Superfund and Mining Megasites: Lessons from the Coeur d'Alene River Basin

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Committee on Superfund Site Assessment and Remediation in the Coeur d' Alene River Basin; Board on Environmental Studies and Toxicology; Division on Earth and Life Studies; National Research Council

SUPERFUND AND MINING MEGASITES

LESSONS FROM THE COEUR D'ALENE RIVER BASIN

Committee on Superfund Site Assessment and Remediation in the Coeur d'Alene River Basin

Board on Environmental Studies and Toxicology

Division on Earth and Life Studies

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Preface

The U.S. Environmental Protection Agency (EPA) was established in 1970 to protect human health and the natural environment. The agency's mission includes enforcing and implementing environmental laws enacted by Congress, assessing environmental conditions, and solving current and anticipating future environmental issues. To assist EPA in addressing risks associated with chemical emergencies as well as abandoned hazardous waste sites, Congress passed the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) in 1980, better known as the Superfund Act. The Superfund program addresses short- and long-term risks of chemical spills and supports the permanent cleanup and rehabilitation of hazardous waste sites.

In 2002, Congress instructed EPA to ask the National Research Council (NRC) to conduct an independent evaluation of the Coeur d'Alene River basin Superfund site in northern Idaho as a case study to examine EPA's scientific and technical practices in Superfund megasites, including physical site definition, human and ecologic risk assessment, remedial planning, and decision making. NRC established the Committee on Superfund Site Assessment and Remediation in the Coeur d'Alene River Basin. In this report, the committee analyzes the record of decision and supporting documents from this Superfund site to assess the adequacy and application of EPA's own Superfund guidance in terms of available scientific and technical knowledge and best practices.

In the course of preparing this report, the committee held five meetings, including public sessions in Washington, DC; Wallace, Idaho; and Spokane, Washington—where local, state, tribal, and federal officials, as well as representatives from the private sector and nongovernmental organizations, including regulated industries and citizen groups, were invited to meet with the committee and present their views on Superfund activities in the Coeur d'Alene River basin. Interested members of the public were also given an opportunity to speak on these occasions. The following individuals spoke at these meetings: U.S. Senator Larry Craig; U.S. Senator Michael Crapo; U.S. Congressman C. L. "Butch" Otter; Brian Cleary, counsel to Coeur d'Alene tribe; Ernest Stensgar, Chairman of the Coeur d'Alene tribe; Phillip Cernera, Coeur d'Alene tribe; Alfred Nomee, Coeur d'Alene tribe; Ian von Lindern, TerraGraphics Environmental Engineering; John Roland, Washington Department of Ecology; Robert Hanson, Mine Waste program manager; Stephen Allred, director, Idaho Department of Environmental Quality; Ron Roizen, Bill Rust, Frank Frutchey, Lee Haynes, Jack Riggs, Bob Hopper, Fred Brackebusch, Ivan Linscott, Shoshone Natural Resources Coalition Science Committee; Fred Kirschner, Spokane tribe; Rogers Hardy, Citizens Against Rail to Trail/Citizens Advocating Responsible Treatment; Thomas Pedersen, University of Victoria; David Moershel, Spokane physician and president of the Lands Council; Allen Isaacson, professor, Spokane Community College and former U.S. Forest Service supervisory hydrologist for the Idaho Panhandle National Forest; Bruce Lanphear, director, Cincinnati Children's Environmental Health; Jerry Cobb, Panhandle Health District; Brad Sample, CH2M Hill; David Fortier, environment protection specialist, Bureau of Land Management; Paul Woods, Laura Balistrieri, Stephen Box, Nelson Beyer, U.S. Geological Survey; Daniel Audet, U.S. Fish and Wildlife Service; and Elizabeth Southerland, Michael Gearheard, Sheila Eckman, Anne Dailey, Mary Jane Nearman, Angela Chung, Marc Stifelman, Cami Grandinetti, Bill Adams, EPA.

In addition to the information from those presentations, the committee made use of the peer-reviewed scientific literature; government agency reports; information submitted to the committee by citizens, advocacy groups, and industry; and unpublished database information as well as related statistics and data directly obtained from EPA and the states of Idaho and Washington.

This report consists of nine chapters. The first chapter provides an overview of the committee's charge, the issues related to this charge, and the approach the committee took in completing its task. Chapters 2 and 3 review the history of the Coeur d'Alene mining district and the relationship between the biologic, human, and physical environments in the river basin. Chapters 4-8 review scientific and technical questions relating to the remedial investigation, human and ecologic risk assessments, and remedial decisions set forth in EPA's record of decision for the site and the supporting documents. Finally, Chapter 9 discusses lessons learned from the Coeur d'Alene experience and suggests a new paradigm for addressing environmental and health concerns at large complex mining sites.

We wish to thank Earl Bennett, University of Idaho, and Teresa Bowers, Gradient Corporation, for their valuable service while they served on the committee. The committee is also grateful for the assistance of NRC staff in preparing this report: Karl Gustavson, study director; James Reisa, director of the Board on Environmental Studies and Toxicology; Ray Wassel, program director; Ruth E. Crossgrove, senior editor; Cay Butler, editor; Mirsada Karalic-Loncarevic and Bryan Shipley, research associates; and Olukemi Yai, program assistant; as well as John Brown, Emily Brady, Dominic Brose, Alexandra Stupple, and others who supported the project as part of the Board's staff.

Finally, I thank the members of the committee for their dedicated efforts throughout the development of this report.

David J. Tollerud, MD, MPH
Chair, Committee on Superfund Site Assessment and
Remediation in the Coeur d'Alene River Basin

Acknowledgment of Review Participants

This report has been reviewed in draft form by individuals chosen for their diverse perspectives and technical expertise, in accordance with procedures approved by NRC's Report Review Committee. The purpose of this independent review is to provide candid and critical comments that will assist the institution in making its published report as sound as possible and to ensure that the report meets institutional standards for objectivity, evidence, and responsiveness to the study charge. The review comments and draft manuscript remain confidential to protect the integrity of the deliberative process. We wish to thank the following individuals for their review of this report: Craig Boreiko, International Lead Zinc Research Organization; Stephen E. Box, U.S. Geological Survey; Gary Diamond, Syracuse Research Corporation; Lorne G. Everett, Lakehead University and Shaw Environmental & Infrastructure, Inc.; Michael C. Kavanaugh, Malcolm Pirnie, Inc.; Phillip E. LaMoreaux, P.E. LaMoreaux & Associates; Bruce P. Lanphear, Cincinnati Children's Hospital Medical Center; Dwayne Moore, Cantox Environmental, Inc.; Darrell K. Nordstrom, U.S. Geological Survey; Dianne Nielson, Utah Department of Environmental Quality; Benjamin Parkhurst, HAF Inc.; Katherine N. Probst, Resources for the Future; Joyce S. Tsuji, Exponent, Inc.; and Stephen Washburn, ENVIRON.

Although the reviewers listed above have provided many constructive comments and suggestions, they were not asked to endorse the conclusions or recommendations, nor did they see the final draft of the report before its release. The review of this report was overseen by Dr. David G. Hoel, Medical University of South Carolina, and Dr. Perry L. McCarty, Stanford University. Appointed by the NRC, they were responsible for making certain that an independent examination of this report was carried out in accordance with institutional procedures and that all review comments were carefully considered. Responsibility for the final content of this report rests entirely with the authoring committee and the institution.

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Summary

In 1983, the U.S. Environmental Protection Agency (EPA) listed the Bunker Hill Mining and Metallurgical Complex in northern Idaho as a Superfund site on the National Priorities List (NPL). The basis for this listing was high levels of metals (including lead, arsenic, cadmium, and zinc) in the local environment and elevated blood lead levels in children in communities near the metal-refining and smelter complex. Initial cleanup efforts focused on the areas with the most contamination and the greatest risk of health effects—a 21-square-mile "box" in the heart of the Coeur d'Alene River basin. Children's blood lead levels in the box have declined remarkably since the 1970s when lead poisoning was epidemic. They now appear to be approaching those of same-age children in the U.S. general population.

In 1998, EPA began applying Superfund requirements[1] beyond the original Bunker Hill box boundaries to areas throughout the 1,500-square-mile Coeur d'Alene River basin project area. Soils, sediments, surface water, and groundwater are contaminated in areas throughout the basin with metals derived from historical mining operations, and a wide variety of studies have indicated that this contamination poses increased risks to humans and wildlife in the basin. In 2002, EPA issued a record of decision

[1]The Superfund requirements are set forth in the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) of 1980 as amended (42 USC §§ 9601-9675 [2001]), and its implementing regulations are set forth in the National Contingency Plan (NCP) (40 CFR 300).

OVERVIEW OF CONCLUSIONS AND RECOMMENDATIONS

The committee found that scientific and technical practices used by EPA for decision making regarding human health risks at the Coeur d'Alene River basin Superfund site are generally sound. The exceptions are minor. However, for EPA's decision making regarding environmental protection, the committee has substantial concerns, particularly regarding the effectiveness and long-term protection of the selected remedy.

In the human health risk assessment (HHRA), EPA estimated potential lead intake by current and future populations of children using currently available risk assessment procedures with a reasonable degree of certainty. The application of the IEUBK model[a] was also reasonable but would have benefited from greater collection and use of additional site-specific information. Recognizing the importance of protecting current and future generations, remedial decisions regarding human health appropriately emphasized residential yard remediations. Given the prevalence of high concentrations of lead in soils of the studied communities and the potential for lead exposure of young children, the committee concludes universal blood lead screening of children age 1-4 years is warranted. This screening should be timed to coincide with other routine pediatric health care screening tests. Barring recontamination of remediated properties, it seems probable that the proposed remedies will reduce the targeted human health risks. However, long-term support of institutional-control[b] programs should be provided to maintain the integrity of remedies intended to protect human health and guard against health risks from recontamination.

For environmental protection, EPA's site characterization provided a useful depiction of the metal concentrations in soils, sediments, and surface water over the large spatial scale in the basin. However, the characterization did not adequately address groundwater—the primary source of dissolved metals in surface water—or identify specific locations and materials contributing metals to groundwater. In addition, the committee has serious concerns about the feasibility and potential effectiveness of the proposed remedial actions for environmental protection. There are no appropriate repositories to hold proposed amounts of excavated materials, and establishing them in the basin will probably be extremely difficult. Furthermore, the potential long-term effectiveness of proposed remedial actions is severely limited by frequent flooding events in the basin and their potential to

(ROD) that addressed the entire project area, excluding the box (which was the subject of earlier RODs). This ROD contained a "final remedy" to address contamination-related human health risks and an "interim remedy" to begin to address ecologic risks. These remedies are estimated to cost $359 million over 30 years—and even this effort will not complete the job.

Congress instructed EPA to arrange for an independent evaluation of the Coeur d'Alene Basin Superfund Site by the National Academy of Sciences (NAS). In response, the National Research Council (NRC) convened the Committee on Superfund Site Assessment and Remediation in recontaminate remediated areas with contaminated sediments. Yet, flooding apparently received little attention in EPA's selection of remedies. Overall, downstream transport of lead-contaminated sediments can be addressed only by removing or stabilizing the contaminated sediments in the river basin. The committee recommends that the specific sources contributing zinc to groundwater (and subsequently to surface water) and the largest, potentially mobile sources of lead-contaminated sediments be ascertained, and priorities set for their cleanup. If zinc loading to groundwater is determined to stem from subsurface sources that are too deep or impractical to be removed, groundwater should be addressed directly. EPA should consider more thoroughly the potential for recontamination and proceed with those remedies that are most likely to be successful and durable. Because of the long-term and uncertain nature of the cleanup process, it is unrealistic to develop comprehensive remedial schemes and assess their effectiveness a priori. Hence, a phased approach to cleanup with defined goals, monitoring, and evaluation criteria (an adaptive management approach[c]) is warranted.

In general, the Superfund process has a number of serious difficulties in addressing the complex contamination problems in mining megasites such as the Coeur d'Alene River basin. Remediation involves long-term undertakings in which remedies will usually need to be developed over time, and efficient responses to the problems may require the implementation of programs outside the Superfund framework. EPA has demonstrated flexibility in applying Superfund to the Coeur d'Alene River basin and other megasites and has established a process in the basin that incorporates some of the characteristics the committee considers important to address the problems at such sites. However, it is unclear whether all the problems can be addressed efficiently and effectively within the constraints that govern the Superfund process.

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[a]The Integrated Exposure Uptake Biokinetic Model for Lead in Children (IEUBK model) was used at the Coeur d'Alene River basin site to select soil lead cleanup levels in residential yards.

[b]Institutional controls are actions, such as legal controls, that help minimize the potential for human exposure to contamination by ensuring appropriate land or resource use.

[c]Adaptive management is an approach where remediation occurs in stages and the consequences of each stage or phase are evaluated and provide feedback for planning of the next phase.

the Coeur d'Alene River Basin. The committee, composed of members with a wide range of backgrounds and expertise, was asked to consider EPA's scientific and technical practices in Superfund site definition, human and ecologic assessment, remedial planning, and decision making. During the study, the committee held public sessions in Washington, DC; Wallace, Idaho; and Spokane, Washington, where local, state, tribal, and federal officials, as well as private sector and citizen groups presented their views to the committee.

An important aspect of the study charge, beyond considering issues specific to the Coeur d'Alene River basin, is to attempt to extrapolate "lessons learned" at this site to other large complex Superfund sites in the nation. In response, the committee developed recommendations to facilitate EPA's mission at other large, geographically complex mining megasites.

Remedial efforts within the Coeur d'Alene River basin will require much time, a great deal of money, and a concerted effort by involved parties. Thus, the question "Is it worth it?" is often raised. This question, however, depends on the requirements of the applicable federal laws and is not germane to the question of how the agency has implemented these laws. The committee has, as specified in its charge, focused on the agency's implementation and has not addressed the broader questions about the financial or societal value of these expenditures. Such questions go beyond matters that science alone can address. EPA undertook this difficult task at a time when knowledge of the disposition and effects of contaminants within the basin was evolving, and approaches to remediating large sites were poorly developed. Much has been learned since then, and it is through hindsight that this report reviews the process.

DECISION MAKING IN THE COEUR D'ALENE RIVER BASIN

EPA's scientific and technical procedures were generally appropriate and in accordance with the agency's standard procedures, as understood by the committee, for assessing risks to human health and the environment in the Coeur d'Alene River basin. EPA has also made substantial efforts to provide the public with information about its activities and to provide opportunities for public comment and input. However, the committee has concerns about several technical aspects of the analyses and has recommended various ways that EPA's standard techniques might be improved.

