling and handling prior to centrifugation and centrifugation was done within a few hours of sampling, eliminating storage effects on metals concentrations. Both oxidation and storage effects on the concentrations and bioavailability of metals in sediments can be profound (De Lange 2008; DeFoe and Ankley 1998; Lasorsa and Casas 1996; Simpson et al. 1998). Porewater results for inorganic COPCs will be evaluated by comparing concentrations to chronic toxicity benchmarks as described above, which will be applicable to benthic macroinvertebrates and fish. For some metals (e.g., cadmium, copper, lead, silver and zinc), the effects of bioavailability on metal toxicity can be considered using the biotic ligand model (BLM). The BLM has been widely used to predict bioavailability trends in surface waters (Di Toro et al. 2001; Santore et al. 2001, 2002; Paquin et al. 2002), with the copper BLM having been incorporated into the WQC for copper (USEPA 2007e). The BLM has also been used to assess metal bioavailability in sediments in the context of the equilibrium partitioning framework (Di Toro et al. 2005). A preliminary evaluation of existing porewater data is in Appendix E; further use of these data will be documented in a technical memorandum for sediment COPC refinement (see Number 2 in Table 4-1).

4.5.1.7 Toxicity Results

Existing sediment toxicity results (i.e., benthic macroinvertebrate bioassays) will also be considered in the COPC refinement technical memorandum (see Number 2 in Table 4-1). Appendix E provides preliminary analyses of the existing sediment toxicity data. A synthesis of the toxicity results with bulk sediment chemistry, SEM-AVS, and porewater chemistry will be conducted in the refinement to determine which COPCs and which locations can be prioritized for further evaluation in the BERA. This analysis will be documented in the technical memo for sediment COPC refinement.

4.5.2 Surface Water

The SLERA identified that data for surface water was lacking for the entire UCR. When surface water data become available, a COPC refinement will be conducted and documented in a technical memorandum. Surface water will be evaluated by CSM unit and depth (surface and bottom water) for risk to fish and other pelagic organisms. Exposure metrics will include analysis of the concentrations of individual samples, central tendency of the distribution and the RME concentration (e.g., 95 percent UCL of the mean). These exposure concentrations will be compared to existing chronic WQC (USEPA 2006e) or secondary chronic water quality toxicity benchmarks developed using the GLI methodology (USEPA 1995; Suter and Tsao 1996; Appendix A of USEPA 2008b). Water-based toxicity benchmarks can be developed using the QICAR and TLM models as described above for sediments. Chemicals for which there are no benchmarks available for COPCs, will be documented in the technical memorandum (see Number 5 in Table 4-1) and discussed in the uncertainty analysis of the BERA.

4.5.3 Soils

Soil data were found to be lacking in the SLERA. When additional soil data become available, a COPC refinement will be conducted and documented in a technical memorandum. Soil data will be evaluated in the upland portions of the Site, relict floodplains, and potential locations of aerially deposited shallow water sediments (i.e., blowing dusts) for risks to terrestrial organisms (plants, terrestrial invertebrates, and wildlife). Exposure metrics will include analysis of the concentrations of individual samples, central tendency of the distribution, and the RME concentration (e.g., 95 percent UCL of the mean). These exposure concentrations will be compared to existing EPA ecological soil screening levels (EcoSSLs) or literature derived soil screening levels using the methodologies EPA developed for the EcoSSLs (USEPA 2003b,c; 2005). The methods to identify and evaluate toxicity literature for use in deriving soil screening levels will be documented in the technical memorandum (see Number 10 in Table 4-1).

4.5.4 Biota

Inorganic bioaccumulative COPCs (e.g., mercury), will be further evaluated and refined against appropriate and applicable toxicity benchmarks or critical body burdens. The organic chemicals can be limited to those that are highly hydrophobic, persistent and lipophilic (e.g., Bruggeman et al. 1984; Fisk et al. 1998, 2000). The SLERA removed from consideration all organic COPCs in tissue with a low bioaccumulation potential (i.e., those chemicals that have a low octanol-water partition coefficient; specifically, those with Log octanol-water partition coefficient [K_{ow}] < 4.0[18]). However, new EPA guidelines (USEPA 2008c) support eliminating from consideration COPCs with Log K_{ow} < 5, because those with Log K_{ow} between 4 and 5 still have low bioaccumulation potential. The Log K_{ow} will not be used for the initial screen of bioaccumulative COPCs. The first sampling event of biota will occur with fish in Phase II studies and will include all organic chemicals with Log K_{ow} > 4, as well as methyl mercury. If no organic chemicals with Log K_{ow} between 4 and 5 are detected in fish tissues, then only organic chemicals with Log K_{ow} > 5 plus mercury will be considered COPCs for benthic tissues (including mussels). Results of this analysis and refinement will be used to assess risks to fish and to their consumers, and will be documented in a data summary report (see Number 3 in Table 4-1).

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[18] The K_{OW} represents the ratio of concentrations in a lipid (fat) substitute, octanol, and in water. For example, a K_{OW} of 10,000 means that the concentration of chemical in octanol would be 10,000 times higher than the concentration in water in a two-phase system at equilibrium. This frequently is reported on a logarithmic scale to the base 10 as "Log_{10} K_{OW}," so 10,000 would be equivalent to a Log K_{ow} of 4.

5 LINES OF EVIDENCE AND MEASUREMENT ENDPOINTS

This section describes the LOEs pertinent to assessing risks to each of the receptor groups identified in the previous section. Each LOE consists of components to estimate exposure (measures of exposure) and components to interpret the potential effects associated with chemical toxicity (measures of effects).

5.1 LINES OF EVIDENCE

An approach that employs multiple LOEs recognizes that a single LOE used in the risk analysis may not provide an acceptable degree of certainty, and therefore evaluates several types of information, including qualitative information, before making final conclusions about risk (USEPA 1998). By using multiple LOEs, the confidence in an estimate of risk can be increased and uncertainty can be clearly stated (USEPA 1998). Combining LOEs may or may not be based on a specific weighting scheme (i.e., Chapman et al. 2002). For this work plan, the LOEs were selected to be closely aligned with the risk questions. The approach iteratively reduces uncertainty as the assessment progresses from general approaches to more specific LOEs directed towards COPCs for each receptor group (Fairbrother 2003; Hull and Swanson 2006).

Initial LOEs will use a comparison of an estimate of chronic exposure to an estimate of chronic effect. As described within Section 4.5.1.1, the estimate of chronic exposure will be the upper 95 percent confidence limit of the mean COPC concentration in a specified medium and pathway. The estimate of chronic effect will be the toxicological benchmark (hereafter benchmark). This LOE may be used in later iterations of the BERA as complete data sets rather than discrete points, thereby generating probabilistic assessments (such as through Monte Carlo analyses—USEPA 1997b). Another set of LOEs will be bioassays that directly provide an estimate of risk based on the responses of the indicator species to a specific soil, sediment or water sample. The final group of LOEs will consist of comparison of estimated risks in the UCR to those calculated for reference sites, and a comparison of COPC concentrations in the UCR to those representing background.

The approaches described below will employ multiple LOEs applied at different phases (tiers) of the BERA. The assessment process for each receptor group is embodied in a simplified process flow diagram. In the next two sections, the LOEs being used for aquatic and terrestrial receptors are described, with corresponding illustrations (i.e., Figures 5-1 through 5-8). Each receptor group analysis follows the same basic approach to sequencing the LOEs. Between each sequential LOE is a SMD where a decision is taken in consultation with EPA of whether or not to proceed to the next step. Consistent with EPA guidance (USEPA 1997b), this may lead to conclude either:

- Acceptable risk
- Insufficient information to make a risk decision (uncertainty), or
- Unacceptable risk.

Depending upon the decision, additional LOEs may or may not be required. However, the general sequence of LOEs (should the risk assessment for any receptor group proceed through all the steps) will be:

- Are there suitable benchmarks for each exposure pathway? If yes, they are used to assess risks and documented in a technical memorandum. If benchmarks are not readily available, this too is discussed in the technical memorandum.
- Which COPCs, media, pathways and receptors do not pose risks based on existing data (and hence are not risk drivers); and which ones require further assessment? Assessment of acceptable or unacceptable risk and uncertainty is discussed in the technical memorandum.
- Which COPCs, media, pathways and receptors do not pose risks based on <u>new</u> data or a combination of new and old data sets (and hence are not risk drivers); and which ones require further assessment? Assessment of acceptable or unacceptable risk and uncertainty is discussed in a technical memorandum.
- For COPCs that appear to pose potentially unacceptable risks, the next to final step is to compare estimated risks to those for reference sites or to background and/or to conduct special risk evaluations, candidates of which are identified below. Regardless, the results of this step are recorded first in the technical memorandum, and then in the BERA, with the latter identifying the COPCs, receptors, media, pathways and locations associated with risks to each receptor.
- The final step will identify the locations (areas), receptors, chemicals of concern, media and pathways where there are / are not risks.

5.1.1 Aquatic Receptors

This section provides an overview of the LOEs for the aquatic receptor groups, including aquatic macrophytes, macroinvertebrates (including mussels), fish, early life stage amphibians, and aquatic-dependent wildlife.

5.1.1.1 Aquatic Macrophytes

As outlined within Figure 4-7 of Section 4.4.1.1, roots of aquatic macrophytes are exposed to COPCs in porewater and the emergent parts are exposed to COPCs in the water column. The first step is to determine whether there are toxicological benchmarks for macrophytes. The literature has come to the conclusion that aquatic plants are protected by toxicological data for fish and invertebrates (Kenaga and Moolenaar 1979;

p. 137 in Rand 1995[19]), so EPA's Tier 1 and Tier 2 WQC will be used as benchmarks. If the benchmarks conclude potentially unacceptable risks from one or more COPCs, then the final evaluation will be based on a determination of whether these risks are greater than those at reference sites.

5.1.1.2 Benthic Macroinvertebrates (Including Mussels)

Benthic macroinvertebrates (including mussels) are known to be exposed to COPCs in sediment and their contact water (porewater, sediment-water interface water [SWIW] and surface water, depending on whether the macroinvertebrates live buried within or on the sediment). The first LOE is comparison of sediment COPC concentrations with toxicological benchmarks (Figure 5-1). Depending on the nature of the COPC, this comparison will consider and include, as appropriate and subject to availability of data, bioavailability adjustments (e.g., SEM-AVS, organic carbon). The next step will assess and consider existing sediment toxicity data and if available, sediment porewater data. Because porewater COPC concentrations will be as high or higher as those in the SWIW, porewater results will be used initially as a conservative surrogate LOE for risks[20]. As for sediments, adjustments for bioavailability will be applied to porewaters as applicable (Di Toro et al. 2001, 2005; Gorsuch et al. 2002; Heijerick et al. 2005; HydroQual 2007). Results will be discussed in the COPC refinement technical memorandum to determine if there is sufficient certainty to make risk-based decisions (see Number 2 in Table 4-1).

The foregoing information will be used to define the scope of the next LOE, namely sediment toxicity bioassays (Figure 5-1) using 28-day exposures with the amphipod *Hyalella azteca*, 10-day exposures with the midge *Chironomus dilutus*, or larval mussel bioassays and reproduction bioassays for *H. azteca* and *C. dilutus*. Sediment bioassays will be conducted to evaluate site-specific relationships between sediment chemistry and sediment toxicity at the Site. A variety of exposure metrics (e.g., total metals, SEM-AVS, SEM-AVS/f_{oc}, BLM parameters) will be considered in this evaluation to maximize the potential for effectively addressing the bioavailability of site COPCs. (HydroQual 2007; Di Toro et al. 2005). Toxicity in sediments from the site will be determined by comparison of endpoints (survival, growth, reproduction) with samples collected from reference areas, using a reference envelope approach. Results will be discussed in a data summary report to determine if there is sufficient certainty to make risk-based decisions (see Number 6 in Table 4-1).

Some elements of a comprehensive sediment toxicity study require additional technical discussion, including the necessity of freshwater mussel sediment toxicity tests (*Lampsilis siliquoidea*) or laboratory measures of uptake of COPCs by oligochaetes (*Lumbriculus* sp.).

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[19] In no EPA water quality criterion are plants the most sensitive taxon.

[20] If there are no risks to benthic infauna from exposure to porewater, it will then be assumed that there are no risks to the epifaunal benthos exposed to SWIW.

5.1.1.3 Bottom-Dwelling Fish

Juvenile and adult bottom-dwelling fish (including white sturgeon) are exposed to dissolved COPCs in the water column and from incidental ingestion of sediments, and their eggs and larvae—including fry—are exposed to water in and just above the bottom. Once they commence feeding, these predators, which comprise all UCR fish species, prey primarily on benthic invertebrates that live on or near the surface of the sediment, including emerging insects. The process for estimating risks from these exposures is shown in Figure 5-2. Exposures to COPCs occur from surface water, porewater, SWIW, incidental sediment ingestion, and prey. Risks from porewater will be estimated prior to measuring SWIW concentrations on the thesis that the latter concentrations will always be higher than those in SWIW. If there are unacceptable risks associated with porewater exposures, then SWIW concentrations will be measured and assessed directly.

The first LOE will compare exposure to COPCs in surface water to EPA Tier 1 National Recommended Water Quality Criteria and Tier 2 criteria numbers (USEPA 1995) benchmarks. Bioavailability adjustments using the BLM will be applied if appropriate. If unacceptable risks remain, COPC concentrations in the UCR will be compared to those in reference (background) areas. If unacceptable risks are still indicated, chronic toxicity tests using sensitive indicators (*Ceriodaphnia dubia* or fathead minnow early life stage) will be conducted. In between each of these steps, results will be discussed in a data summary report to determine if there is sufficient certainty to make risk-based decisions (see Number 7 in Table 4-1, Figure 5-2).

Risks from sediment exposures will be determined for the different species and life stages of bottom-dwelling fish. Bottom-dwelling fish not only include burbot, sculpin, sucker, and white sturgeon, but also the eggs and larvae and fry of many of the fish species that live in the UCR. Most fish begin life on and in the bottom, then the pelagic species (e.g., perch, sunfish, salmonids) gradually live increasingly in the water column as they become less vulnerable to predation. Dissolved COPC concentrations in porewaters will be compared to EPA AWQC (Tier 1) and Tier 2 benchmarks (USEPA 1995). Bioavailability adjustments using the BLM will be applied if appropriate. The next step would be comparisons to COPC concentrations in porewater at reference sites. If unacceptable risks remain, the need for further toxicity testing and community surveys will be considered. In between each of these steps, results will be discussed in a technical memorandum to determine if there is sufficient certainty to make risk-based decisions.

COPC residues in juvenile fish and benthic macroinvertebrates will be measured and compared first to toxicological benchmarks for dietary exposure to larger (older) fish, i.e., toxic doses, to the extent to they are available (e.g., Alves and Wood 2006; Bielmyer et al. 2005; Cockell and Hilton 1988). If unacceptable risks are indicated, COPC residues in fish and benthos from reference areas will be measured, calculated and compared to those estimated for the UCR. If risks remain, dietary toxicity testing (e.g., Hansen et al. 2004; Mount et al. 2006) will be considered in consultation with EPA to further determine causality and comparability to reference locations. For those organic COPCs for which CBBs have been defined, comparisons will be made between the CBBs and fish tissue concentrations using the TLM or empirical data. CBBs will be reviewed by EPA prior to implementation. In between each of these steps, results will be discussed in technical memoranda or data summary reports to determine if there is sufficient certainty to make risk-based decisions.

Some species, particularly suckers, incidentally ingest sediments during foraging. The bioavailability of contaminants in ingested sediments is unknown and will be retained as an uncertainty in the BERA. However, consideration will be given to ingested sediments when assessing risks from dietary exposures.

5.1.1.4 Pelagic Fish

The process for analyzing risks to fish that live in the water column (pelagic region) is very similar to that outlined for bottom-dwelling fish (Figure 5-3). Exposures to COPCs in water is limited to those in surface waters rather than porewater or SWIW. Dietary exposures consist mainly of exposures to COPCs in the fish and zooplankton that live in the water column, although it is acknowledged that a fraction of pelagic fish diets derive from benthic prey. As discussed in Section 2.3, zooplankton, primarily Cladocera, are a prominent food item for many fish species in the UCR, even sculpins (e.g., Black et al. 2003; Lee et al. 2006). However, diets do vary by fish species and life stages. For example, burbot and northern pike minnow eat mostly fish, whereas lake whitefish eat mainly daphnids and mountain whitefish eat mainly benthos; and juveniles of most species eat mainly benthos—including epibenthic zooplankton—when very young and increasingly pelagic zooplankton as they age (Lee et al. 2006). In between each step, results will be discussed in data summary reports to determine if there is sufficient certainty to make risk-based decisions (see Number 7 in Table 4-1, Figure 5-3).

5.1.1.5 Early Life Stage Amphibians

The process for assessing risks to early life stage amphibians is shown in Figure 5-4. Amphibian eggs and larvae are exposed to COPCs in surface water and sediment, and larvae are also exposed through their food. Food concentrations are assumed to be indexed by COPC residues in benthic macroinvertebrates because larval amphibians eat the film of periphyton, larval insects and microorganisms on the sediment and rocks. Toxicological benchmarks for both sediment and water (including porewater) exposures will come from the literature, and if the appropriate benchmarks are unavailable for amphibian eggs or larvae (the most sensitive developmental stage for amphibia), then EPA WQC or Tier 2 criteria will be used, as fish and invertebrates are generally more sensitive than amphibians. If risks to eggs or larval amphibians are identified, then bioassays would be conducted to further identify causality and level of risk (e.g., the Standard guide for conducting whole sediment toxicity tests with amphibians (ASTM 2004).

If toxicity is identified, then further studies will be considered, including a comparison to responses in reference areas. In between each of these steps, results will be discussed in a technical memorandum to determine if there is sufficient certainty to make risk-based decisions (see Number 8 in Table 4-1, Figure 5-4).

5.1.1.6 Aquatic-Dependent Wildlife

Risks to aquatic-dependent wildlife will begin with a comparison of whether doses estimated for COPCs in the diet, drinking water and incidentally-ingested shallow water sediments—termed collectively a total dietary exposure—exceed the corresponding toxicological benchmark for the total diet exposure (Figure 5-5). Dietary doses will be estimated via the food chain models (see below) using all available site-specific data (e.g., sediment, water, and fish tissue concentrations) and available literature-based uptake values for those dietary constituents lacking site-specific data see Section 5.1.2 below). Subsequent LOEs will include measured COPC residues in the diets of the wildlife (first using COPC concentrations in fish and, if indicated, then those in benthic macroinvertebrates). If risks are indicated from the dietary model, then COPC concentrations in the sediments and in surface waters will be compared to background. If unacceptable risks remain in the UCR, then additional investigations will be considered, including tests to adjust for COPC bioaccessibility and bioavailability and surveys to assess the number and composition of the receptors to those at reference sites. In between each of these steps, results will be discussed in a technical memorandum to determine if there is sufficient certainty to make risk-based decisions (see Number 9 in Table 4-1, Figure 5-5).

5.1.2 Terrestrial Receptors

The LOEs that will be used to assess risks to each receptor are set forth below. The types of data needed to reduce remaining uncertainties to acceptable levels will be made in consultation with EPA.

5.1.2.1 Terrestrial Plants

Risks to terrestrial plants initially will be assessed by comparing the upper 95 percent confidence limit of mean COPC concentrations in soil samples to Eco-SSLs (Figure 5-6). If risks in soils are unacceptable, then the bioavailability of each COPC in soil will be evaluated using measures such as pH, total organic carbon, particle size, and cation exchange capacity; and risks re-calculated (USEPA 2007a). If risks continue to be indicated, then COPC concentrations will be compared to background[21]. If soil concentrations exceed background, additional investigations that could be performed would then be evaluated and presented within a technical memorandum. As indicated within Figure 5-6, these investigations may include plant bioassays using sensitive indicator plant species (e.g., lettuce germination [Inaba and Takenaka 2005]). If unacceptable risks remain indicated, further investigations will be undertaken. These may include comparisons of plant communities to reference locations, measurements of COPCs in plant tissues (for comparison to critical tissue levels), or other indications of effects. In between each of these steps, results will be discussed in data summary reports to determine if there is sufficient certainty to make risk-based decisions (Figure 5-6).

5.1.2.2 Terrestrial Invertebrates

Soil invertebrates are exposed to COPCs in soil, and foliar invertebrates to COPCs in the plants upon which they feed (Section 4.4.2). The analytical path for assessing risks to terrestrial invertebrates (Figure 5-7) is similar to that for terrestrial plants (Figure 5-6). Risks to the invertebrates that live in the soil and on plant leaves and stems will be assessed initially by comparing upper 95 percent confidence limit of mean COPC concentrations in UCR soil samples to Eco-SSLs. If risks in UCR soils are unacceptable, then the bioavailability of each COPC in soil will be measured and risks re-calculated (USEPA 2007a). If unacceptable risks remain, COPC concentrations in UCR soils will then be compared to background. If unacceptable risks remain, additional investigations would consider measurements of COPCs in plant tissues (refer to Section 5.1.2.1 above), soil invertebrate bioassays (e.g., using springtails or earthworms), surveys of the invertebrate communities at UCR and reference sites, and/or measurement of COPC residues in indicator species. In between each of these steps, results will be discussed in data summary reports to determine if there is sufficient certainty to make risk-based decisions (Figure 5-7).

5.1.2.3 Terrestrial Birds, Mammals and Herpetofauna (Amphibians and Reptiles)

Birds, mammals, and adult amphibians and reptiles are exposed to Site contaminants primarily through ingestion of food, soil and drinking water; see Section 4.4.2. The analytical methodology for assessing risks to these receptors is shown in Figure 5-8. The first LOE is a comparison of soil concentrations to Eco-SSLs. In the next LOE, areas for which an Eco-SSL is exceeded will assess risks to each of the terrestrial receptors, identified in the terrestrial CSM (Figure 4-6), through modeling the total dietary dose

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[21] As indicated within Appendix H, preliminary background soil concentrations have been identified but do not necessarily represent background concentrations to be used within the BERA. It is expected that additional 'background' soil data will be collected during Phase II soil sampling activities, discussed further in Section 7 of this work plan.

from food, soil and drinking water exposures and comparing it to the corresponding toxicological benchmark. For those species – chemical combinations for which there are no data to generate benchmarks (most notably for the herpetofauna), risks will remain uncertain and will be described as such in the Uncertainty section of the BERA. Initial models will use literature-derived uptake values to estimate concentrations in diets. Soils indicating unacceptable risks are then compared to those in background areas, and if risks remain, then doses will be recalculated based on measured residues in terrestrial plants, invertebrates and small mammals. If the foregoing continue to indicate risks in soil from certain locations, then the need for surveys of animal communities and comparisons with reference locations will be considered in consultation with EPA before finalizing the assessment in the BERA. In between each of these steps, results will be discussed in data summary reports to determine if there is sufficient certainty to make risk-based decisions (Figure 5-8).

5.2 MEASURES OF EXPOSURE AND EFFECTS

Specific measures of exposure, such as concentrations in environmental media or daily ingested doses, and of effects, such as percent mortality in a toxicity test, are used for each receptor group to implement each of the LOEs presented above. Measures are selected to link LOEs to the assessment endpoints and should therefore reflect both the susceptibility of the receptor group and the available or obtainable exposure and response data. This section describes the specific measures of exposure and effects that will be employed in the BERA.

Specific measures that will be used for each of the exposure and response analyses in the BERA are described below and in Tables 5-1 (aquatic) and 5-2 (terrestrial). This section also explains the measures of effects that are available for use in interpretation of exposure estimates.

5.2.1 Measures of Exposure

Exposure pathways and the conceptual framework for consideration of exposure, as well as the exposure routes considered important to each receptor group, are discussed in detail in Section 4. Metrics that will be used to quantify exposures to each receptor group are listed below for both aquatic and terrestrial species. Consistent with EPA guidance (USEPA 1997a), the complete exposure profile that will be provided in the BERA will address both qualitative (how does exposure occur?) and quantitative (what is the mean and variability of exposures?) aspects of exposure.

5.2.1.1 Measures of Exposure to Aquatic Receptor Groups

Aquatic receptor groups are exposed to COPCs through respiration (i.e., via transport of dissolved chemicals across the gills or other surface membranes), ingestion, and direct contact. In many cases, the specific route of exposure cannot be discerned from the available literature, or it is not important to the interpretation of the potential for toxicity, because exposures in the literature are expressed simply as concentrations in water, sediment, or organism tissue. Therefore, measures of exposure selected for the BERA to address aquatic receptor groups include concentrations of COPCs in the following media:

- Surface water (µg/L)
- Water at the sediment-water interface (µg/L)
- Sediment porewater (µg/L)
- Bulk sediment (mg/kg dw)
- Tissue of whole zooplankton, benthic macroinvertebrates, or fish (mg/kg dw; mg/kg lipid weight).

A method for combining several of these metrics for evaluating exposure to fish is described in Section 5.2.1.3. Statistics of these categories of information that will be used to represent exposure point concentrations for each of these media are described generally for aquatic and terrestrial receptor groups in Section 5.2.1.4.

5.2.1.2 Measures of Exposure to Terrestrial Receptor Groups

Terrestrial receptor groups include organisms that are exposed to COPCs through direct contact with soils (invertebrates and plants); or as with aquatic-dependent receptors through ingestion of aquatic media (aquatic organisms, sediments, and water); or ingestion of terrestrial media (terrestrial organisms or soil). Respiration or dermal absorption pathways for wildlife will not be evaluated in the BERA, as discussed in Section 4.4.2. Therefore, measures of exposure selected for the BERA to address terrestrial and aquatic dependent receptor groups will be based on concentrations of COPCs in the following media (generated either from direct measures or by application of uptake and trophic transfer functions derived from the literature; if necessary, site-specific trophic transfer functions may be developed. However, for uptake from soils, it is important to include consideration of bioavailability drivers, such as pH, so that data are not extrapolated inappropriately across the Site.):

- Water (mg/L)
- Shallow water Zone Sediment (mg/kg dw)
- Soils (mg/kg dw)

- Tissue of whole fish (mg/kg dw)
- Tissue of benthic organisms (mg/kg dw)
- Tissue of aquatic plants (mg/kg dw)
- Tissue of terrestrial plants (mg/kg dw)
- Tissue of terrestrial invertebrates (soil and/or foliar) (mg/kg dw)
- Tissues of small birds or mammals (mg/kg dw).

Dose calculations from combinations of one or more of these metrics for evaluation of exposure to terrestrial receptor groups exposed via ingestion of multiple media are described in Section 5.2.1.3. Statistics developed to estimate various exposure point concentrations for each of these media are described for aquatic and terrestrial receptor groups in Section 5.2.1.4.

5.2.1.3 Dietary Exposure Algorithms

Algorithms that will be used in estimating dietary exposure to fish and wildlife (aquatic-dependent or wholly terrestrial) are described in the following sections.

Fish

To evaluate exposure of fish through ingestion of benthos, other fish, and or/sediments, concentrations of COPCs in each ingested medium will be compared to benchmarks expressed as dietary concentrations. Where multiple prey types are likely to be ingested by a fish (e.g., zooplankton and chironomid larvae), the concentration in the overall diet will be calculated using the following algorithm:

$$[COPC]_{diet} = \sum f_1[COPC]_1 + f_2[COPC]_2 + f_n[COPC]_n + f_{soil/sed}[COPC]_{soil/sed} + f_{water}[COPC]_{water}$$

(Eq. 1)

Where:

$[COPC]_{diet}$ = concentration of the COPC in the overall diet (μg/kg dw)

$[COPC]_{1\ldots n}$ = concentration of the COPC in the prey items 1 through n (μg/kg dw)

$[COPC]_{soil/sed}$= concentration of the COPC in incidentally ingested soil or sediments (mg/kg)

$[COPC]_{water}$= concentration of the COPC in ingested water (μg/L)

$f_{1\ldots n}$ = fraction of prey items 1 through n in the overall diet (unit less), based on mass, the sum of which does not exceed 1.

$f_{soil/sed}$= fraction of the diet comprised of incidentally ingested soil or sediment (unit less)

F_{water}= fraction of the diet comprised of drinking water

The fraction of different foods in the diet of fish found in the UCR is summarized in Table 5-5, as provided by the LRFEP (1999 to 2005). These data will be used to populate the fish exposure equation.

Wildlife

To estimate exposures to wildlife, the cumulative daily dose to each wildlife receptor group through ingestion of food and water, including incidental soil or sediment ingestion, will be calculated using the general equation:

$$Daily\ Dose = \left(\left(FIR \times C_{food} \times ABS_{food}\right) + \left(WIR \times C_{waer}\right) + \left(SIR \times C_{sed} \times ABS_{sed}\right)\right) \times AUF \quad \text{(Eq. 2)}$$

Where:

Daily Dose = COPCs ingested per day via food, water, and sediment (mg/kg body weight/day)

FIR = food ingestion rate (kg food dw/kg body weight/day)

C_{food} = concentration in prey items (mg/kg food dw)

ABS_{food} = bioavailable fraction absorbed from ingested prey items (unit less)

WIR = water ingestion rate (L water/kg body weight/day)

C_{water} = concentration in water (mg/L water)

SIR = soil and/or sediment ingestion rate (kg soil and/or sediment dw/kg body weight/day)

C_{sed} = concentration in soil and/or sediment (mg/kg soil and/or sediment dw)

ABS_{sed} = bioavailable fraction absorbed from ingested soil and/or sediment (unit less)

AUF = area use factor (unit less); fraction of time that a receptor group spends foraging in the UCR relative to the entire home range

Exposure factors (e.g., water and food ingestion rates, dietary preferences, body weights) have been evaluated for each receptor group (i.e., using surrogate species) based on data compiled in the EPA's *Wildlife Exposure Factors Handbook* (USEPA 1993) and other ERAs conducted within EPA Region 10 (Table 5-6). Food ingestion rates for species not in the *Wildlife Exposure Factors Handbook* (USEPA 1993) can be estimated using equations presented in Nagy (2001) (Table 5-7). Absorption factors (ABS) will be considered as 100 percent unless there are data that suggest otherwise; when data are lacking, this will be described in the uncertainty section of the BERA. As described above (Section 5.1), dietary exposures will initially be based on measures of abiotic media (soil, sediment, water) and literature-based uptake factors to estimate tissue concentrations (Sample and Suter 1994, 1998). For piscivorous wildlife, measured fish tissue concentrations are available for large fish and will be measured for smaller fish.

Later phases will consider direct measures of COPCs in all food chain organisms for those COPC-receptor pairs that are not screened out.

5.2.1.4 Exposure Statistics

For each of the exposure metrics listed for both terrestrial and aquatic receptor groups, statistics to express exposure may include the following:

- The toxic concentration specific to a sample. This metric is used for interpretation of soil, sediment, and water bioassays.
- An expression of the central tendency (CT) of the data for a COPC in any given media. The best expression of the CT will be dictated by the use to which it will be put. Candidates include the median, arithmetic mean, or geometric mean. Contact exposures (e.g., using soil concentrations to provide exposure estimates for plants) will be calculated using geometric means. Dietary exposures will be estimated using the arithmetic mean. All comparisons to the mean will be made using the 95 percent UCL of the mean as a conservative estimate (reasonable maximum exposure).

Although the primary exposure metric will be a measure of central tendency in the deterministic risk assessment tier of analysis, all data sets will be examined for degree of variability and outliers. This will provide an initial data check for the possibility of localized areas of elevated concentrations and permit probabilistic evaluations of risk

Because fish and wildlife species have different sized foraging ranges, the exposure metric also includes an area use factor (AUF). Some fish, such as kokanee and walleye, are found throughout the Site; other fish, such as sculpins, have very localized foraging areas. Similarly, wildlife may forage widely (e.g., osprey or bald eagles) or more locally (e.g., mink or river otter). Therefore, the exposure area and the associated AUF will be defined as follows.

Aquatic receptors. Water exposures to pelagic fish will be averaged within each CSM unit, using only the data from the upper portion of the water column (i.e., not for water collected within the deep water zone or the deep river channel). Bottom-dwelling fish and benthic macroinvertebrate water exposures will be averaged within each CSM unit using porewater concentrations or SWIW if needed. Sediment and biota exposures will also be averaged within CSM units, with additional differentiation of mean COPC concentrations in shallow and deep water sediments in the lacustrine sections. Because the CSM units are sufficiently large, it will be assumed initially that each species could forage entirely within a single CSM (AUF = 1). This may provide conservative estimates from some species that travel throughout the whole river if one CSM has higher concentrations than the others. Later risk assessment tiers will re-examine the AUF assumptions if needed.

Wildlife. Foraging range sizes for wildlife are available in the EPA's *Wildlife Exposure Factors Handbook* (USEPA 1993) and other ERAs conducted within EPA Region 10. For aquatic dependent wildlife, exposures will be estimated for each CSM unit (riverine, and shallow lacustrine) using AUFs of 1. For upland species, the initial sampling area is sufficiently large to encompass all foraging sizes, so AUF = 1 will be used in the Phase II analysis. If smaller areas then become the focus for more detailed analysis, the foraging area of the species will be taken into account, and the AUF will be calculated as the ratio: foraging area/sample area.

5.2.2 Measures of Effect

Consistent with the assessment endpoints described in Section 4.3, effects on all ecological receptor groups will be measured in terms of endpoints that have obvious links to effects on the survival, growth, or reproduction of the species populations occurring in the UCR, following Gentile and Slimak (1992).

5.2.2.1 Categories of Effects Measures

Measures of effects will be derived for risk analyses in the BERA using the following two broad categories:

- **Site-specific toxicity tests.** For the assessment of risk to benthic macroinvertebrates (in Phase II) and possibly soil invertebrates, plants, or fish (in later Phases), Site-specific toxicity tests (bioassays) will be conducted. Because standard procedures are preferred, the measures of effects will be dictated by the metrics that have been established in the standardized method. Determination of the significance of a response in a Site-specific toxicity test will be made consistent with available EPA guidance (e.g., USEPA 2000). Statistical comparison of responses (e.g., growth as ash-free dry weight, percent survival, reproduction) in test organisms (i.e., surrogates for receptor groups) exposed to Site media with those of corresponding reference area[22] sediments to detect significant differences will be one measure of effect. The use of the bioassay results will be contingent upon achievement of test-specific performance standards (e.g., attainment of minimum performance criteria for the control test groups).
- **Empirical and Modeled Benchmarks.** Data on survival, growth, or reproduction of either aquatic or terrestrial organisms tested in the laboratory (preferred) or field (taking into account confounding effects of mixtures of toxicants or other stressors) will be used to develop effects benchmarks.

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[22] A reference area is defined as a location that is as similar to the Site as possible in all physical, chemical, and biological attributes, without any of the COPCs being present.

5.2.2.2 Site Specific Bioassays

Standard protocols available for conducting bioassays for sediment and soil organisms are listed in Table 5-8.

5.2.2.3 Toxicity Benchmarks

This section provides an overview of the process for developing toxicity benchmarks and applying them in the BERA. The general term "benchmark" is used to describe threshold values as point estimates (or distributions) of chemical doses (or concentrations) that indicate a level of exposure to a defined species and life stage, taxonomic group, or community where the measured response is not statistically different from a control group, or is the lowest exposure level where adverse biological effects occur. Specific types of response statistics include "no-observed-adverse-effect-concentrations or levels (NOAECs or NOAELs)," "lowest-observed-adverse-effect-concentrations or levels (LOAECs or LOAELs)," and percent effect levels (e.g., LC_{20}), as well as values derived for multiple species using a broad array of data sets, such as AWQC, PECs, and Eco-SSLs. Published regulatory benchmarks will be used where available, and others will be developed from literature and toxicity databases as needed in accordance with previous guidance (USEPA 2004a, 2005, 2008b; USACHPPM 2000) and current state-of-the-practice (Allard et al. 2009), refer to Section 4.5.

Benchmark Classes

Benchmarks will be applied in a variety of ways during the risk assessment process. Benchmarks will be used to evaluate COPC concentrations in environmental media (refer to Section 4.5) or to evaluate the total dietary intake of a COPC by a particular receptor group (or a surrogate species that represents a particular receptor group). The types of benchmarks that will be used in the BERA include:

- **Media-based benchmarks for groups of organisms.** Benchmarks (e.g., Tier 1 and 2 AWQC, SQGs, Eco-SSLs) represent a concentration in a specific environmental medium that incorporates information from multiple toxicity data sets for a variety of species. Typically, benchmarks represent a conservative value below which it is highly unlikely for unacceptable risk to occur. AWQC are protective of 95 percent of the species, and "take into account such additional factors as social, legal, and economic considerations" (Stephan et al. 1985). As discussed in Section 4.5, media-based benchmarks will be used as a LOE to refine which COPCs, receptors, pathways and media do and do not pose risks and will be documented in the benchmark technical memorandum (See Number 1 in Table 4-1).