The committee recognizes that substantial controversy surrounds remediation at the Coeur d'Alene River basin site, and EPA's decisions were responsive, at least in part, to concerns of affected parties. For instance, cleanup efforts were strongly opposed both locally and within the Idaho state government, partially stimulated by fear of the economic consequences of having the entire basin declared a Superfund site. In contrast, other groups demanded site remediation and strongly opposed any approaches that would allow metals-contaminated media to remain in the environment following cleanup. Therefore, some decisions the committee considers suboptimal might have resulted from compromise with affected parties, as well as the reality of limited financial resources.

The discussion below is a synopsis of the committee's conclusions and recommendations provided throughout this report.

SITE CHARACTERIZATION AND REMEDIAL INVESTIGATIONS

In completing the remedial investigation (RI), EPA conducted, sponsored, and synthesized substantial research in cooperation with the state of Idaho, other federal agencies, and the Coeur d'Alene Tribe to evaluate the extent of metals contamination in the basin. Some of the research efforts are state of the art and should substantially inform the selection of appropriate remedies. Overall, EPA's evaluations provide a useful depiction of the location of contaminated soils, sediments, and surface waters over the large spatial scale of the basin. The data have been used to estimate average mass loading of metals in the Coeur d'Alene River and Lake and to provide an adequate description of contaminants moving through much of the system.

Nevertheless, the committee has identified some serious weaknesses in the RI. EPA has not adequately characterized the substantial hydrologic and climatic variations that can occur in the basin. Contaminant transport models are based on average flows and conditions, and the RI only minimally characterizes the extreme events (for example, flood events that transport large amounts of contaminated sediments) that substantially affect the fate and transport of metals throughout the basin. In addition, EPA's segmentation of geographic areas within the basin for assessment and remedial actions does not facilitate a basinwide analysis of sources, transport, and fate of contaminants. In particular, remediation of the Bunker Hill box is under a separate administrative structure, yet this area contributes substantially to downstream contamination.

To support remedial decision making adequately, the specific source areas[2] of contamination releasing dissolved and particulate metals should be characterized. Instead, EPA inferred source areas contributions of metals largely from surface-water studies and not, for example, from studies of metal leachability from source materials. EPA's site characterization also did not adequately address groundwater—the primary source of dissolved metals in surface water. Understanding the contamination of groundwater by aquifer materials, the dynamics of groundwater movement, and the complex relationship between surface water and groundwater will require additional study.

Evaluations of chemical speciation and mineralogy were extremely limited in the RI. As metals move though the system, their chemical form can change and affect, for example, their ability to be absorbed by organisms if ingested (bioavailability) or their ability to leach into groundwater from

[2]Source areas are the specific locations of materials that contribute contaminants to environmental media of interest (for example, surface water or groundwater).

aquifer material. For Lake Coeur d'Alene, additional characterization of the behavior of metals in lake sediments and the relationship between eutrophication and metals release is also needed.

Recommendations

In its remedial planning, EPA should incorporate new data that have been made available by the U.S. Geologic Survey (USGS), the Coeur d'Alene tribe, and others since issuance of the ROD and should proceed, as planned, with more thorough source identification before cleanup to verify the location, magnitude, disposition, and contributions from contaminant sources.

A better understanding of dissolved metals, particularly zinc, is needed to account for movement to and from groundwater and surface water. The chemical and hydrologic components of the assessment should be sufficiently rigorous to identify source areas of contaminants and permit evaluation of the consequences of alternative remedies to the transport of dissolved metals through the system.

Understanding the speciation of metals is important to characterize risk more effectively and ascertain the potential effectiveness of remedial actions. Speciation information should be collected and examined to elucidate the potential for metal transport and the effect of transformation processes on the fluxes and bioavailability of metals.

HUMAN HEALTH RISKS AND REMEDIAL DECISIONS

Human Health Risk Assessment (HHRA)

The HHRA sought to estimate risks to human health associated with estimated concentrations of environmental contaminants, particularly lead and arsenic, and to calculate cleanup concentrations that would protect human health.

EPA estimated potential lead intake by current and future populations of children according to current risk assessment procedures with a reasonable degree of certainty. Consequently, the committee concluded that EPA's HHRA is correct in concluding that environmental lead exposure poses elevated risk to the health of some Coeur d'Alene River basin residents. The committee agreed that subsistence activities, if they were to be practiced, would be associated with elevated risk. EPA also applied reasonable methods to apportion risk among exposure sources, including those unrelated to mine wastes. EPA concluded that although lead from old house paint probably contributed to the exposure of some children, lead-contaminated soil was the primary contributor to health risk from lead.

Children of ages 1 to 4 are the group at highest risk from lead exposure. The committee found it inappropriate that the HHRA presented aggregate data on childhood lead screening for children 0-9 years old, as that information is misleading and tends to underestimate the risk among the principal target group. Furthermore, the annual blood lead sampling of children at fixed sites is suboptimal and produces results with too much potential for nonrepresentative sampling to evaluate the effectiveness of public health intervention strategies in the basin. Universal blood lead screening of children 1-4 years old is warranted for Coeur d'Alene River basin communities, given the prevalence of high concentrations of environmental lead.

For arsenic, EPA collected no information about actual human uptake and based its risk assessment on arsenic concentrations in environmental samples. Biological indicators of actual human arsenic exposure would serve to strengthen future risk assessments at sites such as Coeur d'Alene, though the committee recognizes the limitations of the currently available arsenic biomarkers.

The effects of psychological stress on mental health are not considered in the HHRA. However, there is strong scientific evidence that living in or near an area designated as a Superfund site is associated with increased psychological stress and may also cause adverse health effects.

Recommendations

Health surveillance activities conducted or sponsored by local, state, or federal (for example, the Agency for Toxic Substances and Disease Registry [ATSDR] or EPA) entities should include the following:

- Annual blood lead screening of all children 1-4 years old who live in the basin. Screening should be coordinated with local health care providers and timed to coincide with other routine health care screening tests. These data would be useful for evaluating the efficacy of the remedial activities.
- Health interventions that address possible consequences of chronic psychological stress. These may have significant community benefits and should be implemented before or concurrent with cleanup efforts.
- Continued research at the national level on biomarkers of human arsenic exposure to strengthen future HHRAs.

Use of the Integrated Exposure Uptake Biokinetic (IEUBK) Model

A major controversy at the Coeur d'Alene River basin site arose because EPA did not base its risk assessment and remediation decisions on the blood lead levels that had been measured but on the IEUBK model to estimate potential levels and related health risks.

EPA's remediation goal for lead in soil states that a typical child or group of similarly exposed children should not have more than a 5% estimated risk of exceeding a blood lead level of 10 micrograms per deciliter (μg/dL). Because protecting the future, as well as current, residents is important and because measuring attainment of the remediation goal is not possible, the use of a model that predicts such risks is necessary and appropriate. Multicompartment predictive blood lead models, such as the IEUBK model, are powerful tools for assessing pediatric risk from lead exposure, exploring lead risk management options, and crafting remediation strategies.

At the Coeur d'Alene River basin site, EPA's application of the IEUBK model was generally adequate and appropriate, but not optimal. Additional collection and use of site-specific information, particularly site-specific bioavailability and ingestion rates, would have improved the application of the model. The credibility of the results would have been enhanced by greater use of alternative tools (for example, other models and epidemiological studies) to assess the reliability of IEUBK model predictions and better characterization of the physical-chemical properties of the exposure source materials.

The committee also provides several conclusions regarding the model and recommendations for the model's future development and application. The committee concluded that, in general, the design and functioning of the IEUBK model are consistent with current scientific knowledge; however, the committee concluded that there were some technical issues, particularly the uncertainties associated with the default assumptions for bioavailability of soil lead, soil and dust ingestion rates, and the parameter used to extrapolate from a single blood lead estimate to the distribution of concentrations throughout a population.

EPA regulatory guidance on the use of the IEUBK model in conjunction with data from blood lead surveys is incomplete, particularly on actions to take when blood lead studies and IEUBK model results disagree by a substantial margin. The guidance states that model results are to take precedence in those cases; however, a more comprehensive articulation is required. The committee concluded that the model's inherent uncertainties coupled with the need to protect present and future populations necessitate additional information (such as blood lead studies) to help characterize the model's uncertainties. This is particularly true at large mining megasites, such as the Coeur d'Alene River basin, where physical site characteristics and human exposure profiles can vary widely across the large geographic area. At those sites, the IEUBK model results should not be the sole criterion for establishing health-protective soil concentrations because model uncertainty and site complexity may interact in unexpected or unknown ways.

Recommendations

EPA should pursue initiatives to improve the knowledge base for soil and dust ingestion rates and consider whether soil ingestion rates are site specific. EPA should also pursue implementation of a model version that provides a probabilistic distribution of blood lead concentrations in a population.

EPA should require that cleanup levels derived from the IEUBK model be supported by site-specific measures of bioavailability and concentrations of lead in various sizes of soil particles.

EPA should clarify guidance on using the IEUBK model in conjunction with blood lead studies, particularly when reconciling differences between modeled and observed blood lead levels and when considering the uncertainties associated with each.

A comprehensive revision of the 1998 EPA directive on IEUBK model use at large geographically complex sites is needed. The revision should establish a decision-making structure for determining site cleanup concentrations and specifications based on the IEUBK model's predictive capability, blood lead study results, economic feasibility, and long-term remedy protection.

Remedial Decisions Regarding Human Health

The committee concluded that EPA adequately characterized the feasibility of alternative remedial actions for addressing risks to human health; however, the long-term effectiveness of the selected remedy in the Coeur d'Alene River basin is questionable because of the possibility, even likelihood, of recontamination from floods and damage to protective barriers used in residential remediations.

Barring recontamination, it seems probable that the proposed remedies will reduce the human health risks addressed. There are logical reasons to expect that residential yard remediations decrease lead exposure, and available evidence suggests the efficacy of this approach within the Bunker Hill box. Thus, the strategy for yard remediation is supportable even though the scientific evidence supporting substantial beneficial effects is currently weak.

Recommendations

Long-term support of institutional-control programs should be provided to avoid undue human health risks from recontamination and to maintain the integrity of remedies intended to protect human health.

The effectiveness of remedial actions for human health protection needs to be further evaluated. This evaluation should be supported by ongoing environmental and blood lead monitoring efforts.

ENVIRONMENTAL RISKS AND REMEDIAL DECISIONS

Ecologic Risk Assessment (ERA)

EPA's ERA describes the likelihood, nature, and severity of adverse effects on plants and animals resulting from exposure to metals associated with mining operations throughout the study area. The committee found the assessment to be generally consistent with best scientific practices. In some respects, it was substantially more extensive than ERAs at many other sites. However, support for conclusions on different organisms and habitats is highly variable. Conclusions about waterfowl are especially strong because of the wealth of data on dose-response relationships developed by USGS and the U.S. Fish and Wildlife Service, but conclusions on other organisms, particularly in riparian and upland communities, are much less certain. Deficiencies that precluded a thorough assessment of impacts on some biota and on large portions of the basin are also apparent. For example, few measures of community structure and site-specific toxicity tests were used to characterize risks to fish and benthic macroinvertebrates in the lower Coeur d'Alene River. The Lake Coeur d'Alene assessment was not supported by studies to evaluate whether metal concentrations in sediments or overlying waters were impacting ecologic communities. Finally, in considering effects on organisms, the high variability in exposures related to extreme events, including low-flow conditions and flood events, was not considered.

Overall, the committee was surprised at the minimal extent to which EPA used the ERA in subsequent decision making. Preliminary remediation goals (PRGs) (concentrations of metals intended to protect organisms) developed for fish, benthic invertebrates, small mammals, plants, amphibians, and birds other than waterfowl are based on national regulatory criteria, literature-derived values, or background concentrations. PRGs derived in that fashion are highly uncertain and have questionable value for guiding remediation decisions. Of the PRGs, only the national ambient water quality criteria were adopted from the ERA as remediation goals in the ROD.

Recommendations

Further evaluations of the impacts of exposures to metals in the aquatic and terrestrial environment are needed to support remedial actions intended to promote recovery of biota within the basin.

In developing restoration goals and performance metrics, additional consideration should be given to habitat modifications (for example, stream channelization) resulting from human activities that may prevent a return to premining conditions.

Remedial Decisions for Protecting the Environment

EPA used the feasibility study to select, document, estimate the cost of, and compare five alternative strategies for environmental protection. Despite the extensive effort and documentation, none of these alternatives was selected. The remedial strategies in EPA's ROD for protecting the environment are presented as "interim remedies," and the committee is encouraged that EPA took this approach. At a site of this size and complexity, developing comprehensive remedial schemes and assessing their effectiveness a priori is not realistic. The on-the-ground effect of remedial actions is often unknown, as are unforeseen conditions that make solutions that appear feasible on paper, infeasible in the field. EPA is proposing to use adaptive management to implement interim ecologic protection remedies; however, the committee is concerned about the rigor of EPA's adaptive management approach at this site, particularly regarding performance indicators needed to evaluate progress.

The feasibility and effectiveness of EPA's proposed remedial actions to protect fish and wildlife resources have not been adequately characterized. These actions can be roughly described as those intended to stem the influx of dissolved zinc to surface waters and as those intended to reduce the transport of lead-contaminated sediments through the basin and the effect of those sediments on waterfowl. Removal of contaminated materials is a core constituent of both strategies, yet the lack of available repositories (or even identified locations) is particularly problematic. Still, the committee recognizes that contamination problems in the study area will be solved only when the contaminated materials in the river basin have been removed or stabilized.

The threat to aquatic life in the basin results primarily from the influx of high levels of dissolved zinc from groundwater to surface waters. Yet, groundwater has not been targeted for remediation. Removing contaminated materials as a means to curtail fluxes of metals to groundwater and subsequently to surface water is a logical strategy. However, the specific source areas contributing zinc to groundwater throughout the basin are not well understood, so it is not clear if proposed removals will have an effect on surface-water concentrations. Evidence of the effectiveness of prior removals of materials in the basin has not demonstrated a substantive effect in reducing surface-water concentrations of zinc. A major portion of the dissolved zinc in the lower basin results from groundwater seepage through the Bunker Hill box, a source that is not addressed in the ROD for the basin.