- **Critical body burdens.** Benchmarks for selected organic or organometallic compounds that are expressed as a concentration in a whole organism and that indicate exposures associated with adverse effects on growth, survival, or reproduction are referred to as CBBs (McCarty et al. 1985, 1991). Meador (2006) critically reviewed the literature on CBBs, identifying which COPCs reliable benchmarks are and are not available. CBBs based on the whole body concentration of an individual COPC in a particular organism will be used in the BERA to assess risks to fish and, possibly, to plants and higher trophic level wildlife. CBBs will be used if a clear exposure-response relationship has been demonstrated in the literature. The relation of body burden to biological effects has not been validated for metals in aquatic organisms (with the exception of organometals such as tributyltin and methylmercury) and will not be used (Barron et al. 2002; USEPA 2007a).

- **Dietary benchmarks**. Benchmarks for dietary exposure of fish and for wildlife are commonly developed as a daily ingested dose (mg/kg-day) of a COPC that is associated with a NOAEL or LOAEL, or a proportional response (EC_x) for growth, survival, or reproduction. This benchmark can then be compared to a rate of daily ingestion of a COPC from all environmental media within the diet of a wildlife species. Similar dietary benchmarks can be derived for fish exposed to selected metals. Derivation of dietary benchmarks is discussed further below.
- **Media-based benchmarks for individual organisms.** These benchmarks represent concentrations of individual chemicals in abiotic exposure media (water, soil, sediment) or in the food of the tested species. This type of benchmark is expressed as a concentration in an environmental medium that indicates a NOAEC or a LOAEC or an ECx for a specific organism. Unlike the media-based benchmarks for groups of organisms, it addresses a narrow group (aquatic algae) or, more often, a species such as reported in a standard toxicity test. These individual benchmarks may be used for detailed evaluations of surrogate receptors (e.g., fish or mussels) if other benchmarks do not aid in risk evaluation.

It is possible to derive chronic no effect benchmarks for many inorganic and organic chemicals, including SVOCs and pesticides, using a variety of well-established methods available in the literature. Quantitative structure-activity relationships have been widely used for more than three decades to estimate NOAELs for untested chemicals based on physicochemical properties and toxicity data for closely-related chemicals. The methods proposed for use in the BERA—e.g., the target lipid model, equilibrium partitioning, and regressions based on solubility product constants—are described in Section 4.5.1. Detailed approaches for these and derivation of benchmarks for other media (e.g., CBBs, wildlife diets) will be provided in the technical memorandum (See Number 1 in Table 4-1). It is expected that not all COPCs will have toxicity information available for all receptor groups.

5.3 UNCERTAINTIES

Note that we differentiate between uncertainties and data gaps. Data gaps are pieces of information that are needed to assess risk for which it is relatively simple to derive site-specific data (for example, soil concentrations). Uncertainties are those pieces of information for which there are no (or very little) scientific data existing from the Site or elsewhere, and for which it would be difficult to generate information (for example, TRVs for reptiles). Data gaps are addressed in Section 6, and Section 8.4 addresses how uncertainty may be described in the BERA.

Multiple LOEs will be used in the BERA, including risk quotients, modeling results, and field observational studies. For this work plan, the LOEs were selected to be closely aligned with the risk questions. An interactive approach is used that reduces uncertainty as the assessment progresses from general assessments using literature-derived values to more Site-specific LOEs directed towards specific COPCs for each receptor group. The amount of tolerable uncertainty depends upon the significance of the assessment endpoint, from both an ecological and a sociological perspective and also on how close the exposure concentration is to a threshold of effect. If the exposure concentration is far above or below the threshold, then it does not need to be measured as precisely as when it is closer to the threshold, as increased precision is unlikely to change the determination of risk.

The choice of which measure of exposure to use depends on the risk question, the risk analysis method, and the degree of uncertainty in the overall risk calculation that is considered acceptable. In the final iterations of the risk analysis, probabilistic risk calculations may be performed as a way of clearly assessing the effect of uncertainties. In this case, exposure will be expressed as a probability distribution that is appropriate to the level of detail available for the exposure media or exposure metric of interest. Decisions about where to apply probabilistic versus deterministic approaches will be made during the analysis phase of the BERA in consultation with EPA.

6 DATA GAPS AND STUDIES – PHASE II

By defining risk questions and LOEs required to address them, and contrasting them with the available data, specific data gaps can and have been identified. Section 3 described existing information for the UCR, including reviews of the data quality and its relevance to the BERA. In addition it synthesized information to describe what is known and identify where additional data are needed to complete the BERA. Based on the aforementioned data analyses and in consideration of risk questions presented within Section 4 and LOEs presented within Section 5, the following section identifies the resulting data gaps and presents data collection activities that will be completed during Phase II of the RI/FS.

The data collection activities proposed for Phase II are not intended to be exhaustive. Rather, Phase II data collection activities will refine our understanding of risk drivers and determine which LOEs are most important for risk analysis. Therefore, data collected during Phase II will be used to evaluate the potential for unacceptable ecological risks by means of the LOEs laid out within Section 5.1 and shown in Figures 5-1 through 5-8; following this analysis, additional future studies will then be identified and conducted as needed to complete the BERA. The breadth of data collection activities will therefore evolve as the project proceeds and LOEs are collected. Following successive data collection activities, the data will be evaluated and presented in data summary and data gap reports (see Table 4-1).

Aquatic and terrestrial receptor groups for which risk will be assessed are listed in Tables 4-2 and 4-3, respectively. These include: benthic macroinvertebrates (including mussels), aquatic macrophytes, fish (including white sturgeon), amphibians (early life stages and adults), reptiles, birds, mammals, terrestrial plants, and terrestrial invertebrates. Exposure media to be addressed by the BERA for each receptor group are addressed by the measures of exposure listed in Tables 4-3 and 4-4. The following sections are intended to identify the additional data collection that is required, based on the synthesis and evaluation of existing Site data (as described in Section 3.3 and its associated appendices). Refer to Section 9, Figure 9-1 for the sequence, schedule, and dependencies of the Phase II studies.

6.1 SURFACE WATER QUALITY

Most of the current and qualified (1996 to present) surface water quality data for the Site consists of non-COPC measurements (e.g., temperature, conductivity etc.) (e.g., Lee et al. 2006). Extensive monitoring of several metals has been done by Ecology and USGS, but only at Northport, Washington). As noted within Sections 3.2 and 3.3, data from the Northport, Washington water quality monitoring station is limited to a handful of metals (e.g., arsenic, cadmium, chromium, copper, lead, nickel and zinc) and other water quality parameters, and has not measured organic chemicals since September 2000. With the exception of a sample collected at the City of Grand Coulee drinking water intake (USEPA 2003a); and PBDE measurements within Marcus Flats (Johnson et al. 2006), data on concentrations of organic COPCs in surface waters that are of good quality and representative of all CSM units are extremely limited. As a result, primary surface water data gaps that affect completion of the BERA are the absence of data on concentrations of all COPCs 1) Site-wide, and 2) representative of CSM units, and 3) encompassing the range of annual hydrologic flow conditions (low- and high-flow periods).

As illustrated within Figures 4-7 through 4-12, COPC concentrations in surface water can and will be used to characterize one of the exposure routes for a wide range of aquatic and wildlife receptors. These include: fish (white sturgeon inclusive), benthic macroinvertebrates (includes mussels), and aquatic-dependent wildlife[23]. Because these data are presently lacking, surface water data will be collected as part of Phase II data collection activities. Information required to address existing surface water data gaps include measurements of:

- Dissolved and total concentrations of all COPCs (inorganic and organic) identified by the RI/FS work plan (USEPA 2008a) and by the SLERA in representative reaches (i.e., CSM units).
- Dissolved and total concentrations of COPCs spanning annual high flow and water level conditions. Based on an analysis of the annual hydrologic flows (as measured at the upstream boundary of the Site), these conditions will occur during the following time periods: Event No. 1 (October 8 to 22, 2009), Event No. 2 (March 28 to April 8, 2010), and Event No. 3 (May 27 to June 10, 2010).
- Conventional and nutrient water quality data to interpret bioavailability of COPCs (e.g., alkalinity, hardness [as $CaCO_3$], total dissolved solids [TDS], TSS, TOC, dissolved organic carbon [DOC], pH, silica [as dissolved SiO_2]).
- Stable isotopes (i.e., deuterium and ^{18}oxygen) to interpret the homogeneity in mixing of water within the UCR.
- Radionuclides (uranium^{238}) in near beach (shallow water sediment) sampling locations.

In addition to the above-listed analytical requirements, field measurements of water temperature, pH, dissolved oxygen, conductivity, turbidity, and oxidation-reduction potential will also be measured to interpret surface water quality data.

The Phase II surface water sampling program will not only consider the above-listed data gaps, but will also consider receptor-specific exposure routes. For example, at each

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[23] As described in Section 4.4, each receptor group has multiple routes of exposure. Because data gaps and studies are identified by individual media (i.e., single exposure route), it will take multiple studies to complete the information required to completely characterize exposure to the receptor groups

transect (Map 6-1)[24], individual near-surface (≤ 1 m) water samples, and an equivalent number of near-bottom (≤ 1 m) water samples will be collected and analyzed. Similarly, near-shore samples will be collected which can be used to evaluate risks.

Key questions to be addressed by the Phase II surface water sampling program include:

- Do COPC concentrations in surface water exceed state, federal, or Tribal water quality benchmarks?
- Do COPCs in surface water pose an unacceptable risk to aquatic life and wildlife through direct contact, ingestion, or respiration?
- Do COPCs in surface water pose an unacceptable risk to human health through dermal contact and ingestion?
- Do COPCs in surface water pose an unacceptable risk to aquatic life and wildlife through food chain transfer?
- Do COPCs in surface water pose an unacceptable risk to human health through food chain transfer?

At the time of writing of this work plan, the above-listed program has been conducted, following an EPA-approved QAPP. Three sampling events (October 2009, April 2010, and June 2010) were conducted to collect information at different flow regime/pool levels. Water concentrations are compared to ambient WQC (chronic), or equivalent as described in Section 4.5.2, in order to address the above-stated risk questions. Those COPCs without criteria will be carried forward as uncertainties in the BERA (see Section 8.4).

6.2 FISH TISSUE CHEMICAL CONCENTRATIONS

As described within Section 2.3, a large amount of information has been collected on characteristics of the fish community (e.g., species composition) and fish diets throughout the UCR. Similarly, and as outlined within Sections 3.2 and 3.3, a significant amount of fish tissue chemistry is available for a wide range of species and feeding guilds for large-bodied (i.e., >30 cm) fish. Although this information will be extremely useful in evaluating potentially unacceptable risks to people in the HHRA, it is less useful for determining dietary exposures for piscivorous fish and wildlife (e.g., walleye, kingfisher or mink) that consume smaller whole fish (not fillets). In addition, and as identified within the SLERA, although the Phase I fish tissue data set is comprehensive and representative of Site CSM units, a number of COPCs have not been analyzed to date. Specifically, these include: a number of the non-TAL metals, PCB congeners, organochlorine pesticides, PBDEs, and SVOCs. Therefore, additional fish tissue

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[24] Transects have been placed to ensure that CSM units are well characterized.

chemistry for sizes and species that are consumed by fish and wildlife are needed to support the risk questions:

- Are concentrations of COPCs in fish tissue greater than concentrations of COPCs associated with effects on the survival, growth or reproduction of these fish?
- Are concentrations of COPCs in fish tissue greater than dietary benchmarks for survival, growth, or reproduction of piscivorous birds and mammals?

The HHRA also requires additional data collection on COPC concentrations in fish tissues. Previous field sampling efforts targeted the sport fish species commonly consumed by people (e.g., walleye), while other fish (e.g., burbot) that are consumed less often were not sampled or were collected with less frequency during Phase I (see Appendix F).

Based on analyses presented within Section 3.3, a number of spatial and trophic relationships were observed by evaluating Phase I data. Specifically, metal contaminant concentrations were highest in the largescale sucker (a bottom-dwelling species); with whole body concentrations of copper, lead, and zinc significantly ($p \leq 0.05$) declining with increasing distance downstream from Northport, Washington; while a similar decreasing trend was not observed for arsenic, cadmium and mercury. As noted within Section 3.3, similar trends were observed in sediment contaminant concentrations (e.g., copper, lead, and zinc concentrations decreasing as a function of river mile from the U.S.-Canada border).

To address data gaps described above, a fish tissue sampling program will be conducted as part of Phase II activities. As with Phase I fish tissue sampling, samples will be collected throughout the Site representing all CSM units (Map 6-2) and will be completed in September through October 2009.

Phase II fish sampling will target fish species that represent varying feeding guilds and prey for fish, wildlife and/or people. Fish will be collected in three target size classes (<15, ≥15 to ≤30, and >30 cm). The two smaller size classes are intended to collect fish species that provide data for the evaluation of risk to wildlife and fish species, while the largest size class will provide data to support the human health risk assessment and the ecological risk assessment for a few wildlife receptor groups.

For each size class, targeted fish species within respective feeding guilds will include the following:

Size Class ≤15 cm

The goal is to collect six different species from three feeding guilds to achieve representation across guilds.

- Primary species
  - Omnivore—yellow perch

  - Insectivore—rainbow trout
  - Benthivore/detritivore—largescale sucker
- Secondary species
  - Omnivore—bluegill
  - Insectivore—whitefish
  - Benthivore/detritivore—longnose or bridgelip sucker
- Tertiary species (may include)
  - Omnivore—redside shiner, crappie, pumpkinseed, and smallmouth bass
  - Insectivore—pikeminnow
  - Benthivore/detritivore—sculpin.

<u>Size Class >15 to ≤30 cm</u>

Six species from three feeding guilds are targeted to achieve representation across guilds.

- Primary species
  - Benthivore/detritivore—largescale sucker
  - Insectivore—kokanee
  - Piscivore—walleye
- Secondary species
  - Benthivore/detritivore—longnose or bridgelip sucker
  - Insectivore—lake whitefish
  - Piscivore—smallmouth bass
- Tertiary Species (may include)
  - Benthivore/detritivore—sculpin
  - Insectivore—mountain whitefish
  - Piscivore—pikeminnow.

<u>Size Class ≥30 cm</u>

- Walleye - Fillet (with skin) and remainder (i.e., head, viscera, fins, skeleton and musculature not obtained with the fillet) composites, mercury concentrations will be determined on an individual basis as well
- Burbot - Fillet (with skin) and remainder composites
- Smallmouth bass - Fillet (with skin) and remainder composites, mercury concentrations will be determined on an individual basis as well
- Largescale sucker - Fillet (with skin), gut tissue (without gut contents), and remainder composites

- Lake whitefish - Fillet (with skin) and remainder composites
- Rainbow trout - Fillet (with skin) and remainder composites
- Kokanee - Fillet (with skin) and remainder composites.

6.3 SEDIMENTS

As discussed within Section 3.3, despite the large amount of information that exists for COPC concentrations (i.e., bulk chemistry) in surface sediments, there remain data gaps for many COPCs. The SLERA identified the following as data gaps: 1) no concentration data for a number of organic (e.g., PBDEs) and inorganic (e.g., gold, indium, and many of the minor metals) COPCs; 2) spatially limited COPC data (e.g., there are no data on concentrations of dioxins/furans within the deep water zone); or elevated analytical detection limits preventing an evaluation of risks (e.g., some of the SVOCs). Similarly, despite elevated concentrations of SEM metals within deep water zone sediments, the existing data set lacks information to assess and evaluate the bioavailability of these COPCs (e.g., SEM–AVS /f_{OC}).

6.3.1 Spatial Refinements

Three distinct sediment classes can be distinguished within the Site (see Section 3.3)[25]. These distinct classes help to 1) define and delineate the nature and extent of sediment contamination (e.g., presence of granulated slag), and 2) focus future sediment sampling activities. This provides a plausible explanation for the consistently identified longitudinal trends for a number of COPCs (e.g., copper, lead, mercury, and zinc) (see Section 3.3.2). Sediment classes, in conjunction with a number of other properties (e.g., channel morphology, hydrology) and exposure pathways (i.e., receptors) can be used to refine the aquatic CSM (see Section 4.2.1.1). Sediment sampling designs will account for this refinement to ensure that data collected are directly applicable to evaluation of risks to the appropriate receptor groups under reasonably worse case environmental conditions.

6.3.2 COPC Refinements

Phase II sediment sampling activities will consider COPC refinements as discussed within Section 4.5.1. These refinements will evaluate existing data in the context of 1) acceptable ecological risks, 2) insufficient information to make a risk decision (uncertainty), or 3) unacceptable ecological risk, and will be presented in the form of the COPC refinement technical memorandum for inclusion in the Sediment Sampling QAPP

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[25] Sediment classes within the UCR are not only chemically distinguishable based on zinc to vanadium ratios, but can also be effectively distinguished based on sediment grain size (i.e., class I sediments are predominantly coarse sand-sized).

and the BERA. These refinements will focus data collection by optimizing the scope of the studies and focusing on risk drivers. In other words, by performing this analysis, areas within CSM units may be identified as having a greater potential to accumulate certain COPCs due to chemical/physical similarities (e.g., fine, high TOC sediments sorb higher concentrations of metals and highly hydrophobic organics like the dioxins and furans; [e.g., Horowitz 1991; Stone and Haight 2000]). Despite the previous refinements and as previously noted, there are a number of data gaps that can and will be addressed during Phase II sediment sampling activities. These include measurements of:

- Nature and extent of COPCs not analyzed in any portion of the Site within either shallow or deep water zone sediments[26]
- Concentrations of specific COPCs (e.g., SVOCs) to evaluate risks to ecological receptors and people within shallow and deep water zone sediments
- Organic COPCs (e.g., dioxins/furans) in deep water zone sediments within the transitional and lucustrine CSMs
- SEM, AVS and supporting parameters to assess bioavailability and risk associated with SEM metals in the deep water zone of lacustrine sediments.

6.3.3 Sediment Toxicity Refinements

In addition, there are remaining uncertainties (data gaps) associated with sediment toxicity results from Phase I studies. As presented within Section 3.3 and detailed in Appendix E, although historical and Phase I sediment toxicity bioassays identified effects in shallow water zone sediments located within and upstream of Marcus Flats, it remains unclear if these observations are due to chemical toxicity because the results are associated with: 1) questionable representativeness of reference sites and 2) questionable health of some of the control batches of test organisms (Appendix E). As a result, additional information is required. This data gap will be addressed through additional sediment toxicity testing and synoptically collected sediment / porewater chemistry data during Phase II, as outlined within Section 5.1.1.2.

6.3.4 Sediment and Porewater Studies

Although historical and Phase I sediment toxicity bioassays identified effects in sediments located in the shallow waters (<80 ft) within and upstream of Marcus Flats, the data are insufficient for characterizing risk throughout the Site. Previous studies did not collect sufficient samples to establish concentration-response relationships for all sediments or porewaters within the Site, nor did they incorporate information required

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[26] COPCs not sampled within surface sediment samples to date include but are not limited to: PBDEs, gold, indium and many other exotic inorganics. The reader is referred to the draft SLERA for a complete list of COPCs that have not yet been sampled within surface sediments.

for making adjustments for bioavailability of metals within the different sediment types. Furthermore, no long-term bioassays (e.g., 42-day Hyalella and 53-day Chironomus studies) have been conducted to assess effects on reproduction of these organisms.

Specifically, sediment toxicity bioassays will be performed using 28-day exposures with the amphipod *H. azteca* and 10-day exposures with the midge *C. dilutus* to measure growth and survival, and 42-day and 53-day tests for these two species, respectively to assess reproductive endpoints. Some elements of a comprehensive sediment toxicity study require additional technical discussion, including the necessity of freshwater mussel sediment toxicity tests (*Lampsilis siliquoidea*) or laboratory measures of uptake of COPCs by oligochaetes (*Lumbriculus* sp.). A variety of other properties within the sediment and associated porewater from synoptically collected samples will be measured to evaluate correlates of any observed effects. These properties will include SEM–-AVS, f_{OC}, bulk (sediment) and dissolved (porewater) concentrations, and parameters needed to evaluate and assess bioavailability (e.g., alkalinity, pH, DOC, TOC, Na^+, sequential extractions, etc.). In addition, appropriate reference areas (e.g., upstream in Canada and within the Site) will be identified and included as part of the testing in order to interpret test results. Collecting information that addresses the above-listed data gaps will:

- Provide information on bioavailability of COPCs in bulk sediments and porewater
- Evaluate sediment bioassay results in terms of COPC bioavailability (i.e., does a concentration-response relationship exist between bioavailable forms of COPCs and observed toxicity?)
- Identify if COPC concentrations—total, dissolved, bioavailable—in whole sediments and porewater pose unacceptable risks to ecological receptors.

Sediments and porewaters will be collected during two (or three) sampling events. The first event will include samples from representative areas throughout the Site, and will be used to further refine gradients of COPCs and associated chemistries within the Site. Bioassays will be conducted with a subset of these samples to select the most sensitive bioassays and adjust standard protocols, as necessary for application in this Site. The second (and third) sampling events will employ the methods deemed most suitable as a result of the first round of sampling, and will further refine the concentration-response relationships through additional toxicity tests. Toxicity Evaluation and Identification methods will be employed to verify statistical cause-and-effect relationships.

A reference envelope approach will be used to evaluate the sediment toxicity generated under this sampling program. Therefore, the sampling program will target acquisition of matching sediment chemistry and sediment toxicity data at about 15 reference locations in the study area (i.e., within the Site and/or at outside Site reference locations, with at least two-thirds (10) of these located outside the Site).

Specifics about the design (sample locations, numbers, methods, criteria for qualifying reference locations, etc.) will be included in the sediment QAPP.

6.4 BEACH (SHALLOW WATER ZONE) SEDIMENTS

Although the 2009/2010 beach sediment sampling plan[27] was primarily developed to address additional data gaps for the human health risk assessment, the data can and will be used to 1) address COPC data gaps within shallow water zone sediments (see above), and 2) further refine our understanding and development of the Phase II soil sampling, see Section 6.6 below. Shallow water zone sediment sampling will occur September 9 to 11, 2010 at the following locations (Black Sand Beach, Upper Columbia R.V. Park, Northport Boat Launch, Onion Creek, and the Dalles Orchard); while the remaining 29 locations will be sampled during low-pool (i.e., April 2011), refer to Map 6-3 for sampling locations.

6.5 SOIL CHEMISTRY

Soil data for the Site are limited and are insufficient for evaluating unacceptable risks from exposure of wildlife to upland soils. As identified within Section 3.3, soil contamination may exist within the UCR due to the following three distinct processes 1) smelter stack emissions, 2) hydrologic transport and deposition (relict floodplains), or 3) wind-blown shallow water sediments. As illustrated within Figure 4-12, soil data are required to estimate exposure of terrestrial receptors (plants, invertebrates, wildlife) through direct contact, food chain uptake, or incidental soil ingestion. Filling these data gaps is therefore critical for assessment of exposures to a wide range of terrestrial receptors (including people), and to define the nature and extent of upland soil contamination.

To address these data gaps soil samples will be collected during Phase II activities. The goal is to collect data on concentrations of COPCs in soils from the UCR Site that are of acceptable quality to conduct an assessment of risk to people, wildlife, plants, and terrestrial invertebrates in

- Upland areas within and near areas potentially influenced by smelter stack emissions
- Potential relict floodplains along the river
- Upland areas with the highest probability of potentially being influenced wind-blown shallow water zone sediments.

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[27] EPA approved the 2009/2010 beach sediment study design in March 2009.

Soil concentration information will be used to assess risks to terrestrial invertebrate and plant communities, or to wildlife (amphibians[28], reptiles, mammals, birds). Soil data will be used in the BERA to determine which COPCs may pose an unacceptable risk to terrestrial receptors and which locations within the Site are of most concern for each receptor group.

Specific risk-related questions that will be addressed through collection of data on concentrations of COPCs in soil and associated soil chemistry parameters will be

- What are the concentrations and spatial distributions of COPCs in soils that have potentially been influenced by smelter stack emissions, deposition of sediment onto relict floodplains, or deposition of wind-blown shallow water zone sediments?
- Do COPC concentrations in soil pose unacceptable risks to human health and terrestrial receptor groups?
- Are concentrations of COPCs in soils greater than benchmarks for survival, growth or reproduction of plants or invertebrates?
- Are concentrations of COPCs in soil or diets (from water, soil and prey) of amphibians and reptiles (herpetofauna) greater than benchmarks for survival, growth or reproduction?
- Are concentrations of COPCs in soil or diets (from water, soil and prey) of birds and mammals greater than benchmarks for survival, growth or reproduction?

In addition to collecting data on COPC concentrations, data on soil parameters that affect bioavailability of chemicals will also be collected. This includes moisture content (percent moisture), f_{OC}, pH, grain size distribution, and cation exchange capacity. COPC concentration results will be reported on a dry weight basis for comparability and, if necessary, carbon-normalized (organics). Given the limited soil data set, it is anticipated that soil samples will be evaluated for the following COPCs: metals and metalloids, pesticides, SVOCs, PAHs, PCBs, dioxins/furans, and PBDEs.

As noted within Section 6.4.1, shallow water zone sediment data results will be used to refine sampling locations for soils potentially influenced by wind-blown sediments. Therefore, in consideration of this data collection activity and access by field-sampling crews (i.e., not during snow covered or frozen ground conditions) soil sampling activities will be targeted between July through August 2010.

6.6 WHITE STURGEON

Due to poor recruitment of white sturgeon (*Acipenser transmontanus*) in the UCR since the 1970s, this species has been identified as a special status species (refer to

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[28] Adult life stage only.

Section 2.3.4). While both spawning activity of adult sturgeon and occurrence of eggs and early larval stages have been frequently reported during the past years, young-of-the-year numbers have been limited. Juveniles hatched from resident adults and reared in a hatchery have been released into the UCR as part of the Lake Roosevelt White Sturgeon Recovery Plan and adapt well to the natural habitat. Existing data and experts have identified a survival bottleneck that could occur subsequent to the initiation of exogenous feeding (Howell and McLellan 2006; UCWSRI 2002). Subject to EPA's approval, the work proposed to be conducted as part of this RI/FS will focus on the potential effects of contamination on white sturgeon and will be consistent with the Agreement (USEPA 2006a).

6.6.1 Sturgeon Toxicity Tests (water)

In 2008 and 2009, Teck conducted studies to evaluate the toxicity of COIs in surface water collected from the UCR by conducting stream-side toxicity tests using water drawn from the UCR. Teck also assessed the toxicity of selected COIs (cadmium, copper, lead, and zinc) to white sturgeon by conducting spiked-water toxicity tests (acute and chronic). The EPA has not yet reviewed the results of these studies in detail, and these studies have not been approved by EPA for use in the RI/FS process.

Additional acute (96 hr) and chronic (56 days post hatch) studies of water-only exposures to early life stage sturgeon are being conducted by both Teck and the USGS. These studies will fill in the remaining data gaps regarding threshold effect concentrations for cadmium, copper, and zinc.

6.6.2 Sturgeon Toxicity Tests (sediment)

Given the demersal nature of white sturgeon, our present understanding of home range within the Site (i.e., Marcus Flats up to the U.S.-Canada border), early-life stage behavior (e.g., hiding in the substrate), and existing sediment COPC distributions and concentrations (refer to Section 3.3), there is uncertainty if granulated slag is chemically toxic to early-life stages of white sturgeon. Therefore, to address this data gap, subchronic (i.e., 40+-day) toxicity tests of sturgeon fry to sediment containing granulated slag will be conducted by the University of Saskatchewan in 2010 as part of Phase II activities. Although this work is dependent on the availability of eggs and hatching, testing is scheduled to commence in July 2010.

7 ADDITIONAL PHASES

As described in Section 6, additional data collection is required to complete the BERA. This requires an iterative process such that initial studies will inform the scope and requirements of subsequent studies, consistent with EPA guidance (USEPA 1997a). Currently known data gaps and associated Phase II studies are addressed in the previous section (Section 6). Additional studies (Phase III) will be conducted; the spatial extent, receptor groups, media, pathways and COPCs for these studies will be defined by the results of Phase II studies. Based on the results of the Phase II and Phase III studies, other studies may be required to develop further LOEs and reduce the uncertainty in risk estimates. Studies in Phase III and beyond will include, but are not limited to, collection of tissue chemistry data from additional biota, further analysis of bioavailability factors, receptor-specific bioassays, and comparison of Site data to background or reference areas. The studies described below include those that will be considered when assessing data gaps after each iteration of data collection (see Figure 7-1 for study sequence and Figures 5-1 through 5-8 for sequencing related to specific receptor groups). The need for additional studies beyond those described for Phase III will be determined in consultation with EPA. It should be noted that the following discussion does not include potential data collection efforts in support of the HHRA (e.g., Tribal or recreational use surveys). As outlined within the RI/FS work plan (USEPA 2008a), tasks described in the HHRA will be completed and can be used to inform additional data collection activities discussed herein (e.g., terrestrial resource study) or within the Settlement Agreement.

7.1 SEDIMENT STUDIES – PHASE III

Following Parts 1 and 2 of the Phase II sediment collection activities, a scientific management decision point (SMDP) memorandum will be generated evaluating and documenting if there is adequate information to conclude 1) that ecological risks are acceptable, 2) there is insufficient information (uncertainty) to make a decision, or 3) information indicates unacceptable ecological risk. In areas within the lacustrine reaches of the UCR where sediment toxicity was not performed during Phase II, sediment toxicity bioassays using 28-day exposures with the amphipod *H. azteca* and 10-day exposures with the midge *C. dilutus* may be necessary, either to resolve data gaps (e.g., SEM-AVS /f_{OC} > “no toxicity” thresholds), or reduce uncertainty. This work would be completed during Phase III data collection efforts.

Based on the results of the Phase II sediment sampling program and as documented within a SMDP memorandum, should areas be identified where there is insufficient information to make a risk-based decision, or the data indicates unacceptable ecological risk due to chemical toxicity, additional sediment (including subsurface and background) chemistry, and porewater data may be required.

7.1.1 Sediment-Water Interface Water – Phase III

The sediment-water interface water, (i.e., water within 4 cm of the sediment or substrate's surface) will be measured in any of the CSMs if potentially unacceptable risks are identified from Phase II COPC measurements in the porewater. Dissolved and particulate COPCs will be measured in the SWIW. The COPCs measured will be determined following the COPC refinement for sediments.

7.1.2 Sediment Weathering – Phase III

If sediments are found to pose potentially unacceptable risks to aquatic life and there is evidence of elevated COPC concentrations in porewater of SWIW, then tests to evaluate weathering of sediments influenced by granulated slag will be conducted to characterize the potential for release of COPCs from *in situ* sediment deposits. These tests would document *in situ* COPC leaching rates under conditions in the UCR. Results of this study will supplement other measures (LOEs) of bioavailability when assessing risk to benthic or water column organisms.

7.2 AQUATIC RECEPTORS – PHASE III

Studies of tissue chemistry and effect response of aquatic biota will be conducted in Phases III and IV. These include studies with dependencies on results of Phase II for study design or for determination of whether or not the study needs to be conducted. Phase IV studies will follow from Phase III, as needed to reduce uncertainties in risk estimates. Decisions for initiating Phase III or IV studies will be documented in the data summary reports.

7.2.1 Benthic Macroinvertebrates Tissue Chemistry – Phase III

As identified within Section 2.3.2, although benthic macroinvertebrates (including mussels) play key roles in transforming organic matter in sedimentary environments and are an important food source for numerous fish (including white sturgeon) and some aquatic-dependent wildlife species within the UCR, no *in situ* tissue concentrations of COPCs in benthic macroinvertebrates are available that are of good quality and representative of current conditions[29]. Because of the lack of data on concentrations of COPCs in benthic macroinvertebrate tissues, they will be measured as part of Phase III data collection activities. Primary questions to be addressed by these data include: 1) what are concentrations of COPCs in benthic macroinvertebrates (including freshwater mussels), and 2) do COPC concentrations in benthic macroinvertebrates

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[29] The 1994 M.S. thesis of Tielens (1994) will not be used to compute risks because the data are not indicative of current conditions. However, his study will be used for comparison in the BERA.

(including freshwater mussels) pose unacceptable risks to fish, aquatic-dependent wildlife and people?

Although COPC concentrations within benthic macroinvertebrate tissues is a known data gap, results from Phase II data collection activities will be used to inform the sampling design for benthic tissues. The list of COPCs to be measured can be refined. New EPA guidelines (USEPA 2008a) support eliminating from consideration organic COPCs in biota with Log K_{ow} < 5, because those with Log K_{ow} between 4 and 5 still have low bioaccumulation potential (see Section 4.5.3.1). Additionally, fish tissue data for a wide range of size classes and feeding guilds will be collected during Phase II. If no chemicals with Log K_{ow} between 4 and 5 are detected in fish tissues, then only chemicals with Log K_{ow} > 5 will be considered COPCs for Phase III benthic tissues (including mussels).

Results of Phase II studies will provide data on spatial distributions of COPCs of concern that will guide the locations for benthic macroinvertebrate tissue collections. Evaluation of COPC concentrations in bottom-dwelling fish species with small home ranges such as sculpin will be considered for inclusion in the design of Phase III benthic tissue data collection. Benthic tissue samples will be collected during the late summer months (August and September 2011) when biomass is likely to be the greatest.

If insufficient benthic tissue can be collected on Site, consideration will be given to conducting a laboratory uptake study. Teck, in consultation with EPA, will determine the appropriate species and methodology for such a study, noting that there are limitations of being able to extrapolate results among species and sediment types, particularly for inorganic substances.

7.2.2 Benthic Macroinvertebrate Abundance or Biomass (includes mussels) – Phase III

Sediment toxicity tests provide evidence of the likely presence of adverse effects on benthic macroinvertebrates, but they may not accurately represent *in situ* exposures if few if any organisms live in some areas because of inadequate habitat. If the toxicity test results collected in Phase II do not lead to a definitive conclusion about the presence or absence of adverse biological effects, or conflicting LOEs exist, additional measures will be used to assess the potential for adverse impacts on the benthic community. Evaluation of results of this study will consider the potential confounding effects of reservoir drawdown and other factors that can influence benthic communities. Therefore pending the results from Phase II sediment sampling, and in association with Phase III data collection efforts to determine COPC concentrations within benthic macroinvertebrates (see Section 7.1.1), a biological survey of the benthic macroinvertebrate community (including freshwater mussels) will be conducted. Rapid bioassessment protocols designed for lakes and reservoirs will be used for this study (e.g., Barbour et al. 1999 and Hayworth 2004).

7.3 PLANKTON TISSUE CONCENTRATIONS – PHASE III

As discussed within Section 2.3.2 of this work plan, there is Site-specific information confirming that zooplankton are a major component of the aquatic community, forming a substantial fraction (e.g., ≥90 percent) of the diet for several pelagic fish (kokanee and rainbow trout); and lesser contributions for many other species. Despite data confirming that many fish species within the UCR consume and are largely dependent on plankton consumption, there are no data to characterize COPC concentrations within plankton (herein referred to as plankton tissue). As illustrated within Section 4.4.1.3 and Figure 4-9, data on COPC concentrations in plankton tissue will be useful for evaluating chemical transport pathways in the aquatic food web, and the potential for risk to fish that prey on zooplankton.

To address this data gap, zooplankton will be collected during Phase III sampling activities, if results from fish tissue analyses (Phase II) are inconclusive regarding uptake of contaminants. The primary question to be addressed by this data collection effort will be: Do COPC concentrations within zooplankton pose unacceptable risks to the species known to feed upon them (e.g., Lee et al. 2006) in the UCR?

As indicated within Section 2.3.2 and Appendix A, zooplankton in the UCR consist largely of two predominant taxonomic groups: the orders Copepoda (copepods) and Cladocera (cladocerans), are abundant in the lacustrine CSM (including Marcus Flats), having the highest densities ($\mu g/m^3$) during the summer and early fall. As a result, plankton tissue data collection activities during Phase III will coincide with anticipated maximum plankton densities (i.e., end of August to mid-September), and will be spatially distributed within representative reaches of the lacustrine and transitional (Marcus Flats) CSMs (Map 6-2).

All samples will be analyzed for total TAL metal (including uranium, molybdenum, and fluoride) concentrations, and percent moisture. In addition, and if sufficient plankton mass is collected, samples will also be analyzed for the following in the order listed: 1) total non-TAL metals (bismuth, boron, cerium, cesium, dysprosium, erbium, europium, gadolinium, gallium, germanium, gold, holmium, indium, lanthanum, lithium, lutetium, molybdenum, neodymium, niobium, praseodymium, rubidium, samarium, scandium, strontium, tantalum, tellurium, terbium, thorium, thulium, tin, titanium, tungsten, ytterbium, yttrium, zirconium); and 2) organic COPCs with preference towards measures of dioxins/furans, PCBs, PBDEs, PAHs, DDT and its metabolites, and SVOCs with Log $K_{ow} > 4.0$.

These data on concentrations of COPCs in zooplankton, in conjunction with surface water quality information (Section 6.1 above) and fish tissue data (Section 6.2 above), will be used to evaluate risks to planktivorous fish (see Section 4.4.1).