The Coeur d'Alene River basin is a system where floods have a fundamental role in the resuspension and distribution of contaminants and particularly in the potential recontamination of remediated areas, including wetlands and river banks, by contaminated sediments. An understanding of the source areas of these contaminated sediments is evolving. Although impacts to waterfowl in the lower basin are severe, the durability of proposed remedial efforts to protect waterfowl is highly questionable. In addition, recontamination of wetlands by flood waters containing lead-contaminated sediments would quickly undo the benefits of remediation. The committee sees the need for such measures as restoring wetlands on agricultural lands in the lower basin and upgrading the quality of the habitat in existing wetland areas that have the least likelihood of being recontaminated.

Recommendations

EPA should improve its planned adaptive management approach by establishing unambiguous links between management objectives, management options, performance benchmarks, and quantitative monitoring indicators for the habitats and ecologic communities addressed in the ROD.

Remedial Efforts to Address Zinc in Surface Water

As part of its remediation planning, EPA should seek to locate those specific sources contributing zinc to groundwater (which is subsequently discharged to surface water) and set priorities for their remediation. If it is determined that loading to the groundwater stems from subsurface materials too deep or impractical to be removed, groundwater should be addressed directly.

EPA should continue to support research on and demonstration of lower-cost innovative groundwater treatment systems. In particular, EPA should place a high priority on identifying possible methods of reducing metal loading in groundwater from the Bunker Hill box and highly-affected basin tributaries.

Remedial Measures to Address Transport and Effects of Particulate Lead

Recontamination of remediated areas from flooding is a major concern. In selecting sites for remediation, EPA should consider the potential for recontamination and proceed with remedies that are most likely to be successful and durable. To the extent that water yield and flooding can be managed through land-use practices, it is important to include these practices in schemes designed to protect human and ecosystem health.

Remedial measures should address the largest potentially mobile sources of lead-contaminated sediments and seek to address those sources with the highest potential for contributing such sediments to the system. To facilitate such measures, EPA should develop a quantitative model for sediment dynamics, deposition, and geochemistry for the basin watershed. In designing and implementing remedies, consideration should be given to possible unintended effects, such as impacts to fluvial behavior and migration of resuspended sediments.

MINING-RELATED MEGASITES

Superfund megasites are often defined as those sites with projected cleanup costs expected to exceed $50 million. In this section, the committee restricts its conclusions to mining-related megasites that, in addition to their high costs for remediation, include massive amounts of wastes resulting from many years of mining activities. Wastes at these sites are dispersed over a large area and deposited in complex hydrogeochemical and ecologic systems that often include human communities and public natural resources.

The committee concludes that an effective program for mining megasites should emphasize long-term adaptive management. The desirable program components are a stable management structure, long-term monitoring components, active state and local involvement in the remediation process, a broad perspective regarding what actions should be undertaken in addition to cleanup, and long-term funding.

Most of the committee's recommendations regarding mining megasites can be implemented within the Superfund framework; some reflect actions that EPA has already undertaken to some extent in the Coeur d'Alene River basin; and some probably cannot be implemented under the current framework, at least not without private or nonprofit partnerships.

Recommendations

Design the data collection, evaluation, and decision-making process at mining megasites so that the remediation program focuses on establishing a durable process for long-term management of the sites, as final remedies may not be realistic at some megasites.

Be ready to waive specific "applicable or relevant and appropriate requirements" (ARARs) if an effective monitoring program demonstrates that those numeric standards are not necessary to achieve the basic goals of protecting human health and the environment.

Where final remedies cannot be realistically implemented, establish a rigorous and responsive adaptive management process for environmental remediation. ERAs at such sites should be designed to support remedy selection, and move beyond documentation of the presence or absence of risks. In particular, the ERA should be a source of performance metrics and restoration goals for use in an adaptive management strategy.

Establish an independent external scientific review panel with multidisciplinary expertise to provide ongoing evaluations and advice to the relevant agencies on remediation decisions at mining megasites. Although this recommendation may appear to add to the bureaucratic process, at particularly complex sites it may well speed cleanup, avoid excess costs, and provide a mechanism for resolving technical disagreements.

Broaden the goals of the cleanup to include restoration of habitat for ecologic resources to the extent required to meet biological performance goals. For affected communities, provide economic assistance and comprehensive medical support services that acknowledge the broad effects that toxic waste sites have on health.

Encourage development of alternative and innovative technologies, including responsible re-mining as remedial strategies. Consider offering indemnification to private or nonprofit entities that participate in cleanup, agreeing that their liability will be limited to problems resulting from the remediation activity.[3]

Look for opportunities to provide long-term support for implementing and maintaining the cleanup activities and stewardship of the land. Possible sources of such support might include special appropriations by Congress, trust funds, or partnerships with private organizations.

Both risk assessment and risk management activities should be structured according to the natural environmental system boundaries; they should not represent the aggregation of policies previously used at smaller, simpler locations.

[3]Such relief should not be afforded to any responsible party at the site who has not entered into a binding settlement agreement with EPA regarding its cleanup liability.

1

Introduction

For more than 100 years, the Coeur d'Alene River basin has earned its cognomen as "The Silver Valley" by being one of the most productive silver, lead, and zinc mining areas in the United States. Its history is as rich as the millions of tons of ore that have been extracted and processed there. But that history has left a legacy of contamination that extends 166 miles across the state of Idaho, through Lake Coeur d'Alene and down the Spokane River into the state of Washington. A U.S. Environmental Protection Agency (EPA) plan to clean up this contamination under Superfund[1] proposes spending hundreds of millions of dollars over three decades—and even this effort is not expected to complete the job. As might be expected of any undertaking of this magnitude, the plan has created substantial controversy and confusion. This report reviews and evaluates many of the issues and concerns that have been raised regarding EPA's decisions.

COEUR D'ALENE RIVER BASIN

The headwaters of the South Fork of the Coeur d'Alene River begin in the Bitterroot Mountain Range at the Idaho-Montana border, and the river flows westward as a high-gradient mountain stream past the town of Mullan

[1]The Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) of 1980 (P.L. 96-510) established a "Superfund" to identify contaminated sites, determine responsible parties, and finance cleanups when responsible parties could not. EPA administers the Superfund program in cooperation with individual states and tribal governments.

to Wallace, Idaho, where it joins two large tributaries, Canyon and Ninemile Creek. Below Wallace, the valley broadens, the channel gradient begins to diminish, and the river increases in flow as it passes the Idaho communities of Osburn, Kellogg, Smelterville, and Pinehurst. Below Pinehurst, the South Fork joins the North Fork, and the valley widens to several miles, with the floodplain containing thousands of acres of wetlands and small lakes that provide a valuable stopping place for migratory waterfowl. Some 70 miles from its source, the river empties into the 25-mile-long Lake Coeur d'Alene, which in turn is drained by the Spokane River at its northern end.

In the late 1800s and through most of the 20th century, the upper and middle portions of the basin were a major mining region—the "fabulous Coeur d'Alene" (see Chapter 2 of this report). The area had more than 100 mines and ore processing operations producing silver, lead, zinc, and other metals. The Bunker Hill Mine and Smelting Complex, located in Kellogg, Idaho, was the largest of these, and, when the Bunker Hill smelter was built, it was the largest smelter in the world. The Coeur d'Alene mines produced and processed an estimated 130 million metric tons (more than 140 million U.S. tons) of ore during their first century of operation (Long 1998). Today, although a few mines continue to operate, most have closed; the smelting complex is shut down and most of its facilities have been demolished.

The mining, processing, and smelting of such a huge volume of ore resulted in widespread environmental contamination. Many of the mine tailings throughout the region were discharged directly to Coeur d'Alene River and its tributaries until 1968 when the practice was prohibited. Smelting operations at Bunker Hill also discharged large quantities of sulfur dioxide, lead, and other metals that affected local communities and the environment. During operation of the smelter—particularly in the early 1970s when its pollution-control devices failed—large numbers of nearby residents, especially children, had highly elevated blood lead levels (BLLs) (IDHW 1976). The wastes produced by the milling and processing operations pose risks to residents of the area and to the wildlife—particularly fish and migratory birds—that depend on the basin's natural resources.

SUPERFUND DESIGNATION

In 1983, EPA listed the Bunker Hill Mining and Metallurgical Complex on the National Priorities List (NPL).[2] This site encompasses a 21-square-mile rectangular area (commonly called "the box") surrounding the Bunker Hill smelter complex. The site was divided into two operable units (OUs):

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[2]The National Priorities List is intended primarily to guide EPA in determining which sites warrant further investigation under Superfund.

OU-1 covered the "populated areas" of the box and OU-2 covered the "nonpopulated areas," including the former smelter and industrial facility. Cleanup began in earnest after EPA issued the record of decision (ROD) for OU-1 in 1991 and for OU-2 in 1992. Although much of the area within the box has been cleaned up, remedial activities are still under way.

In February 1998, EPA announced that it would extend its Superfund remedial authorities outside the box. Until then, the agency had attempted to address contamination problems outside the box without invoking the formal Superfund process. The agency concluded, however, that the authorities it had been applying to address the widespread contamination and risks to human health and the environment posed by the mining-related wastes outside the box were insufficient (EPA 2004).

This action resulted in the addition of OU-3 that covers all the contaminated areas in the basin, Lake Coeur d'Alene, and the Spokane River, outside the original box. This controversial extension created a large degree of contention among residents within the basin, as many new communities were given the "Superfund" designation. Not surprisingly, many residents were concerned and angry over the designation of their community as a Superfund site and the perception that the designation and associated stigma would be long-lasting and further depress an economy already suffering severely from the loss of mining-related jobs. This fear was bolstered by the reality that the box has remained on the NPL since its listing in 1983, and the ROD for OU-3 established a 30-year "interim" remedial plan. Furthermore, confusion about the OU-3 site designation was magnified by the inexact nature of the site boundaries.[3] This situation is understandably stressful and confusing for residents and landowners within the basin, as there is no straightforward mechanism to determine whether property is located within the Superfund site.

COEUR D'ALENE RIVER BASIN AS A MINING MEGASITE

Cleaning up the Coeur d'Alene River basin is a major challenge for EPA's Superfund program. The amount and wide distribution of waste materials preclude complete remediation with traditional cleanup approaches such as removal and capping. Large portions of the communities

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[3]The Superfund site is considered to be "all areas of the Coeur d'Alene Basin where mining contamination has come to be located." Although areas where contamination does not exist are not included in the site, this designation has led to the widespread notion that the Superfund site encompassed the entire 1,500-square-mile watershed of the Coeur d'Alene River between the Montana border and the confluence of the Spokane River with the Columbia River (for discussion, see Villa 2003). This issue is addressed by EPA in the ROD, Part 3, Responsiveness Summary (EPA 2002), under: "General comment: Concerns about the boundaries of the Superfund site," p. 2-4.

are built on top of mining wastes, and infrastructure, such as the embankment of Interstate 90, is built out of them. Every flood distributes these wastes further, and the contaminants undergo chemical changes—which can increase or decrease the risk they pose—as they travel through the river basin. Thousands of people living in multiple political jurisdictions are involved, and some cleanup efforts are expected to take centuries to achieve ambient environmental protection standards even after hundreds of millions of dollars are spent on cleanup activities.

This site is not, however, an isolated case. There are thousands of abandoned hardrock mining areas throughout the country, particularly in the western states[4] (see Chapter 9). EPA has already listed 63 of these on the NPL, and some have many of the same characteristics as the Coeur d'Alene River basin—they are extensive, expensive, complex, and controversial, with private parties that may be unable or unwilling to accept responsibility for the cleanup. EPA has come to call sites like the Coeur d'Alene River basin "megasites"[5] and is increasingly concerned about how to handle them with the diminishing cleanup funds it has available. Experience at the Coeur d'Alene River basin provides some useful insights into this question.

THE COMMITTEE'S CHARGE

To evaluate scientific and technical aspects of the Superfund designation to OU-3, Congress instructed EPA to arrange with the National Academy of Sciences (NAS) to undertake an independent evaluation of the Coeur d'Alene River basin Superfund site.[6] The study was funded by a Congressional appropriation in the 2003 Consolidated Appropriations Resolution (P.L. 108-7). The corresponding bill report (Report 107-740) from the U.S. House of Representatives Appropriations Committee indicated that it wanted NAS to consider:

> EPA's scientific and technical practices in Superfund site definition, human and ecologic assessment, remedial planning, and decision making. NAS is further expected to assess the adequacy and application of EPA's own Superfund guidance in terms of currently available scientific and technical knowledge and best practices, as well as to provide guidance to facilitate scientifically based and timely decision making for the Coeur d'Alene site.

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[4]Hard rock mines exclude coal and certain industrial mineral mines, such as sand and gravel mines.

[5]The general definition of a megasite is that it probably will cost more than $50 million dollars to clean it up to the standards called for in the Superfund legislation.

[6]Designated on the NPL as the Bunker Hill Mining and Metallurgical Complex.

In making this request, Congress made it clear that it did not expect "NAS to recommend a specific remedial strategy for this site" and that it did not intend "that ongoing and planned remediation activities within the original 21 square mile NPL site be disrupted or adversely impacted in any way" because of the study.

In response, the Committee on Superfund Site Assessment and Remediation in the Coeur d'Alene River Basin was convened by the National Research Council (NRC) of NAS. The committee, composed of members with a wide range of backgrounds and expertise, was charged to consider the specific tasks provided in the statement of task (see Appendix A for the statement of task and committee member biosketches). The topics within the task roughly parallel the Superfund evaluation process and pertain to the various decision documents relating to OU-3, including site characterization in the remedial investigation, the ecologic risk assessment, the human health risk assessment, the integrated exposure uptake biokinetic model (a model used by EPA to evaluate soil cleanup levels for lead in the human health risk assessment), and remedial decisions covered in the feasibility study and the ROD. Finally, the statement of task directs the committee to develop "lessons learned" from the evaluation of this site that can be extrapolated to other sites and considered at the national level. The chapters of this report reflect the components of the statement of task.

NATIONAL RESEARCH COUNCIL AND THE COMMITTEE PROCESS

The NRC of NAS is a nonfederal, nonprofit institution that provides objective science, technology, and health policy advice generally by producing consensus reports authored by committees. The NRC exists to provide independent advice; it has no governmental affiliation and is not regulatory in nature. The committee was constituted only to review and evaluate the scientific and technical aspects of the remedial proposals and whether these proposals conformed to the relevant regulatory guidance.

There is no direct oversight of a committee by the study sponsor or any other outside parties. In this regard, EPA and other interested parties have no more input or access to committee deliberations than the general public. This arrangement permits the committee complete independence in conducting its study. The committee members represent a wide range of backgrounds and expertise and conduct their work solely as a public service, volunteering to the NRC and the nation, cognizant of the importance of providing timely and objective scientific advice.

In conducting its review and evaluation, the committee relied on the Superfund site decision documents and supporting materials, other scientific studies including those conducted in the Coeur d'Alene River basin, technical presentations made to the committee by investigators, presentations to the committee by the public, other information submitted by individuals and interest groups (including expert witness reports from the natural resources damage assessment case currently under way in federal court),[7] and the committee's observations while visiting and touring the site. The committee presented written questions and information requests to EPA, the state of Idaho, and the state of Washington when further clarification was needed. All information that was received by NRC staff was made available to committee members and is available to the public through NRC's public access records office.

The committee held five meetings. Three of the meetings included open, information-gathering sessions where the committee heard from invited speakers and from interested members of the public. The first public session (in January 2004) was in Washington, DC. Two meetings (one in April and one in June 2004) were held in the Coeur d'Alene region, and the committee toured a length of the Coeur d'Alene River basin from Burke, Idaho, to Spokane, Washington, and held public comment sessions in Wallace, Idaho, and Spokane, Washington. The entire final two meetings were closed, deliberative sessions attended only by committee members and NRC staff.

Issues at the Coeur d'Alene River basin site are complex and have a long history; as such, this review addresses some issues in greater detail than others. For example, the statement of task (Appendix A) requests the committee to review the adequacy and adherence to guidance on a scientific and technical basis. The committee was not asked to provide a legal review and therefore the report does not provide a clause-by-clause review of compliance with the National Contingency Plan and the Comprehensive Environmental Response, Compensation, and Liability Act. There were also numerous concerns expressed by the public that are outside the purview of the committee. Some of these relate to limitations in the legislation establishing Superfund, some to issues outside EPA's responsibility, some to policy decisions made by the agency, and some to statements agency personnel have made explaining these decisions.

One question often raised to the committee was whether the benefits expected to result from the cleanup are worth the high costs required to achieve them. Certainly this is an expensive project. EPA projected the

[7]In the natural resources damage assessment court case, the Coeur d'Alene tribe and Federal Trustees (U.S. Fish and Wildlife Service and others) are suing a consortium of mining companies for damages to the environment in the Coeur d'Alene River basin. The committee did not engage in or follow this legal process as it is not within its purview. The committee did have access to expert witness reports (which are public documents) from this case that were relevant to aspects of the Coeur d'Alene River basin environment related to their statement of task.

discounted costs over the first three decades to be approximately $360 million, including approximately $92 million to protect human health in the basin and approximately $250 million primarily for environmental protection (EPA 2002, Table 12.0-1). The current population of children in the basin (the primary intended beneficiaries of remedial efforts in residential areas) is small, and it remains unclear how much conditions will actually be improved for the fish and waterfowl by the interim measures being proposed. Thus, the question "Is is worth it?" is often raised. This question, however, pertains to the requirements of the applicable federal laws and is not germane to the question of how the agency has implemented these laws. The committee has, as specified in its charge, focused on the agency's implementation and has not addressed the broader questions about the value of these expenditures.

In this and other ways, the committee has focused on addressing issues within the statement of task. The committee attempted to strike a balance in addressing the larger issues while providing sufficient detail to explain its conclusions and recommendations. It became clear to the committee that the evaluation and remediation process are continuing. New information is being gathered, experiments on possible remedial approaches are being conducted, and proposed remedies are being revised. This process will continue for decades and perhaps centuries. Thus, the committee does not consider its review to be the last word, but hopes that its findings and recommendations will assist government agencies and other stakeholders in improving the approaches to address large complex mining megasites such as the Coeur d'Alene River basin.

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2

Historical Background

The Coeur d'Alene region, named after the Indian tribe that originally inhabited the area, lies in northwestern Idaho, east of Spokane, Washington (Figure 2-1). The region remained relatively isolated and pristine until late 1883 when the Northern Pacific Railroad, in an effort to stimulate passengers to ride its newly opened branch looping north of Coeur d'Alene, published a brochure entitled "In the Gold Fields of the Coeur d'Alenes." Two decades earlier, Captain John Mullan had spent 4 years opening up the valley by constructing a military wagon road "through swamps, over hundreds of ridges, and bridging many streams" from Fort Benton, Montana, to the shore of Lake Coeur d'Alene (Rabe and Flaherty 1974, p. 12). This route, however, was too difficult and the winters too severe for it to attract the railroads that were opening the West, and few settlers followed the track, which was becoming overgrown. However, A.J. Prichard's discovery of gold in a creek feeding the North Fork of the Coeur d'Alene River in the fall of 1883, broadcast to the world by the Northern Pacific, drastically changed all that. Within a few months, an estimated 5,000 prospectors and others looking for a quick buck had streamed into the valley (Hart and Nelson 1984).

Until then, the few thousand residents of the area, most of whom were members of the Coeur d'Alene tribe living along the shore of Lake Coeur d'Alene, were able to enjoy the natural riches that this area provided. The river was described as "transparent as cut glass," the mountains "clothed in evergreen forests" of white pine, grand fir, douglas fir, and spruce; the riparian areas thick "with the cottonwoods and silver beeches on both

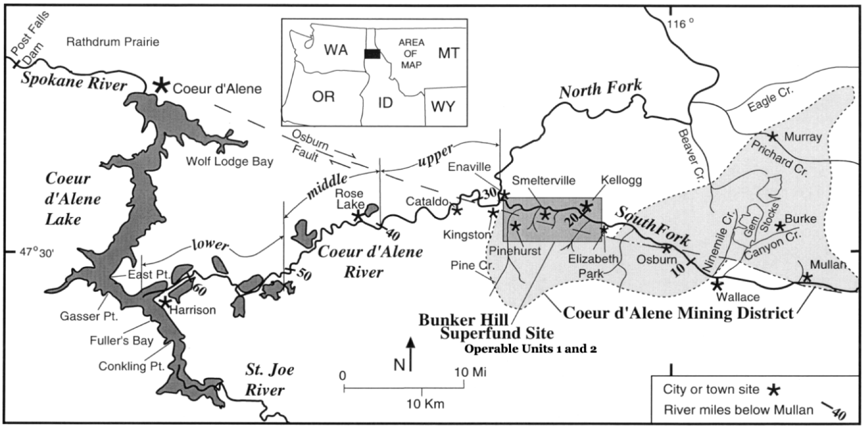

FIGURE 2-1 Location of Coeur d'Alene River basin. SOURCE: Adapted from Bookstrom et al. 2001.

banks almost forming an arch overhead" of the deep channel; and the stream "alive with trout and other fish" that "could be seen by the thousands in the clear water" (Rabe and Flaherty 1974). Deer, beaver, muskrat, otter, mink, wolves, weasels, mountain lions, badgers, wolverines, bear, and moose, along with numerous species of birds and vast schools of "salmon-trout," were abundant. Father Nicholas Point, who ran the Coeur d'Alene Mission, claimed that "Perhaps nowhere else does so small an area contain such a variety [of wildlife]" and described the tribal members filling their canoes with fish in a couple of hours of fishing, and 100 braves returning from a hunt with 600 deer (Rabe and Flaherty 1974). Even at the beginning of the mining era, one prospector could boast of having caught 247 trout in one day's fishing in Placer Creek, a tributary of the South Fork (Rabe and Flaherty 1974, p. 46).

The gold rush was relatively short lived, for much of the gold was buried under 25 feet of gravel or embedded in quartz seams in the bedrock. In either case, the gold was inaccessible to individual prospectors using hand labor and simple placer mining techniques, and many left. Those who stayed used more capital-intensive techniques and continued extracting gold from the North Fork basin for half a century (Hart and Nelson 1984).

THE EARLY YEARS

The gold, however, is not what made the Coeur d'Alene region one of the richest mining areas in the world. That resulted from the discovery of rich silver-lead-zinc–bearing ores along the tributaries and main stem of the

South Fork of the river. The first lead-silver mine in the district was the Tiger, discovered in May 1884 near what would become Burke, Idaho. By the end of 1885, 3,000 tons of ore had been extracted from this mine (Quivik 2004, p. 87). This discovery was followed within a few months by the discovery of many of the richest and most productive mines in the district, including the Morning, Gold Hunter, Poorman, Sullivan, and many others (Cook 1961). The biggest mine of all, the Bunker Hill (named after the Revolutionary War battle), was discovered by Noah Kellogg in the fall of 1885. By 1891, 26 of the 40 developed properties along the South Fork were productive (Rabe and Flaherty 1974). The silver that attracted the miners gave the South Fork the name "the Silver Valley," but the ores were also rich in lead and zinc along with lesser amounts of other metals.

Getting the Metals Out

Placer mining, however, was not an option for extracting the metals along the South Fork. The ores were contained in veins that ran through the bedrock of the mountains through which the South Fork and its tributaries flowed. The miners had to tunnel into the mountains following the veins. This was arduous and dangerous work. The tunnels were formed by drilling or "jacking" holes into the rock by hand and then blasting out the rock. The tunnels would be cut under the veins with angled tunnels, called stopes, cut up into the vein. The ore blasted from the stopes would fall into carts placed in the tunnel below, where it could be hauled out of the mine. During the first couple of years, after being sorted by hand, the raw ore was hauled by pack train out of the valley for shipment to processing facilities.

Within a couple of years, however, the Bunker Hill and other mines were building mills to concentrate the ore, separating the metal-rich materials from those that were less valuable. The first concentrators, called jigs, used a process that involved crushing the ore in stamp mills until it was primarily the size of coarse sand. The crushed ore was mixed with water and run over a "jig-table" or through a "jig cell" that allowed the heavier particles, containing the higher concentration of metals, to collect in grooves cut across the bottom of the table while the lighter particles, containing less metal, were carried over the tail of the jig to become "jig tailings."

The jigging process was relatively inefficient, recovering less than 75% of the metals (Bennett 1994). As a result, the jig tailings and slimes (the mud resulting from the water mixing with the finely powdered rock), which were often disposed of by being dumped into or adjacent to streams, contained relatively high percentages of lead and other metals. The rich ore recovered from the jig was shipped to out-of-state smelters to be converted into ingots of silver and lead. Construction of a narrow-gage railroad in Idaho between Kellogg and Cataldo in 1887 eased the shipping process, but it still involved hauling ore from the mills in the region to a loading area in Kellogg. At Cataldo, the ore was loaded on steamships to be hauled to Coeur d'Alene where it was transferred to the Northern Pacific for transit to a smelter (in Montana or Washington). The narrow-gage railroad, which was associated with the Northern Pacific, was superseded by a standard-gage railroad built in 1888 by the Union Pacific that ran from Tekoa, Washington, up to Wallace (Hart and Nelson 1984). Two years later, the Northern Pacific built its line into the Coeur d'Alene Valley from Missoula, Montana, which traveled over a famous S-shaped bridge that was completed in 1890.

The process of developing underground mines, building ore processing facilities, and constructing railroads required large amounts of capital and organization, and was not one to be undertaken by individual prospectors. Eastern and western capital flooded into the region, generating a conflict between the miners and the mine owners that colored much of the region's history through the early 1900s.

The Miners and Their Settlements

Because transportation was so difficult and the miners worked underground in 10-hour shifts, the miners initially tended to live as close to the mines as they could. Thousands of them lived in shacks and rooming houses crowded in communities such as Burke, Gem, Mace, Mullan, and Wardner jammed in the narrow valleys near the entrances to the mines (see Box 2-1). These mining towns, like mining towns throughout the West, contained many more saloons and bordellos than churches (see Magnuson 1968). Many of the early settlements were abandoned "when the ore ran out or the towns were bypassed by transportation" (Hart and Nelson 1984).

One town that stayed was Wallace. Wallace was located not at a mine mouth but on a cedar swamp near the conflux of Canyon Creek and the South Fork, on the banks of which were the sites of numerous mining operations. Colonel W.R. Wallace built a log cabin there in 1884 and set about building a town (which he initially called Placer Center) that, he predicted, would become the "center of one of the richest mining sections of this continent." Indeed the town did prosper and become the commercial center for the upper basin. Colonel Wallace, however, was less fortunate. The scrip he used to acquire the land turned out to be worthless, and, one day in February 1889, all of his land was claimed by other residents. Although the town was well located for commercial purposes, it suffered from severe flooding and several fires during its first few decades.

Laboring in the mines was tough and dangerous and the mine workers soon demanded better pay and better working conditions. By 1891, they had secretly organized unions in all the major mines in the district. They

BOX 2-1 The Town of Burke, Idaho

The canyon that held Burke is so deep that the sun could reach the town only for 3 hours a day in the winter. It is so narrow that the town's only street had to carry wagons, two railroads, and Canyon Creek when it overflowed its banks. S.D. Lemeux pulled the awnings on his grocery back to allow the daily freight through on the Northern Pacific tracks that ran down the middle of the street and straight through the center of the Tiger Hotel. The four-story hotel, originally built as the boarding house for the Tiger-Poorman mines, had 150 rooms and a "beanery" that served 1,200 meals a day. It burned down in a grease fire in 1896 but was rebuilt. The railroad tracks were built through the hotel in 1906, when Harry Day of the Hercules mine convinced the Northern Pacific to construct a spur track up to his loading platform below Gorge Gulch. The hotel covered the canyon floor that the railroad had to be built on. The Federal Mining and Smelting Company, which owned the Tiger-Poorman and its hotel, agreed to Day's request providing that "the portion of the hotel under which you pass is to be lined with sheet or corrugated iron as fire protection."

Source: Hart and Nelson 1984.

petitioned for better health care, safer working conditions, and a daily wage of $3.50 (Hart and Nelson 1984, p. 50). The Bunker Hill Mine resisted and organized the mine owners into the Mine Owners Protective Association to fight the unions.[1] The Coeur d'Alene mining wars, which continued over the next decade, involved armed fights, assassinations, lockouts, the dynamiting of mine properties, the imposition of martial law, the use of federal troops to suppress the "insurrection," and the internment of hundreds of miners in squalid concentration camps. The miners were a tough lot (see Box 2-2) and their unions were at the peak of their power in early 1899. Within 6 months, however, the unions were broken and the federal troops required every miner to obtain a work permit before working again in the mines. They could obtain a permit only after "swearing to an anti-union pledge." During the ensuing year, 2,000 miners worked under this system, only 130 of whom had previously worked in the Coeur d'Alene district and only 99 of whom had ever been a union member (Hart and Nelson 1984). These Coeur d'Alene mining wars form an important chapter in the history of American labor movements.