7.4 OTHER AQUATIC BIOASSAYS (INCLUDES MUSSELS) – PHASE IV

If there is insufficient information or uncertainty to make a decision about risks, or if the data indicate potentially unacceptable ecological risk due to chemical toxicity, then it may be necessary to conduct additional aquatic toxicity bioassays (including early life stages of mussels or amphibians). The need for additional bioassays has been included as contingencies in the assessments of surface water data (Phase II); sediment chemistry (Phase II); sediment toxicity bioassay data (Phase II); and benthic macroinvertebrate data (tissue and aquatic community survey information) collected during Phase III sampling efforts. The decision to test and details of the proposed studies will be documented in data summary reports. Examples of candidate tests include:

- Tests for toxicant interactions (e.g., additivity, antagonism) using tests of specific mixtures, concentrations and bioavailability of COPCs (e.g., Konemann and Pieters 1996)
- Chronic toxicity tests of *Ceriodaphnia dubia* to evaluate surface water toxicity (USEPA 2002d)
- ASTM sediment toxicity test with amphibians (ASTM 2004)
- Freshwater mussels (ASTM 2009).

Some elements of a comprehensive sediment toxicity study require additional technical discussion, including the necessity of freshwater mussel sediment toxicity tests (*Lampsilis siliquoidea*) or laboratory measures of uptake of COPCs by oligochaetes (*Lumbriculus* sp.).

7.5 AQUATIC SPECIALTY STUDIES – PHASE IV

If preliminary risks to fish or aquatic-dependent wildlife receptor groups are identified after measuring COPC concentrations in most components of their diets (sediment, macroinvertebrates, plants, and fish), and if it is deemed necessary to further refine these risk estimates, additional studies will be done. This includes, but is not limited to, studies of bioavailability of COPCs in fish and macroinvertebrate tissue, direct dietary tests for fish (to account for bioavailability of biotically-incorporated chemicals) and measurement of COPC concentrations in emergent forms of aquatic macroinvertebrates (eaten preferentially by some fish, bird, and amphibian species).

7.5.1 Bioavailability of COPCs from Fish and Macroinvertebrate Tissue – Phase IV

If preliminary risk estimates based on Phase III COPC tissue concentrations (see Section 7.1.1) indicate that unacceptable risk to aquatic receptor groups from metals may result from ingestion of fish or benthic macroinvertebrates (including freshwater mussels), a study may be conducted to measure metal bioaccessibility in fish and benthic macroinvertebrates during Phase IV data collection activities. Bioaccumulated metal residues in benthic invertebrates can be elevated due to normal, homeostatic physiological processes. For example, many aquatic and benthic invertebrates, especially bivalve and gastropod molluscs, store metals bound to proteins and in calcium granules in chemical forms that limits bioavailability of metals to the organism as well as to fish and wildlife that consume them (Campbell et al. 2005; Desouky 2006; Dumas and Hare 2008; Dubois and Hare 2009a,b). Consequently, risks to fish, wildlife and humans from consumption of macroinvertebrates that are based on measurement of total metal concentrations in macroinvertebrate tissues may be erroneous. As an alternative, recent studies advocate measuring bioavailable and inert forms of metals in tissues (Campbell et al. 2005; Desouky 2006; Dumas and Hare 2008; Dubois and Hare 2009a,b), including the cytosol, heat-stable proteins like metallothionein, heat-denaturable protein fractions, and granules. The need for this study will be dictated by risk analyses through Phase III; estimations of unacceptable risk in fish or wildlife from food-chain exposure could be refined by quantifying bioavailability of COPCs in diets (initial assumptions generally are based on 100% bioavailability). If conducted, results of this study will be used to adjust bioavailability estimates in food chain models used to assess risks to fish and aquatic-dependent wildlife.

7.5.2 Dietary Toxicity to Fish – Phase IV

If preliminary risk calculations or comparisons to benchmarks indicate that there is unacceptable risk to fish in the UCR, based on data collected during Phase II surface water, fish and plankton tissue sampling; Phase III benthic macroinvertebrate tissue sampling;, then direct measurements of dietary toxicity to fish may be conducted in lieu of or addition to Phase IV measures of bioavailability of COPCs in macroinvertebrate tissue. The decision for this testing will be documented in a technical memorandum.

For most COPCs, and in particular for metals, toxicity benchmarks based on tissues of organisms or bulk sediments reported in the literature do not explicitly address bioavailability. Moreover, toxicant interactions often cannot be addressed solely using toxicity for single COPCs; mixtures must be tested (Marking 1977; Konemann 1981). Critical body burdens for divalent metals are an unreliable indicator of adverse effects because the metals, even some of the nonessential ones, are homeostatically regulated to varying extents (Barron et al. 2002; Campbell et al. 2005; Desouky 2006; Meador 2006). For these reasons, direct measurement of toxicity through fish dietary bioassays (Hansen et al. 2002, 2004; Erickson 2003; Mount et al. 2006) will be considered.

7.5.3 White Sturgeon – Phase IV

EPA identified an additional investigation that may need to be conducted to evaluate the effects of COI exposures on white sturgeon in the UCR:

- Investigation 4: Evaluation of the toxicity of COIs in the diet of sturgeon (i.e., modeling or dietary toxicity tests conducted in the lab)

The EPA expects that development of the sturgeon dietary model parameters (e.g., exposure and physiological factors), that are needed to fulfill the objectives of Investigation 4, will be done, unless the outcome of the previous studies deem this unnecessary, with the understanding that model simulations will not be run until Teck has collected the appropriate empirical data. Should the model not be satisfactory, or if there is an interest in further reducing uncertainty of this exposure route, then empirical laboratory feeding studies may be done with the sturgeon.

7.5.4 Emergent Insect Tissue Chemistry – Phase IV

Invertivorous aquatic-dependent wildlife, including birds (e.g., swallows) and mammals (e.g., bats), consume both aquatic and terrestrial aerial insects, thus forming a link between the two ecosystems. Preliminary risk estimates for these wildlife receptors will use tissue COPC concentrations measured in benthic macroinvertebrates (Phase III) to estimate their dietary exposure. Initial risk calculations will assume equivalence of COPC concentrations in larval and adult forms of insects. However, pupation may lead to an alteration of COPC concentrations in the adult forms, if COPC concentrations in the discarded exoskeleton differ substantially from those in the larval form. For those COPC-aerial invertivore receptor pairs for which the 1) calculated hazard quotient is judged to be significant, and 2) for which literature information indicates differential sequestration of those COPCs in the discarded exoskeleton, the risk analysis will be iterated after measuring residues in emergent insect larvae in Phase IV. The decision of whether or not to pursue this level of refinement will be documented in the data summary report.

7.6 TERRESTRIAL STUDIES – PHASE IVa

As identified within Section 6.5, soil chemistry data will be collected during Phase II to define the nature and extent of upland soil contamination via three potential pathways 1) smelter stack emissions, 2) hydrologic transport and deposition (relict floodplains), and 3) wind-blown shallow water zone sediments. In addition to defining the nature and extent of upland soil contamination, these data will be used in the BERA to evaluate which COPCs may pose an unacceptable risk to terrestrial receptors (e.g., terrestrial plants, soil invertebrates, birds, mammals, and herptofauna).

Following the collection of soils data during Phase II, and as outlined within Section 5.1.2 and shown in Figures 5-1 through 5-8, risks to the terrestrial receptors will be evaluated by comparing the upper 95 percent confidence limit of mean COPC concentrations in soil to conservative benchmarks (Eco-SSLs). In addition to comparing individual data points to Eco-SSLs, it is anticipated that results would also be evaluated using standard quantitative methods to estimate by interpolation the concentrations of metals in surface soils between sampling locations; comparison of the measured and interpolated concentrations to Eco-SSLs will result in a spatial representation of areas exceeding benchmark concentrations. Performing such analyses would ensure that the extent of potential soil contamination has been defined, and by default, will identify additional potential spatial data gaps. Results of these analyses would be documented within the data summary report.

7.6.1 Soil Sampling and Bioavailability

If the analysis identifies data gaps in the extent of soil contamination, additional soil samples would be collected during Phase III. If the analysis confirms that the extent of soil contamination has been adequately delineated and identifies areas in which risks in soils are unacceptable, then the bioavailability of each COPC will be considered in the risk calculation. It is important to note and as outlined within Section 6.5, soil parameters used to account for bioavailability (as collected during Phase II) would include, but not necessarily limited to: pH, TOC, particle size, cation exchange capacity, and/or *in vitro* gastrointestinal measures. Results of this analysis would also be documented in a technical memorandum. Should conservative benchmarks not be available for COPCs, a review of the scientific literature would be conducted to develop a TRV appropriate for the receptor class(es) potentially at risk. Results of the literature review and findings would also be documented in the a technical memorandum.

7.6.2 Soil Background

If unacceptable risks are identified following evaluation of the above-mentioned LOEs, COPC concentrations will then be compared to background. A few data have been identified with which preliminary background soil concentrations can be defined (Holmgren et al. 1993; USGS 2004; Weston 2005: Sanei et al. 2007). It is important to note that given the present uncertainty associated with the nature and extent of upland soil contamination, soil samples collected during Phase II will be positioned not only to capture, but extend beyond, the anticipated zone of influence due smelter stack emissions (refer to Appendix G). As such, Phase II soil sampling activities will provide additional background soil data.

7.6.3 Upland Lakes and Wetlands

If areas of soil contamination are identified that pose unacceptable risk to terrestrial organisms, then the potential for contamination of lakes or wetlands within that area will be evaluated. This may require sampling of sediment, vegetation, or other matrices. Decisions about whether to require such analyses will be made in consultation with EPA.

7.7 TERRESTRIAL RECEPTOR STUDIES – PHASE IVb

If soil concentrations exceed background and food chain analysis suggests site-related risk to terrestrial organisms, additional investigations that could be performed (as illustrated within Figures 5-6 through 5-8) would then be evaluated. Initially a terrestrial resource survey would be conducted (see below for details). Following the terrestrial survey, additional LOEs that could be evaluated for plants include: plant bioassays using sensitive indicator plant species (e.g., lettuce germination; Inaba and Takenaka 2005), comparisons of plant communities to reference locations, measurements of COPCs in plant tissues (for comparison to critical tissue levels). Similarly, additional LOEs that could be evaluated for soil invertebrates include: bioassays using sensitive indicator species, comparisons of invertebrate communities to reference locations, or measurements of COPCs in tissues (for comparison to critical tissue levels). General descriptions of these additional studies are presented below

7.7.1 Terrestrial Resource Study

This study would include the collection of additional data on the presence and spatial distribution of terrestrial resources (e.g., plants, invertebrates) and receptors (wildlife) in the areas potentially affected by COPCs. The information may be obtained from reviews of relevant literature, interpretation of remote sensing data, or field surveys. Results of this study would then be used to refine the problem formulation, to assist in the identification and design of further studies of terrestrial exposures, and for the calculation of risks to terrestrial ecological receptors.

7.7.2 Plant Tissue Chemistry

This study would involve the collection and analysis of plant tissue for specific COPC concentrations. This information would further inform the BERA about bioavailability of COPCs in soils, specifically the amount taken up by plants compared to co-located soil concentrations, and provide Site-specific data for dietary risk analyses for wildlife.

7.7.3 Plant Bioassay

Plant bioassays would be conducted according to standard protocols using species known to be sensitive to the COPCs. The kinds of soil toxicity tests that may be conducted include plant germination tests, or early seedling growth.

7.7.4 Invertebrate Tissue Chemistry

This study would involve the collection and analysis of invertebrate tissue for specific COPC concentrations. This information would inform the BERA about bioavailability of COPCs in soils, specifically the amount taken up by invertebrates compared to co-located soil concentrations, and provide Site-specific data for dietary risk analyses for wildlife.

7.7.5 Terrestrial Invertebrate Toxicity Studies

Terrestrial invertebrate bioassays would be conducted according to standard protocols using invertebrate species known to be sensitive to the COPCs and that are expected to be on-site or be representative of species that are expected to be on-site. The bioassay would be conducted using Site soil collected from areas where benchmarks and background were exceeded and comparisons made to reference site or standard soil bioassays. The kinds of soil toxicity tests that may be conducted include earthworm or springtail toxicity tests.

As indicated within Figure 4-6 of Section 4.2.1.2, should data collected for, any or all of the above-listed additional studies be required, this information could then be used to calculate Site-specific uptake and dietary exposure to evaluate risks to terrestrial birds, mammals, reptiles and/or adult amphibians. It should be noted however, these data might not be required for the aforementioned receptors as literature based uptake and dietary exposure assumptions may be sufficient to evaluate risks.

7.8 TRANSPORT AND FATE EVALUATIONS

If the results of the BERA establish that COPCs in one or more UCR media pose unacceptable risks to ecological receptors, a number of studies related to the transport and fate of those COPCs within the UCR may be required (including mercury). This requirement will be evaluated based on LOEs and in consultation with EPA.

7.9 ADDITIONAL STUDIES

Additional studies may be necessary due to the iterative nature of the BERA and the results of the studies summarized in the preceding sections. If data gaps are identified based on a review of the data from the proposed studies and in consultation with EPA, additional studies will be developed to provide supporting information to complete the BERA, HHRA or RI/FS. These are anticipated to occur (if necessary) following the completion of the studies described above.

8 ANALYSIS AND RISK ESTIMATION PLAN

The currently available data and the data to be developed by future studies provide information necessary to evaluate the LOEs for aquatic and terrestrial ecological receptors. This section provides a description of the methods that will be used to interpret and integrate the LOEs (i.e., characterize risks) for each receptor, and discusses methods for analysis of uncertainties and interpretation of ecological significance. The approach is based on the current problem statements, EPA guidance, and established practice, and incorporates a process by which new approaches can be incorporated into these steps.

8.1 RISK CHARACTERIZATION

LOEs for each receptor group are described in Section 5.1. Initial LOEs for each receptor group are comparisons between simple measures of exposure (i.e., water, soil, sediment, or biota concentrations) and either established or literature-based benchmarks. Methods for characterizing risk from these and other LOEs are outlined below, using specific receptor groups as examples. However, if a similar LOE is used for another receptor group, the approach to characterizing risk will be the same. For example, benthic macroinvertebrates are used to illustrate how bioassay data may be used to characterize risk.

8.1.1 Comparison to Benchmarks

Comparison of COPC concentrations in exposure media (water, sediment, soil, etc.) to ecologically meaningful benchmarks will be made in both a deterministic and partially probabilistic manner. Benchmarks include applicable regulatory limits (such as ambient water quality criteria) and site-specific values such as those derived from synoptic sediment chemistry – sediment toxicity tests (see Section 5.2.2). Deterministic comparisons will be made using the reasonable maximum exposure (i.e., the 95 percent UCL of the mean) averaged over a CSM unit or appropriate area representative of receptor foraging locations (see Section 5.2.1.4). A geostatistically based surface-area weighted concentration may be used instead of the mean, if the data are geographically biased and non-normal (see Section 8.1.3 for discussions about visualization of the data). All tests of significance will have a target value for Type I error rate of 0.05 and a target value for Type II error rate of 0.80. Data will also be examined spatially to see if there are clusters of above-benchmark concentrations or any other spatial attributes that might indicate risk within each of the units, regardless of whether the overall RME exceeds benchmark values. This will result in characterization of areas of similarity and location of areas of potential concern for sessile organisms (see Section 8.1.3 below). Comparisons will be made through derivation of cumulative distribution functions (CDF) of exposure concentrations in comparison with various benchmarks of the different receptor groups as another way of assessing both FOD and frequency of exceedance, regardless of the RME value. Use of CDFs allows for a statement of the probability of exceedance of the benchmark within the exposure area. .

8.1.2 Bioassays

Evaluation of risk to benthic macroinvertebrates will include Site-specific sediment toxicity tests as outlined within Section 5.1.1.2. A similar LOE may be used for fish or soil organisms (i.e., plants or terrestrial invertebrates), refer to Section 5.1.2.2. Toxicity tests produce quantitative indicators of survival, growth, and reproduction of sensitive and representative species.

Should bioassays be required as determined via the LOE decisional process (refer to Section 5.1.1.2), standardized methods will be used to conduct and interpret toxicity tests (Table 5-8). To determine whether growth, survival, or reproduction of benthic macroinvertebrates are impaired in Site sediment, for example, toxicity responses of *C. dilutus* and *H. azteca* will be compared statistically between organisms exposed to Site sediments with those of organisms exposed to sediments from one or more suitable reference areas. Suitable reference areas will be selected both from within and outside the Site to develop a reference envelope (see Sections 5.2 and 6.4.4). Suitability is based on TOC content, grain size, and mPECQ of the sediments, consistent with guidance (USEPA 2000). Test acceptability will be judged according to performance criteria provided by the standard methods as compared to laboratory control sediments.

As outlined within Section 5.1.1.2, sediments toxicity test results will be related to and concomitantly evaluated with concentrations of COPCs in sediment, porewater (adjusted for bioavailability). The relationship between potential adverse effects and COPC concentrations in sediments will be evaluated using quantitative or categorical methods. Quantitative methods will include regression analyses (using parametric or nonparametric methods, possibly including data transformation). Categorical methods would include evaluation of Site data relative to national, regional, or Site-specific SQGs.

8.1.3 Spatial Analysis

Risks will be evaluated and interpreted separately for each distinct set of exposure conditions (e.g., CSM unit; combination of physical and biological conditions; and adjustments of bioavailability). Maximum likelihood estimates of the toxicity test responses and benchmark exceedances will be interpreted in terms of the probability of adverse effects on the receptor groups in each exposure condition. Visualization or GIS-based statistical methods will be used to characterize the spatial distributions of COPC concentrations or receptor responses within each CSM. Interpolation methods for mapping may include krieging, inverse distance weighted averaging (IDWA), splining, or trend surface analysis (TSA), depending upon the amount of data, their variability, and their spatial distribution (clustered, regularly distributed, irregularly distributed). These are all methods of spatial interpolation which predict unknown values from data observed at known locations. A variogram analysis will be used to look for spatial autocorrelation and to assist in selection of the appropriate methodology, prior to interpolating or performing geostatistical analysis of the data. If spatial patterns are found, these will be addressed either by refinement of the CSMs or by consideration with the uncertainty analysis of the BERA.

8.1.4 Tiered Approach to Reduce Uncertainty

Characterization of risk to fish and wildlife will be conducted in an iterative manner, beginning with simple and conservative approaches and simple exposure assumptions (e.g., ingestion rates for different media, habitat use patterns), proceeding to more complex and realistic approaches and assumptions, as well as incorporating additional data as necessary. This process will include refinement of the COPCs and exposure areas through iterative improvement of risk estimates. This process will iteratively include more Site-specific and species-specific information to improve realism and reduce the uncertainty of exposure estimates and the reliability of the benchmarks and TRVs, including steps such as:

- Use of Site-specific chemistry data for surface water, sediment, prey tissue (i.e., zooplankton, benthic invertebrate, and fish) and whole body fish tissue
- Use of receptor-specific diet information
- Use of information on the relative bioavailability of metals in ingested media (using literature data, sensitivity analyses, and possibly including Site-specific studies of COPC bioavailability from invertebrate tissue)
- Use of receptor species-specific life history information that might affect duration, life stage, or spatial extent of exposures.

To ensure that the toxicity information used to interpret exposure estimates is the most appropriate for the receptor group and exposure scenario, and is technically sound, the source information for each benchmark will be critically evaluated. When realistic exposure estimates clearly exceed a benchmark, additional LOEs will be examined, such as bioavailability of the COPC in each medium or comparison to reference areas, or bioassay test results. LOEs from higher tiers (e.g., bioassays) will be given greater weight than those from lower tiers (i.e., exceedance of screening benchmark values). Monte Carlo simulations may be employed to develop risk probability statements, when comparing a distribution of exposures with a range of possible effect threshold values. Uncertainties in the approaches, assumptions, benchmarks, and Site-specific data will be addressed in a discussion of the overall uncertainty of the results.

8.2 INTEGRATION OF LINES OF EVIDENCE

The general concept of a WOE approach will be used to interpret LOEs. This WOE approach may follow a quantitative likelihood method (e.g., Burton et al. 2002; Royall 1997) or be more qualitative in nature and will be more thoroughly described in the forthcoming Problem Formulation plan. This approach will use statistical evaluations of Site data relative to reference data and allow quantitative integration and weighting of different LOEs (Royall 1997). Weighting of different LOEs will be based on evaluations of the ecological relevance of each LOE (i.e., to growth, reproduction, and survival) and on evaluations of relative uncertainties in the different LOEs. Relative uncertainties will be based on both quantitative assessments of data variability and, if necessary, qualitative assessments of data quality. Evaluation of limiting cases and Monte Carlo methods will be used as necessary to evaluate the results of different assumptions or judgments regarding quantitative parameters (i.e., conduct a sensitivity analysis).

The quantitative likelihood procedure will produce an expected probability of impairment for each category of receptor and set of exposure conditions (i.e., CSM unit). Sensitivity analyses may be carried out to identify the LOEs that are most important in driving these probability estimates and to evaluate concordance among different LOEs. The results of the risk characterization will be a description of the relative risks to different receptor categories in different locations; the importance of different LOEs; and the types of population effects that may be expected from the LOEs that affect the growth, reproduction, and survival of ecological receptors.

8.3 INTERPRETATION OF ECOLOGICAL SIGNIFICANCE

A determination of the ecological significance of risk estimates for multiple receptors will be made qualitatively, at the scale of the CSM unit or other appropriate ecological scale, and will consider the following:

- Exposure, toxicity, and/or risk within the reference envelope.
- The types of effects that could occur for each receptor or receptor group. This will be based on toxicological endpoints reported by studies from which benchmarks are derived, and the endpoints employed by Site-specific toxicity tests.
- Receptor groups that are potentially affected. If risk estimates are high for a broad range of receptors, the ecological significance should be considered greater than if only one receptor group is potentially affected.
- Relationships of affected receptors to the rest of the ecosystem. For example, if the potentially affected receptor group represents keystone predators or provides a significant and irreplaceable food resource to other species, the significance of the risk would be greater.

The discussion of the ecological significance of a set of risk assessment results for a CSM unit will be a qualitative synthesis of all the available information to provide a general statement of ecological risk.

8.4 UNCERTAINTY

All estimates of risk have associated uncertainty. By moving through the LOEs, from general estimates to specific measures, uncertainty can be reduced. However, even the final risk estimate has uncertainties associated with it. Uncertainties may be stated in either a qualitative manner (high, medium, or low) or quantified through a variety of statistical measures (e.g., standard error of the mean; Monte Carlo probability statements). For each LOE conducted within each of the receptor groups, the associated uncertainty will be clearly stated and documented as a specific section of the risk evaluation (i.e., technical memorandum). These uncertainties will be collated and addressed collectively in the uncertainty section of the BERA to provide receptor- or chemical-specific uncertainty analyses. The final risk management decision can therefore be made with a stated amount of uncertainty.

9 SCHEDULE

As outlined within Section 8, data collection activities beyond Phase II are dependent on the results and interpretation of data collected during Phase II. As a result, the following schedule outlines Phase II activities and the mechanisms (i.e., technical memoranda) which will be used to assess and determine the scope and breadth of Phase III activities, see Figure 9-1. A schedule that extends beyond Phase II activities and to the completion of the RI/FS (i.e., issuance of the Record of Decision) is provided within the RI/FS work plan (USEPA 2008a). The schedule provides timeframes for the development of study specific QAPPs, field implementation, data reporting; and technical memoranda which will be instrumental in evaluating and integrating LOEs, thereby identifying data gaps and technical requirements for subsequent sampling activities (Phases).

The sequencing and timing of the investigations to support the BERA will be determined by the nature of the studies undertaken (including constraints such as the need for sampling during certain seasons or hydrologic conditions), the complexity of the analyses required, and the decisions that are made based on the results of completed studies. A number of field investigations have been sequenced to allow for information from preceding studies to inform later studies (i.e., an iterative process). Studies to be initiated during Phase II (i.e., 2009 and 2010) are largely independent of one another.

The schedule (Figure 9-1) in conjunction with the overall project schedule identified within the RI/FS work plan (USEPA 2008a) will be updated periodically and additional tasks will be documented as new information becomes available and as tasks and Phases are refined.

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11 GLOSSARY

$(\Sigma\text{SEM}-\text{AVS})/f_{oc}$—The sum of two parameters—the concentration of simultaneously extractable metals minus the concentration of acid volatile sulfides—normalized for the concentration of total organic carbon in the sediment.

Acid volatile sulfide (AVS)—A fraction of the sulfides in sediment that form a complex with heavy metals like copper, lead, and zinc.

Anadromous—Pertaining to fish that live their lives in the sea and migrate to a freshwater river to spawn.

Benthic—The living organisms that are found at or near the bottom (i.e., sediments) of the UCR.

Bioaccessibility—Refers to the amount of environmentally available metal that actually interacts with the organism's contact surface (e.g., membrane) and is potentially available for absorption (or adsorption if bioactive upon contact). Environmentally available metal is the total amount of metal that is available for physical, chemical, and biological modifying influences (e.g., fate and transport) and is not sequestered in an environmental matrix.

Bioavailability—Refers to the extent to which bioaccessible metals absorb onto, or into, and across biological membranes of organisms, expressed as a fraction of the total amount of metal the organism is proximately exposed to (at the sorption surface) during a given time and under defined conditions.

Biotic ligand model (BLM)—An analysis tool used to evaluate differences in the availability and toxicity of metals that occur as a result of changes in water chemistry from site to site, and at a given site over time.

Chemical of potential concern (COPC)—A chemical that has been determined to pose an unacceptable risk to a certain receptor group under a certain set of conditions.

Conceptual site model (CSM)—A written description and visual representation of the known, expected, and/or predicted relationships between the site chemicals and the ecological receptors.

Congener—In chemistry, variations or configurations of a common chemical structure.

Critical body burden—The threshold chemical concentration within a whole organism above which toxic effects will occur.

Daphnia—A genus of sensitive plankton within the family Daphnidae. One of the planktonic species consumed by a number of fish in Lake Roosevelt.

Daphnids—A group (family) of crustaceans that are filter-feeding plankton. Daphnids belong to the family Daphnidae.

Deepwater zone—For the purposes of the UCR RI/FS this has been defined to coincide with annual water management activities and as such, includes areas (sediments) located below 1,210 ft amsl.

Detritivore—Organisms that feed on dead and decaying animal or plant material.

Dioxin—A group of chlorinated organic compounds with similar chemical structures. Some dioxins have harmful properties, depending on the number and position of chlorine atoms. Dioxins are formed unintentionally and released as by-products of human activities such as waste incineration, fuels combustion, chlorine bleaching of pulp and paper, or pesticide manufacturing. They are also formed by natural processes such as forest fires and volcanoes.

Dissolved organic carbon—The concentration of organic (not inorganic) carbon dissolved in water or porewater.

Drawdown—The distance that the water surface of the reservoir is lowered from a given elevation as water is released from the reservoir.

ECx—Effect concentration, the concentration resulting in a specific biological effect in a certain percentage of organisms. For example, an EC_{20} in a growth experiment indicates the concentration where 20 percent of the test organisms experienced reduced body weight.

Effluent—The polluted discharge from a man-made structure that flows into a body of water.

Exotics—Metals/metalloids not typically evaluated in environmental investigations (i.e., not analyzed as part of EPA’s Target Analyte List for Superfund).

Furan—An organic compound produced when wood, especially pine-wood, is distilled. Furan is a clear, colorless, very volatile and highly flammable liquid with a boiling point close to room temperature.

Granulated slag—Granulated slag is a coarse- to medium-grained sand-sized particle that is black in color and has the appearance of obsidian. Chemically, Trail granulated slag is a calcium-iron rich silicate with a fairly consistent composition dominated by silica (~31 percent composition), lime (~15 percent), and iron (~30 percent).

Hyalella azteca—*H. azteca* is a ¼-inch-long crustacean that is common in aquatic systems. It is a widely used organism in standard sediment toxicity tests.

In situ—Latin phrase meaning in its original position or place.

Lacustrine—Pertaining to or living in lakes or ponds.

Lentic—Of or pertaining to still waters such as lakes, reservoirs, or ponds.

LOAEC/LOAEL—Represents the lowest observable adverse effect concentration or level (e.g., dose) evaluated in a toxicity test that has a statistically significant adverse effect on the exposed organisms compared with unexposed organisms in a control or reference site.

Log K_{ow}—The K_{ow} represents the ratio of concentrations in a lipid (fat) substitute, octanol, and in water. For example, a K_{ow} of 10,000 means that the concentration of chemical in octanol would be 10,000 times higher than that in water in a two-phase system at equilibrium. This frequently is reported on a logarithmic scale to the base 10 as "Log_{10} K_{ow}", so 10,000 would be equivalent to a Log K_{ow} of 4.

Lotic—Of or relating to actively moving water such as rivers or streams.

Macroinvertebrates—A broad term used to refer to invertebrates large enough to be seen with the naked eye (includes mussels).

Near-bottom water—Refers to surface waters collected approximately 1 meter above the sediment bottom of the reservoir.

NOAEC/NOAEL—No observable adverse effect concentration or level, represents the highest effect concentration or level (e.g., dose) evaluated in a toxicity test that has no statistically significant adverse effect on the exposed organisms compared with unexposed organisms in a control or reference site.

Oligotrophic—An oligotrophic lake or reservoir low in nutrients and organic productivity. Oligotrophic lakes are usually deep, with nutrient poor sediments, few macrophytes and large amounts of dissolved oxygen.

Periphyton—A complex mixture of algae, cyanobacteria, heterotrophic microbes, and detritus that is attached to submerged surfaces (e.g., rocks, gravel, sediment, logs) in most aquatic ecosystems.

Phytoplankton—Small, usually microscopic plants (such as algae), found in lakes, reservoirs, and other bodies of water.

Physiographic—Of or pertaining to the study of physical features of the Earth's surface.

Polychlorinated biphenyls (PCBs)—Mixtures of up to 209 individual chlorinated compounds (known as congeners) that were manufactured and used mostly in electrical equipment prior to 1977 when their production was stopped in the United States due to environmental and health concerns.

Porewater—Water that fills the interstitial space between sediment grains in sedimentary deposits. Porewater may be displaced due to the activities of benthic fauna (animals), or by physical processes such as compaction.

Reference area—A geographic area (that is not the focus of the risk assessment) consisting of similar physical and chemical characteristics (i.e., of natural or anthropogenic origins) as the area under investigation. The primary difference is that the reference area does not receive inputs from the primary contaminant source(s) that are the focus of the risk assessment.

Relict floodplain—A relict floodplain is defined as an area that may have been subjected to flooding under past flow conditions of the UCR but that is not expected to be flooded under present flow of the UCR and reservoir pool level management controls.

Richness—The number of species identified in a sample or area.

Scientific Management Decision Point (SMDP)—A point in the risk assessment process where the risk managers decides whether the information in hand is sufficient to conclude that ecological risks are acceptable, that information is not adequate to make a decision, or that information indicates a potentially unacceptable ecological risks.

Sediment class—A type of sediment that is characterized by unique physical and chemical properties.

Shallow water zone— For the purposes of the UCR RI/FS it has been defined to coincide with annual water management activities to includes area (sediments) exposed within the annual drawdown (i.e., 1210 to 1290 ft amsl).

Simultaneously extractable metals (SEM)—The dissolved concentrations of certain divalent metals extracted from sediment using a weak acid. SEM analyses are completed in conjunction with analyses of acid volatile sulfide.

Slag – Material produced during the smelting or refining of metals by reaction of the flux with impurities (e.g., calcium silicate formed by reaction of calcium oxide flux with silicon dioxide impurities). The liquid slag can be separated from the liquid metal because it floats on the surface. Slag produced in smelting copper, lead, and other metals can also contain iron silicate and oxides of other metals in small amounts.

Stratified sampling design—A sampling program where multiple groups (strata) are investigated by determining the characteristics of a percentage of each group's members chosen at random. For example, a sediment sampling program may be designed to examine the characteristics of varying sediment classes (strata) by randomly collecting sediment samples from each stratum.

TAL metals—From EPA's 2008 Target Analyte List for Superfund, Contract Laboratory program: statement of work with inorganic Superfund methods. (http://www.epa.gov/superfund/programs/clp/mtarget.htm). Metals include: aluminum, antimony, arsenic, barium, beryllium, cadmium, calcium, chromium, cobalt, copper, iron, lead, magnesium, manganese, mercury, molybdenum, nickel, potassium, selenium, silver, sodium, thallium, vanadium, and zinc.

Total organic carbon—The concentration of organic (not inorganic) carbon measured in sediment or a particle.

Total organic carbon normalized (TOC normalized) – Expressing concentration of metals or chemical constituents in biotic or abiotic matrices on the basis of the amount of organic carbon present in the sample (e.g., mg/kg_{oc})

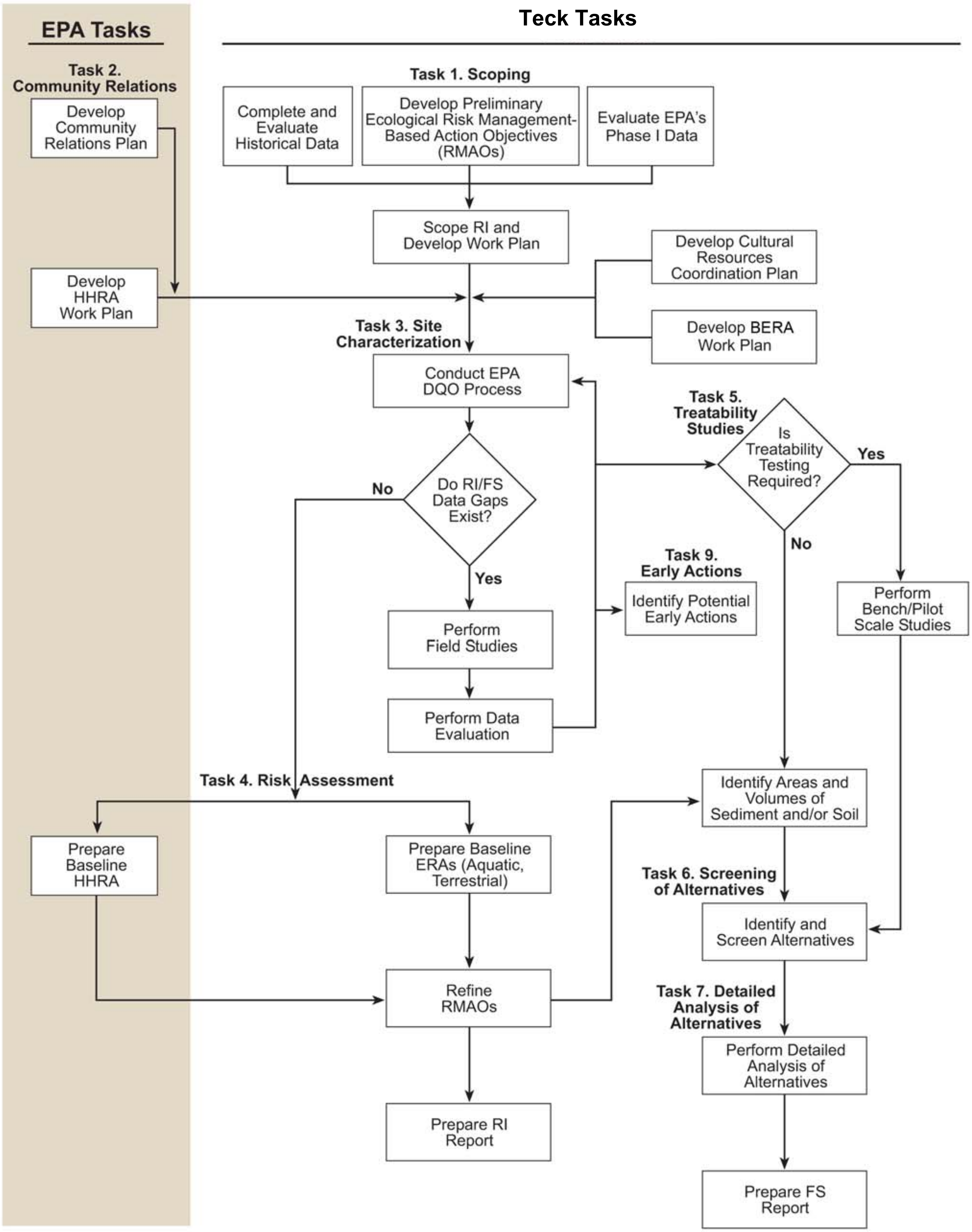
Total suspended solids (TSS)—The portion of the sediment load suspended in the water column. The grain size of suspended sediment is usually less than 1 millimeter in diameter (clays and silts). High TSS concentrations can adversely affect primary food production and fish feeding efficiency. Extremely high TSS concentrations can impair other biological functions such as respiration and reproduction.

Water at the sediment-water interface—Refers to water just above the sediment layer where sediment-dwelling organisms may be exposed to COPCs originating from the sediments or sediment porewater.

Zooplankton—Microscopic and macroscopic animals that swim in the water column. These invertebrates include chiefly three groups: rotifers, cladocerans, and copepods.