[1] Another purpose of this association was to fight against the high tariffs that the railroads charged for hauling ore out of the valley.

BOX 2-2 The Coeur d'Alene Miner

Mining has always been hot, rough, dirty, wet, and often dangerous work. At the turn of the century, it was physically exhausting labor done in dark, narrow passageways with a short supply of air, a great deal of dust, and few exits to the surface. The conditions, and especially the dust, limited the number of productive working years of the miner in the mines and reduced his lifespan if he survived underground. The miners were paid between $3.00 and $3.50 a day, working thirteen ten-hour shifts every two weeks, with the shift starting when they arrived at the work place inside the mine and with a day off on alternate Sundays.

The miners in the Coeur d'Alene region were a mixed bag of nationalities, representing the last remnants of the restless, independent men who roamed the frontier and the first generation of European immigrants searching for jobs in their new land. Only one-quarter of them were native-born Americans; the others were predominantly British, Italian, and Scandinavian. All foreign nationalities were represented except Orientals, who were banned from the district by the miners who feared the competition of their cheap labor.

Regardless of background, all who worked as hard rock miners had the same 10-hour work day, day after day, with a Sunday off every other week. Their nonworking life was not much more flexible. They woke at 5:30 AM to get dressed, eat breakfast, and have time to get to their stopes in the mines by 7:30 AM to begin work. After working ten hours, traveling back and forth to the portal and on to their jobs inside the mines for three or four hours, sleeping eight hours, and eating for another one or two hours, the miners had little or no time left for recreation, family, or community activities.

Source: Hart and Nelson 1984.

Environmental Impacts

When mining and mineral processing began in the Coeur d'Alene mining district, environmental protection was not a concern. The mine operators relied on the ability of the Coeur d'Alene system to get rid of mine wastes, most of which were dumped into the Coeur d'Alene River or its tributaries without restriction until well into the next century. Mills located on hillsides deposited their tailings in gullies so that gravity and surface-water drainage could move them down to the floodplains while winds winnowed the fine-grained particles and spread them over adjacent slopes and flat areas. Tailings from mills located in the floodplains were dumped near the mills or directly into the South Fork of the Coeur d'Alene River (Long 1998).

The rapid growth of the mining industry was accompanied by extensive logging to provide timbers to support the roofs of the mining tunnels, to construct railroads, to provide fuel, and to build the towns and mill facilities that were springing up throughout the basin. The logging resulted in deforestation that increased the rate of runoff from the hills, and this, combined with the large amount of tailings that clogged the channels, raised stream levels so that overbank flooding occurred each year and drove flood water to higher and higher levels (Box et al. 1999).

Major spring floods followed in 1893 and 1894. By 1903, tailings covered the broad floodplains at Woodland Park, Osburn, and Smelterville Flats. These deposits and the frequent floods caused a number of channel changes where the South Fork runs through the flats (Box et al. 1999) (see Figure 2-2).

By 1900, the results of dumping the waste tailings in the river were being observed in the agricultural areas in the lower basin. Residents complained that the tailings made the water and sediment toxic to livestock and vegetation. They called the animals poisoned by these materials "leaded horses, leaded cows, leaded dogs, leaded chickens, or leaded fish" (quoted by Casner 1991). One resident described in her diary how the "family cat would go into 'fits' after drinking 'the bad water'" (Casner 1991). By 1900, mill tailings had reached Lake Coeur d'Alene and had affected as much as 25,000 acres along the South Fork and main stem of the Coeur d'Alene River (Long 1998).

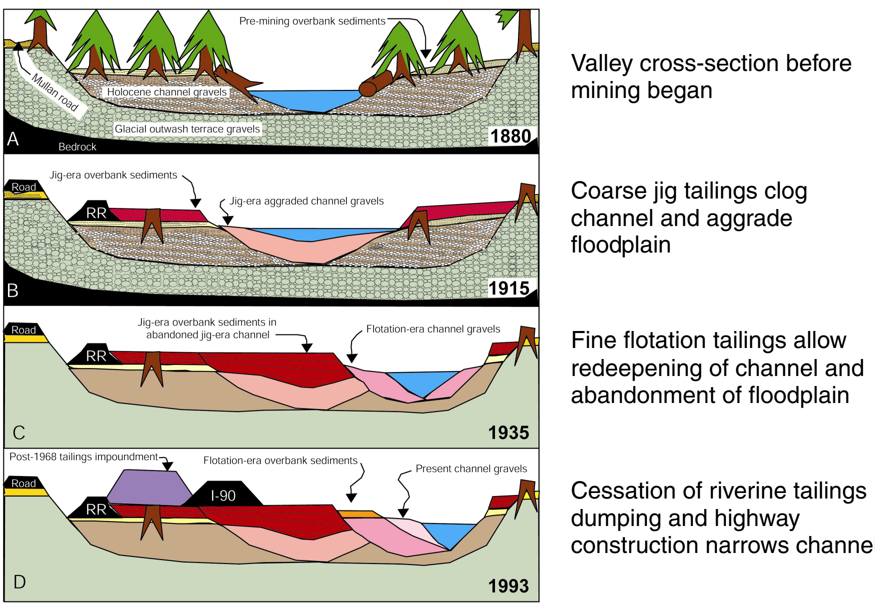

FIGURE 2-2 Changes in the channel of the Coeur d'Alene River at Cataldo Flats, 1880–1995. SOURCE: Box 2004.

Beginning in 1901, the mining companies installed pile and plank dams to reduce the amount of suspended load carried down the Coeur d'Alene River. Although the increasing complaints from downstream landowners were probably a major stimulus for this action, the mine owners also realized that the trapped tailings would contain substantial amounts of metal that might be reclaimed. The dams were located at Woodland Park on Canyon Creek and at Osburn and the Pinehurst Narrows on the South Fork of the Coeur d'Alene River. The Osburn dam created a reservoir that covered approximately 300 acres (Casner 1991).

In spite of these efforts, several downstream farmers filed court suits against the mining companies. The complainants claimed that mine wastes being deposited on their lands by the river were killing crops, hay, and other vegetation and that horses, and to a lesser extent cattle, dogs, and chicken, were being poisoned by residues deposited on grass and along the shore of the river after the floods. They also claimed that, when deposited on land, the material brought down by the river was made more toxic by reacting with air and that the resulting substance produced speedy death if ingested by horses (Ellis 1940). These were the first in a series of lawsuits what would become a protracted effort to get the mining companies to stop discharging mine wastes into the river system. The farmers' problems undoubtedly were exacerbated by the damming of the Spokane River at Post Falls in 1906, which raised the level of Lake Coeur d'Alene, flooding the lower reaches of the Coeur d'Alene River and, as a result, increasing the rate of deposition and causing the river to flood over its banks and deposit tailings on the surrounding lands more frequently.

The Mine Owners Association (MOA) "successfully defended the preferential status of miners' water rights in organized mining districts, claiming that the waste was harmless, and offered the economic importance of mining as a justification for their dumping policies" (Casner 1991). To avoid further court suits, the MOA began buying "pollution easements" on lands along the lower Coeur d'Alene River valley and "overflow easements" on the floodplains from Kellogg to Lake Coeur d'Alene (Grant 1952). These easements released "the mines from all past and future pollution claims" resulting from any possible damage to crops or domestic animals that mining operations might cause.

THE MIDDLE YEARS

During the first half of the 20th century, life in the Silver Valley settled down. Union problems dissipated, working conditions improved somewhat, and improved transportation allowed miners—and their families—to live in homes located in more stable communities on the flats. In 1910, a major wildfire ripped through the region destroying forests and towns alike (Hart and Nelson 1984; Pyne 2002). However, because the economy was booming, most towns quickly rebuilt, often improving over the former layout, and there was apparently little impact on mining operations. The denuded hillsides likely did increase the severity of floods, but this was already a common problem in the basin. The population in the valley increased (Figure 2-3), although not as much as mining output (Figure 2-4). Much of the increased output resulted from improvements in mining and ore-processing technologies rather than from the employment of more workers.

Improvements in Technology

Advances in mining and ore-processing technologies introduced after the turn of the century allowed the Coeur d'Alene area mines to substantially increase their production of metals. A dry pneumatic drill, the Wiggle-Tail, had largely replaced hand jacking for drilling blasting holes. These machines increased the productivity of the miners but did not improve mining conditions. They were frequently termed "widow makers" because

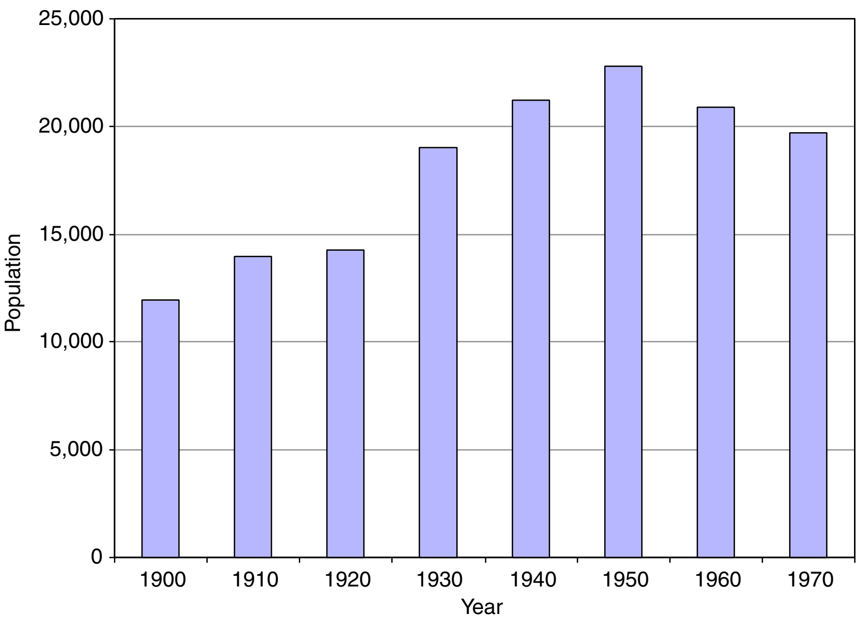

FIGURE 2-3 Population of Shoshone County, Idaho: 1900-1970. SOURCE: Forstall 1995.

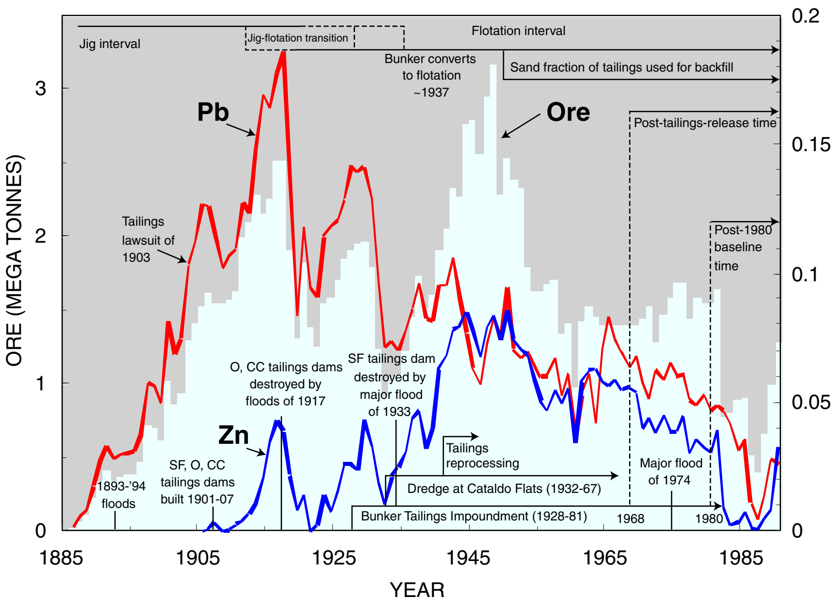

FIGURE 2-4 Annual production, Coeur d'Alene mining district, 1885-1990. (1 megatonne equals approximately 1.1 million tons.) SOURCE: Bookstrom et al. 2004; Box 2004.

in addition to creating large amounts of dust which could cause silicosis (a potentially fatal condition of the lungs), they had a tendency to loosen the rock in the tunnel and stope ceilings while in operation (Hart and Nelson 1984). In 1918, an improved pneumatic drill was introduced that was more stable and had a water line as well as a compressed air line (Hart and Nelson 1984). The water, forced through a hollow drill bit, cleaned out the blasting hole as it was being drilled and suppressed the dust. The larger supply of compressed air helped ventilate the workings. These new drills both increased productivity and improved safety and working conditions for the miners.

With this new equipment and better ventilation, the miners were able to tunnel farther and deeper. The massive Bunker Hill Mine, for instance, has about 150 miles of mining tunnels ranging from 3,600 feet above to about 1,600 feet below sea level (about 1 mile deep) (University of Idaho 2005).

Another major technological advance was the introduction of a new method of concentrating the ore. The Wilfley table (invented in 1903) adopted at some mills to supplement the jigs, increased recovery rates for lead and silver to more than 80% (Bennett 1994). An even more efficient and selective "flotation" process, which could recover additional metals, was introduced to the Coeur d'Alene mines, and by 1930 ores were being concentrated by this method exclusively (Long 2001). This process involves grinding the ore very finely and blowing air through a mixture of this finely ground ore and water mixed with a frothing agent (usually pine oil or cresylic acid) and a collection agent. The froth attracted the sulfide-bound minerals and this metals-rich froth was collected for further processing (Bennett 1994). The process was much more efficient than the jig-tables in removing metals, reaching extraction efficiencies of around 85% by the 1930s and 95% by the late 1950s (Bookstrom et al. 2004). The more efficient recovery also made it economical to process lower-grade ores.

The tailings from the flotation process were quite different from the jig tailings. They contained much lower concentrations of metals but, being much finer, were more mobile. These frothing "slimes" could not be stockpiled and the river easily carried them over the plank dams. Consequently, they were transported for longer distances downstream (Long 1998, pp. 90-91). When left to dry on the floodplains by receding flood waters, they were also easily picked up and transported by winds.

Because ores of lower grade could be handled profitably by the flotation process, the amount of rock flour that was added to the mine runoff was significantly increased over that of the jig system, which relied on relatively high-grade ores. Besides the frothing and collection agents, the flotation process also used various other reagents such as sodium carbonate, copper sulfate, zinc sulfate, and potassium dichromate (Fahrenwald 1927).

Another change in ore processing in the valley involved the Bunker Hill Mine's construction of a smelter in 1917. This smelter began with three blast furnaces, four roasters, a lead refinery, and a silver refinery. With a capacity of only 1,000 tons of ore per day, the facility produced mostly lead and silver from concentrates produced at the Bunker Hill Mill located about a mile to the east. The smelter continued to expand and by 1936 was the largest lead-producing facility in the world (Bennett 1982, p. 19).

Because the flotation process recovered zinc and other metals in addition to the silver and lead that were collected from the jig tables, facilities were also built to process these metals. An electrolytic zinc plant was constructed by Sullivan Mining Company at Government Gulch near Kellogg in 1928, and it was the first facility in the United States to produce zinc with 99.99% purity in commercial quantities (Murray 1982, p. 6). In 1943, a zinc fuming plant was added to facilitate the recovery of zinc from smelter slags. A cadmium plant was annexed to the smelter at the Bunker Hill Mine in 1945, and high-grade cadmium began to be recovered from smelter by-products.

All these advances allowed the valley to increase metal production substantially (see Figure 2-4). During their periods of production, the mines processed an estimated 130 million metric tons[2] of ore and produced about 7 million metric tons of lead, 3 million metric tons of zinc, and 30,000 metric tons of silver, approximately 17%, 6%, and 18% of the nation's production of these metals, respectively (Long 1998). Ore production peaked around World War I at approximately 2.5 million metric tons per year and again peaked in 1948 at 3.2 million metric tons per year (see Figure 2-4) (Bookstrom et al. 2004, Figure 7a).

Waste Management

As production increased, the tailings became more of a problem. The Page and Bunker Hill Mines built the first tailings impoundments in 1904, but these were small and captured only the coarser materials (Casner 1991; Bennett 1994). The processing of lower-grade ores also resulted in substantially increased waste tailings.

The more efficient concentration technologies also supported the recovery of metal from some of the earlier wastes. The reprocessing of tailings began as early as 1905, and the tailings impoundments behind the dams at Canyon Creek and Pine Creek began to be reprocessed around 1919, although the presence of sewage, garbage, and other contaminants created problems (Long 2001, p. 89).

Although the tailings entrapped behind the plank dams were reprocessed, the dams were not maintained. Major floods in the spring of 1917 destroyed the Osburn and Canyon Creek dams, and the dam at Pinehurst was breached by floodwaters in 1933 (Long 1998, p. 8). Figure 2-5 shows the breached dam and substantial tailings behind it at Osborn in 1920. There was little reason to replace the dams after they were breached, because the impoundments were already full of sediment—they would not be effective in capturing the flotation tailings even if they had room. Also, they had not been successful in eliminating the court suits by farmers whose land was being contaminated downstream (Casner 1991). These cases continued up until 1930, although the mining companies were generally successful in defending their rights (Casner 1991).

During the 1920s, some mines began to use tailings ponds in an attempt to control the increasing waste problem. The flotation tailings were discharged into these ponds where they were allowed to settle before the water was discharged to the river. By 1923, wastes from selective flotation at Page Mill were being discharged into a tailings pond constructed within a swampy area on the western side of the Smelterville Flats known as Page pond (MFG 1992, pp. 1-26). Between 1926 and 1928, the Bunker Hill

[2] 1 metric ton equals approximately 1.1 U.S. tons.

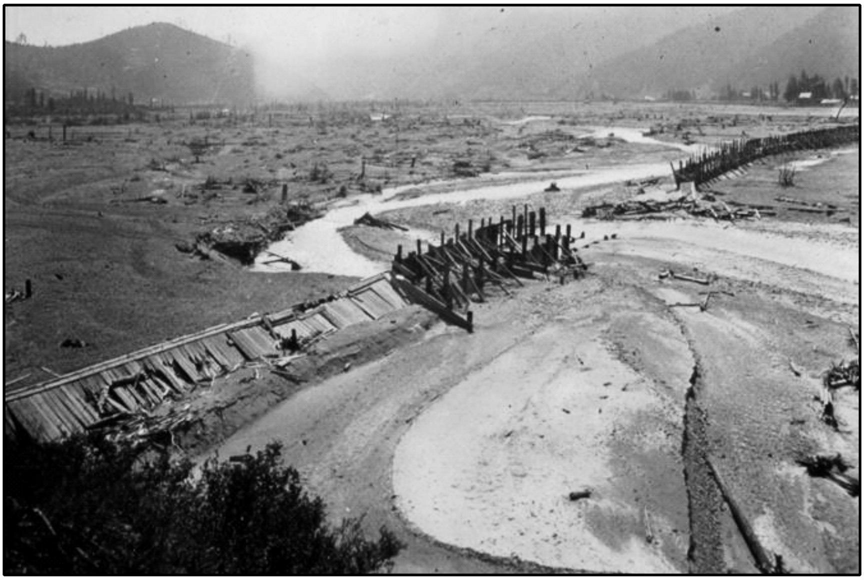

FIGURE 2-5 Tailings Dam at Osburn, Idaho, 1920. SOURCE: Richard 1921, as cited in Bennett 1994.

Company built a larger tailings pond west of Kellogg that expanded over the years to become the central impoundment area, which received most of the flotation wastes discharged since 1928 (Casner 1991; Long 1998).

In 1932, the MOA, in response to substantial concern being raised by residents in the city of Coeur d'Alene and other downstream areas, and to preclude possible government restrictions on the discharge of tailings into the river, constructed a suction dredge near Cataldo to remove tailings from the river (Grant 1952). At Cataldo, the river system converts from a high- to a low-gradient system, and solids settle out in this natural depositional area. The suction dredge pumped about 7,000 gallons of water a minute, excavating an estimated 500 tons of sediment per hour at 5% sediment load and ran approximately 22.5 hours per day from June through December. Over the life of the dredge, it removed an estimated 34.5 million U.S. tons of tailings, which were deposited in a tailings pond on Mission Flats (URS Greiner, Inc. and CH2M Hill 2001, p. 2-7). This pond ultimately covered an area of about 2,000 acres to a depth of 25-30 feet (Casner 1991). The dredge operated during the summers from 1932 to 1968 (Long 2001). Although it removed substantial amounts of tailings from the river, apparently no effort was made to determine how much it actually reduced the deposition of tailings on the lands downstream.

BOX 2-3 Remembrance of the 1930s

"We never saw blue sky when I was there in the 1930s," a former resident recalled a few years ago. "We never saw the sun. Right after we moved there, I put my baby daughter on the porch one morning. A neighbor came running over and said, 'Don't you know any better? You can't put a new baby out on the porch in the morning! It's real bad of a morning here!' I remember another night my daughter had been very ill; we didn't know what it was. She was just gasping for breath. The next morning, the clothing that had been hanging on the clothesline all night went to pieces as I got ready to iron it. We wore rayon in those days. It was the sulfur dioxide that had destroyed the fibers."

Source: Tate 1981.

Tailings were not the only wastes of concern. As the mines were excavated into the mountains, groundwater migrating downward through permeable rock fissures was encountered. When groundwater enters the mine tunnels, chemical reactions can occur that greatly hasten the degradation of the sulfide minerals and result in acidic waters with high dissolved metals concentrations. Such waters are called "acid rock drainage." The Bunker Hill Mine had the most serious problem.

The Bunker Hill smelter also emitted substantial amounts of sulfur dioxide and other air pollutants that were discharged directly to the atmosphere. Years later, valley residents still had vivid memories of this smoke (see Box 2-3). In an attempt to counter these problems, the Bunker Hill Company built a "solarium" with ultraviolet lights that workers and children living in the valley could use to obtain doses of substitute sunlight (Tate 1981).

The company also recognized that these pollutants were likely to cause environmental problems and responded in the same way that the mine owners had responded to the farmers. It bought "smoke easements" for the lands likely to be affected by its emissions. By 1940, these smoke easements covered more than 7,000 acres of private land (Casner 1991). The deposition of pollutants emitted from the smelter caused the death of trees in the area and contaminated the soil such that little vegetation could grow there. Even as late as the 1980s and 1990s, extensive efforts undertaken by the company and the government to replant seedlings to reestablish the forest and control erosion off these slopes were unsuccessful (Tate 1981; EPA 2000).

Increased Community Concern

Because the mining companies were, as discussed above, so successful in defending themselves against the farmers' court suits, downstream residents began to seek redress through the political system. The residents of Coeur d'Alene City echoed their concerns as the flotation tailings began to reach the city in the mid-1920s (Casner 1991). In 1929 and 1930, John Coe, editor of the *Coeur d'Alene Press*, published a series of dramatic articles detailing the history and dimensions of the pollution problem. Casner (1991) indicated that John Coe and three politicians representing the lower-valley residents had toured the river and observed (and had become stuck in) the "yellow muck," smelled the "stifling stench," and saw "a picture of desolation . . . a veritable 'Valley of Death' . . . in a 'Paradise Lost' . . . created by the 'sublime indifference of the octopus of heartless wealth'" (Casner 1991). The paper followed up on this series by lobbying for action by the state legislature and showing that Canadian mines were operating profitably even though that country prohibited the dumping of wastes into streams.

According to Casner (1991), the mining companies responded by sponsoring their own studies that identified little or no problem, stimulating articles in local newspapers that attacked the downstream politicians for threatening the existence of the mining industry and opposing any government action in testimony before the state legislature and Congress. Nevertheless, in March 1931, the state legislature established and provided emergency funding for a "Coeur d'Alene River and Lake Commission" to investigate the issue and report back to the legislature in 1933. The commission requested the assistance of federal experts, writing "Our river is gone, for the time at least, but we would really like to save our lake. Will you help?" (Casner 1991).

Although studies undertaken for the commission by the U.S. Bureau of Mines generally supported the position of the mine owners, other studies by the U.S. Biological Survey, Bureau of Fisheries, and the Public Health Service did not. Dr. M. M. Ellis of the Bureau of Fisheries authored one of the best known of these studies. He investigated the effects of mine wastes on fisheries and other aquatic organisms in the region in 1932. He found that

> The polluted portion of the Coeur d'Alene River, that is the South Fork from a short distance above Wallace, Idaho to its junction with the North Fork above Cataldo, and the main Coeur d'Alene River from the junction of the forks to its mouth near Harrison, Idaho was found (July 1932) to be practically devoid of fish fauna, bottom fauna or plankton organisms. …Thompson Lake and Swan Lake, both rather heavily polluted by recent backwaters from Coeur d'Alene River were almost without plankton fauna. The plankton fauna of Coeur d'Alene Lake as a whole was rather sparse, and particularly poor at the south end. No plankton were taken off Harrison and at the mouth of Coeur d'Alene River; and very few as far up the lake as East Point. (Ellis 1940, p. 55)

By comparison, Ellis noted that the unpolluted small lakes nearby and the tributaries to the Coeur d'Alene River between Cataldo and Harrison supported normal fish populations and abundant plankton and aquatic vegetation. In experiments, he exposed some fish and plankton species to mine slimes, mine water, mill effluents, and Coeur d'Alene River samples and showed that they were lethally toxic to all the test organisms. Native fish in cages placed in the river died within 72 hours. Ellis concluded "There is but one solution for this pollution problem as far as fisheries are concerned, namely the exclusion of all mine wastes from the Coeur d'Alene River" (Ellis 1940). Before coming to this conclusion, he had also inspected and carried out experiments at the same Canadian mine that Coe had visited and found a healthy fish population there.

The Biological Survey evaluated several birds found dead and concluded that they died of metal poisoning attributed to pollution in the river and from the smelters (Casner 1991). The problem of swan mortality had been observed in 1924 with an account of 25 swans sickening and dying in the wetlands between Medimont and Harrison (Chupp and Dalke 1964).

John Kurtz Hoskins of the U.S. Public Health Service had 296 water samples from several locations in Lake Coeur d'Alene analyzed and found average lead concentrations ranging from 0.08 to 0.22 milligrams/liter (mg/L), with the concentration generally decreasing from the mouth of the Coeur d'Alene River to Coeur d'Alene City. One of the samples at Harrison had a lead content of 2.25 mg/L and another at Coeur d'Alene showed lead at 1.75 mg/L (Hoskins 1932, as cited in Casner 1991). He concluded that, under normal conditions, the lake water was practically saturated with lead in solution and pointed out that the concentrations were above the guideline for potable water on interstate carriers, which was 0.1 mg/L at that time. The mining industry aggressively challenged the Hoskins report with results of their own investigation which found lead at only 0.027 mg/L in water samples taken from the Coeur d'Alene City pumping station (O'Keefe and Ziegler 1930, as cited in Casner 1991).

Although the commission's reports raised public awareness of the problems in the valley, the commission made only two recommendations. The first was to support the use of the dredge that the mines had already begun operating at Cataldo. The second was that a flume or pipeline be built down the length of the South Fork to carry the mining slimes to settling beds at Mission Flats.

In contrast to the frequent public statements by mine owners that their wastes created no significant public health or environmental problems, by 1930 the occupational hazards and public health risks in the production of lead and its compounds had been well known (Markowitz and Rosner 2002). The mine owners had substantial evidence that there were problems in Coeur d'Alene associated with mining. In addition to the sickened and dying animals, the death rate among miners in Idaho averaged 2.47 per thousand per year between 1903 and 1908[3] (Hart and Nelson 1984). By 1920, Bunker Hill management realized that their smelter could be causing some health risks for its employees and initiated an unproven electrolytic treatment for removing the lead from their bodies (see Box 2-4 and Figure 2-6).

Nevertheless, the depression of the 1930s and then World War II diverted attention from possible public health and environmental concerns. During the 1940s, the Idaho Fish and Game Service and the U.S. Fish and Wildlife Service became sufficiently concerned about the death of migratory waterfowl feeding in the lower basin that they tried to use flares, gunshots, and boats to keep swans and geese away from the lethal feeding grounds, but they abandoned this effort because it was unsuccessful (Chupp and Dalke 1964).