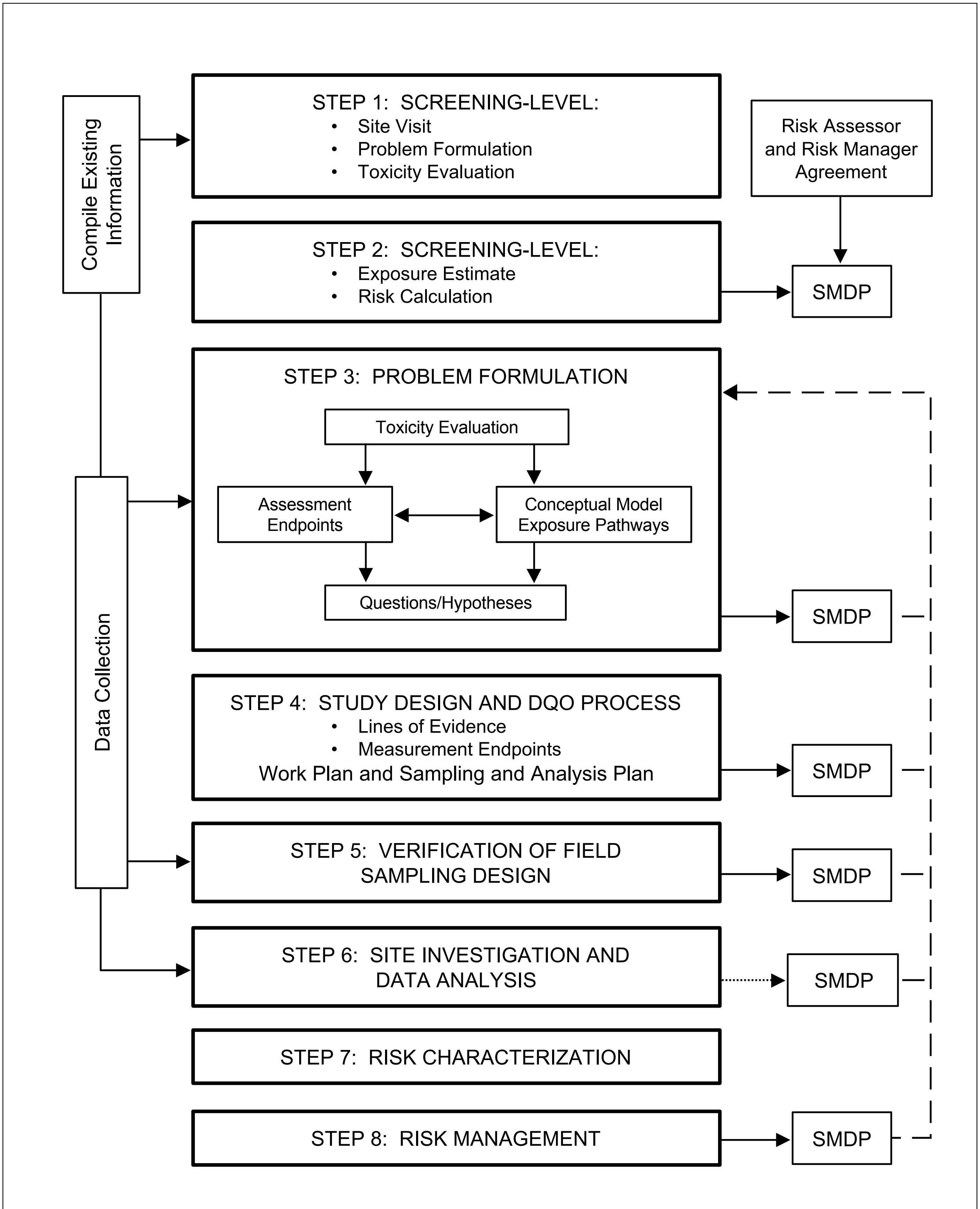
Zooplanktivory—Descriptor for aquatic biota which feed on zooplankton.

FIGURES



Note: Task 8 in the Settlement Agreement is "Project Schedule," which will be ongoing throughout the RI/FS.

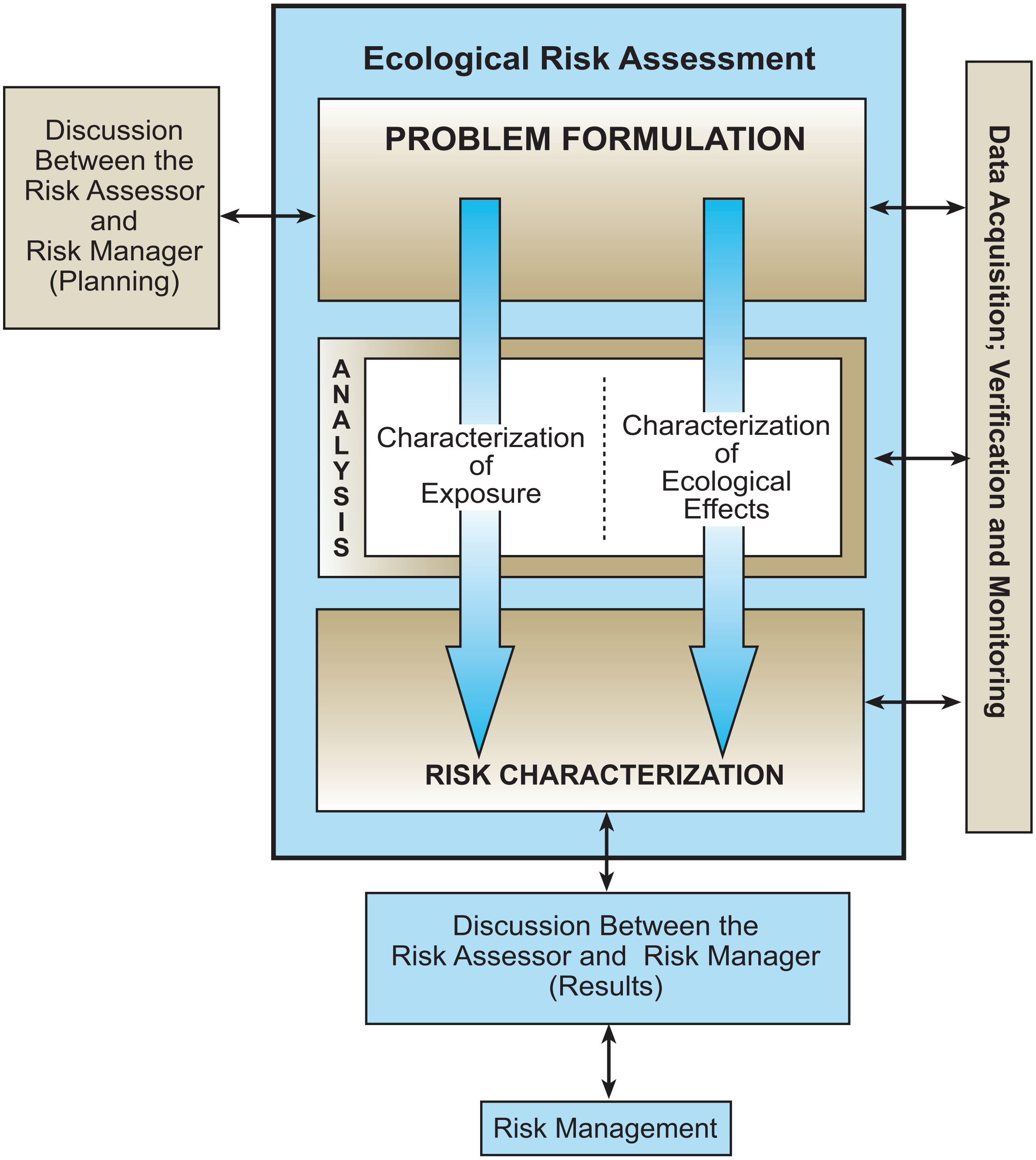
Figure 1-1. Relationships Among Major Tasks in the UCR RI/FS



Scientific Management Decision Point (SMDP)

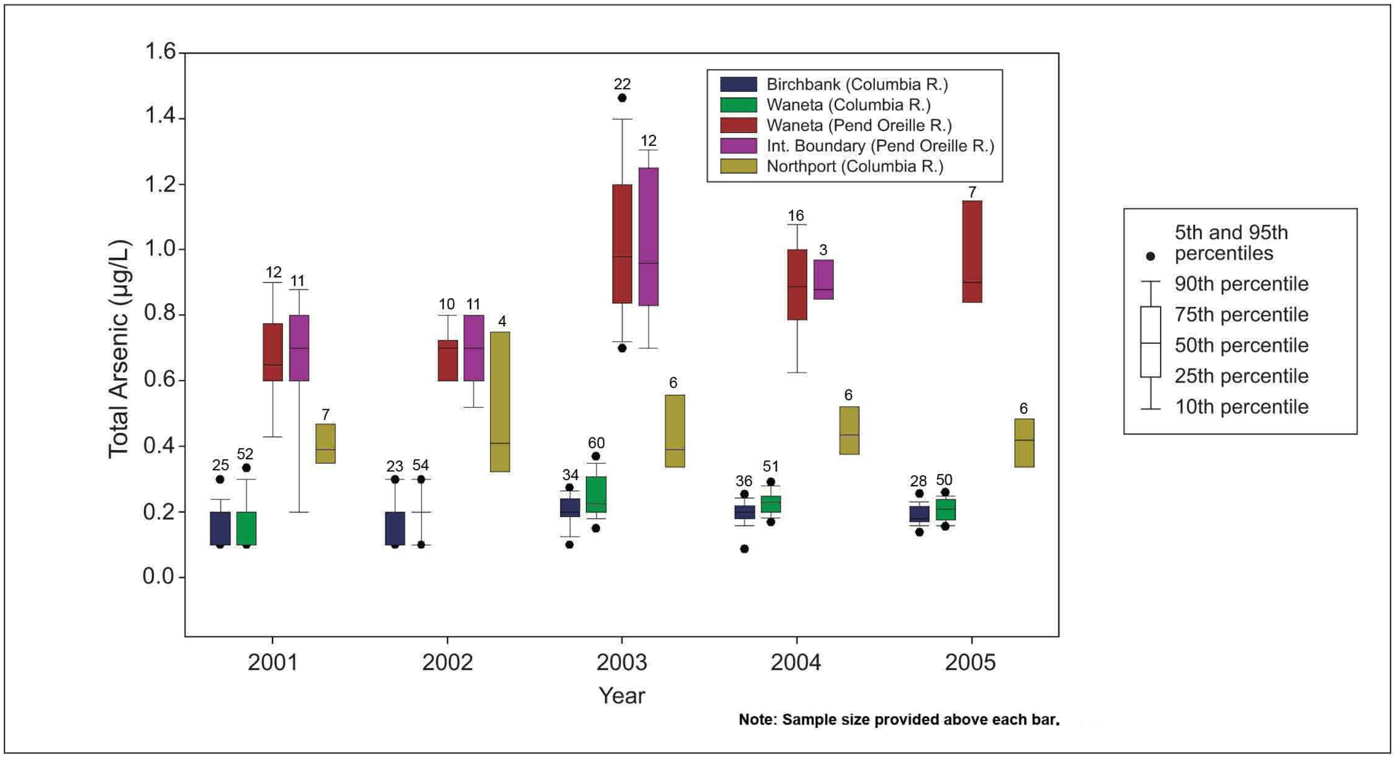
Source: Modified from USEPA 1997a.

Figure 1-2 Eight-step Ecological Risk Assessment Process for Superfund



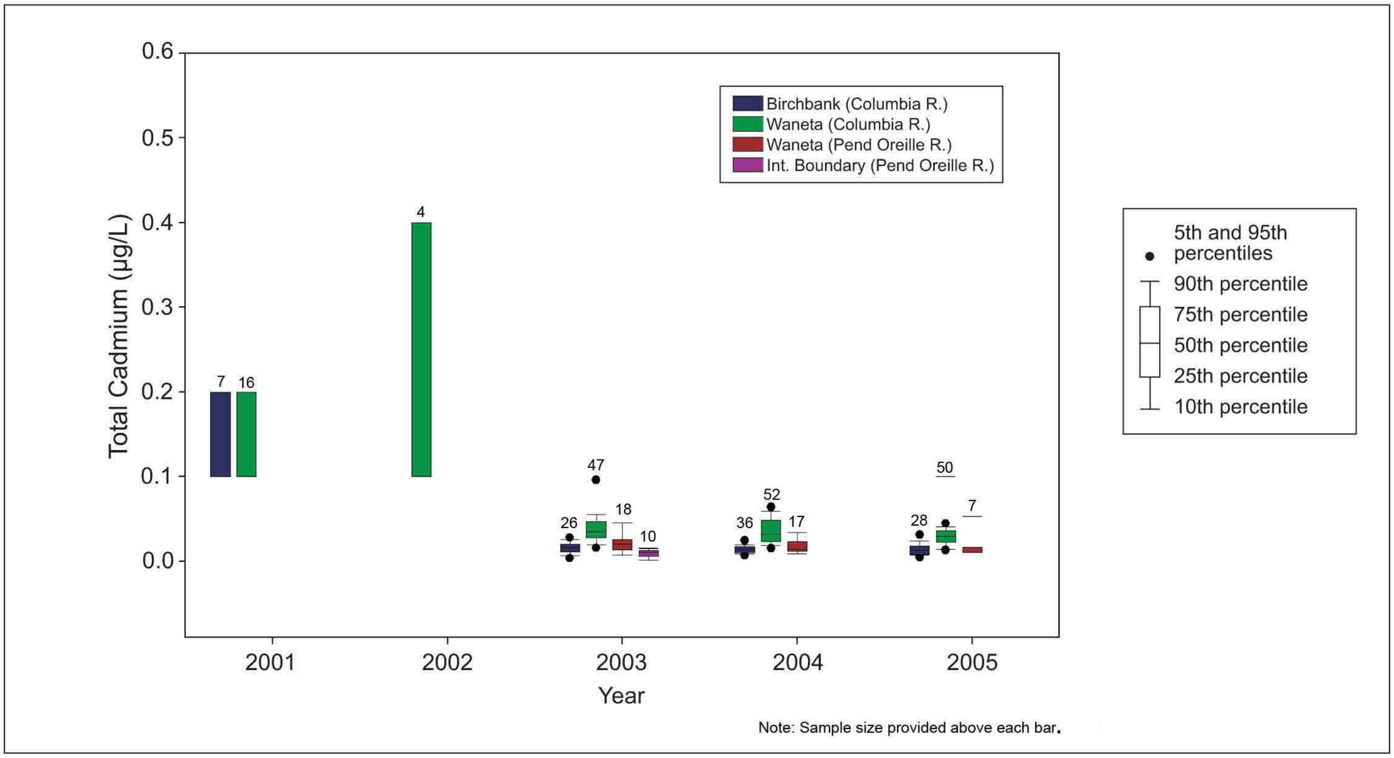
Source: USEPA 1998

Figure 1-3. EPA Ecological Risk Assessment Framework



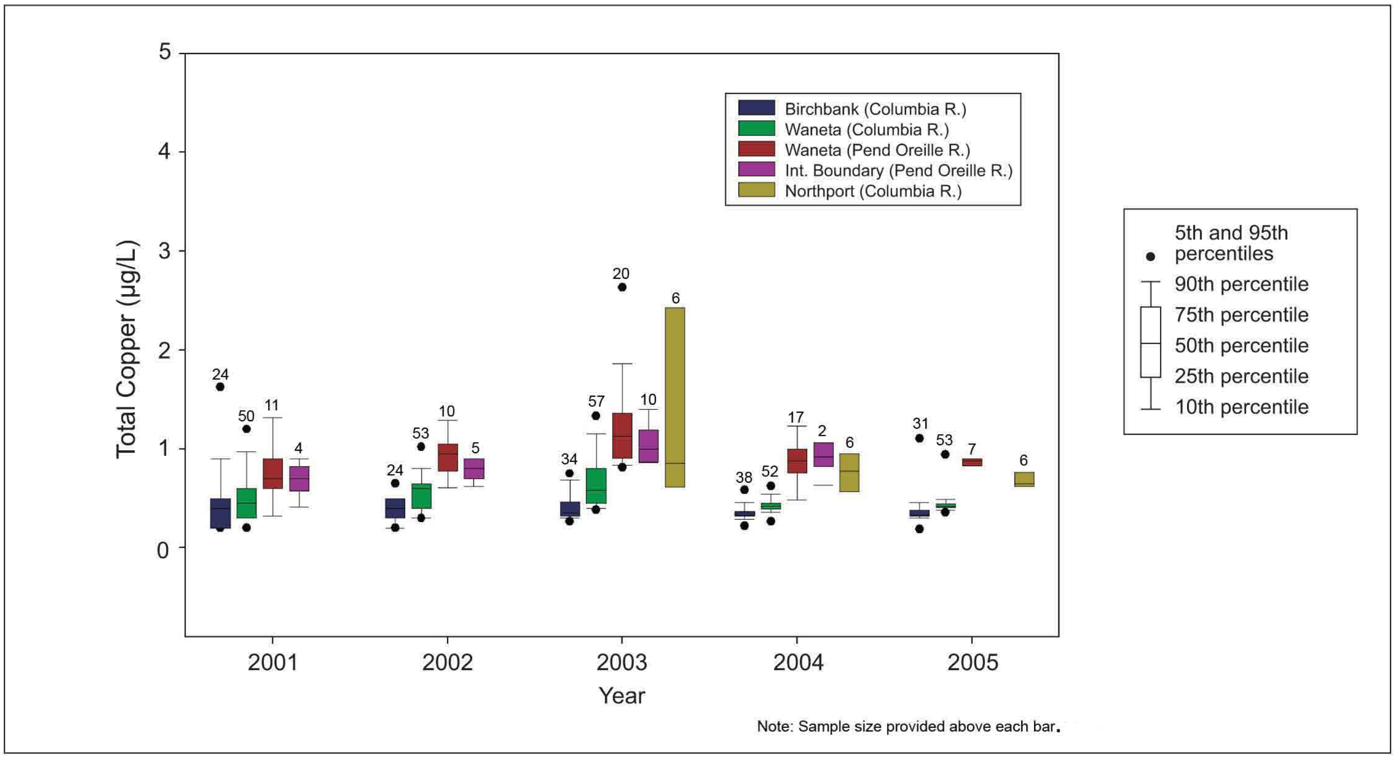
Note: Sample size provided above each bar.

Figure 3-1. Total Arsenic: Comparison of Surface Water Concentrations at Birchbank, Waneta, the U.S.-Canada Border, and Northport (2001-2005)
Source: Environment Canada (http://waterquality.ec.gc.ca), USGS (http://waterdata.usgs.gov)
Note: Box plots based only on detected concentrations.



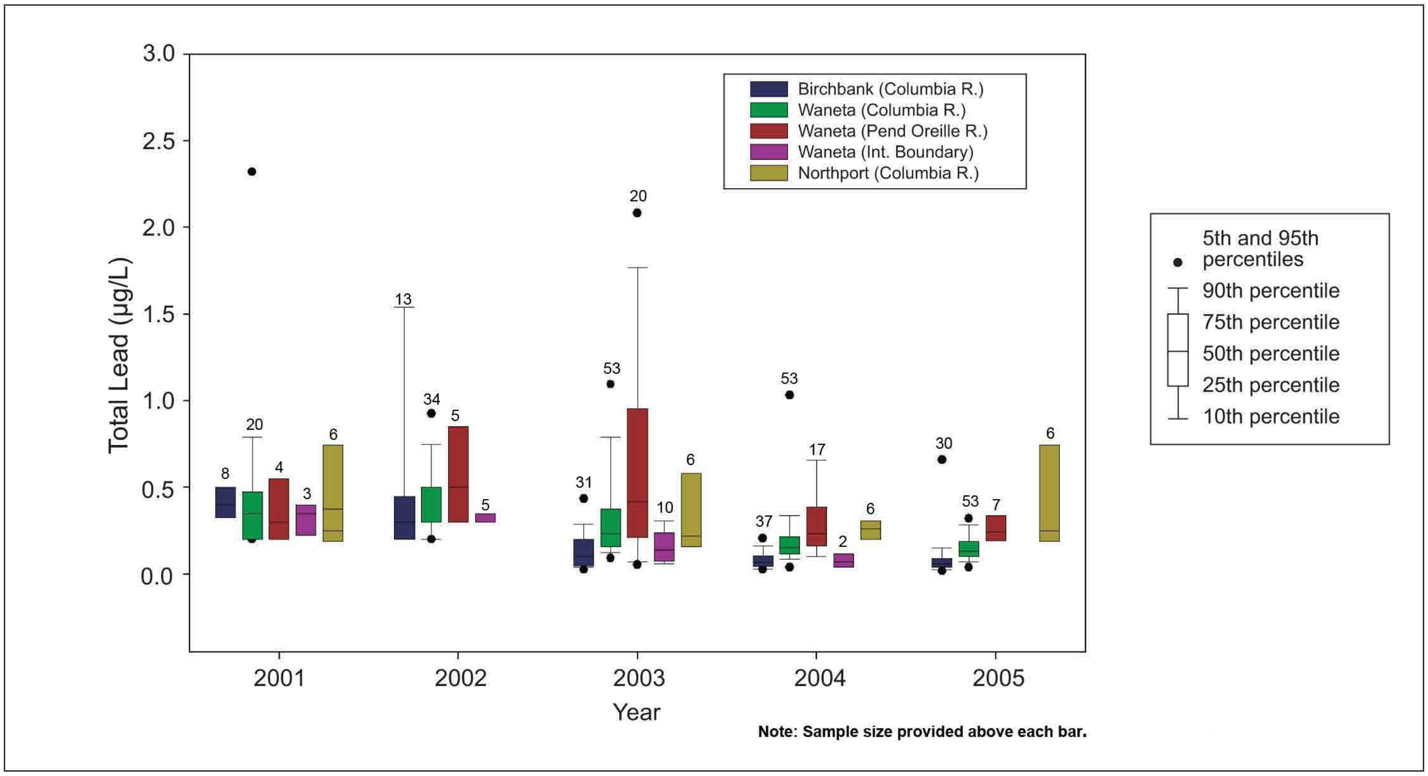
Note: Sample size provided above each bar.

Figure 3-2. Total Cadmium: Comparison of Surface Water Concentrations at Birchbank, Waneta, and the U.S.-Canada Border (2001-2005)
Source: Environment Canada (http://waterquality.ec.gc.ca), USGS (http://waterdata.usgs.gov)
Note: Box plots based only on detected concentrations. Cadmium was detected in only 1 of 26 samples at Northport from 2001-2005 (detection limit of 0.1 µg/L). Data not available for Pend Oreille in 2001 and 2002.



Note: Sample size provided above each bar.

Figure 3-3. Total Copper: Comparison of Surface Water Concentrations at Birchbank, Waneta, the U.S.-Canada Border, and Northport (2001-2005)
Source: Environment Canada (http://waterquality.ec.gc.ca), USGS (http://waterdata.usgs.gov).
Note: Box plots based only on detected concentrations. Copper was not detected at Northport in 2001 and only twice in 2002 (0.49 and 0.78 µg/L).



Note: Sample size provided above each bar.

Figure 3-4. Total Lead: Comparison of Surface Water Concentrations at Birchbank, Waneta, the U.S.-Canada Border, and Northport (2001-2005)
Source: Environment Canada (http://waterquality.ec.gc.ca), USGS (http://waterdata.usgs.gov).
Note: Box plots based only on detected concentrations.

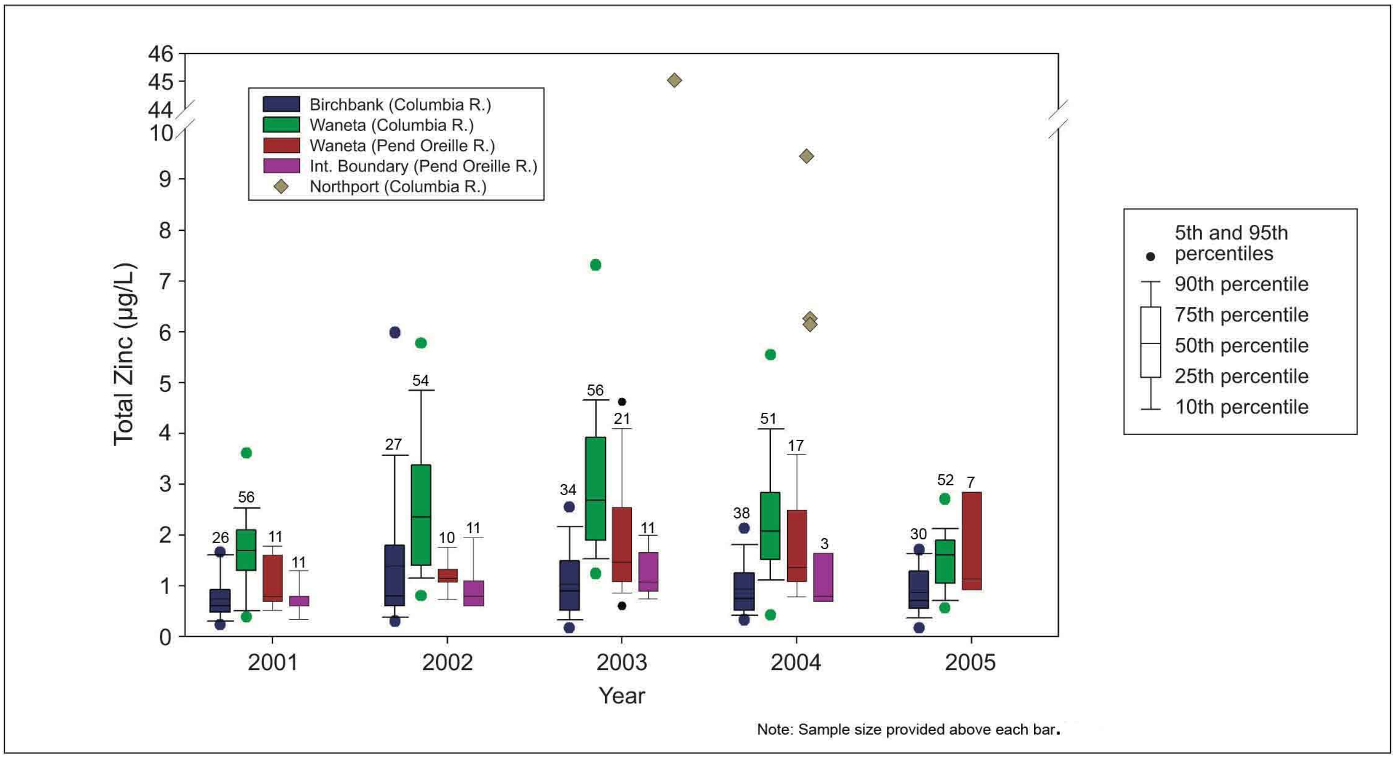


Figure 3-5. Total Zinc: Comparison of Surface Water Concentrations at Birchbank, Waneta, the U.S.-Canada Border, and Northport (2001-2005)
Source: Environment Canada (http://waterquality.ec.gc.ca), USGS (http://waterdata.usgs.gov).
Note: Zinc was infrequently detected at Northport at a detection limit of 5 µg/L. Detected concentrations are show as individual points. Box plots based only on detected concentrations.