The depression initially brought depressed metal prices, leading to the closure of many mines. However, they were saved by passage of the federal Silver Purchase Act in 1934, which guaranteed that the government would buy all the silver produced by American mines at twice the existing world price (Bennett 1994). This act encouraged every mine that could produce silver to reopen. Particularly fortunate was the Sunshine Mine, which had discovered a very rich silver bearing ore in 1931. The Sunshine became the most productive silver mine in the world and by itself produced more than one-third of all the silver produced in the Silver Valley (Bennett 1994).

The advent of World War II increased the demand for metals, particularly lead. But it also created a labor shortage, with many of the miners joining the armed services. In spite of efforts by the government and the mine owners to overcome these labor problems, production from the mines never reached the levels it had during World War I and actually decreased during the war years. Instead, the mines began to reclaim some of the old tailing and waste ore stockpiles. A reprocessing mill at the old Sweeney Mill processed some 1.2 million tons of tailings, producing 24 million pounds of lead and 8.4 million pounds of zinc, along with over half a million ounces of silver. Another built at Osburn Flats processed 4.4 million tons of jig tailings to produce 54 million pounds of lead, 77 million pounds of zinc, and 2.8 million ounces of silver (Bennett 1994). In total, 12 new mills were built to remine waste piles as well as stockpiles of tailings. Long (1998, p. 2) estimated that, in total, about 6 million metric tons (6.6 million tons) of tailings have been reclaimed from creeks and dumps for reprocessing. Of course, the reprocessing also produced tailings that again were discharged into the rivers, so the overall environmental benefit was limited.

[3]Most of these deaths probably resulted from mine accidents and respiratory diseases and not from lead poisoning. This is approximately twice the national death rate for males under the age of 65 during this period (Bell and Miller 2002).

BOX 2-4 The Clague Electrolytic Treatment

The Bunker Hill management recognized that the smelting process posed a threat to the health of some of its workers. By 1920, the company had engaged in medical experiments to counteract the effects of lead poisoning. In 1921, mining historian T. A. Rickard wrote that the company made "beneficent use of electricity" by providing the "Clague electrolytic method for the treatment of lead poisoning." As many as forty smelter workers at a time took the treatment—which consisted of placing the patients' arms and legs in a salt-water solution and then passing a 110-volt current through their bodies—at the Wardner hospital. The process was intended to attract lead to the electrodes in the water.

Source: Casner 1991.

THE LATER YEARS

With the return of the miners from the war and the continued high metal prices resulting from the economic boom in the United States, combined with reduced competition from abroad, ore-processing facilities were expanded and metal production in the Coeur d'Alene region increased, reaching a peak in the mid-1960s (see Figure 2-4). The Bunker Hill Mining Company, for instance, increased its smelter capacity to 100,000 tons per

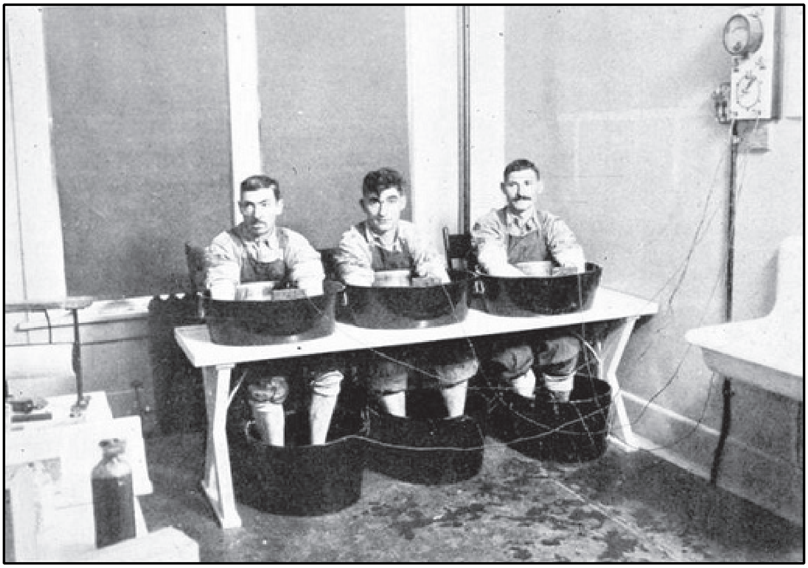

FIGURE 2-6 Workers taking the Clague electrolytic treatment in the 1920s. Photograph courtesy of Richard Magnuson, Wallace, ID.

day and added additional recovery units so that by 1972 it was recovering six different metals (Bennett 1982).

These were boom years for the valley. Another major project was the construction of Interstate 90 in the early 1960s, which was built on embankments and road beds constructed from tailings excavated from Cataldo Flats, the central impoundment area, and other locations.

But as the economy recovered, so did concerns about the public health and environmental contamination dangers resulting from mining. Not much had improved in the Silver Valley (Box 2-5). Congress passed two laws in 1948, the Water Pollution Control Act and the Mining Waste Pollution Control Act, which began to put pressure on the country's mining industry. The large mines began to address some of their pollution problems. An acid plant was added to the zinc plant in 1954 to collect sulfur dioxide from stack gases and a second one was added in 1966 (MFG 1992, p. 1-22). Bunker Hill built a new smoke stack on its smelter in 1958 (Bennett 1982). In the late 1940s, some of the mines began separating the sand-sized fractions from the other tailings and returning the coarser materials to fill abandoned workings (Long 1998).

By 1968, in response to state and federal pressure, all the mill tailings were being disposed of in settling ponds rather than being discharged directly into the river[4] (Rabe and Flaherty 1974). In that year, Bunker Hill also began diverting its contaminated adit drainage to the central impoundment area, although it was then allowed to flow into the river without treatment, and added an acid plant to the lead smelter. In 1969, Bunker Hill installed an improved "bag house" for controlling air emissions, and this along with several other improvements resulted in a 90% reduction in sulfur dioxide emissions (Bennett 1982, p. 21). The company also built a wastewater treatment plant to treat acid mine drainage in 1974.

Passage of the Clean Air Act in 1970 and the Federal Water Pollution Control Act in 1972 substantially increased the environmental pressures. But public attention was particularly aroused in September 1973 when the primary pollution-control device at the Bunker Hill smelter, the bag house, was partially destroyed in a fire. The new owners of the facility, Gulf Resources, decided that they would continue to operate the facility without this pollution control. This continued until August 1974.[5] During this time

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[4]In some cases, these settling ponds, built without liners and often on top of old tailings deposits, may have increased the flow of dissolved metals into the river while reducing the amount of suspended sediment (Rabe and Flaherty 1974).

[5]Company records made public in subsequent court proceedings indicated that this was a very cynical decision based solely on economic considerations. The company was generating substantial profits as a result of high metal prices, and it estimated that, based on the results of a court case in Texas, it would probably not have to pay more than $7 million to settle any lead poisoning lawsuits resulting from its actions (Bennett 1994).

BOX 2-5 Living in the Valley

"Pam Nichols, an amiable florist who's spent most of her 33 years [in the Valley], remembers that when she was a child her blond hair would sometimes turn green because of all the sulfur in the air. Others recall that, for days on end, there would be blue skies and sunshine on the hills above town and haze so thick in Kellogg you had to drive with your lights on. The South Fork was as white as lye with industrial and municipal wastes. 'Lead Creek,' it was called, and children were warned to stay away from it. Dogs that drank out of puddles after a rain sometimes died. You couldn't keep a lawn or raise a garden."

Source: Tate 1981.

period, the smelters main stack emitted up to 160 tons per month of particulate emissions containing 50-70% lead compared to 10-20 tons per month prior to the fire (TerraGraphics 1990). Average monthly emissions at this time contained 73 tons of lead (ATSDR 2000), and ambient air concentrations of lead measured as high as 30 micrograms per cubic meter ($\mu g/m^3$) (IDHW 1986).

After noting increasing levels of lead in ambient air in Kellogg, Idaho, the Idaho Department of Health and Welfare quickly initiated a public health investigation. This study (IDHW 1976) showed that in Smelterville, adjacent to the smelter, 99% of the children tested had blood lead levels (BLLs) greater than or equal to 40 µg per deciliter (dL) (the Centers for Disease Control and Prevention [CDC] BLL of concern in 1974). Overall, about 46% of the 919 children aged 1-9 years who were tested had BLLs greater than or equal to 40 µg/dL (IDHW 1976). Although these were some of the highest BLLs ever recorded, many of the basin residents remained unconcerned (see Box 2-6).

In responding to these increased pressures, Bunker Hill spent more than $21 million upgrading its wastewater treatment plant, installing hoods over its blast furnaces and scrubbers on the sintering plant, and building two tall smoke stacks (715 and 610 feet high) to further disperse its emissions and thereby decrease ambient air concentrations of lead and other contaminants in the valley (Bennett 1994). At the same time, metal prices began to fall, government price supports had disappeared, and Bunker Hill was facing increased competition from newer, more efficient smelters (Bennett 1994). As a result, the smelter was shut down in 1981 with a loss of 2,100 jobs—approximately three quarters of the total mining employment in the district at the time (Bennett 1994, 2004).

By 1983, when a second large human health study was conducted, the proportion of children living closest to the smelter site with BLLs of 30 µg/

BOX 2-6 "I Don't Like People Poking at My Kids."

"There's nothing wrong with my kids," one mother told a journalist in the early 1980s. She, her husband, and their two children lived in a small, tidy house on the main street of Smelterville—a community with some of the highest concentrations of lead found in the Kellogg area. Her children, ages nine and 13, both had lead levels higher than 70 micrograms when tested during the CDC survey. She refused to have them participate in any of the numerous follow-up surveys and declined several offers to have them tested for neurologic or psychologic abnormalities. "I don't see any need for it," she says. "I don't like all these people poking at my kids, sticking their noses in where they don't belong." She pauses. "I don't know. Maybe there is more wrong than I realize, but I don't think so."

Many other residents agreed. Although the company had bought and demolished all the residences within one-half a mile of the smelter, the citizens of Smelterville protested the proposed closing of the Silver King Elementary School which was also located within this area, even though monitors at the school showed lead levels in the atmosphere 10 times higher than the ambient air standard. There wasn't enough evidence showing the high lead levels would harm their children they argued, and when the question was put to a vote, 996 of the 1,127 ballots cast were in favor of keeping the school open.

Source: Tate 1981.

dL or greater declined from 99% in 1974 to 19% (IDHW 1986, Table 81). Since this time, the area around the former smelter has seen declining BLLs, and by 2003 only 2% of children had BLLs greater than 10 μg/dL.

SUPERFUND

The final blow to the district's mining industry was passage of the Superfund legislation (more formally entitled the Comprehensive Environmental Response, Compensation, and Liability Act) in 1980. Although much of the impetus for the law came from a desire to clean up industrial hazardous waste sites in the East, the Bunker Hill Mining and Metallurgical Complex was quickly (1983) placed on the National Priorities List for cleanup.

The site, commonly referred to as the box, encompasses a rectangle, 3 miles wide and 7 miles long, running from the vicinity of Kellogg on its eastern end to Pinehurst on its western end. This was the area most seriously affected by airborne pollution from the Bunker Hill smelter (Long 2001). The U.S. Environmental Protection Agency (EPA) did not begin cleanup actions until 1986 when they instigated a "fast-track" cleanup targeting public areas, such as parks and playgrounds. In 1991, a record of decision (ROD) covering the populated portions of the area (designated as operable unit [OU] 1) was issued; in 1992, an ROD was produced covering the nonpopulated areas (designated as OU-2).[6]

During the same time period, the state of Idaho sued the existing mining companies for $50 million in damages in a natural resources damage (NRD) lawsuit. This suit was settled for $4.5 million, which went into a trust fund to finance cleanup efforts (Long 1998). In 1991, the Coeur d'Alene tribe filed another NRD lawsuit against eight mining companies. One company, the Coeur d'Alene Mines Corporation, settled with the tribe. In 1996, the United States joined the Coeur d'Alene tribe in this suit. At the time of writing, this case is ongoing.

EPA officials said that they intended to address the environmental problems that existed outside of the box using programs other than Superfund. However, they found their other tools to be inadequate, and, in 1998, the agency announced that it was initiating the Superfund process for contaminated areas within the 1,500 square mile Coeur d'Alene River basin reaching from Montana to Spokane, Washington—one of the largest Superfund designations in the country—to be designated as OU-3 of the Bunker Hill Superfund site (Villa 2003).

The economic conditions and environmental pressures that had forced the closure of Bunker Hill, the largest facility in the valley, affected many other mines as well. During the 1980s, the population of the valley's communities fell by a quarter, incomes tumbled, and poverty rates soared. New owners attempted to reopen Bunker Hill but declared bankruptcy in 1991 (Bennett 1994). A few mines remained in operation, but the Silver Valley would never be the same again.

During its history, the Silver Valley could claim a number of achievements (Bennett 2004). It was the largest and richest silver-producing region in the world, producing more than 1 billion ounces, with the Sunshine Mine being the richest silver mine ever developed. Bunker Hill was the largest lead and zinc mine in the United States, but was only 1 of 18 mines in the district that produced more than a million tons. As indicated above, the valley accounted for 18% of all the silver that has been produced by U.S. mines, 17% of all the lead and 6% of all the zinc (Long 1998). More than 100 mines have operated in the district, including some of the deepest and largest in the country. The total value of the metals produced by valley mines exceeded $26 billion in 1997 dollars (Long 1998). But the legacy of this history is also immense—environmental problems spread over hundreds of square miles creating one of the largest and most expensive cleanup challenges in the nation, a challenge that is likely to take longer to overcome than it did to create.

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[6]For a useful chronology of mining and Superfund related events, including remedial activities, at the Bunker Hill Superfund site, see Figure 1 in EPA 2000.

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3

The Coeur d'Alene System

OVERVIEW

The Coeur d'Alene River basin is a large complicated system with tremendous topographic, hydrologic, and biological variability. This chapter summarizes the components of the Coeur d'Alene system that the committee considers most important in understanding the system and evaluating the likely effectiveness of proposals for the basin's cleanup. The information presented here forms the basis for the analyses contained in the subsequent chapters.