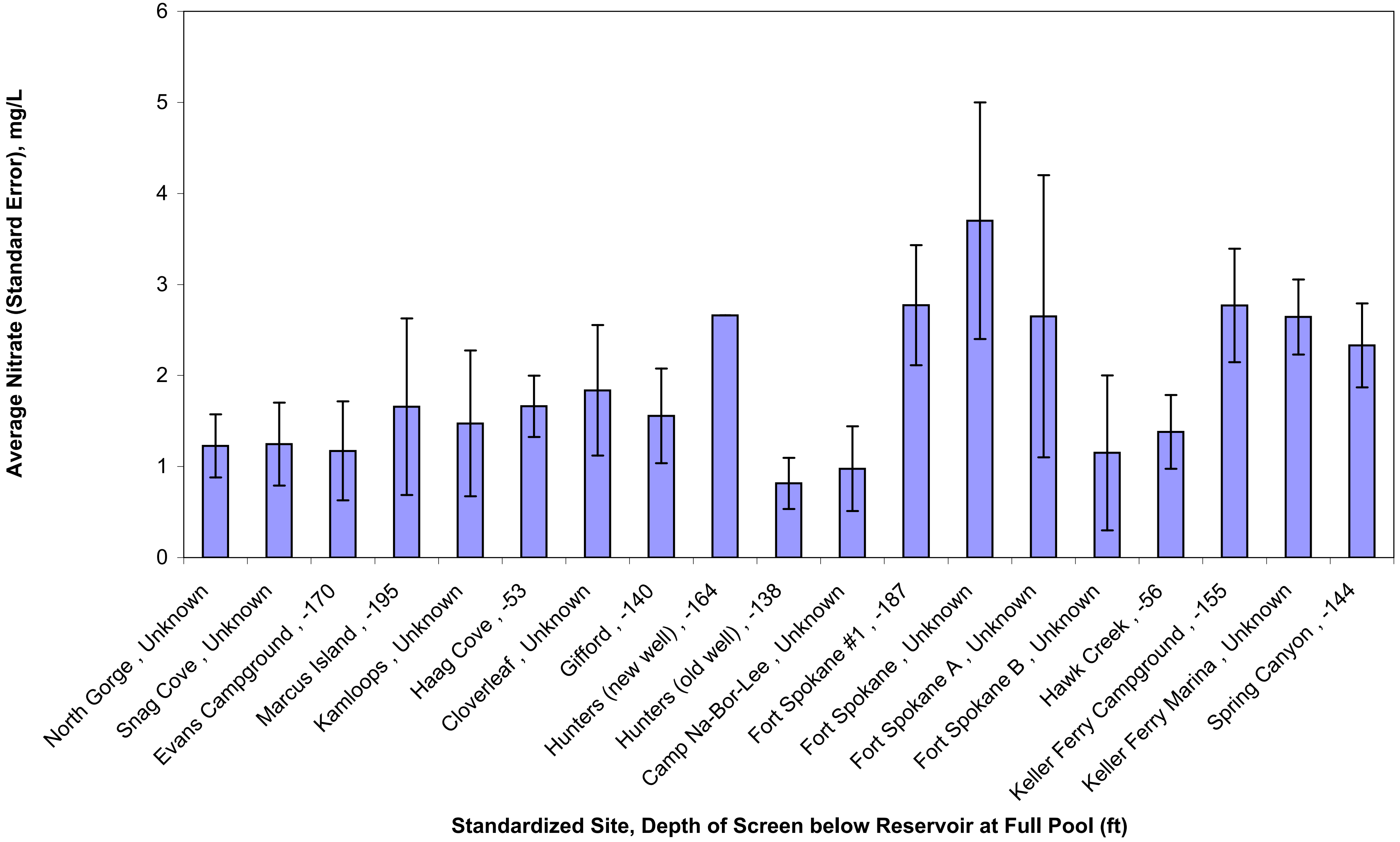


Figure 3-6. Average Nitrate Concentrations as a Function of Well Location

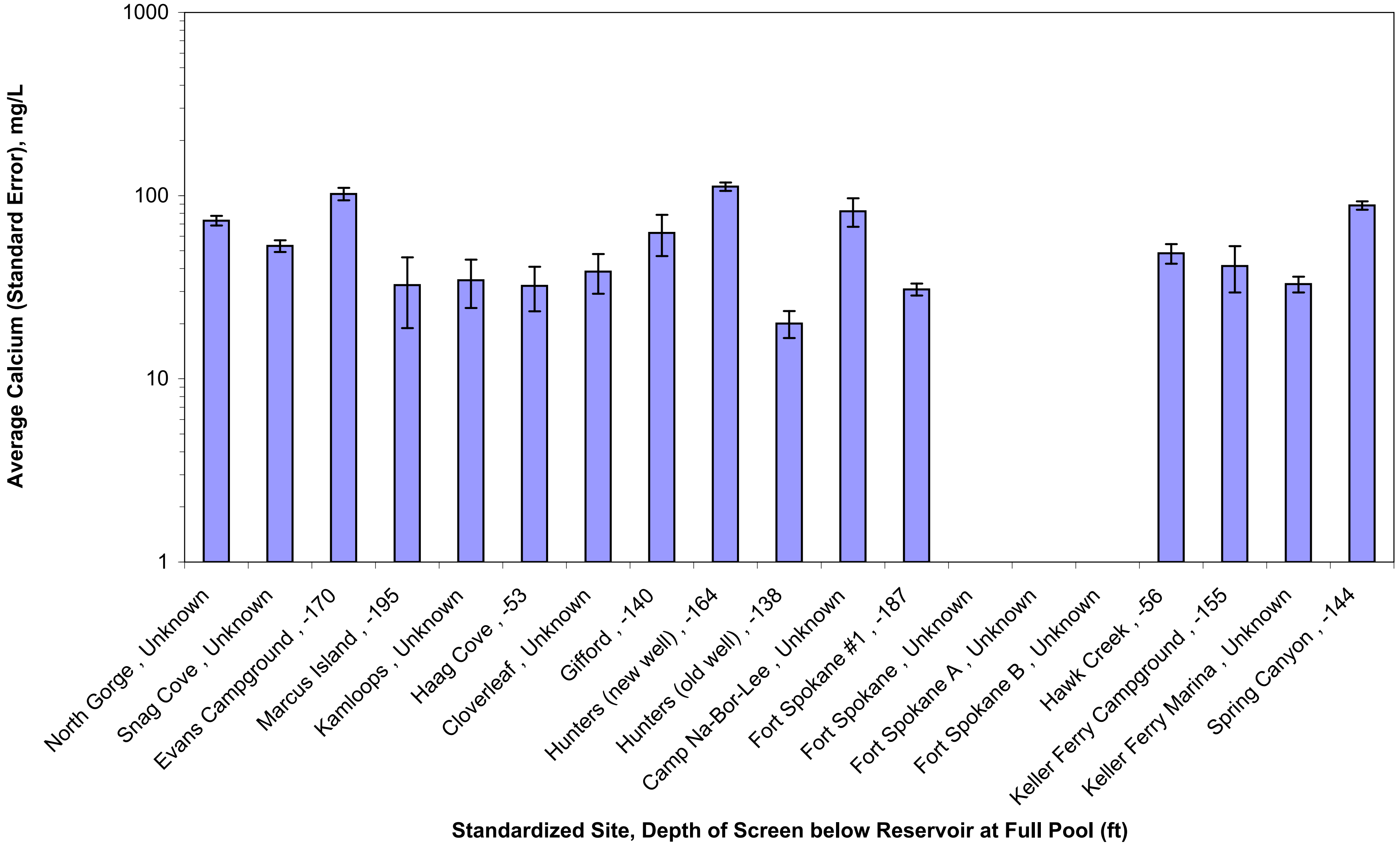


Figure 3-7. Average Calcium Concentrations as a Function of Well Location

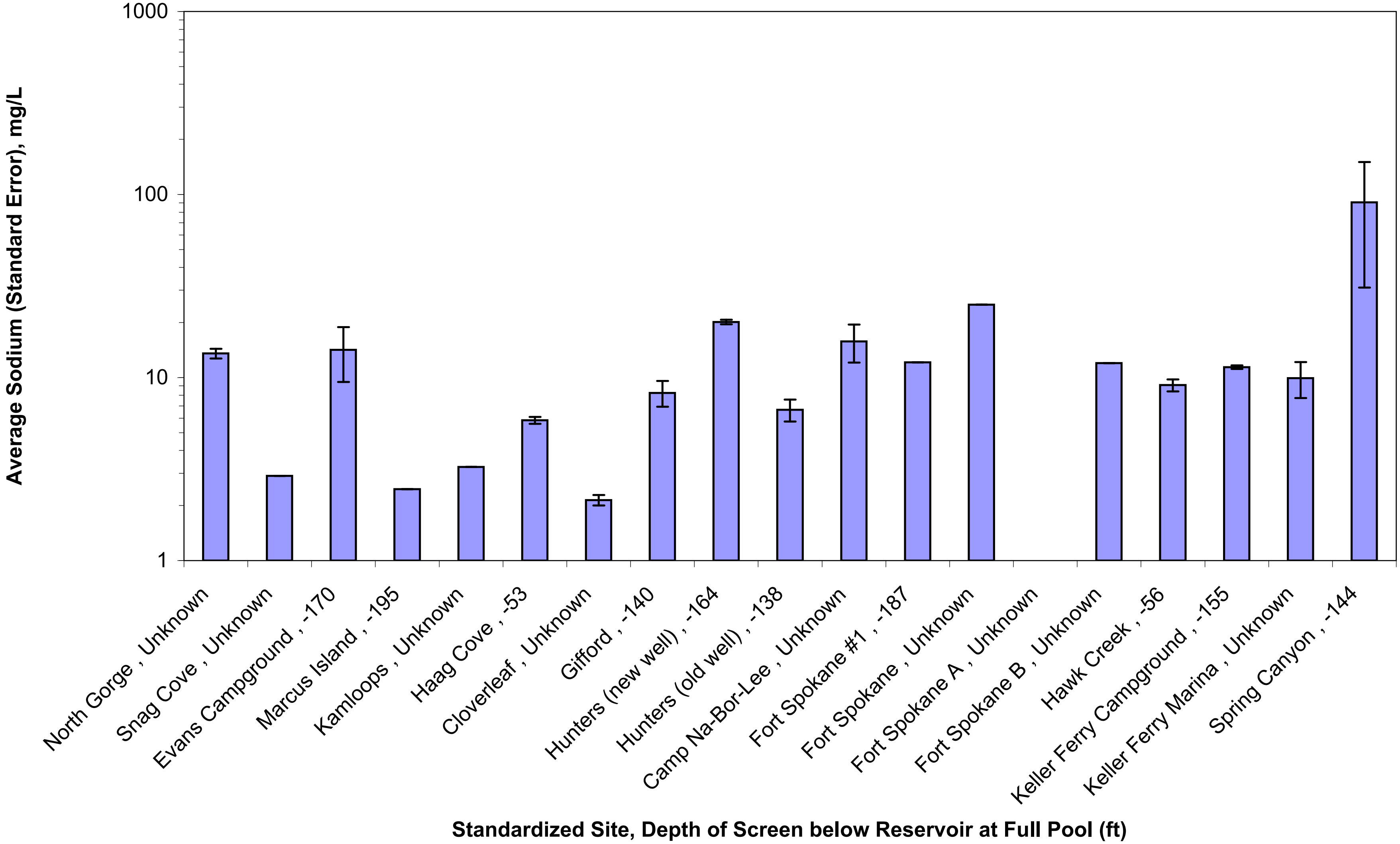


Figure 3-8. Average Sodium Concentrations as a Function of Well Location

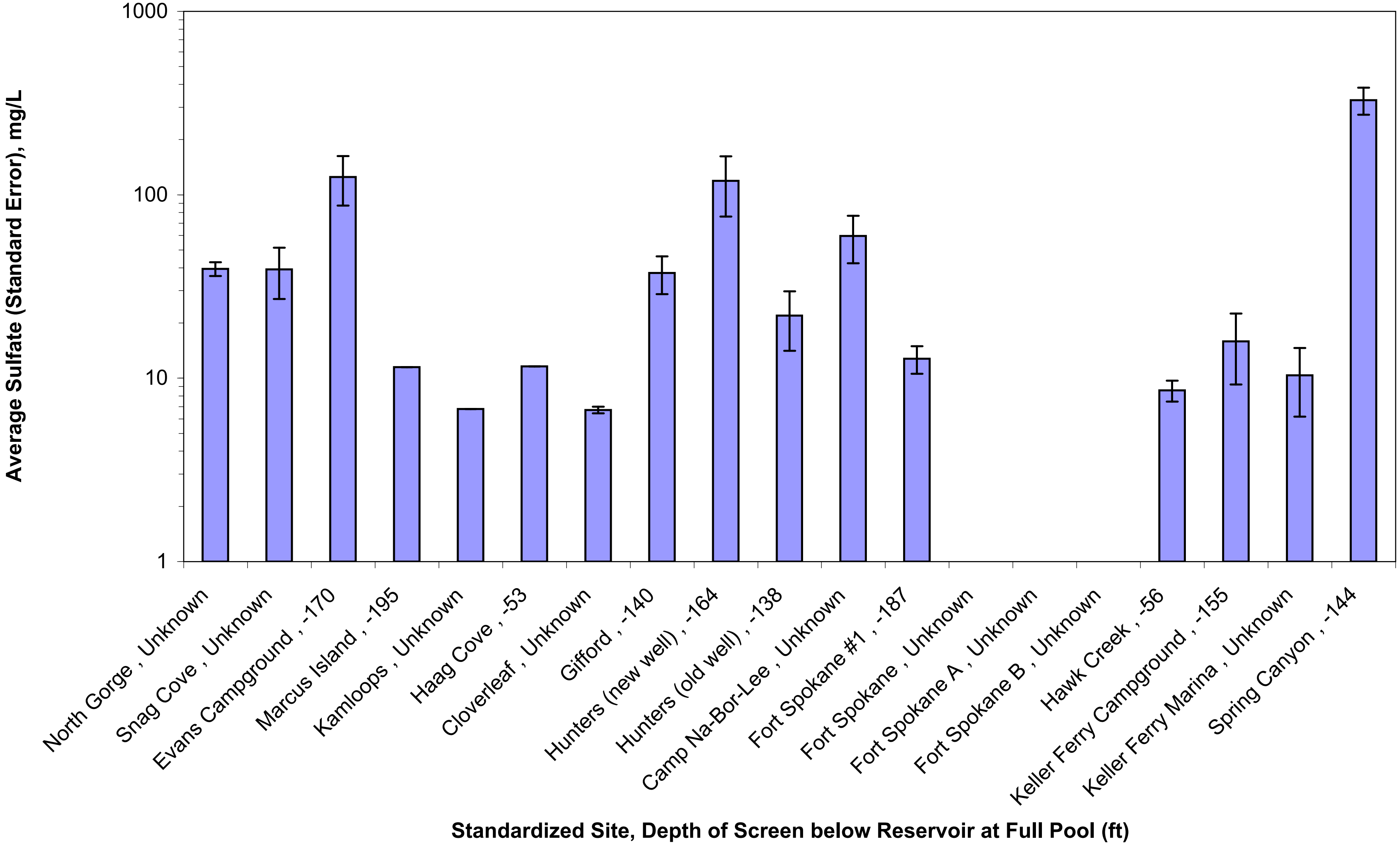


Figure 3-9. Average Sulfate Concentrations as a Function of Well Location

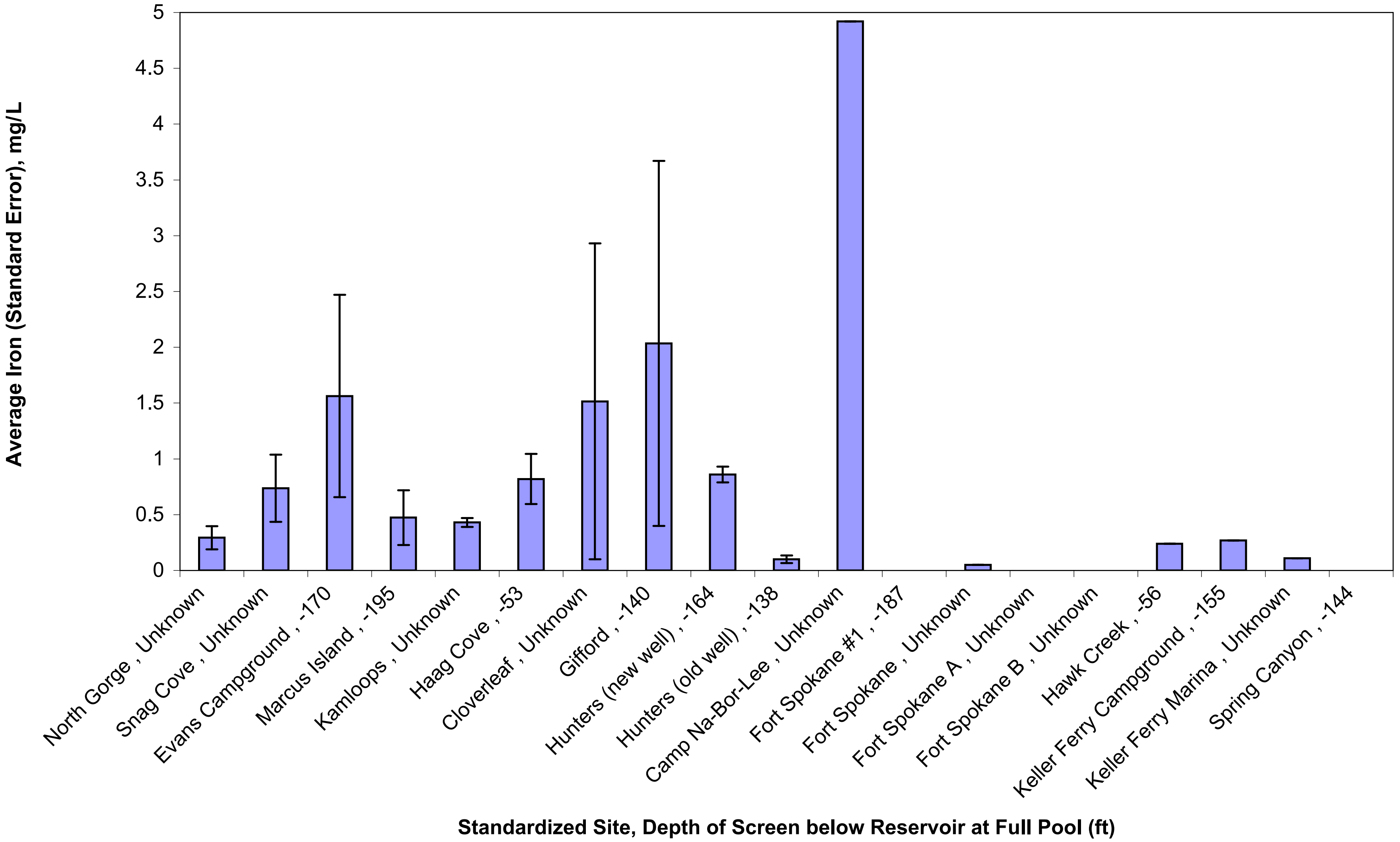


Figure 3-10. Average Iron Concentrations as a Function of Well Location

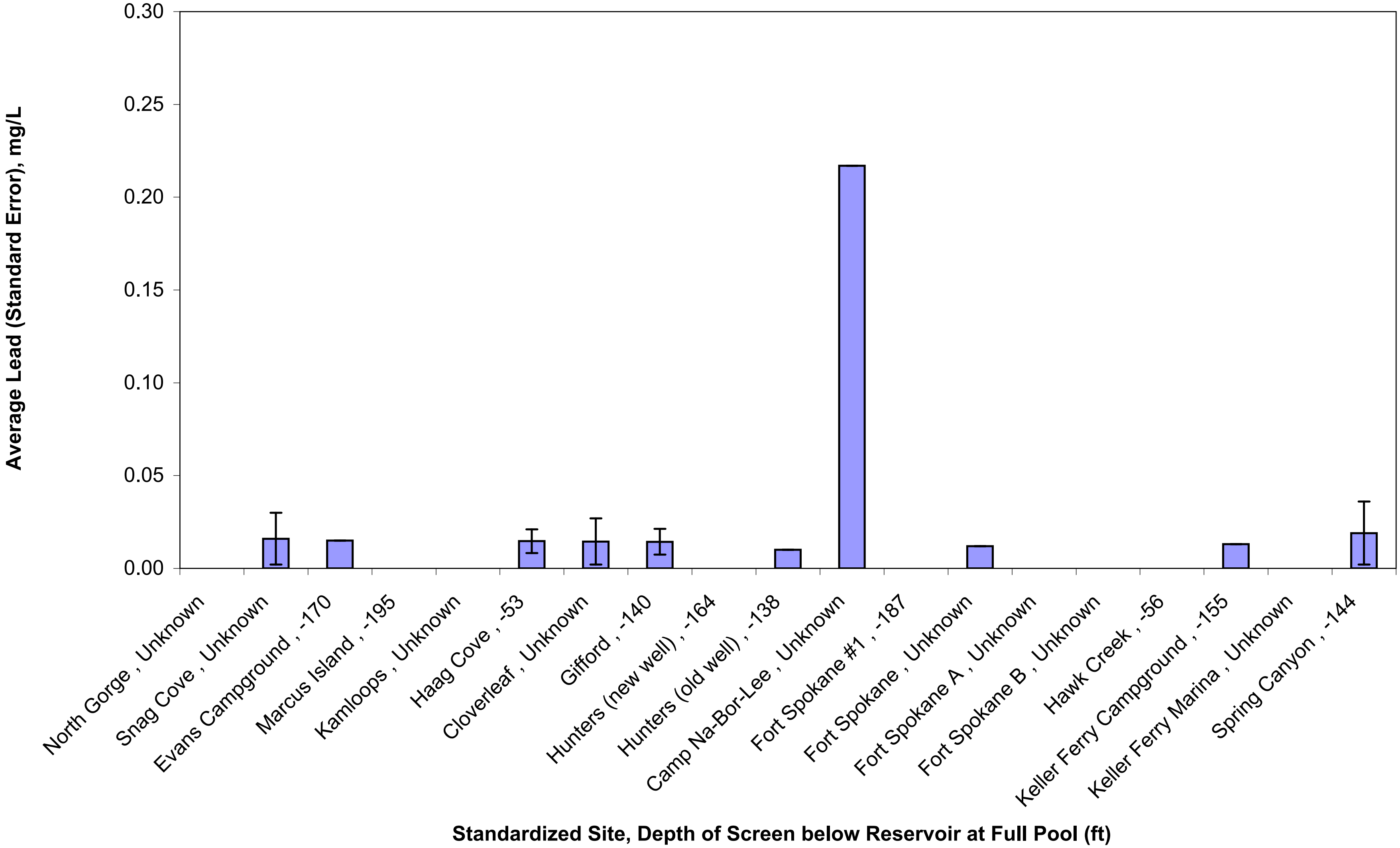


Figure 3-11. Average Lead Concentrations as a Function of Well Location

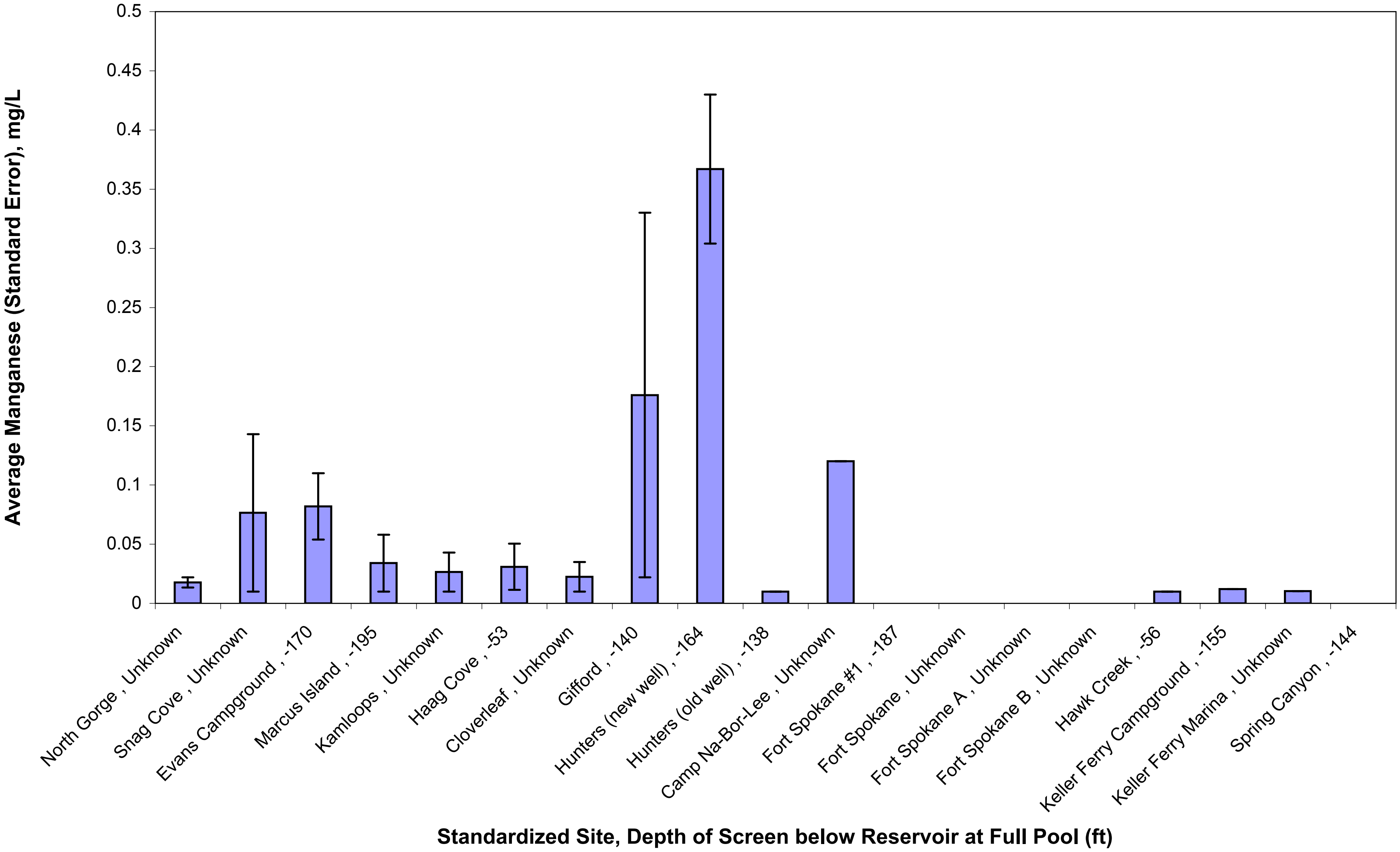


Figure 3-12. Average Manganese Concentrations as a Function of Well Location

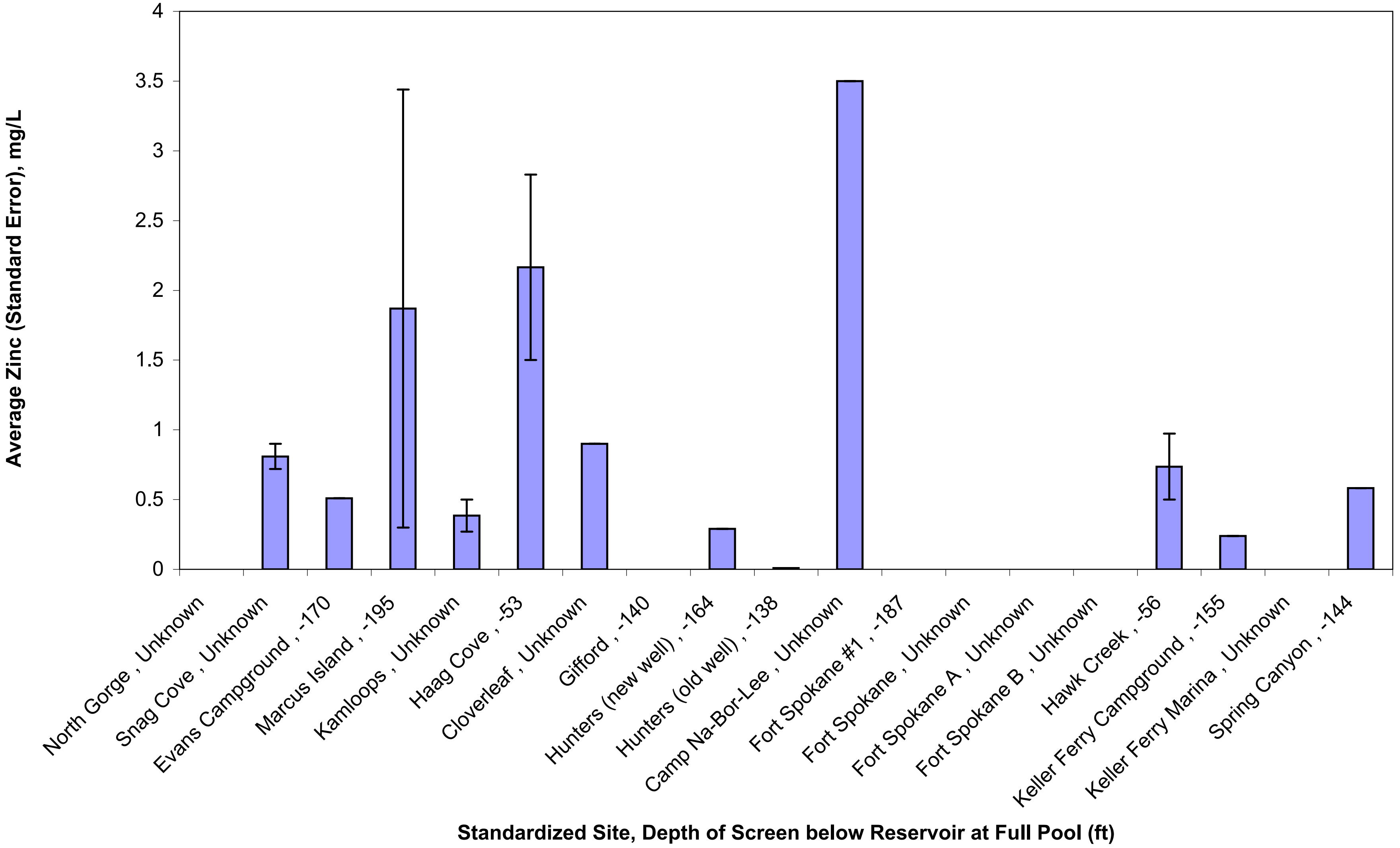


Figure 3-13. Average Zinc Concentrations as a Function of Well Location

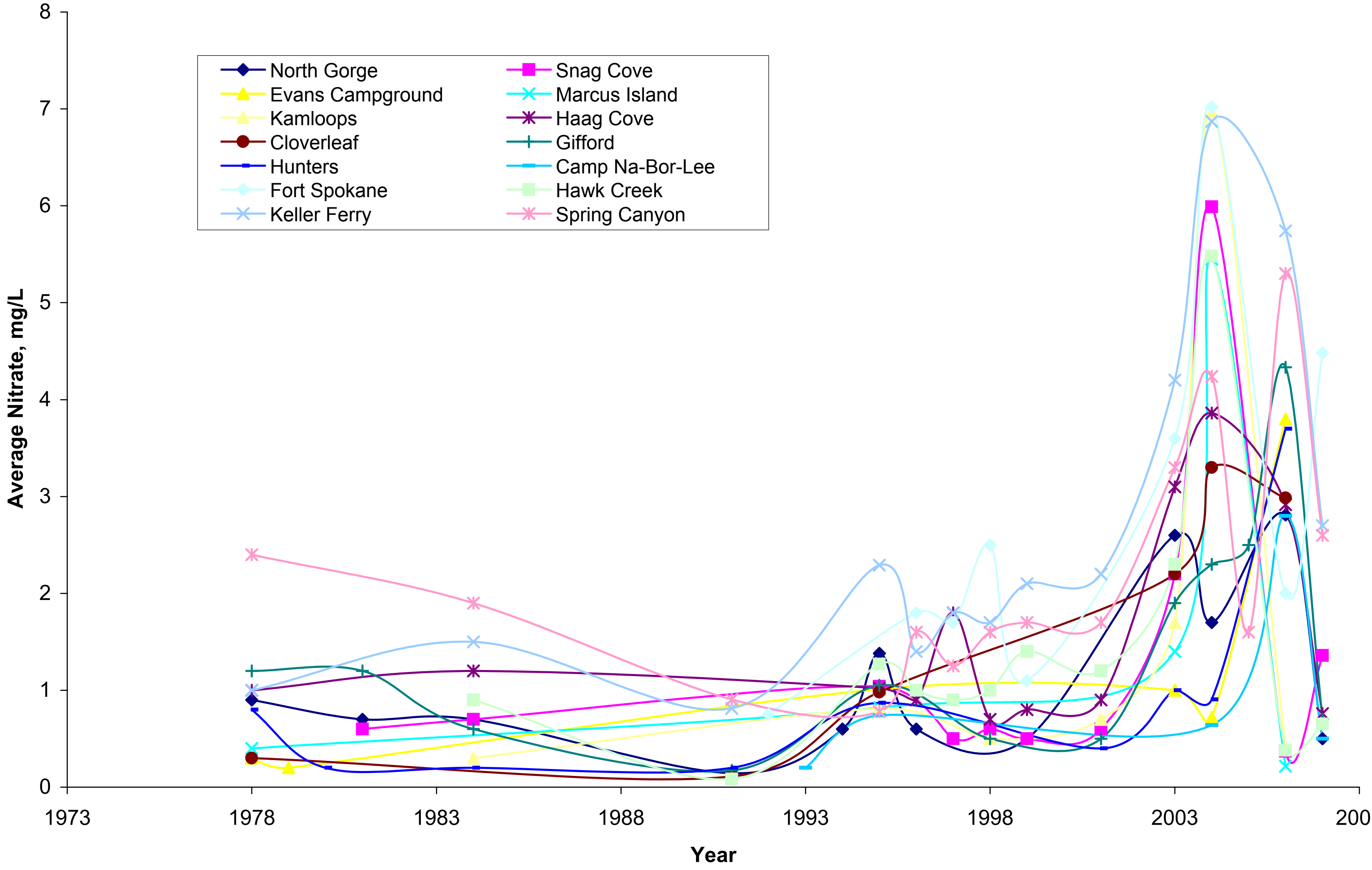
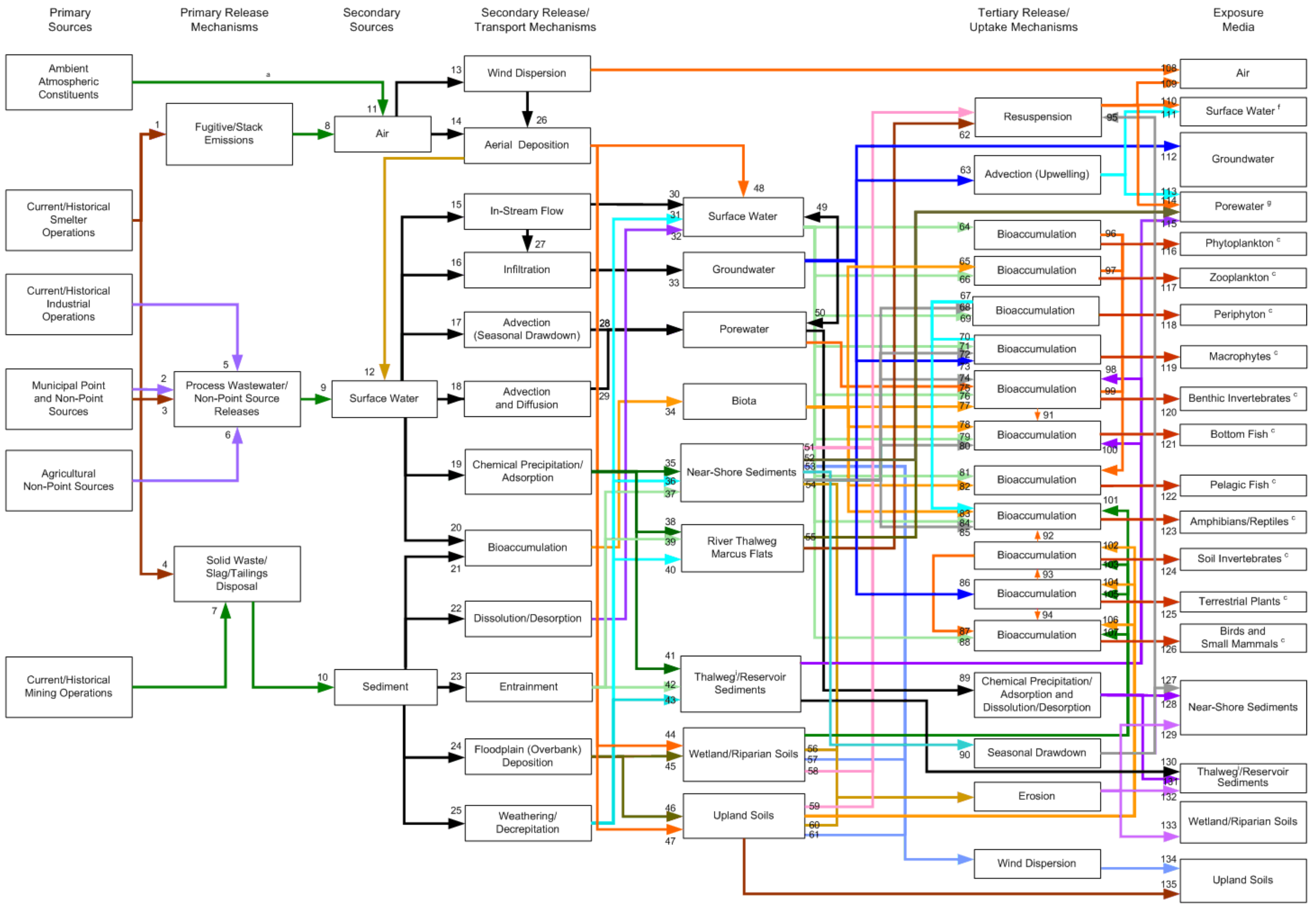


Figure 3-14. Average Nitrate Concentrations as a Function of Time



Exposure Media	Exposure Pathway	Ecological Exposure Pathways																								Human Exposure Pathways [j]
		Aquatic Foodweb [b]																Riparian/Upland Foodweb								
		Phytoplankton	Zooplankton	Periphyton	Aquatic Macrophytes	Benthic Invertebrates [h]	Bottom Fish	Sturgeon	Pelagic Fish	Amphibians (Eggs and Larvae)	Amphibians (Juveniles and Adults)	Aquatic Reptiles	Piscivorous Birds and Mammals	Omnivorous Birds and Mammals	Invertivorous Birds	Herbivorous Birds and Mammals	Aerial Insectivorous Birds and Mammals	Soil Invertebrates	Terrestrial Plants	Terrestrial Amphibians (Juveniles and Adults)	Terrestrial Reptiles	Herbivorous Birds and Mammals	Insectivorous Birds and Mammals	Omnivorous Birds and Mammals	Carnivorous Birds and Mammals	
Air	Inhalation	N	N	N	N	N	N	N	N	N	●	●	●	●	●	●	●	●	N	●	●	●	●	●	●	●
	Foliar Uptake	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	●	N	N	N	N	N	N	N
Surface Water [f]	Ingestion	N	●	N	N	●	●	●	●	●	●	●	●	●	●	●	●	●	N	●	●	●	●	●	●	●
	Direct Contact	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	○	●	●	●	●	●	●	●	●
Groundwater	Root Uptake	N	N	N	●	N	N	N	N	N	N	N	N	N	N	N	N	N	●	N	N	N	N	N	N	N
	Ingestion	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	●
	Direct Contact	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	●
	Inhalation	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	● [e]
Porewater [g]	Ingestion	N	○	N	N	●	●	●	●	○	○	○	○	○	○	○	○	N	N	N	N	N	N	N	N	○
	Direct Contact	○	○	○	●	●	●	●	●	●	○	○	○	○	●	○	○	N	N	N	N	N	N	N	N	○
Phytoplankton [c]	Ingestion	N	●	N	N	●	N	N	●	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Zooplankton [c]	Ingestion	N	●	N	N	●	●	●	●	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Periphyton [c]	Ingestion	N	●	N	N	●	●	●	○	●	○	●	○	●	○	●	○	N	N	N	N	N	N	N	N	N
Macrophytes [c]	Ingestion	N	●	N	N	●	●	●	○	●	○	●	○	●	○	●	○	N	N	N	N	N	N	N	N	N
Benthic Invertebrates [c]	Ingestion	N	N	N	N	●	●	●	●	○	●	●	●	●	●	○	●	N	N	N	N	N	N	N	N	●
Bottom Fish [c]	Ingestion	N	N	N	N	N	●	●	●	○	○	●	●	○	○	○	○	N	N	N	N	N	N	N	N	●
Pelagic Fish [c]	Ingestion	N	N	N	N	N	●	●	●	○	○	●	●	○	○	○	○	N	N	N	N	N	N	N	N	●
Amphibians/Reptiles [c]	Ingestion	N	N	N	N	N	○	○	○	○	●	●	●	●	●	○	○	N	N	●	●	○	●	●	●	●
Soil Invertebrates [c]	Ingestion	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	●	●	○	●	●	●	N
Terrestrial Plants [c]	Ingestion	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	●	N	N	N	●	○	●	○	●
Birds and Small Mammals [c]	Ingestion	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	●	○	○	●	●	●
Near-Shore Sediments	Ingestion	N	N	N	N	●	●	●	●	N	●	●	●	●	●	●	○	N	N	●	●	●	●	●	●	●
	Direct Contact	● [d]	●	● [d]	● [d]	●	●	●	●	●	●	●	●	●	●	●	○	N	●	●	●	●	●	●	●	●
	Inhalation	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	●
Thalweg/Reservoir Sediments	Ingestion	N	N	N	N	●	●	●	●	○	○	○	●	●	○	○	○	N	N	N	N	N	N	N	N	●
	Direct Contact	○ [d]	●	○ [d]	●	●	●	●	●	○	○	○	○	○	○	○	○	N	N	N	N	N	N	N	N	●
Wetland/Riparian Soils	Ingestion	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	●	N	●	●	●	●	●	●	●
	Direct Contact	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	●	●	●	●	●	●	●	●	●
	Inhalation	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	●
Upland Soils	Ingestion	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	●	N	●	●	●	●	●	●	●
	Direct Contact	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	●	●	●	●	●	●	●	●	●
	Inhalation	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	●

Legend:

●	Exposure Pathway Potentially Complete
○	Incomplete Exposure Pathway
N	Pathway Not Applicable

Notes:

a The different colors used for the arrows linking the boxes were selected to help distinguish the various linkages visually, and have no technical meaning.

b Differential exposure pathways exist for different life stages of some receptors (fish and amphibian eggs, larvae, adults). These will need to be considered/outlined in the final CSM.

c Upon death, receptor contributes, as solid-phase and liquid-phase detritus to the dietary, dermal, and incidental ingestion pathways...dietary is probably of importance

d Chemisorption onto external organic material

e Inhalation of COCs contained in media via Sweatlodge pathway

f Surface water may be affected by groundwater discharge from the side banks during pool drawdown.

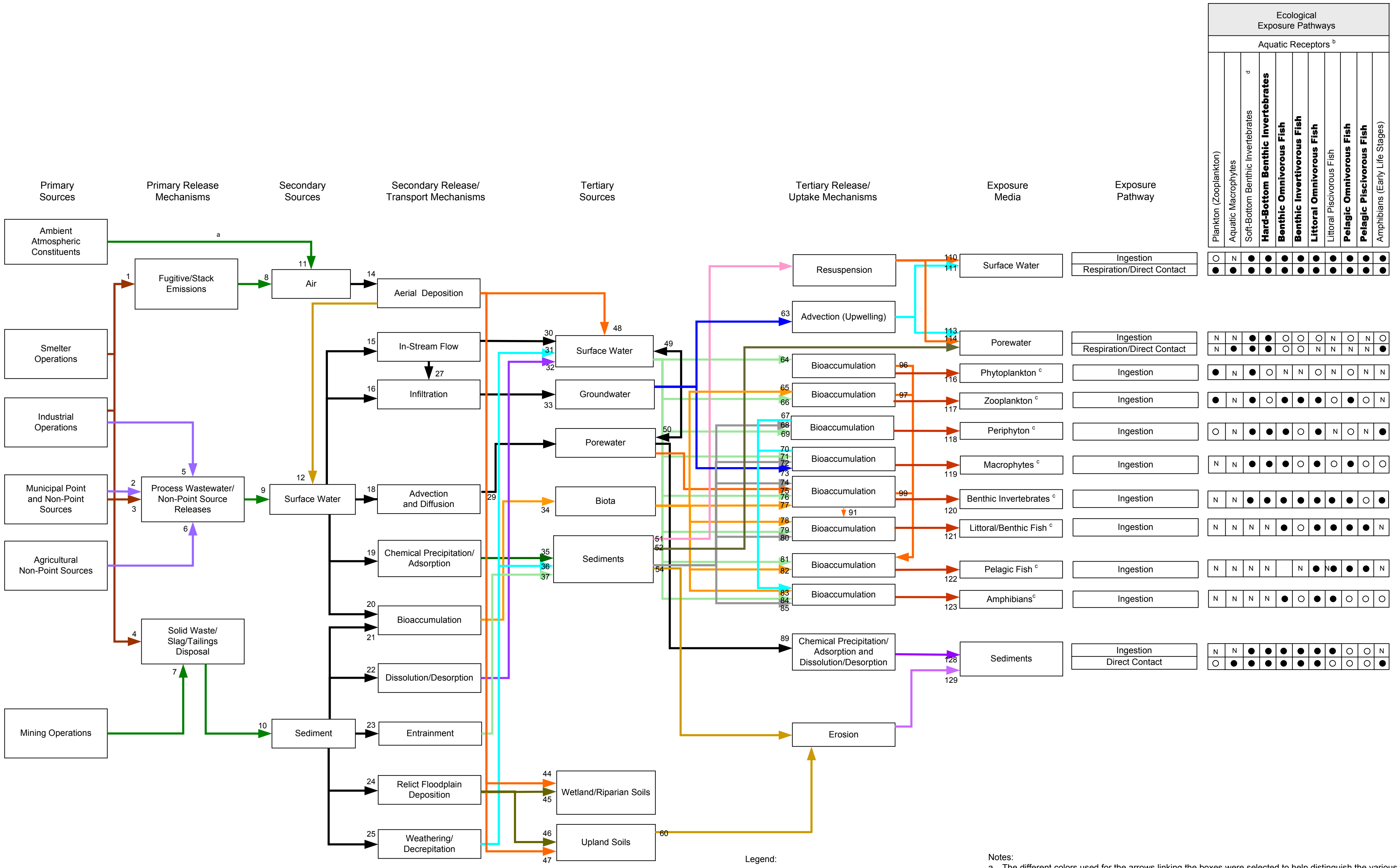
g Porewater may be affected or replaced via groundwater advection.

h Includes mussels

i Thalweg refers to the pre-reservoir channel.

j The human health risk assessment will be performed by the EPA. This CSM represents the current understanding of the human health exposure pathways at the time the work plan was prepared. The human exposure media (biotic and abiotic) and pathways will be refined dependent upon the results of the planned Tribal Exposure Survey and more detailed input from the Spokane Tribes of the Colville Indian Reservation.

Figure 4-1. Sitewide Conceptual Site Model



Ecological Exposure Pathways

Exposure Media	Exposure Pathway	Plankton (Zooplankton)	Aquatic Macrophytes	Soft-Bottom Benthic Invertebrates [d]	**Hard-Bottom Benthic Invertebrates**	**Benthic Omnivorous Fish**	**Benthic Invertivorous Fish**	**Littoral Omnivorous Fish**	Littoral Piscivorous Fish	**Pelagic Omnivorous Fish**	**Pelagic Piscivorous Fish**	Amphibians (Early Life Stages)
		Aquatic Receptors [b]										
Surface Water	Ingestion	○	N	●	●	●	●	●	●	●	●	●
	Respiration/Direct Contact	●	●	●	●	●	●	●	●	●	●	●
Porewater	Ingestion	N	N	●	●	○	○	○	N	○	N	○
	Respiration/Direct Contact	N	●	●	●	○	○	N	N	N	N	●
Phytoplankton [c]	Ingestion	●	N	●	○	N	N	○	N	○	N	N
Zooplankton [c]	Ingestion	●	N	●	○	●	●	●	○	●	○	N
Periphyton [c]	Ingestion	○	N	●	●	●	○	●	N	○	N	●
Macrophytes [c]	Ingestion	N	N	●	●	●	○	●	○	●	○	○
Benthic Invertebrates [c]	Ingestion	N	N	●	●	●	●	●	●	●	○	●
Littoral/Benthic Fish [c]	Ingestion	N	N	N	N	●	○	●	●	●	●	N
Pelagic Fish [c]	Ingestion	N	N	N	N		N	●	N●	●	●	N
Amphibians[c]	Ingestion	N	N	N	N	●	○	●	●	○	○	○
Sediments	Ingestion	N	N	●	●	●	●	●	●	○	○	N
	Direct Contact	○	●	●	●	●	●	●	○	○	○	●

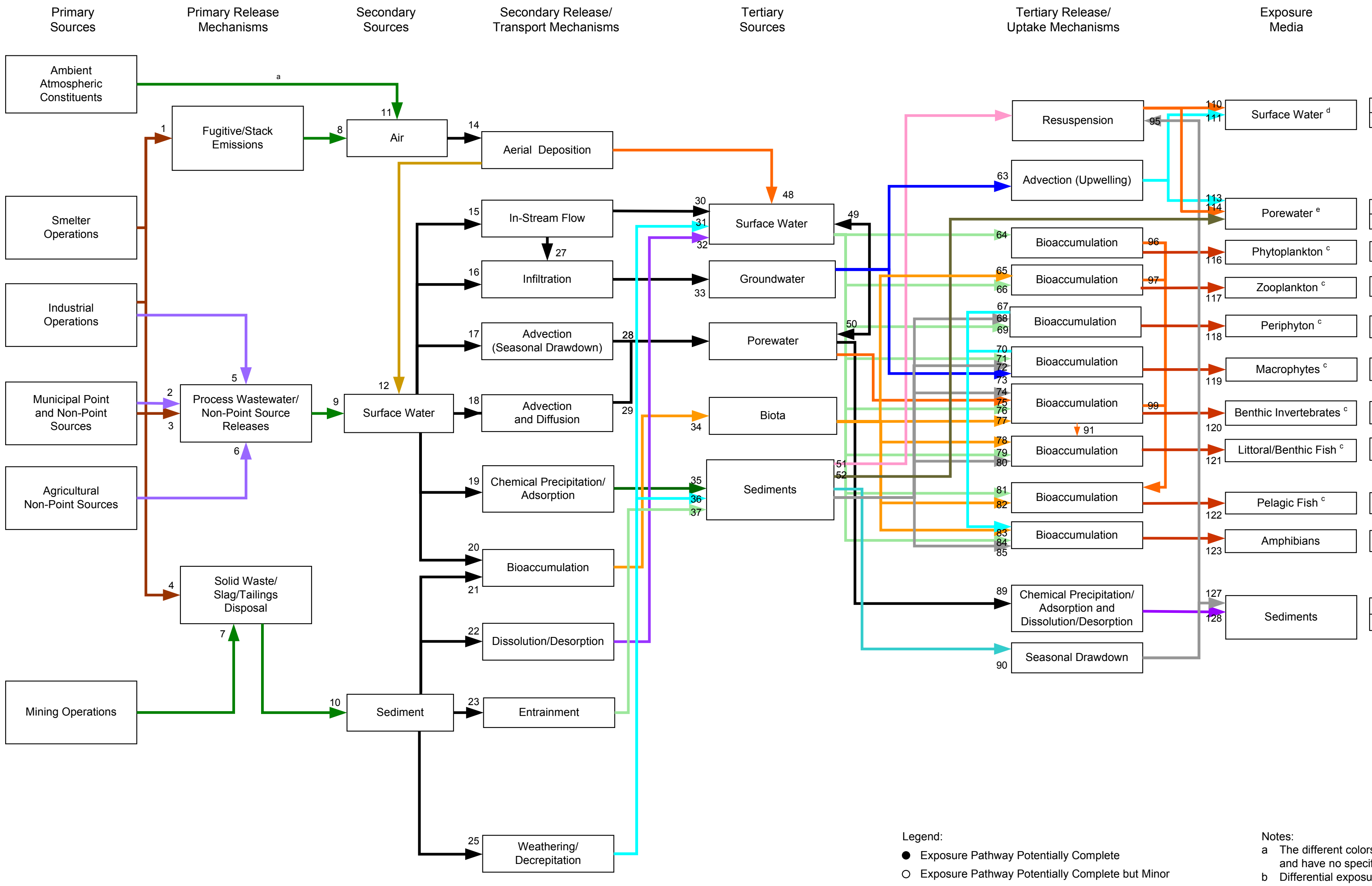
Legend:

- ● Exposure Pathway Potentially Complete
- ○ Exposure Pathway Potentially Complete but Minor
- N Pathway Not Applicable/Incomplete

Notes:

a The different colors used for the arrows linking the boxes were selected to help distinguish the various linkages visually, and have no specific technical connotation.
b Differential exposure pathways exist for different life stages of some receptors (fish and amphibian eggs, larvae, adults).
c Upon death, organism contributes to detritus.
d Includes mussels

Figure 4-2. Conceptual Site Model for the Riverine Areas (Reaches 1 and 2)



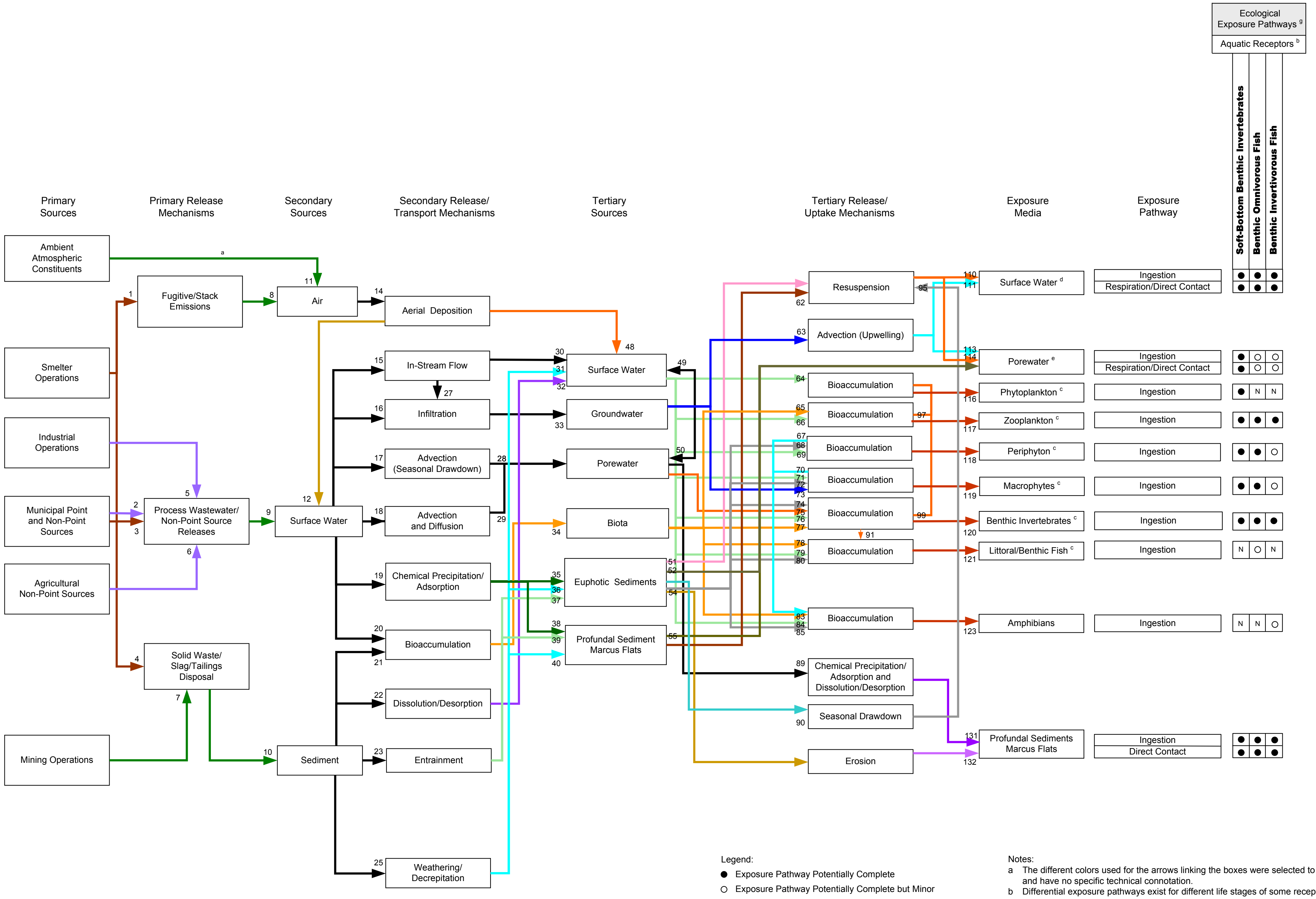
Ecological Exposure Pathways g

Exposure Media	Exposure Pathway	Aquatic Receptors b Plankton (Zooplankton)	Aquatic Macrophytes	Soft-Bottom Benthic Invertebrates f	**Benthic Omnivorous Fish**	**Benthic Invertivorous Fish**	**Littoral Omnivorous Fish**	Littoral Piscivorous Fish	**Pelagic Omnivorous Fish**	**Pelagic Piscivorous Fish**	Amphibians (Early Life Stages)
Surface Water d	Ingestion	○	N	●	●	●	●	●	●	●	●
	Respiration/Direct Contact	●	●	●	●	●	●	●	●	●	●
Porewater e	Ingestion	N	N	●	○	○	○	N	○	N	○
	Respiration/Direct Contact	N	●	●	○	○	N	N	N	N	●
Phytoplankton c	Ingestion	●	N	●	N	N	○	N	○	N	N
Zooplankton c	Ingestion	●	N	●	●	●	●	○	●	○	N
Periphyton c	Ingestion	●	N	●	●	○	●	N	○	N	●
Macrophytes c	Ingestion	N	N	●	●	○	●	○	○	○	○
Benthic Invertebrates c	Ingestion	N	N	●	●	●	●	○	○	○	●
Littoral/Benthic Fish c	Ingestion	N	N	N	N	○	●	●	●	●	N
Pelagic Fish c	Ingestion	N	N	N		N	●	N ●	●	●	N
Amphibians	Ingestion	N	N	N	N	○	●	●	○	○	○
Sediments	Ingestion	N	N	●	●	●	●	○	○	○	N
	Direct Contact	N	●	●	●	●	●	●	○	○	●

Legend:
- ● Exposure Pathway Potentially Complete
- ○ Exposure Pathway Potentially Complete but Minor
- N Pathway Not Applicable/Incomplete

Notes:
- a The different colors used for the arrows linking the boxes were selected to help distinguish the various linkages visually, and have no specific technical connotation.
- b Differential exposure pathways exist for different life stages of some receptors (fish and amphibian eggs, larvae, adults).
- c Upon death, organism contributes to detritus.
- d Surface water may be affected by groundwater discharge from the side banks during pool drawdown.
- e Porewater may be affected or replaced via groundwater advection.
- f Includes mussels
- g Higher phytoplankton and zooplankton abundances may occur during full pool (when unit transitions to lacustrine hydrology.

Figure 4-3a. Conceptual Site Model for Transitional Shallow Water Area (Reach 3)



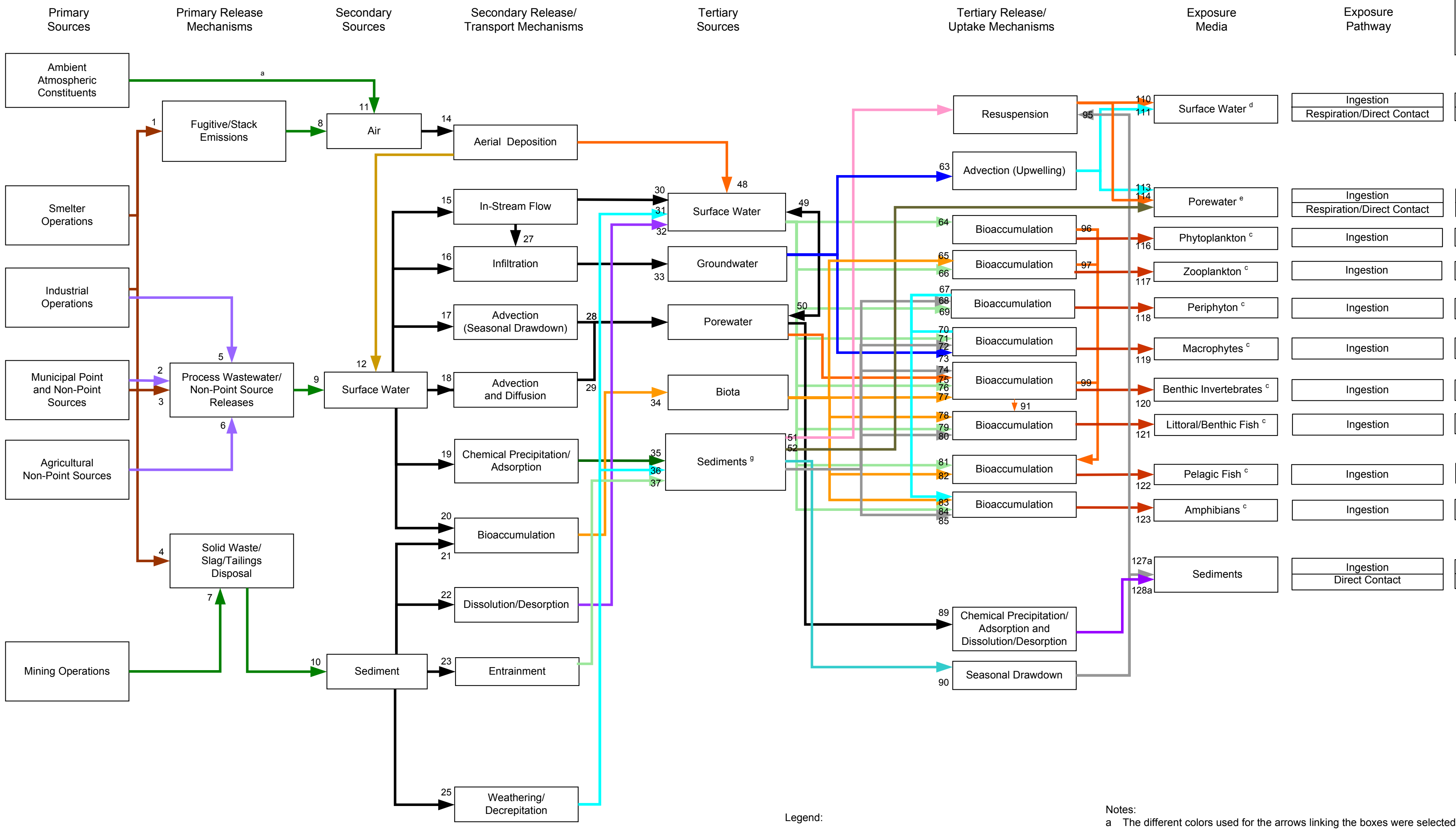
Legend:

- ● Exposure Pathway Potentially Complete
- ○ Exposure Pathway Potentially Complete but Minor
- N Pathway Not Applicable/Incomplete

Notes:

a The different colors used for the arrows linking the boxes were selected to help distinguish the various linkages visually, and have no specific technical connotation.
b Differential exposure pathways exist for different life stages of some receptors (fish and amphibian eggs, larvae, adults).
c Upon death, organism contributes to detritus.
d Surface water may be affected by groundwater discharge from the side banks during pool drawdown.
e Porewater may be affected or replaced via groundwater advection.

Figure 4-3b. Conceptual Site Model for Transitional Deep Water Area (Reach 3)

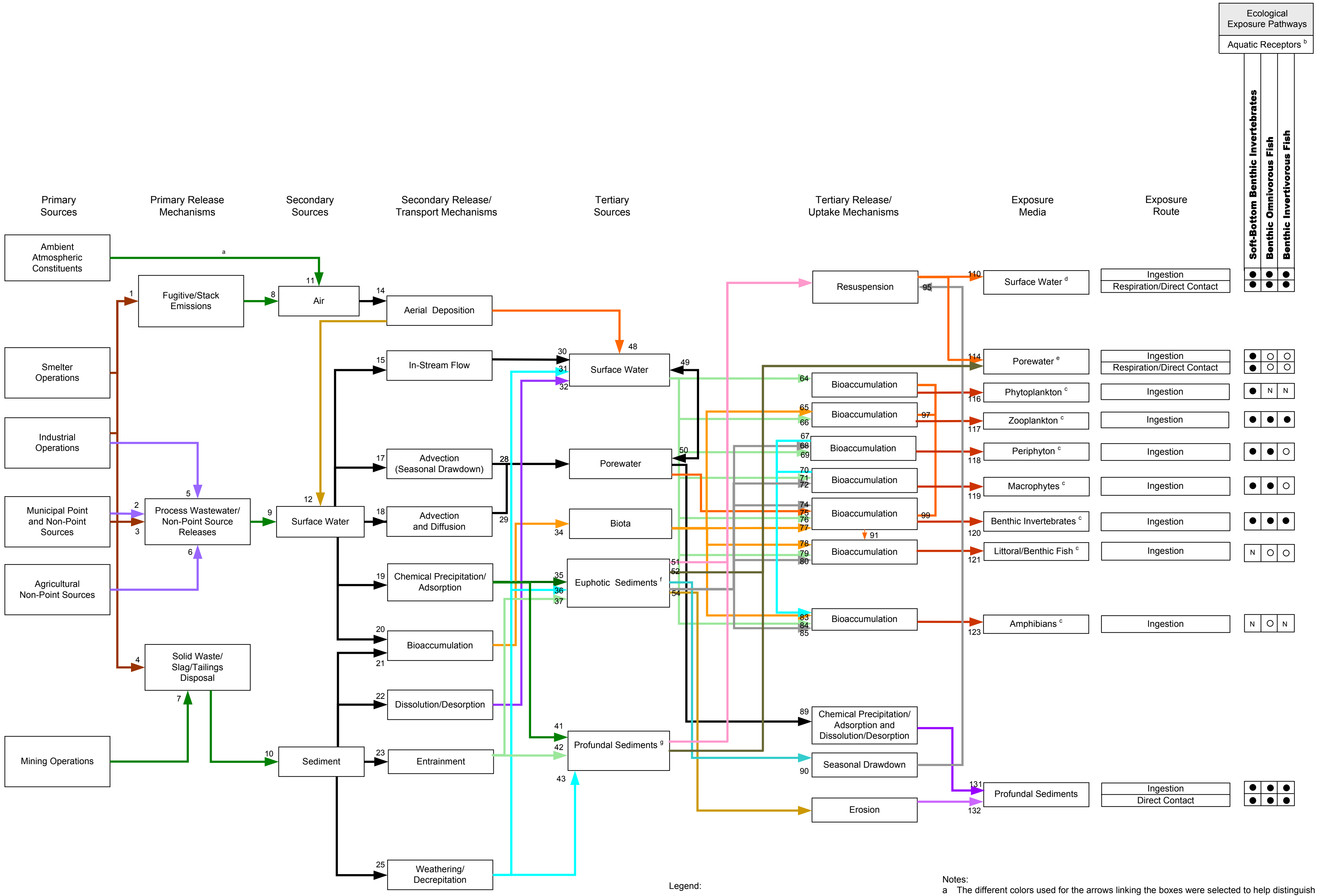


		Ecological Exposure Pathways [g]									
		Aquatic Receptors [b]									
Exposure Media	Exposure Pathway	Plankton (Zooplankton)	Aquatic Macrophytes	Soft-Bottom Benthic Invertebrates [f]	**Benthic Omnivorous Fish**	**Benthic Invertivorous Fish**	Littoral Omnivorous Fish	Nearshore Piscivorous Fish	**Pelagic Omnivorous Fish**	**Pelagic Piscivorous Fish**	Amphibians (Eggs and Larvae)
Surface Water [d]	Ingestion	○	N	●	●	●	●	●	●	●	●
	Respiration/Direct Contact	●	●	●	●	●	●	●	●	●	●
Porewater [e]	Ingestion	N	N	●	○	○	○	N	○	N	○
	Respiration/Direct Contact	N	●	●	○	○	○	○	○	○	●
Phytoplankton [c]	Ingestion	●	N	●	N	N	○	N	○	N	N
Zooplankton [c]	Ingestion	●	N	●	●	●	●	○	●	○	N
Periphyton [c]	Ingestion	●	N	●	●	○	●	N	○	N	●
Macrophytes [c]	Ingestion	N	N	●	●	○	●	○	○	○	○
Benthic Invertebrates [c]	Ingestion	N	N	●	●	●	●	○	○	○	●
Littoral/Benthic Fish [c]	Ingestion	N	N	N	N	○	●	●	●	●	N
Pelagic Fish [c]	Ingestion	N	N	N		N	●	N●	●	●	N
Amphibians [c]	Ingestion	N	N	N	N	○	●	●	○	○	○
Sediments	Ingestion	○	N	●	●	●	●	N	N	N	○
	Direct Contact	N	●	●	●	●	●		N	N	N○

Legend:
- ● Exposure Pathway Potentially Complete
- ○ Exposure Pathway Potentially Complete but Minor
- N Pathway Not Applicable/Incomplete

Notes:
- a The different colors used for the arrows linking the boxes were selected to help distinguish the various linkages visually, and have no specific technical connotation.
- b Differential exposure pathways exist for different life stages of some receptors (fish and amphibian eggs, larvae, adults).
- c Upon death, organism contributes to detritus.
- d Surface water may be affected by groundwater discharge from the side banks during pool drawdown.
- e Porewater may be affected or replaced via groundwater advection.
- f Includes mussels
- g Euphotic sediments in the lacustrine portion of the UCR are defined as <approximately 80 ft below water surface.

Figure 4-4a. Conceptual Site Model for Lacustrine Shallow Water Areas of the Site (Reaches 4, 5, and 6)

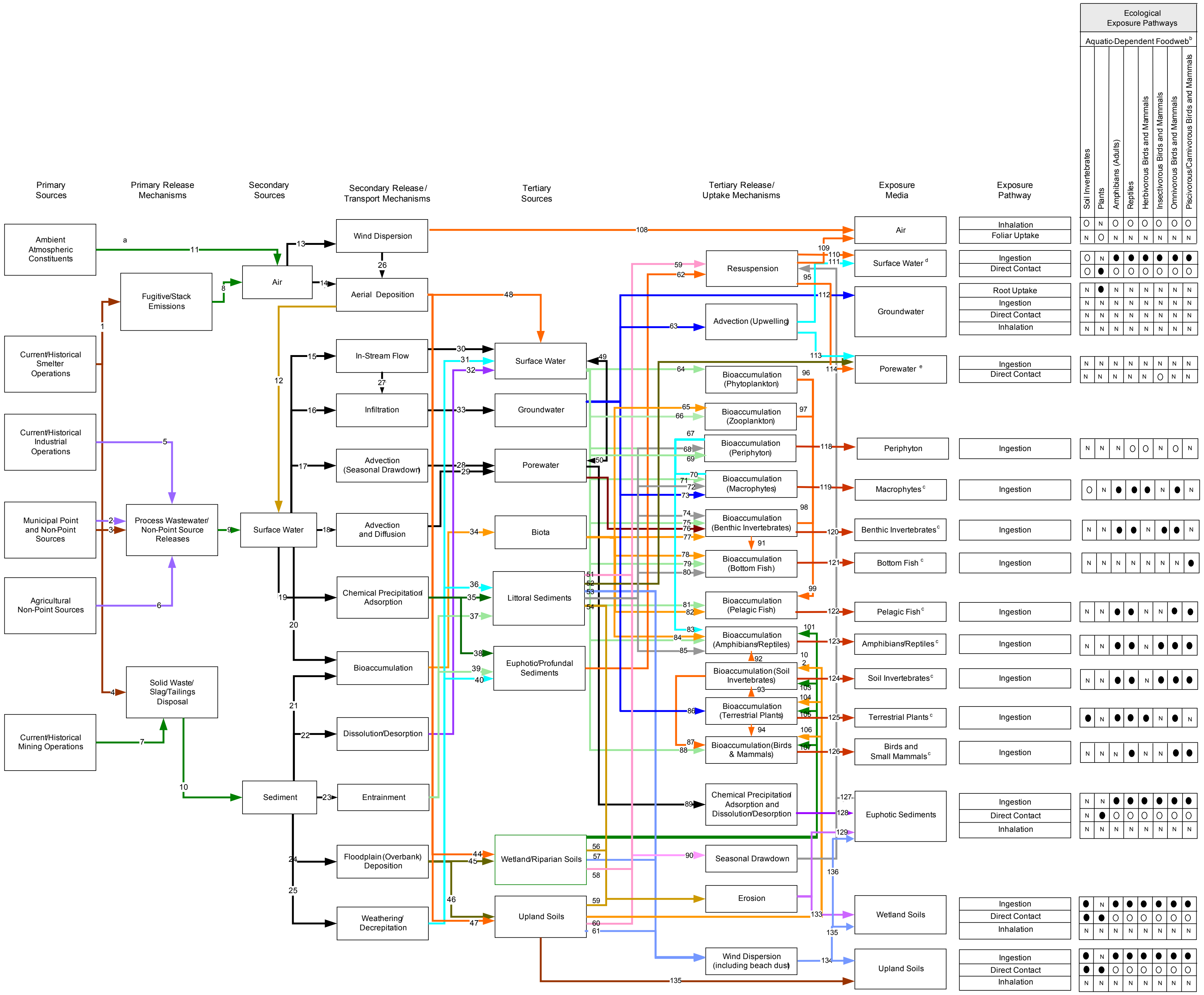


Exposure Media	Exposure Route	Soft-Bottom Benthic Invertebrates	Benthic Omnivorous Fish	Benthic Invertivorous Fish
Surface Water d	Ingestion	●	●	●
	Respiration/Direct Contact	●	●	●
Porewater e	Ingestion	●	○	○
	Respiration/Direct Contact	●	○	○
Phytoplankton c	Ingestion	●	N	N
Zooplankton c	Ingestion	●	●	●
Periphyton c	Ingestion	●	●	○
Macrophytes c	Ingestion	●	●	○
Benthic Invertebrates c	Ingestion	●	●	●
Littoral/Benthic Fish c	Ingestion	N	○	○
Amphibians c	Ingestion	N	○	N
Profundal Sediments	Ingestion	●	●	●
	Direct Contact	●	●	●

Legend:
- ● Exposure Pathway Potentially Complete
- ○ Exposure Pathway Potentially Complete but Minor
- N Pathway Not Applicable/Incomplete

Notes:
a The different colors used for the arrows linking the boxes were selected to help distinguish the various linkages visually, and have no specific technical connotation.
b Differential exposure pathways exist for different life stages of some receptors (fish and amphibian eggs, larvae, adults).
c Upon death, organism contributes to detritus.
d Surface water may be affected by groundwater discharge from the side banks during pool drawdown.
e Porewater may be affected or replaced via groundwater advection.
f Euphotic sediments in the lacustrine portion of the UCR are defined as <approximately 80 ft below water surface.
g Profundal sediments in the lacustrine portion of the UCR are defined as >80 ft water depth.

Figure 4-4b. Conceptual Site Model for Lacustrine Deep Water Areas of the Site (Reaches 4, 5, and 6)

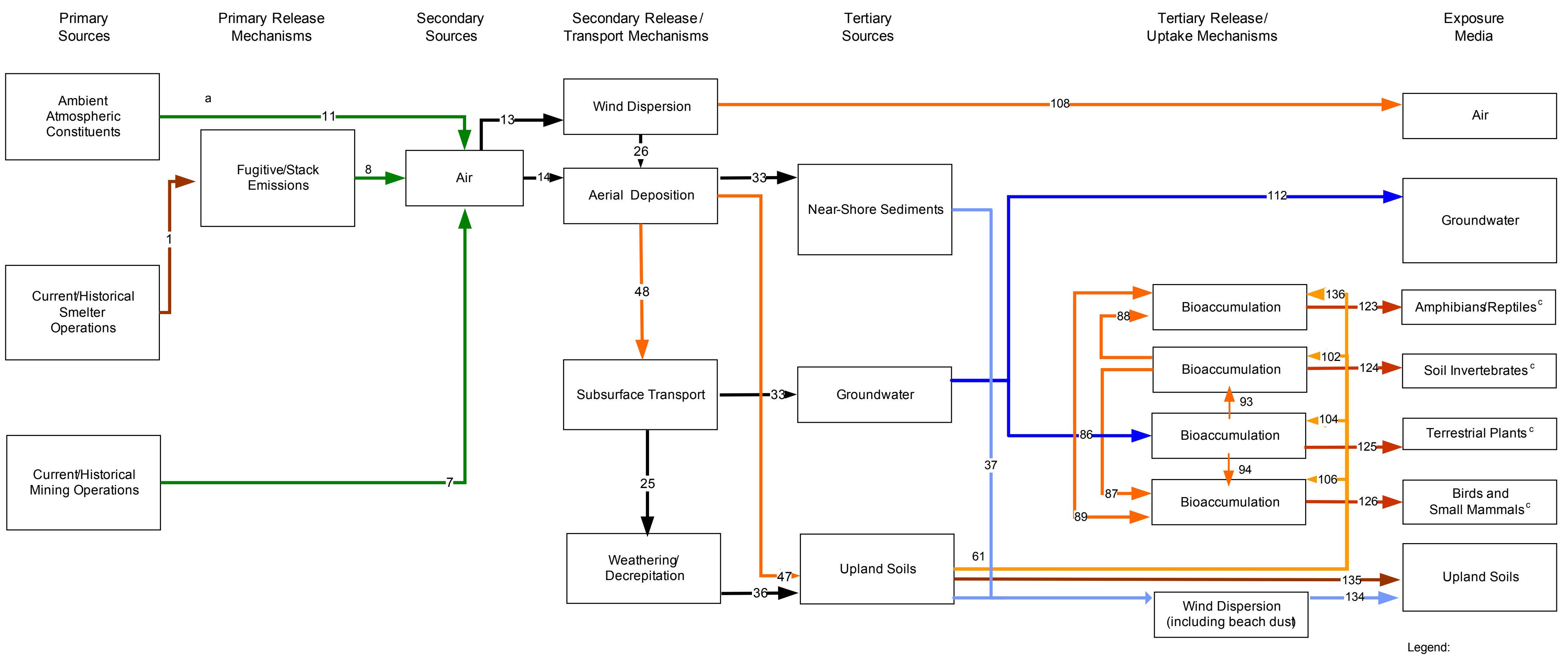


Exposure Media	Exposure Pathway	Soil Invertebrates	Plants	Amphibians (Adults)	Reptiles	Herbivorous Birds and Mammals	Insectivorous Birds and Mammals	Omnivorous Birds and Mammals	Piscivorous/Carnivorous Birds and Mammals
Air	Inhalation	○	N	○	○	○	○	○	○
	Foliar Uptake	N	○	N	N	N	N	N	N
Surface Water d	Ingestion	○	N	●	●	●	●	●	●
	Direct Contact	○	●	○	○	○	○	○	○
Groundwater	Root Uptake	N	●	N	N	N	N	N	N
	Ingestion	N	N	N	N	N	N	N	N
	Direct Contact	N	N	N	N	N	N	N	N
	Inhalation	N	N	N	N	N	N	N	N
Porewater e	Ingestion	N	N	N	N	N	N	N	N
	Direct Contact	N	N	N	N	N	○	N	N
Periphyton	Ingestion	N	N	N	○	○	N	○	N
Macrophytes c	Ingestion	○	N	●	●	●	N	●	N
Benthic Invertebrates c	Ingestion	N	N	●	●	N	●	●	N
Bottom Fish c	Ingestion	N	N	N	N	N	N	N	●
Pelagic Fish c	Ingestion	N	N	●	●	N	N	●	●
Amphibians/Reptiles c	Ingestion	N	N	●	●	N	●	●	●
Soil Invertebrates c	Ingestion	N	N	●	●	N	●	●	●
Terrestrial Plants c	Ingestion	●	N	●	●	●	N	●	N
Birds and Small Mammals c	Ingestion	N	N	N	●	N	N	●	●
Euphotic Sediments	Ingestion	N	N	●	●	●	●	●	●
	Direct Contact	N	●	○	○	○	○	○	○
	Inhalation	N	N	N	N	N	N	N	N
Wetland Soils	Ingestion	●	N	●	●	●	●	●	●
	Direct Contact	●	●	○	○	○	○	○	○
	Inhalation	N	N	N	N	N	N	N	N
Upland Soils	Ingestion	●	N	●	●	●	●	●	●
	Direct Contact	●	●	○	○	○	○	○	○
	Inhalation	N	N	N	N	N	N	N	N

Legend:
● Exposure Pathway Potentially Complete
○ Exposure Pathway Potentially Complete but minor
N Pathway Not Applicable/Incomplete

Notes:
a The different colors used for the arrows linking the boxes were selected to help distinguish the various linkages visually , and have no specific technical connotation .
b Differential exposure pathways exist for different life stages of some receptors (amphibian eggs, larvae, adults).
c Upon death, organism contributes to detritus.
d Surface water may be affected by groundwater discharge from the side banks during pool drawdown
e Porewater may be affected or replaced via groundwater advection

Figure 4-5. Conceptual Site Model for Aquatic-Dependent Plants and Wildlife



Exposure Media	Exposure Pathway	Ecological Exposure Pathways							
		Upland Foodweb							
		Soil Invertebrates	Plants	Amphibians (Adults)[b]	Reptiles	Herbivorous Birds and Mammals	Insectivorous Birds and Mammals	Omnivorous Birds and Mammals	Carnivorous Birds and Mammals
Air	Inhalation	○	N	○	○	○	○	○	○
	Foliar Uptake	N	○	N	N	N	N	N	N
Groundwater	Root Uptake	N	●	N	N	N	N	N	N
	Ingestion	N	N	N	N	N	N	N	N
	Direct Contact	N	N	N	N	N	N	N	N
	Inhalation	N	N	N	N	N	N	N	N
Amphibians/Reptiles[c]	Ingestion	N	N	●	●	N	●	●	●
Soil Invertebrates[c]	Ingestion	N	N	●	●	N	●	●	●
Terrestrial Plants[c]	Ingestion	●	N	N	N	●	N	●	N
Birds and Small Mammals[c]	Ingestion	N	N	N	●	N	N	●	●
Upland Soils	Ingestion	●	N	●	●	●	●	●	●
	Direct Contact	●	●	○	○	○	○	○	○
	Inhalation	N	N	N	N	N	N	N	N

Legend:
● Exposure Pathway Potentially Complete
○ Exposure Pathway Potentially Complete but minor
N Pathway Not Applicable/Incomplete

Notes:
[a] The different colors used for the arrows linking the boxes were selected to help distinguish the various linkages visually , and have no specific technical connotation .
[b] Differential exposure pathways exist for different life stages of some receptors (amphibian eggs, larvae, adults).
[c] Upon death, organism contributes to detritus.

Figure 4-6. Conceptual Site Model for the Terrestrial Upland Habitat

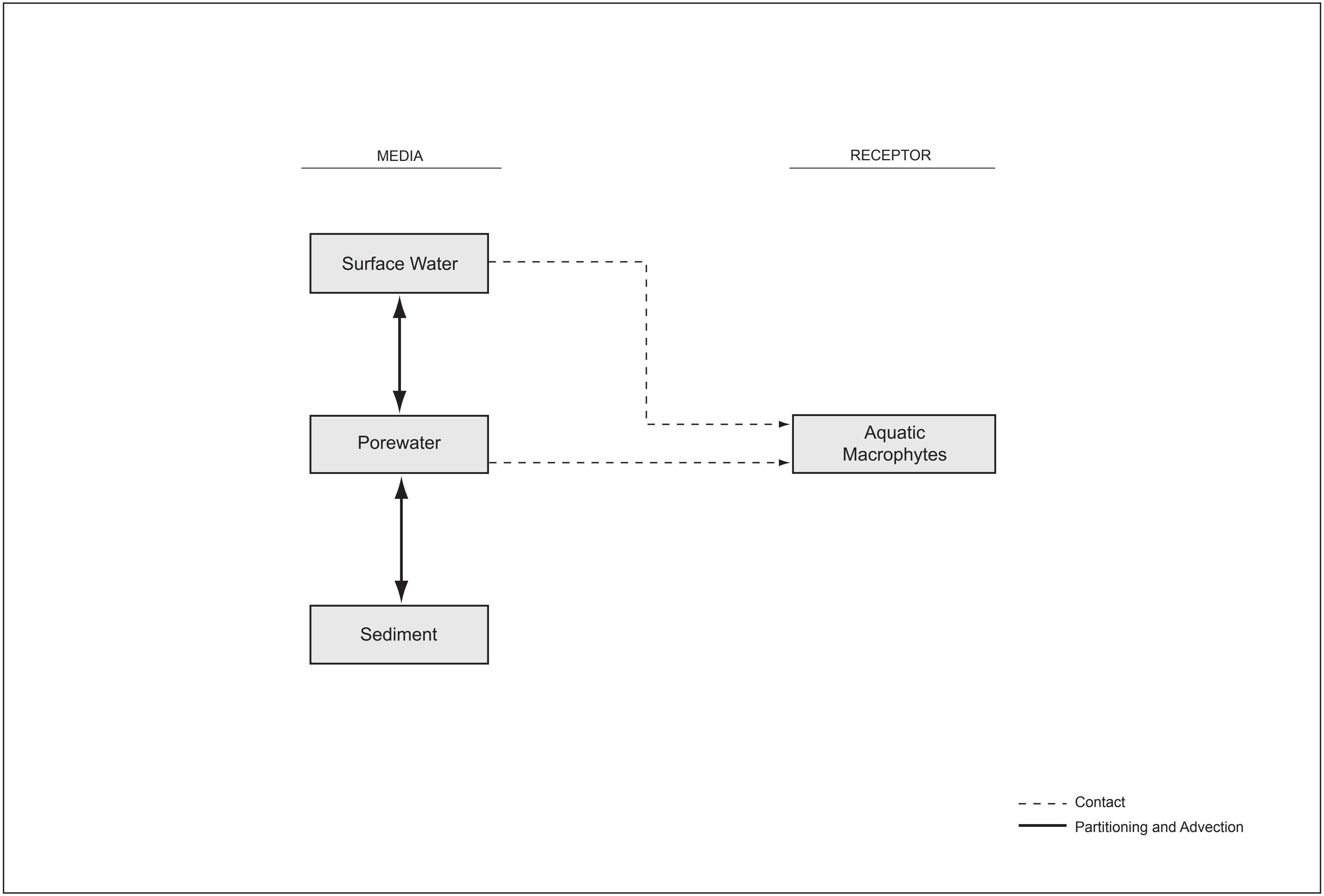


Figure 4-7. Aquatic Macrophyte CSM

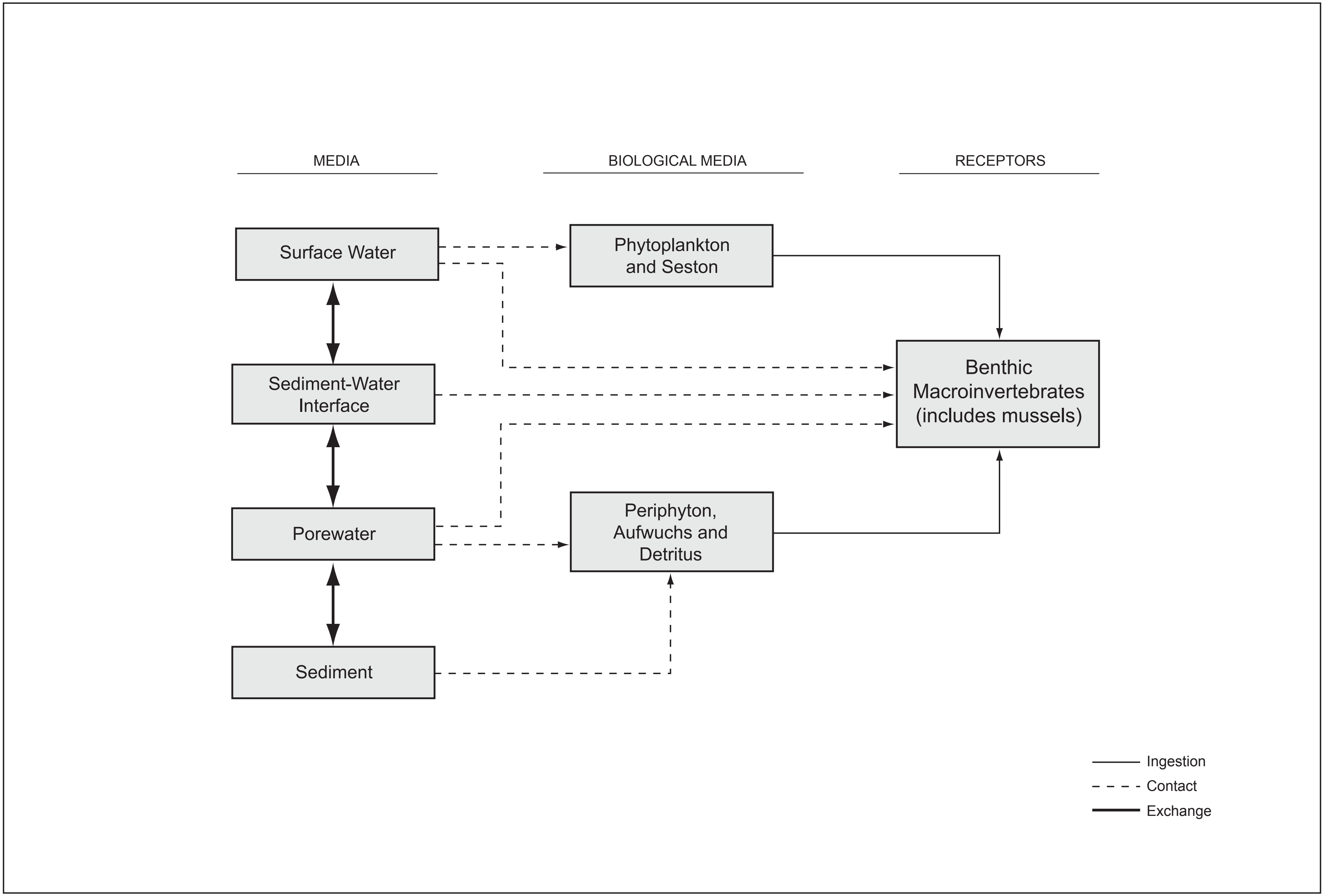


Figure 4-8. Benthic Macroinvertebrate CSM

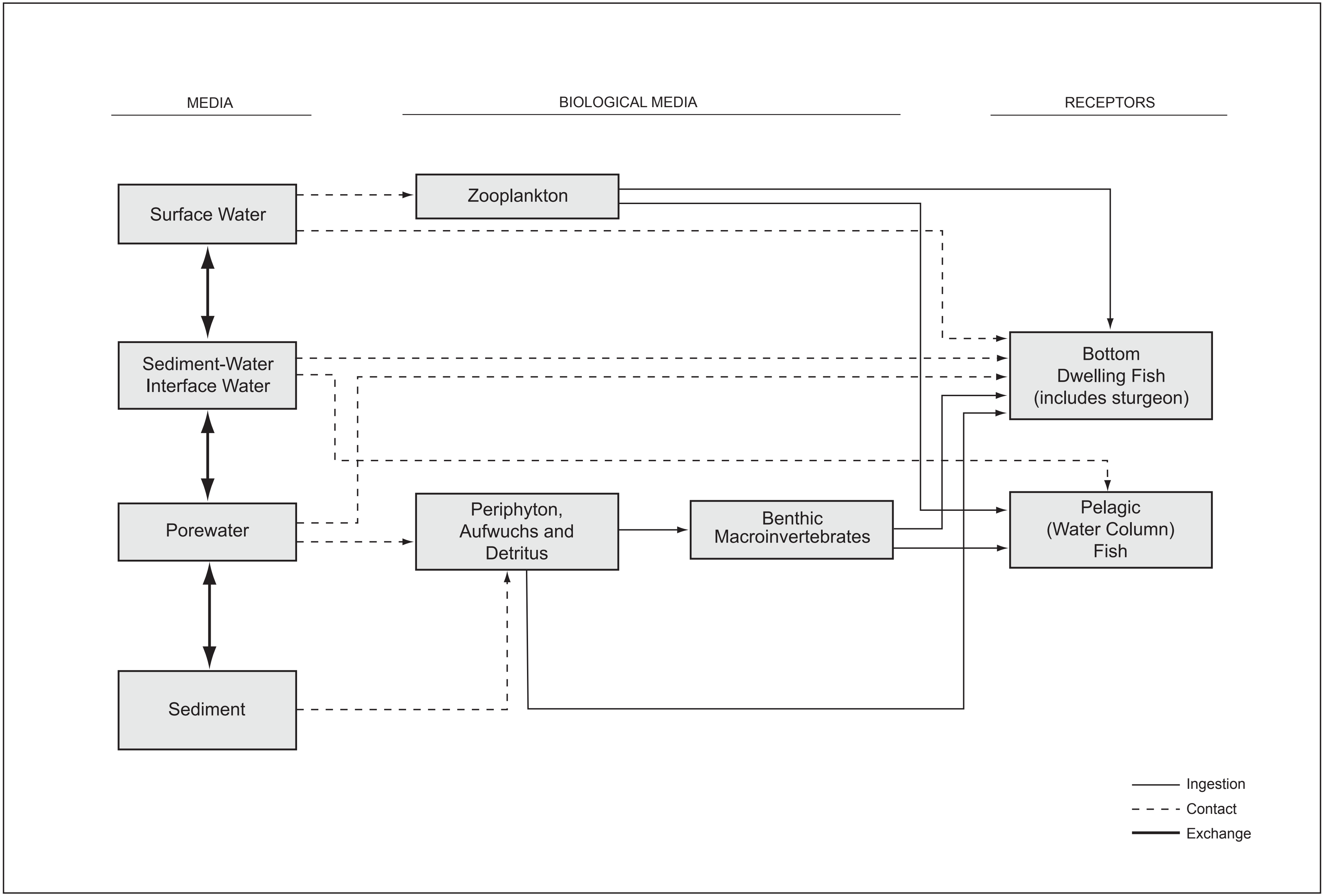


Figure 4-9. Fish CSM

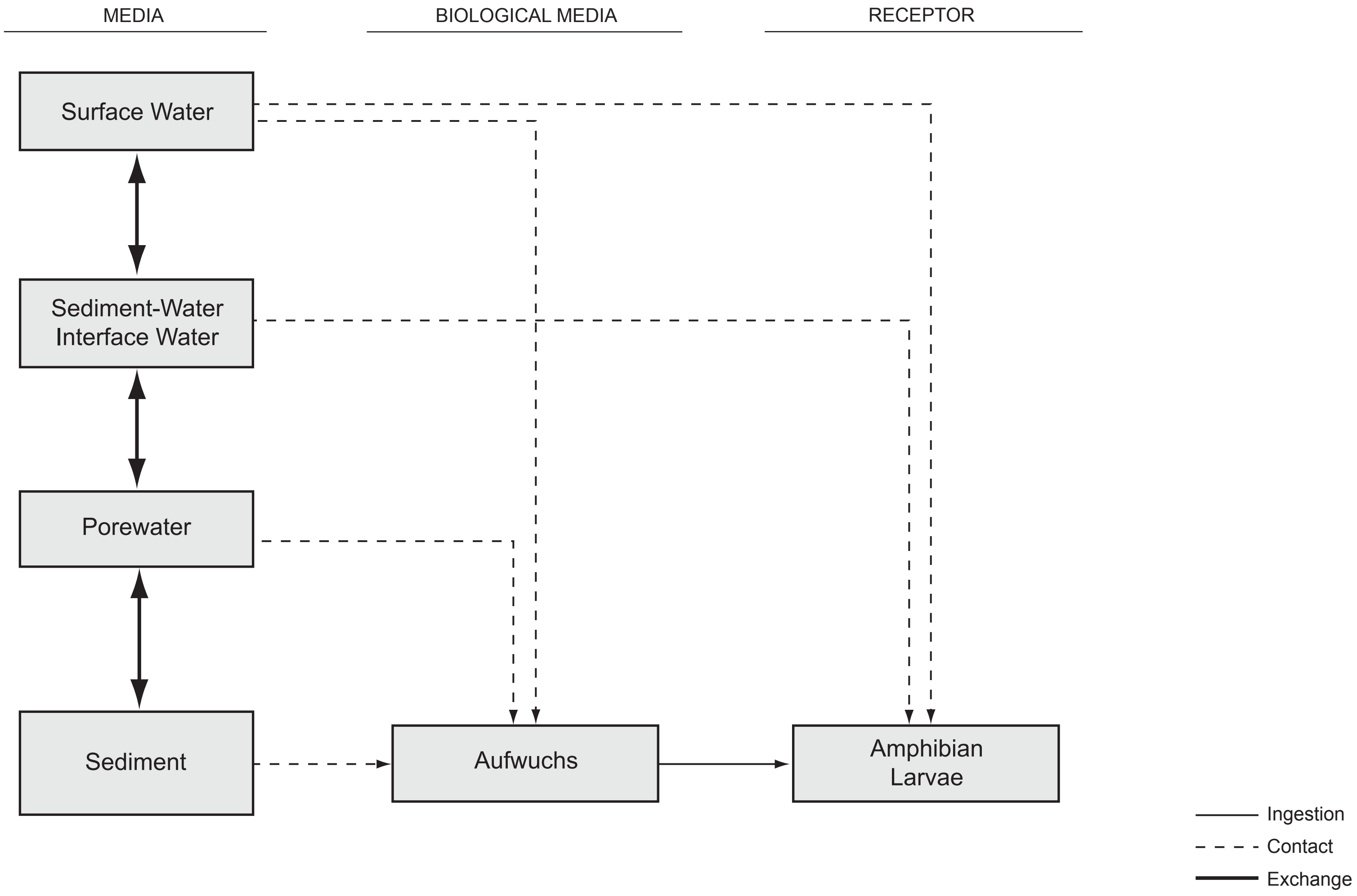
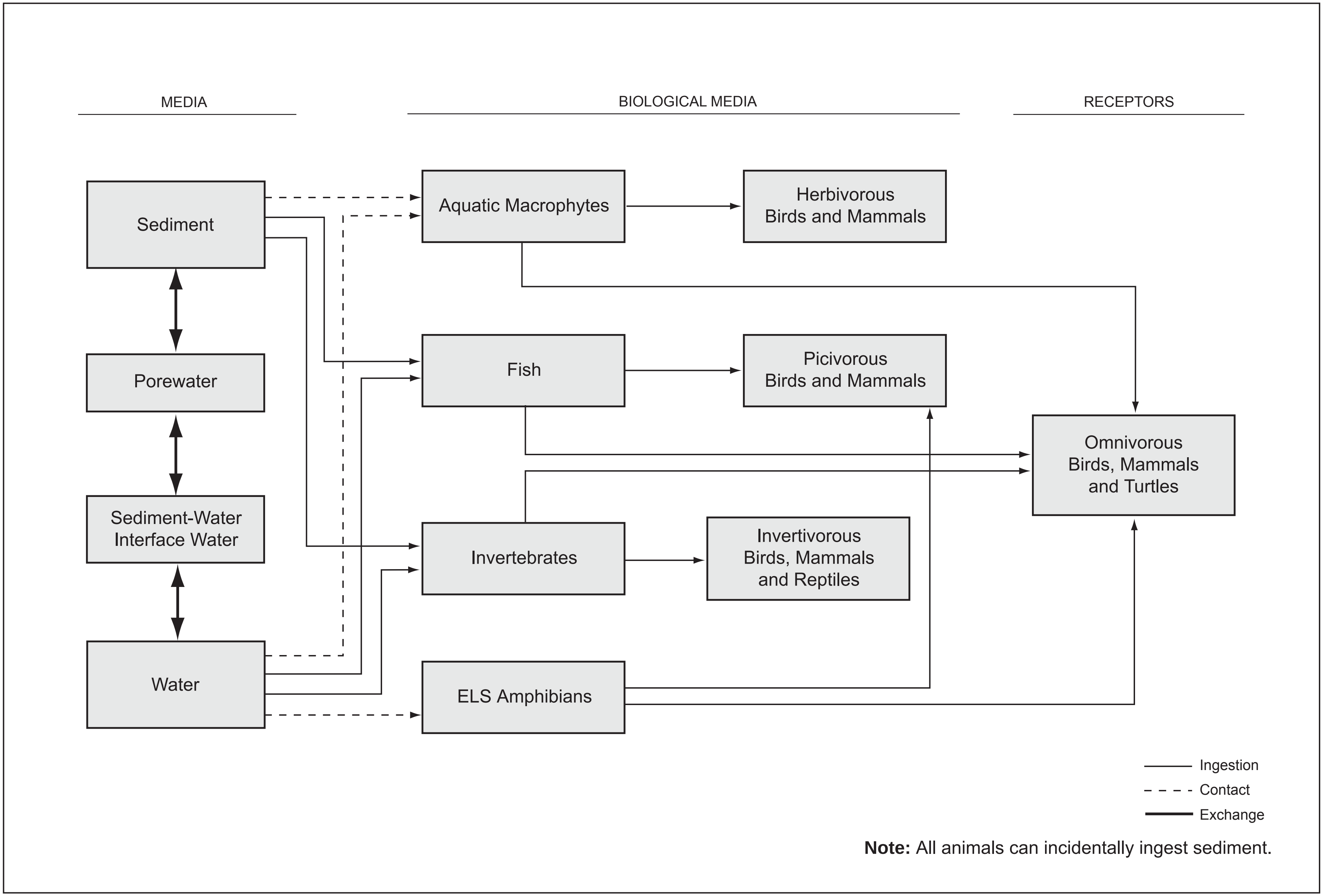


Figure 4-10. Early Life Stage Amphibian CSM



Note: All animals can incidentally ingest sediment.

Figure 4-11. Aquatic Dependent Wildlife CSM

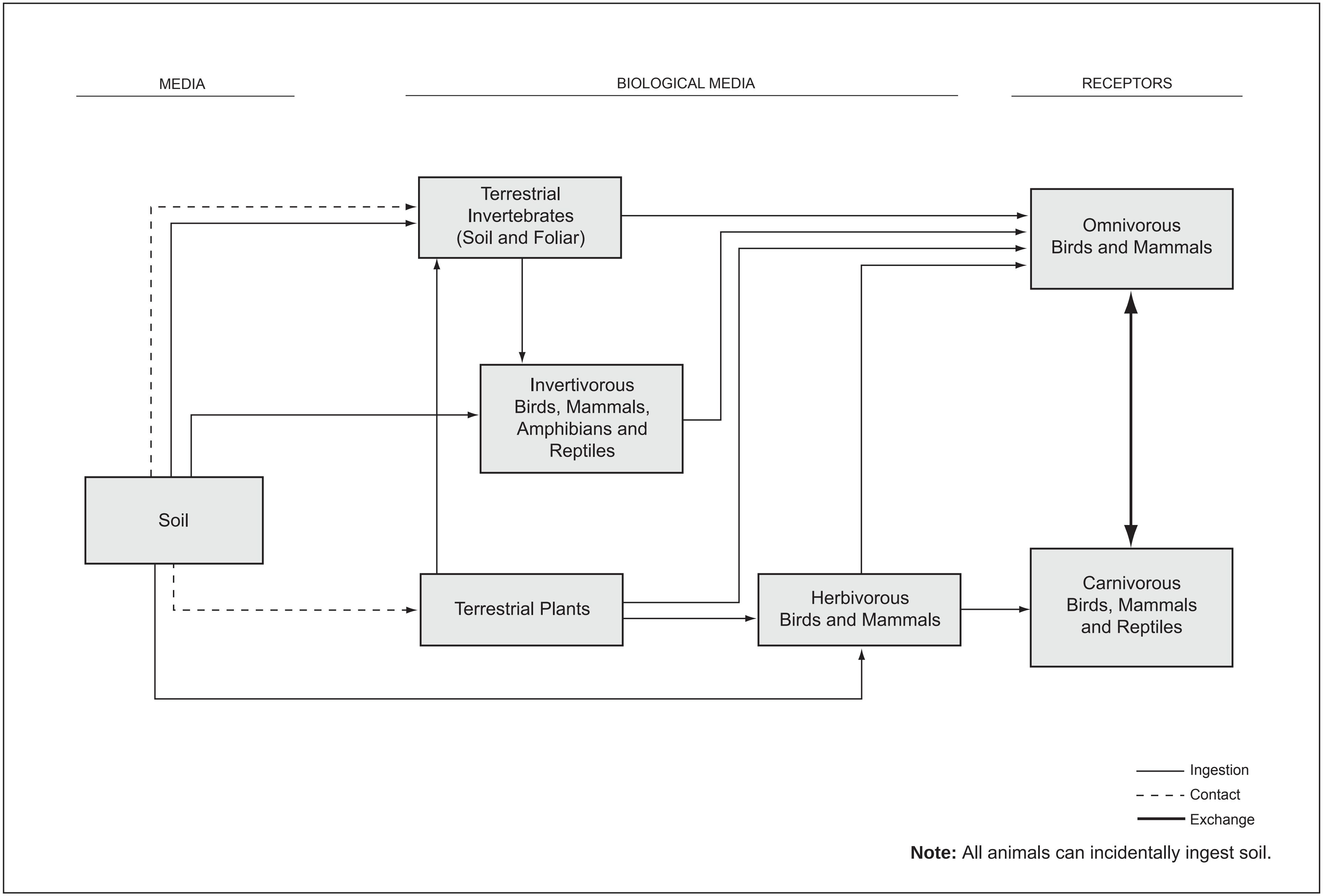


Figure 4-12. Terrestrial Plant, Invertebrate, Bird, Mammal, Amphibian and Reptile CSM

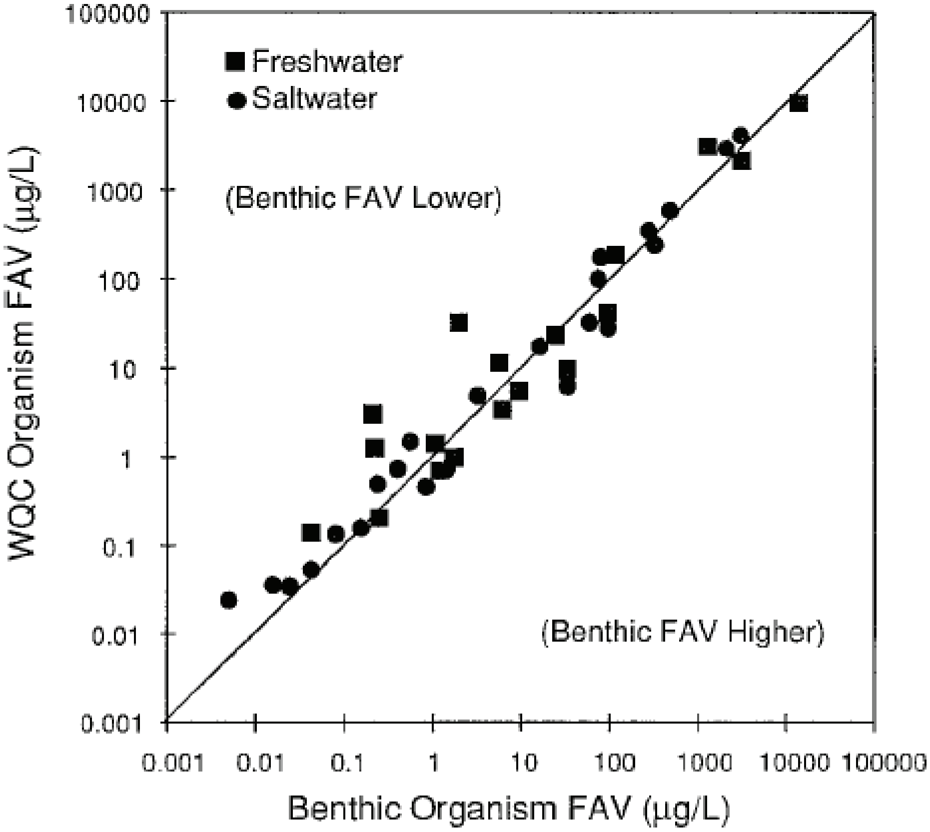


Figure 4-13. National Water Quality Criteria Final Acute Values (FAVs) for all Species Versus Values for Benthic Species only

Source: Ankley et al. (1996)

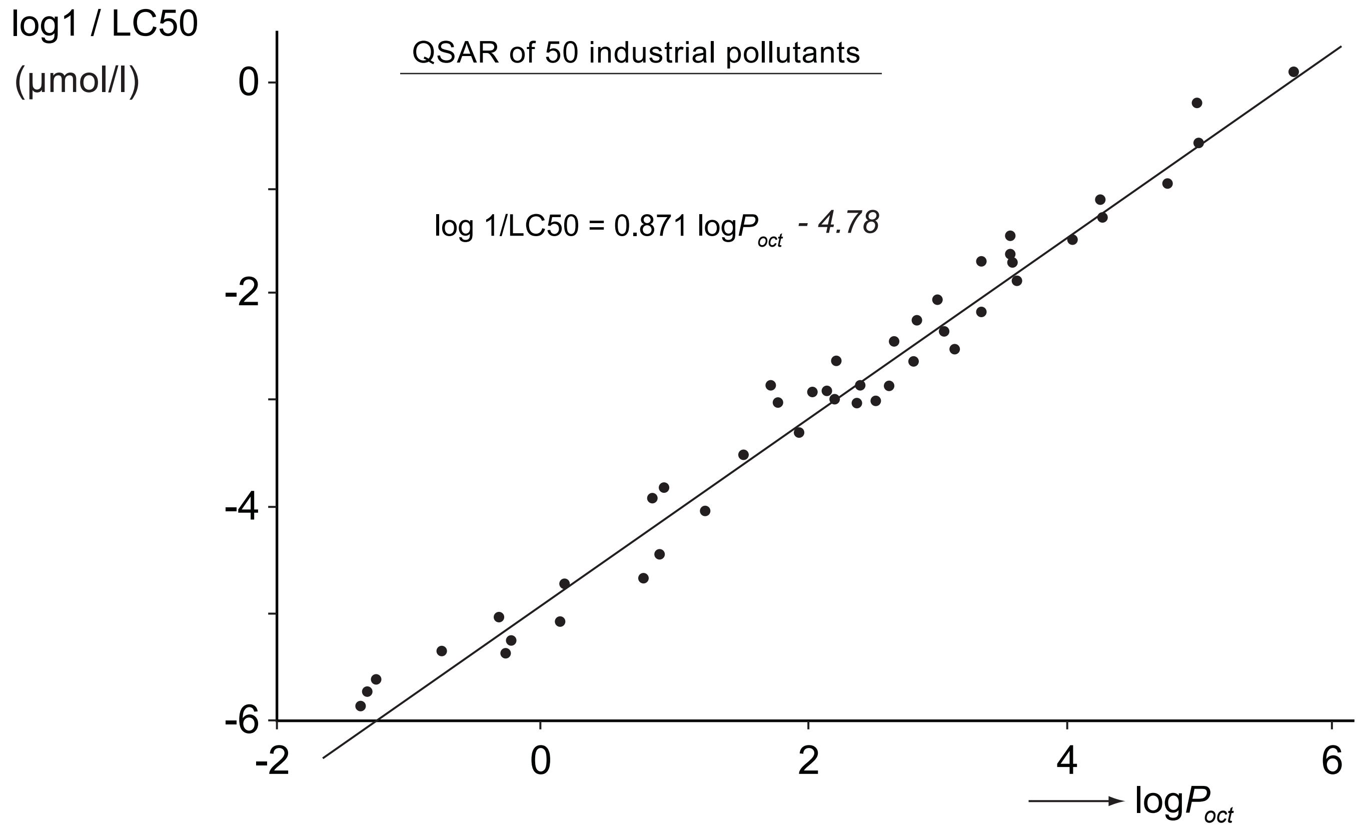
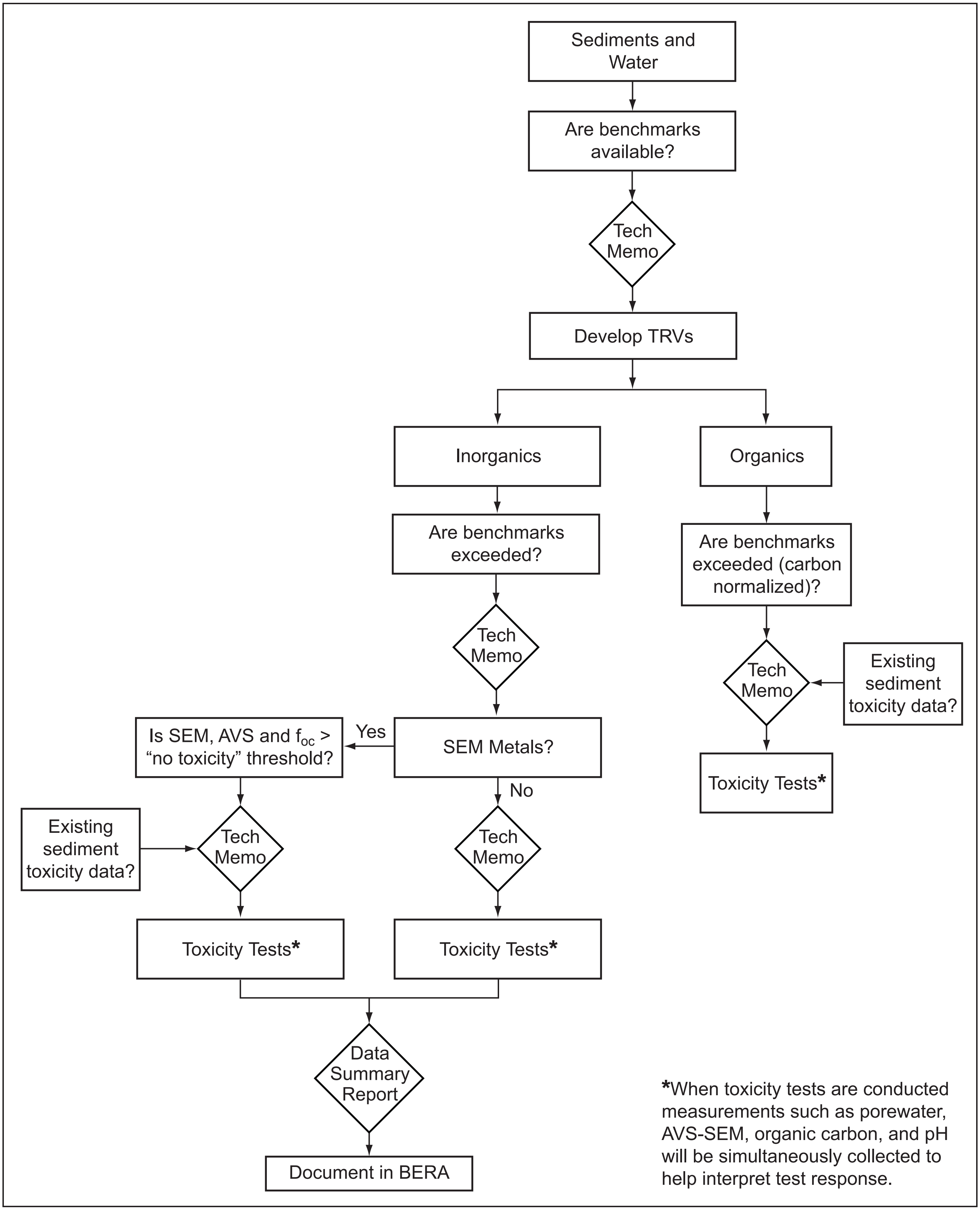


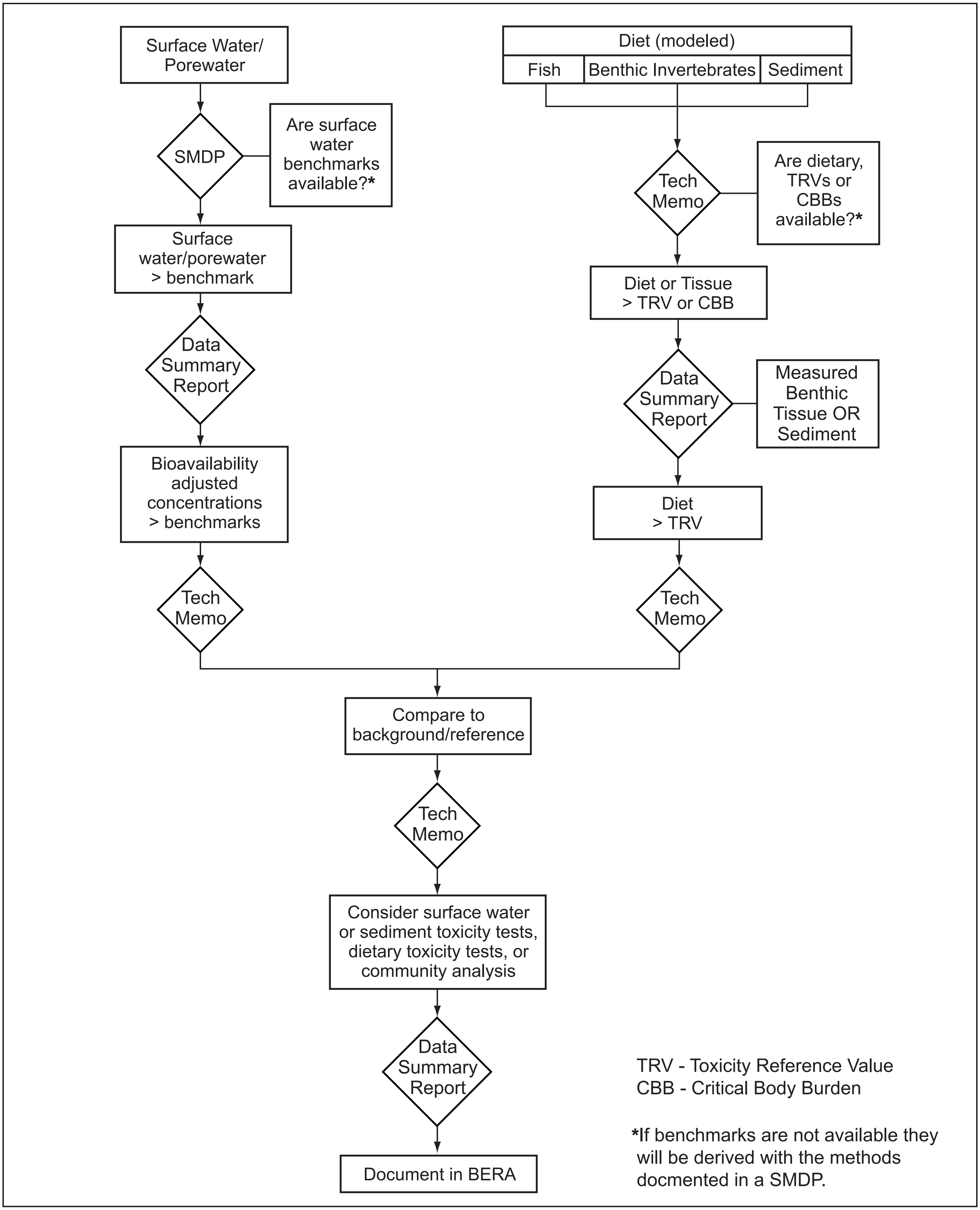
Figure 4-14. Relationship between the toxicity of 50 nonpolar organic chemicals and acute toxicity to an indicator fish (*Lebistes reticulates*). Organics included chlorotoluenes, aliphatic chlorohydrocarbons, alcohols, glycol derivatives and similar compounds
Source: Konemann (1981)



*****When toxicity tests are conducted measurements such as porewater, AVS-SEM, organic carbon, and pH will be simultaneously collected to help interpret test response.

Figure 5-1. Lines of Evidence for Assessing Benthic Macroinvertebrates

Note: "Tech Memo" as illustrated herein may represent a single technical memorandum (see Table 4-1 of this document) or a point for consultation with EPA.

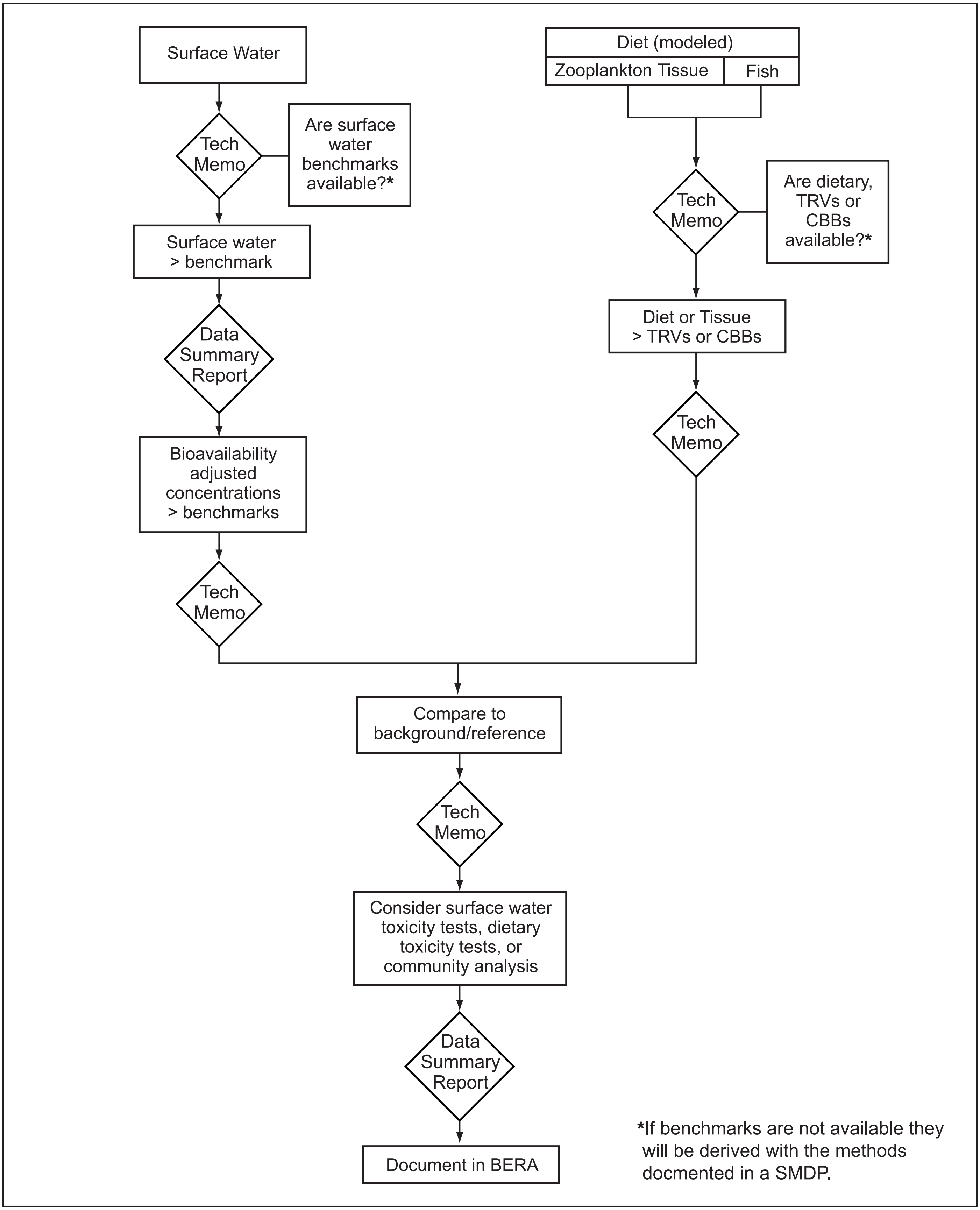


TRV - Toxicity Reference Value
CBB - Critical Body Burden

*****If benchmarks are not available they will be derived with the methods docmented in a SMDP.

Figure 5-2. Lines of Evidence for Assessing BERA Process for Bottom-dwelling Fish

Note: "Tech Memo" as illustrated herein may represent a single technical memorandum (see Table 4-1 of this document) or a point for consultation with EPA.



*****If benchmarks are not available they will be derived with the methods docmented in a SMDP.

Figure 5-3. Lines of Evidence for Assessing Pelagic Fish

Note: "Tech Memo" as illustrated herein may represent a single technical memorandum (see Table 4-1 of this document) or a point for consultation with EPA.

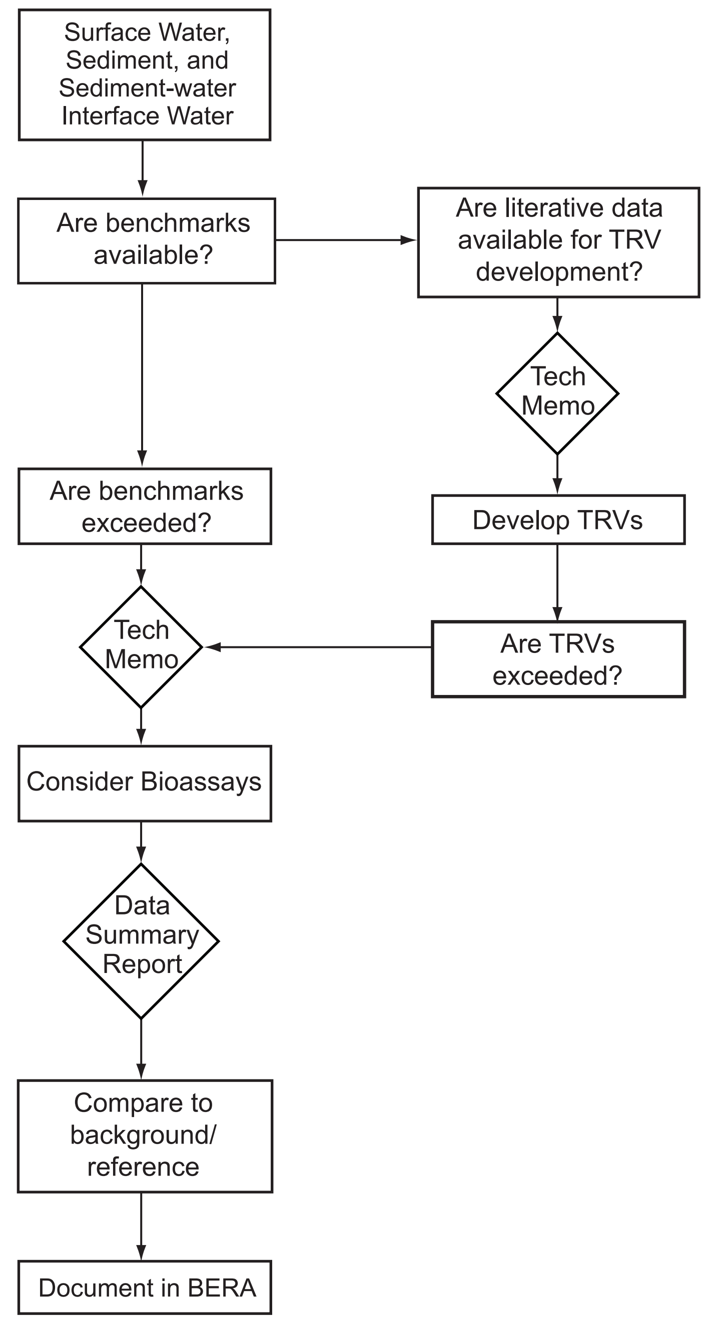


Figure 5-4. Lines of Evidence for Assessing Early Lifestage Amphibians
Note: "Tech Memo" as illustrated herein may represent a single technical memorandum (see Table 4-1 of this document) or a point for consultation with EPA.

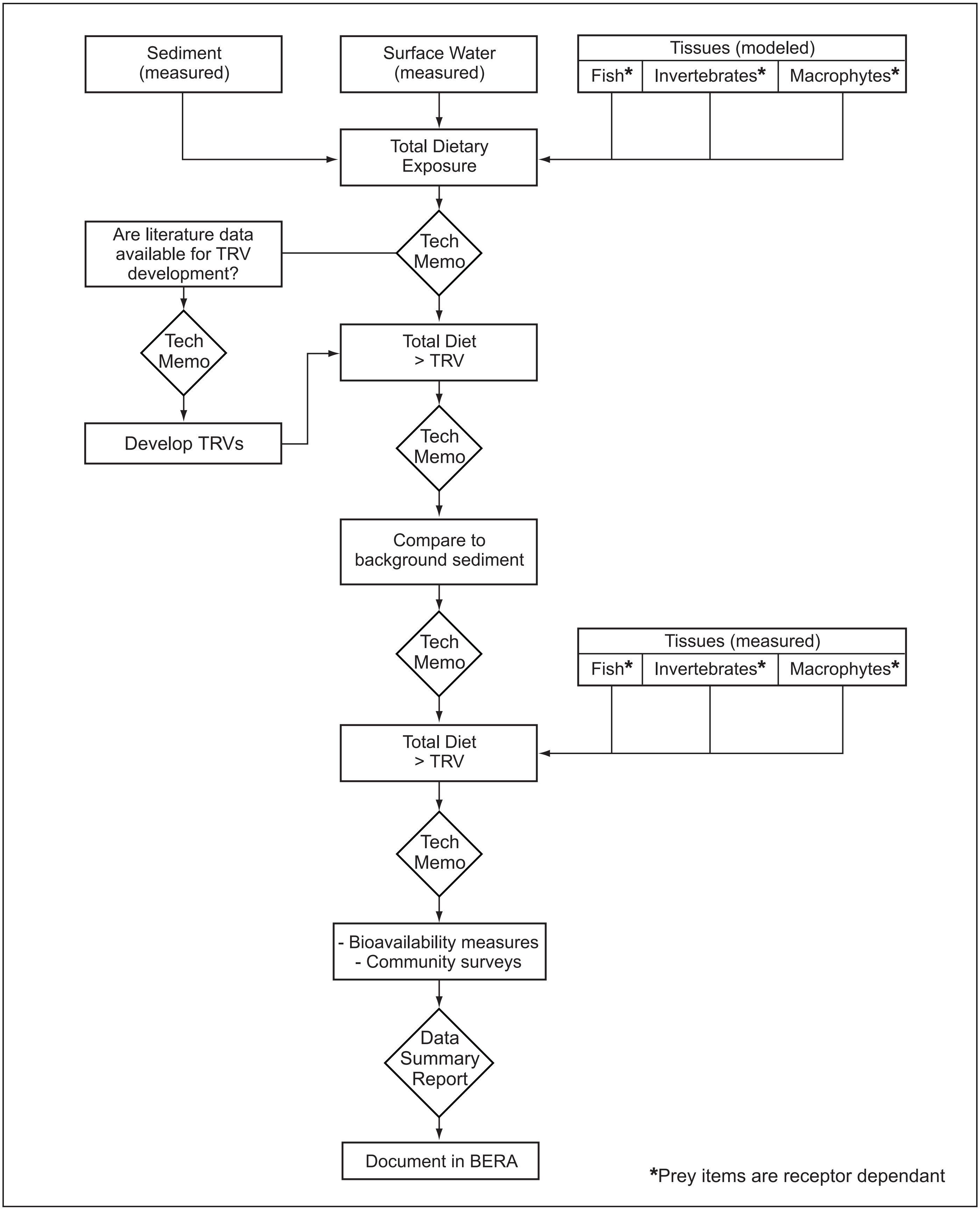


Figure 5-5. Lines of Evidence for Assessing Aquatic Dependent Wildlife
Note: "Tech Memo" as illustrated herein may represent a single technical memorandum (see Table 4-1 of this document) or a point for consultation with EPA.

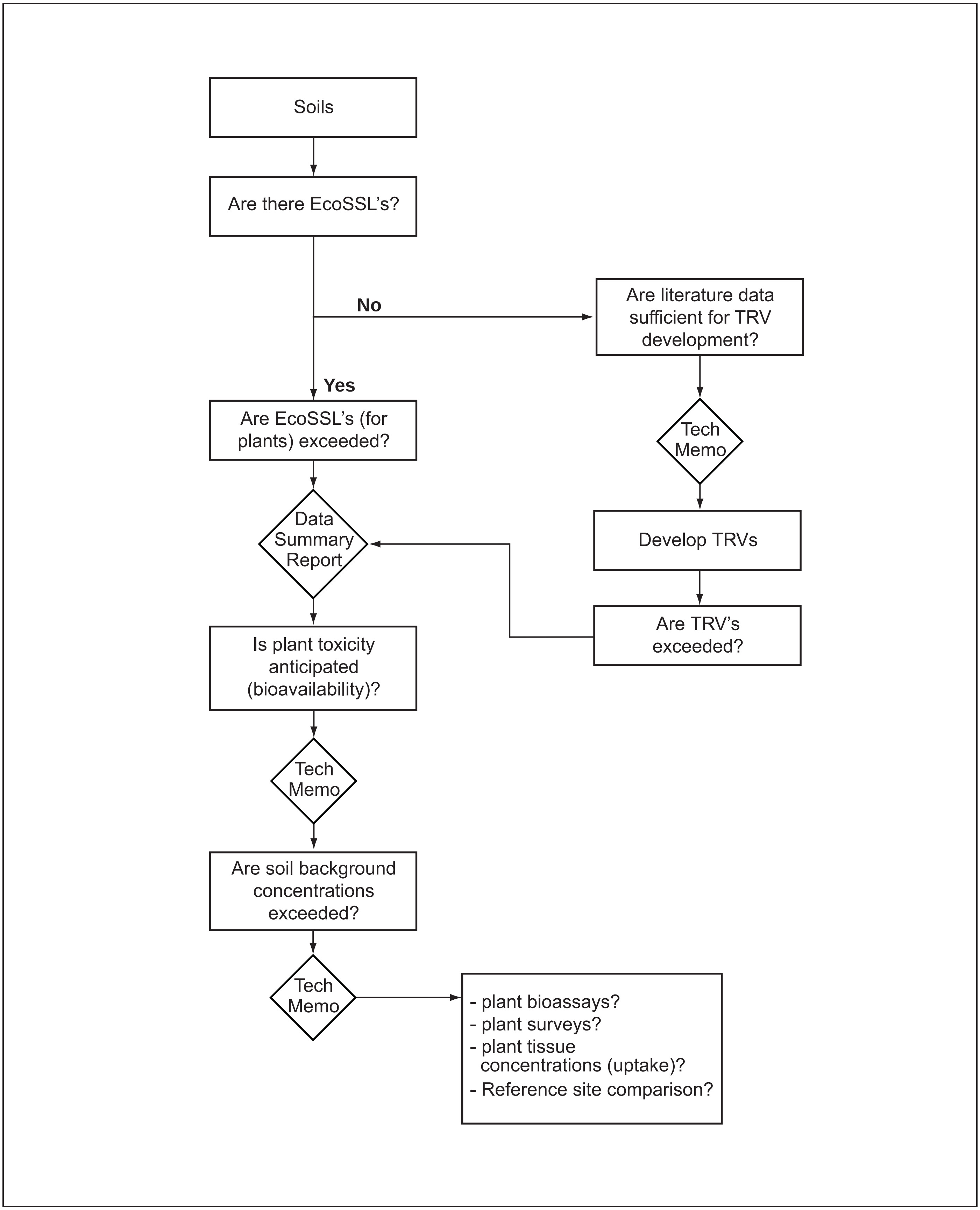


Figure 5-6. Lines of Evidence for Assessing Terrestrial Plants

Note: "Tech Memo" as illustrated herein may represent a single technical memorandum (see Table 4-1 of this document) or a point for consultation with EPA.

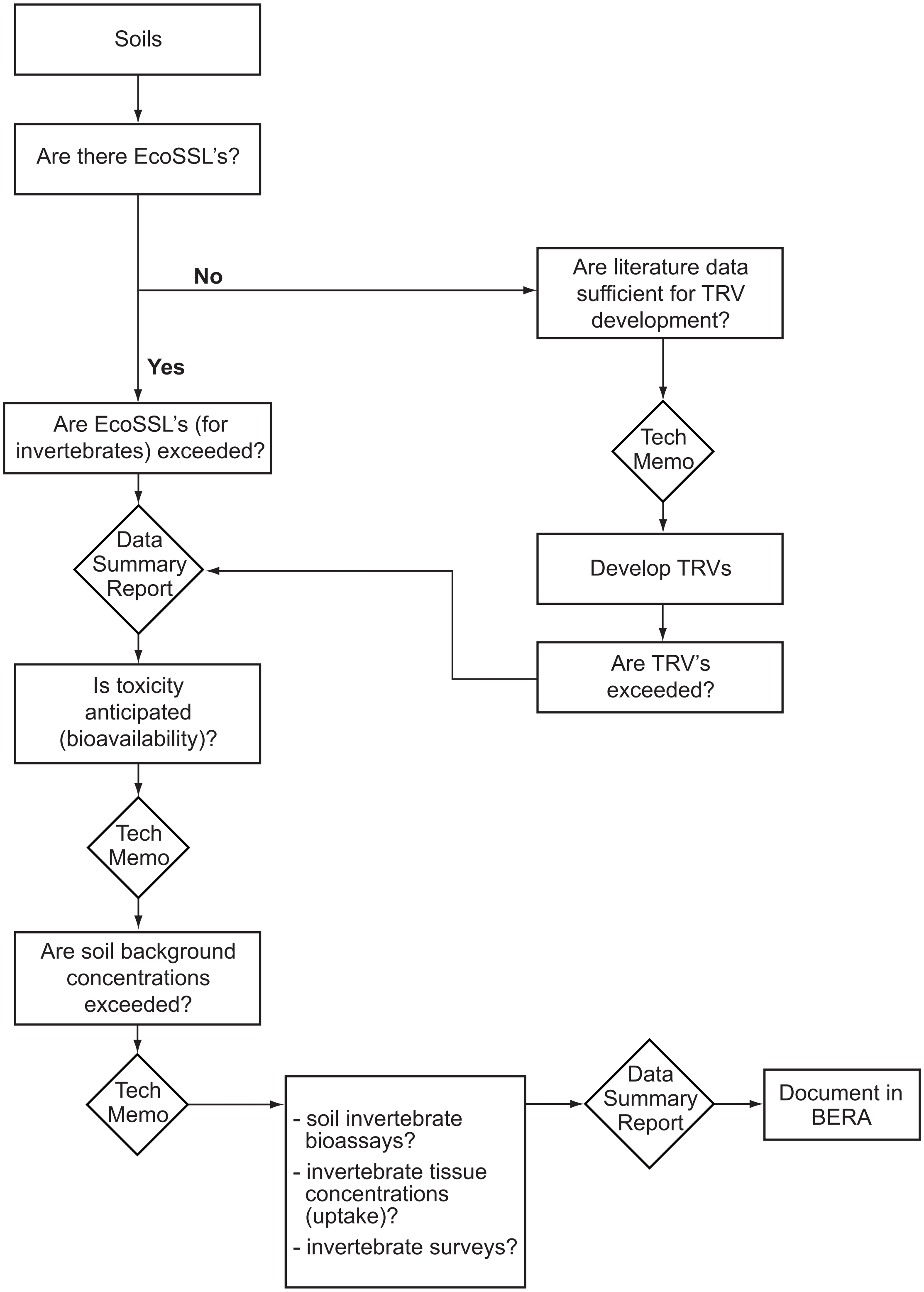


Figure 5-7. Lines of Evidence for Assessing Foliar/Soil Invertebrates

Note: "Tech Memo" as illustrated herein may represent a single technical memorandum (see Table 4-1 of this document) or a point for consultation with EPA.

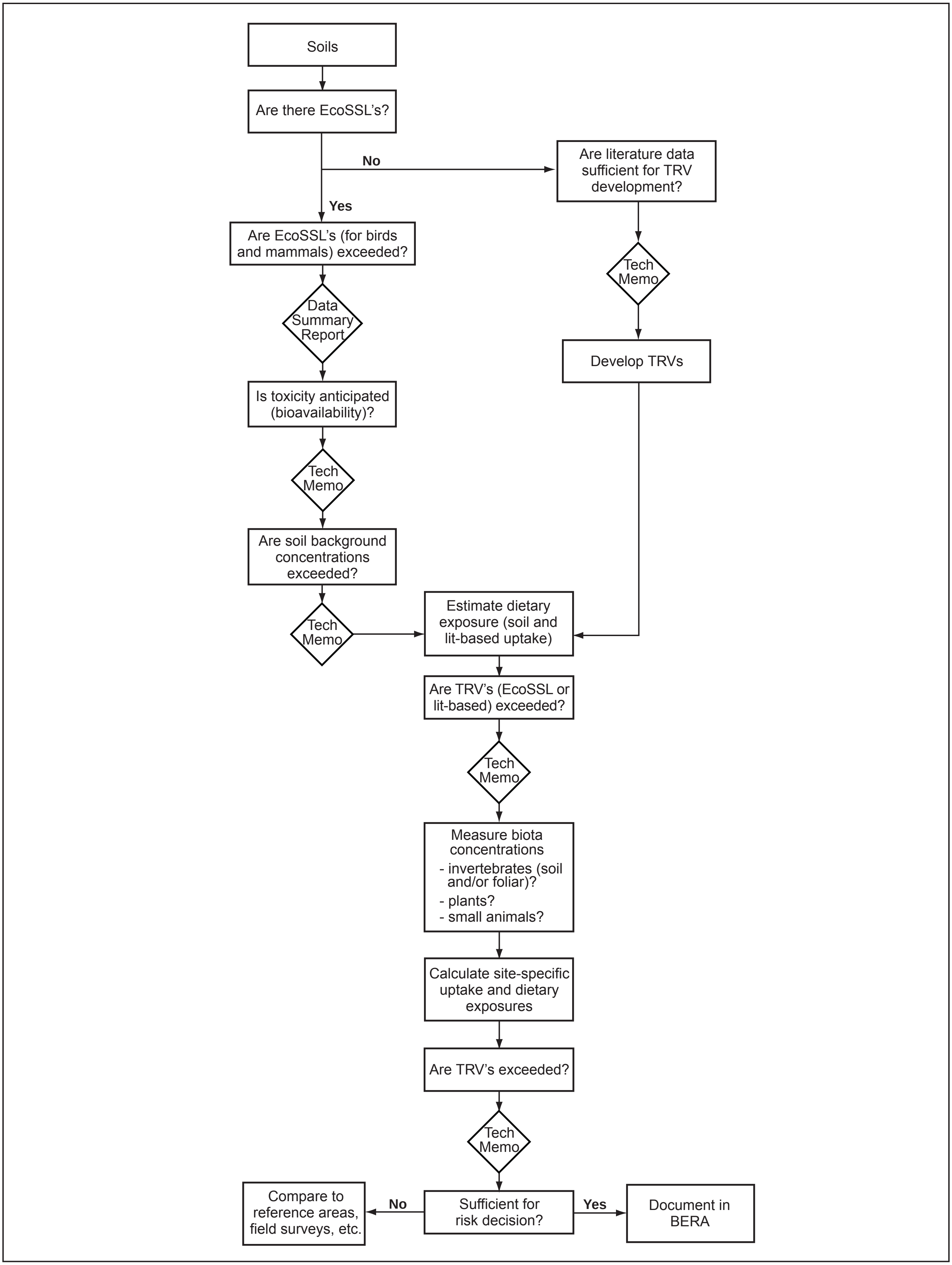


Figure 5-8. Lines of Evidence for Assessing Terrestrial Birds, Mammals, Reptiles and Amphibians

Note: "Tech Memo" as illustrated herein may represent a single technical memorandum (see Table 4-1 of this document) or a point for consultation with EPA.

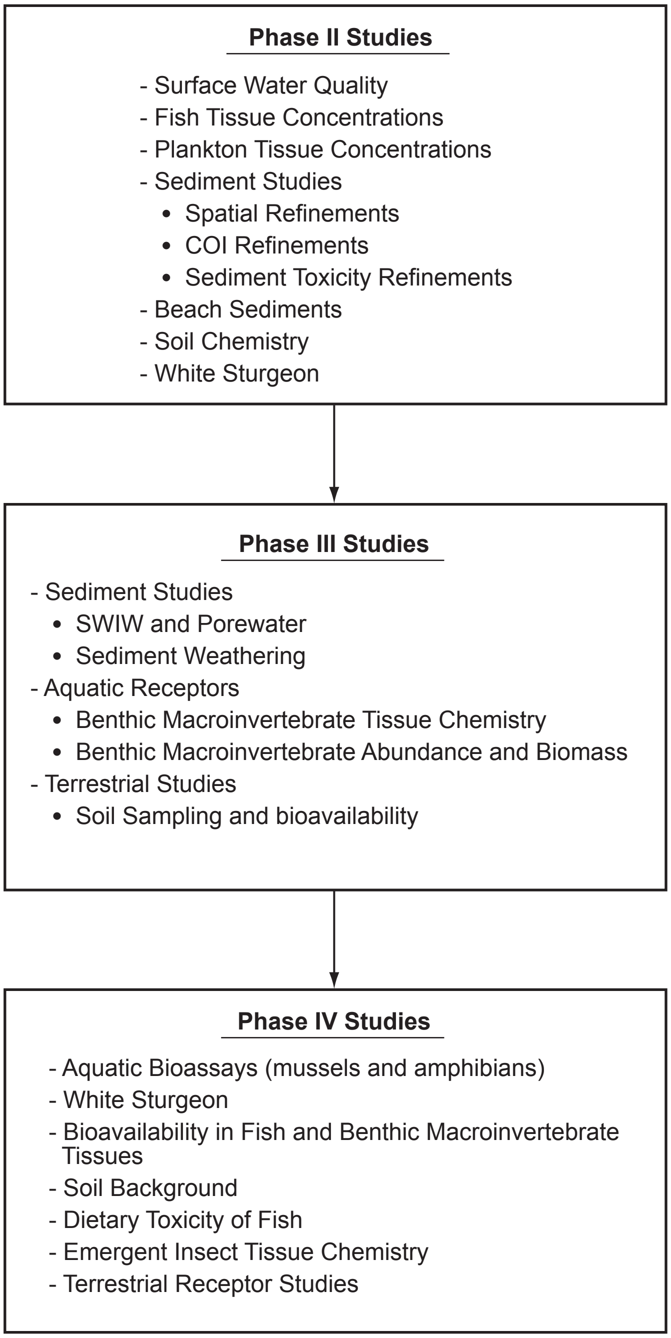
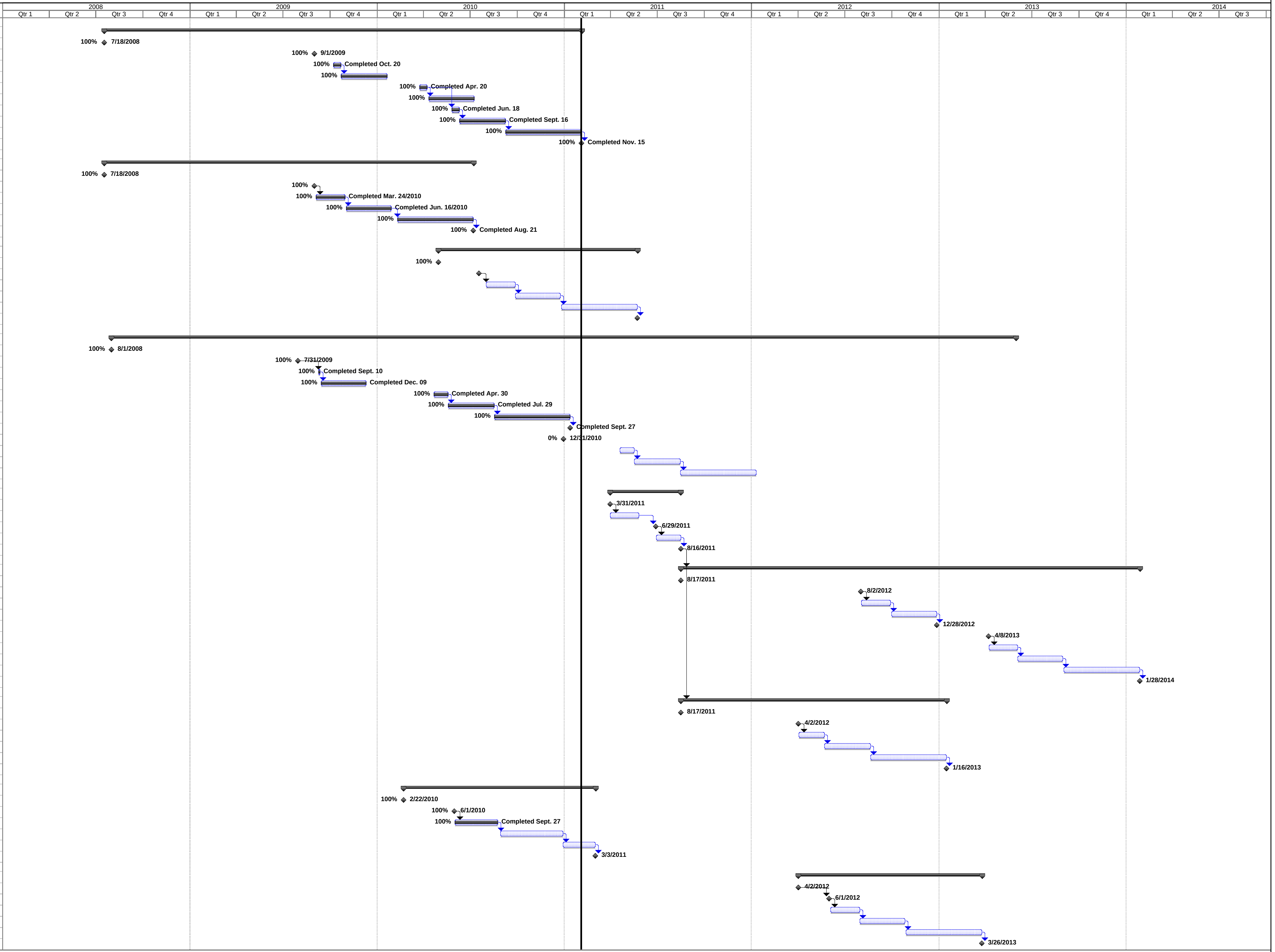


Figure 7-1. Study Sequencing

ID	Task Name	Start	Finish	Predecessors
1				
2	**Surface Water**	**Fri 7/18/08**	**Fri 2/4/11**	
3	Submit Draft SAP	Fri 7/18/08	Fri 7/18/08	
4	Begin Sampling Program Implementation	Tue 9/1/09	Tue 9/1/09	
5	Field Work (Event 1)	Thu 10/8/09	Thu 10/22/09	
6	Laboratory Analysis and Validation (Event 1)	Fri 10/23/09	Wed 1/20/10	5
7	Field Work (Event 2)	Thu 3/25/10	Thu 4/8/10	
8	Laboratory Analysis and Validation (Event 2)	Mon 4/12/10	Fri 7/9/10	7
9	Field Work (Event 3)	Thu 5/27/10	Thu 6/10/10	7
10	Laboratory Analysis and Validation (Event 3)	Fri 6/11/10	Wed 9/8/10	9
11	Data Summary and Data Gaps Report (All Events)	Thu 9/9/10	Thu 2/3/11	10
12	Submit Data Summary and Data Gaps Report	Fri 2/4/11	Fri 2/4/11	11
13				
14	**Fish Tissue Sampling**	**Fri 7/18/08**	**Thu 7/8/10**	
15	Submit Draft SAP	Fri 7/18/08	Fri 7/18/08	
16	Begin Sampling Program Implementation	Tue 9/1/09	Tue 9/1/09	
17	Field Work	Fri 9/4/09	Fri 10/30/09	16
18	Laboratory Analysis and Validation	Mon 11/2/09	Thu 1/28/10	17
19	Data Summary and Data Gaps Report	Wed 2/10/10	Wed 7/7/10	18
20	Submit Draft Data Summary and Data Gaps Report	Thu 7/8/10	Thu 7/8/10	19
21				
22	**Plankton Sampling**	**Sat 5/1/10**	**Tue 5/24/11**	
23	Submit Draft SAP	Sat 5/1/10	Mon 5/3/10	
24	Begin Sampling Program Implementation	Mon 7/19/10	Mon 7/19/10	
25	Field Work	Mon 8/2/10	Mon 9/27/10	24
26	Laboratory Analysis and Validation	Tue 9/28/10	Fri 12/24/10	25
27	Data Summary and Data Gaps Report	Mon 12/27/10	Mon 5/23/11	26
28	Submit Data Summary and Data Gaps Report	Tue 5/24/11	Tue 5/24/11	27
29				
30	**Beach (Shallow Sediment) Sampling**	**Fri 8/1/08**	**Fri 5/31/13**	
31	Submit Draft SAP	Fri 8/1/08	Fri 8/1/08	
32	Begin Sampling Program Implementation	Fri 7/31/09	Fri 7/31/09	
33	Field Work (Event 1)	Wed 9/9/09	Fri 9/11/09	32
34	Laboratory Analysis and Validation (Event 1)	Mon 9/14/09	Thu 12/10/09	33
35	Field Work (Event 2)	Thu 4/22/10	Wed 5/19/10	
36	Laboratory Analysis and Validation (Event 2)	Thu 5/20/10	Tue 8/17/10	35
37	Data Summary and Data Gaps Report (Events 1 and 2)	Wed 8/18/10	Wed 1/12/11	36
38	Submit Draft Data Summary and Data Gaps Report (Events 1&2)	Thu 1/13/11	Thu 1/13/11	37
39	Submit QAPP Amendment	Fri 12/31/10	Fri 5/31/13	
40	Field Work (Event 3)	Wed 4/20/11	Tue 5/17/11	
41	Laboratory Analysis and Validation (Event 3)	Wed 5/18/11	Mon 8/15/11	40
42	Data Summary and Data Gaps Report	Tue 8/16/11	Tue 1/10/12	41
43				
44	**Problem Formulation Expansion**	**Thu 3/31/11**	**Tue 8/16/11**	
45	Submit Problem Formulation Expansion (Draft)	Thu 3/31/11	Thu 3/31/11	
46	Problem Formulation Expansion (Draft) Review	Fri 4/1/11	Thu 5/26/11	45
47	Submit Problem Formulation Expansion (Final)	Wed 6/29/11	Wed 6/29/11	46
48	Problem Formulation Expansion (Final) Review	Thu 6/30/11	Tue 8/16/11	47
49	Problem Formulation Expansion Approval	Tue 8/16/11	Tue 8/16/11	48
50				
51	**Sediment Sampling**	**Wed 8/17/11**	**Tue 1/28/14**	**49**
52	Submit Draft SAP - Part 1	Wed 8/17/11	Wed 8/17/11	
53	Begin Sampling Program Implementation	Thu 8/2/12	Thu 8/2/12	
54	Field Work - Part 1	Fri 8/3/12	Fri 9/28/12	53
55	Laboratory Analysis and Validation - Part 1	Mon 10/1/12	Thu 12/27/12	54
56	Submit QAPP Amendment - Part 2	Fri 12/28/12	Fri 12/28/12	55
57	Begin Sampling Program Implementation	Mon 4/8/13	Mon 4/8/13	
58	Field Work - Part 2	Tue 4/9/13	Mon 6/3/13	57
59	Laboratory Analysis and Validation - Part 2	Tue 6/4/13	Fri 8/30/13	58
60	Data Summary and Data Gaps Report (Parts 1 and 2)	Mon 9/2/13	Mon 1/27/14	59
61	Submit Data Summary and Data Gaps Report	Tue 1/28/14	Tue 1/28/14	60
62				
63	**Soil Sampling**	**Wed 8/17/11**	**Wed 1/16/13**	**49**
64	Submit Draft SAP	Wed 8/17/11	Wed 8/17/11	
65	Begin Sampling Program Implementation	Mon 4/2/12	Mon 4/2/12	
66	Field Work	Tue 4/3/12	Tue 5/22/12	65
67	Laboratory Analysis and Validation	Wed 5/23/12	Mon 8/20/12	66
68	Data Summary and Data Gaps Report	Tue 8/21/12	Tue 1/15/13	67
69	Submit Data Summary and Data Gaps Report	Wed 1/16/13	Wed 1/16/13	68
70				
71	**White Sturgeon Toxicity Testing**	**Mon 2/22/10**	**Thu 3/3/11**	
72	Submit Draft SAP	Mon 2/22/10	Thu 4/1/10	
73	Begin Sampling Program Implementation	Tue 6/1/10	Tue 6/1/10	
74	Field and Laboratory Work	Wed 6/2/10	Tue 8/24/10	73
75	Laboratory Analysis and Validation	Mon 8/30/10	Wed 12/29/10	74
76	Data Summary and Data Gaps Report	Thu 12/30/10	Wed 3/2/11	75
77	Submit Data Summary and Data Gaps Report	Thu 3/3/11	Thu 3/3/11	76
78				
79	**Benthic Tissue Sampling (includes Mussels)**	**Mon 4/2/12**	**Tue 3/26/13**	
80	Submit Draft SAP	Mon 4/2/12	Mon 4/2/12	
81	Begin Sampling Program Implementation	Fri 6/1/12	Fri 6/1/12	80
82	Field Work	Mon 6/4/12	Mon 7/30/12	81
83	Laboratory Analysis and Validation	Tue 7/31/12	Fri 10/26/12	82
84	Data Summary and Data Gaps Report	Mon 10/29/12	Mon 3/25/13	83
85	Submit Data Summary and Data Gaps Report	Tue 3/26/13	Tue 3/26/13	84



Project: UCR RI/FS
Date: Wed 2/2/11

Task ▭ Split ⋯ Progress ▬ Milestone ◆ Summary ▬ Project Summary ▭ External Tasks ▭ External Milestone ◆ Deadline ⇩

* Durations indicated here are intended for planning purposes only and are subject to change.

BERA Schedule_rev_2011_2_2_mg

Figure 9-1. Projected Schedule for BERA Phase 2*