The area covered by the proposed cleanup efforts being reviewed includes the Coeur d'Alene River basin (outside of the Bunker Hill box), Lake Coeur d'Alene, and the upper reaches of the Spokane River, which drains Lake Coeur d'Alene (see Figure 3-1). The total length of this system is 166 miles (267 kilometers [km]), and the study boundary includes an area of approximately 1,500 square miles (almost 4,000 km^2) (URS Greiner, Inc. and CH2M Hill 2001a, p. 4-9). The final project area, however, is much smaller, including only the contaminated portions of the basin, lake, and Spokane River.

Socioeconomic Considerations

Historically, the growth and vitality of the communities of the Coeur d'Alene River basin have been closely linked to the natural resources of the region. The most obvious example is the relationship between the changes

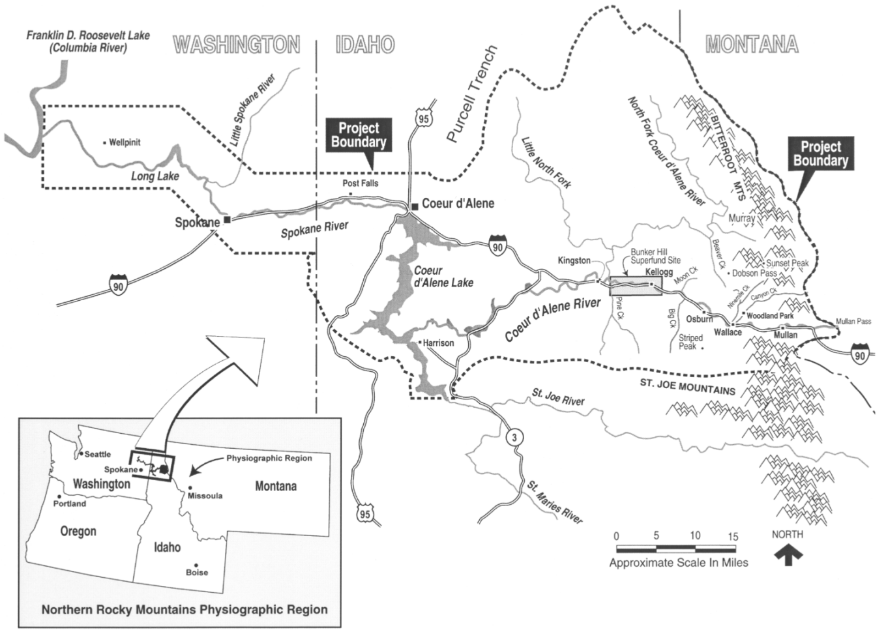

FIGURE 3-1 Map of the Coeur d'Alene River basin. SOURCE: URS Greiner, Inc. and CH2M Hill 2001b.

in the mining industry over time and the status of the associated mining communities. The forest resources have supported the lumber industry, and Lake Coeur d'Alene is developing a strong recreation and tourism economy. In addition, some members of the Coeur d'Alene tribe historically relied on the resources of the basin to support a subsistence lifestyle.

There are also important relationships between the socioeconomic attributes of the basin communities and potential risks from environmental contaminants. The mining communities have large stocks of older housing. Older houses are more apt to have lead-based paints, which constitute an indoor source of lead exposure. They typically also have greater air infiltration rates than new houses, which can result in larger inputs of airborne contaminants to the indoor environment. Households in the basin tend to have low incomes, and basin communities exhibit high poverty rates. Research on the relationships between blood lead in children and environmental and social factors has shown that blood lead levels (BLLs) tend to increase as measures of socioeconomic status decrease (Bornschein et al. 1985). A final factor affecting human health risks for the types of contaminants found in the basin is the age of the people exposed. Very young children (less than 5 years old) are most susceptible to the neurological effects of lead (Koller et al. 2004).

Topography

The Coeur d'Alene River basin is located in the western part of the Northern Rocky Mountain physiographic province, extending from the Bitterroot Mountains that run along the border between Idaho and Montana westward to Lake Coeur d'Alene, which lies near the border of Idaho and Washington.

The river basin consists of the South Fork (299-square-mile [774 km^2] drainage area) and the larger North Fork (895-square-mile [2,318 km^2] drainage area), which merge 4 miles above the community of Cataldo. Downstream from this confluence is the main stem of the Coeur d'Alene River, which flows 29 miles (47 km) to Lake Coeur d'Alene. The lake then drains through the Spokane River (see Figure 3-2).

The river basin contains three topographical types differentiated on the basis of their stream gradients and floodplain characteristics. The first type includes the upper reach of the South Fork from the Bitterroot Mountains to the town of Wallace, the upper reach of the North Fork, and all the

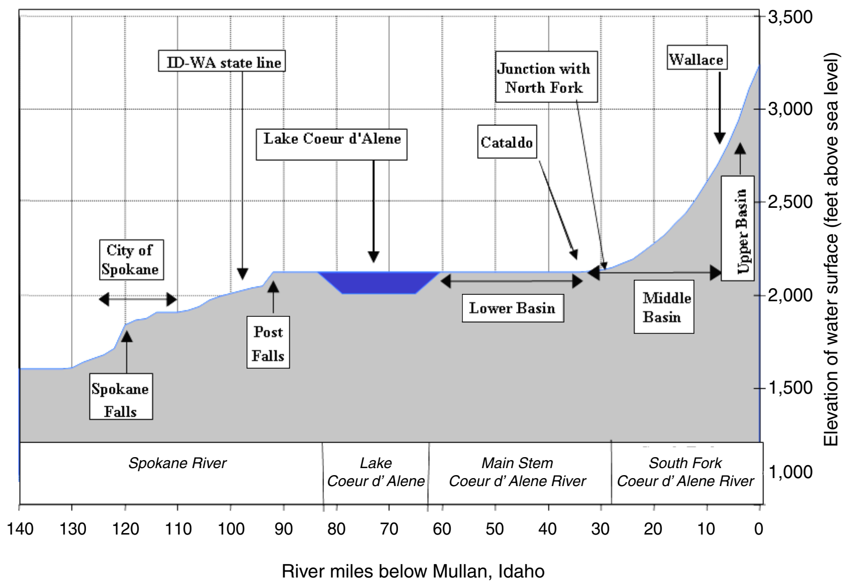

FIGURE 3-2 Longitudinal profile of Coeur d'Alene-Spokane River drainage. SOURCE: Box 2004.

tributaries of the South and North Forks. These areas, which typically have steep stream gradients and limited floodplains, are termed the upper basin.

The middle reach of the South Fork of the Coeur d'Alene River from Wallace to Cataldo and the middle reach of the North Fork are the second type of stream topography. In these reaches, collectively called the middle basin, the valley has wider floodplain areas bordered by steep valley walls, and the river gradient is more moderate.

The third type is the lower basin, containing the main stem of the Coeur d'Alene River, which runs from Cataldo to Harrison. In this reach, the river system is actually deltaic and the channel is backflooded by the waters of Lake Coeur d'Alene. Here, the river channel takes on a meandering pattern and, for most of the year, has an imperceptible gradient. The floodplain in this section is quite broad containing wetlands, "lateral lakes," and agricultural lands.

At the bottom (western end) of the lower basin, the Coeur d'Alene River flows into Lake Coeur d'Alene. This large and relatively deep lake is the ultimate sink for much of the contaminated sediment being carried down the Coeur d'Alene River.

The Spokane River drains Lake Coeur d'Alene at its north end. A dam constructed at Post Falls near the beginning of the river controls the water level in the lake. The Spokane River flows westward through the city of Spokane and on to the Columbia River at Lake Roosevelt behind Grand Coulee Dam.

Although the system can be divided into these different components on the basis of topography, it is important to remember that this is one interactive system, and it needs to be viewed as such if cleanup plans are to be successful (for an example, see Box 3-1).

Climate

Data concerning the climate in the Coeur d'Alene River basin are limited. The Coeur d'Alene River basin is typical of a "highland climate" with substantial variations in temperature and precipitation both from year to year and from higher to lower elevations.

Temperature and Precipitation

The upper basin experiences very high precipitation, averaging 55 inches (1.4 meters [m]) a year, of which 75-80% is in the form of snow (Isaacson 2004). The U.S. Forest Service has recorded up to 100 inches (2.5 m) of precipitation, with the depth of snow exceeding 18 feet (5.5 m). In the middle basin at Kellogg, during the 30-year period of record, the highest temperature recorded was 111°F (44°C), and the lowest was –36°F

BOX 3-1 Riverine Systems and Fish

The fish species in the Coeur d'Alene River basin represent a valuable resource for recreation and subsistence living. As in most Rocky Mountain headwater streams, salmonids, including various species of trout and salmon, are a dominant species, but a number of other important species are found there as well (CH2M-Hill and URS Corp. 2001, Table 2-3).

For many of these species, the river continuum theory (Vannote et al. 1980) demonstrates the importance of the entire hydrologic system to the health of their populations. In general, as mountain rivers grow in size, the size of the fish, the number of small fish, and the range in fish sizes all increase (Minshall et al. 1992). The nature of the food available to the fish and the biotic and abiotic interactions change along the path of the river as it moves downstream. As a river becomes larger, there are more microhabitats and more pathways for obtaining food, and, as a result, the range of sizes and the number of species generally increase downstream.

The river continuum is particularly important to salmonids in that upstream migration patterns are an integral part of their usual life history pattern (Baxter and Stone 1995), and this pattern links fish in a lower subbasin to habitat, prey abundance, and type in an upper basin. For example, in the Coeur d'Alene River basin, cutthroat and bull trout adults inhabit a wide variety of river habitats; however, they return upstream to tributary streams to spawn (Woodward et al. 1995).

Connected habitats in the Coeur d'Alene basin tie upstream biotic communities to those in downstream segments (Vannote et al. 1980; Minshall et al. 1992). High-quality riparian habitats and substrates for benthic invertebrates (an important food source) lead to "quality" trout stream fisheries.

For all these reasons, establishing high-quality riparian zones and desirable channel characteristics, as well as improving water quality along the length of the Coeur d'Alene River and its tributaries, is important to establishing and maintaining healthy and diverse fish populations.

(–38°C). The average was 47°F (8.3°C) (URS Greiner, Inc. and CH2M Hill 2001b, p. 3-2).

The average annual precipitation at Kellogg was 31 inches (0.79 m). The town of Wallace, at a somewhat higher elevation, had an average of 37 inches (0.94 m). Most (70%) of the precipitation occurs in the form of snow in October through April. As an indication of how variable the weather can be, the minimum annual snowfall—16 inches (0.41 m)—occurred in 1995, and the maximum—124 inches (3.15 m)—occurred the following year. The average annual snowfall over the period of record was about 52 inches (1.32 m) (URS Greiner, Inc. and CH2M Hill 2001b, p. 3-2).

Normally, the snowfall melts off slowly in late spring and early summer. However, this area can experience warm winter Pacific storms that bring a sudden onset of above freezing temperature and heavy rains on top of the preexisting snow pack. These "rain-on-snow" events result in rapid snowmelt and produce an abrupt increase over the usual low winter base flows in the river (Box et al. in press, p. 9). The basin is also subject to intense local storms that are characteristic of mountainous areas. These summer thunderstorms are of short duration, but they can cause significant rill erosion, mass wasting (downslope movement of rock and soil under the influence of gravity), and transport of colluvium and mine waste from steep slopes as turbid water or debris flows.

Winds

The most common wind patterns in the basin are typical of the mountain valley drainage phenomena. The winds flow parallel to the axis of the valley—typically flowing gently down the valley (from east to west) at night and in the early morning, as a result of the higher elevations cooling faster than the lower elevations, and then reversing direction in late morning as the sun warms the land, and the warm air begins to flow up the valley (TerraGraphics 1990). This is almost a daily pattern if there are clear night skies and no overriding regional weather patterns. Temperature inversions frequently occur at night and in the early morning before the valley warms up. However, during late summer, the area can experience strong (as much as 70 miles per hour [113 km/hour]) dry winds. Such winds seriously exacerbated the spread of the large forest fires experienced in 1910 and 1967 (Pyne 2001).

The winds on Lake Coeur d'Alene are less predictable, with the most common patterns being from either the north or the south along the axis of the lake (URS Greiner, Inc. and CH2M Hill 2001b, p. 3-3).

Mining-Related Wastes

An estimated 109 million metric tons (121 million U.S. tons) of contaminated mine tailings were produced by the mines and mills that operated in the Coeur d'Alene River basin (Long 1998). Most of these tailings—56 million metric tons (62 million U.S. tons)—were discharged to the basin's streams. These discharged wastes contained an estimated 800,000 metric tons (880,000 U.S. tons) of lead and more than 650,000 metric tons (720,000 U.S. tons) of zinc. These and other mining wastes that were discharged to the river systems intermixed with uncontaminated soils and sediments to produce what the U.S. Environmental Protection Agency (EPA) estimates to be more than 91 million metric tons (100 million U.S. tons) of contaminated materials (EPA 2002, p. 2-1). Another 53 million metric tons (58 million U.S. tons) of wastes containing 350,000 metric tons (386,000 U.S. tons) of lead and at least 650,000 metric tons (717,000 U.S. tons) of zinc "were stockpiled along the floodplain of the Coeur d'Alene River, placed in one of several tailings impoundments, or used as stope fill" (Long 1998).

Four basic types of wastes were discharged in the basin. The first is "waste rock," which is relatively unmineralized rock that is removed in uncovering the ore veins. This waste, most of which was dumped at the mine mouth, is relatively uncontaminated. The second type consists of the "jig tailings" disposed in the early mining era. These are generally coarse[1] materials with relatively high metal content. They were commonly dumped into the basin streams or in waste piles near the ore-processing facilities. The third type of waste consists of "flotation tailings," left over from the flotation method for processing ores, which came into use in the early 1900s. These tailings are much finer than the jig tailings and contain lower concentrations of most metals. The flotation tailings also were commonly dumped into the streams. The fourth type of waste includes a wide variety of wastes discharged to the air, water, and land by the smelters and other mining operations. The smelting facilities were located in the middle basin in the 21-square-mile (54 km^2) area addressed in operable units 1 and 2 (OU-1 and OU-2) of the Superfund site. These wastes can have a wide range of physical and chemical characteristics.

Metals in these wastes are the contaminants of greatest concern, particularly compounds of lead, arsenic, and zinc. The risks that these contaminants pose to human health and the environment depend not only on their concentration and the exposure to them but also on their chemical form or speciation. Some compounds are more biologically available and, therefore, pose higher risks than others.

Chemical Transformations and Toxic Effects

Metals in the environment exist in a variety of chemical forms or "species." For instance, zinc, a metal of primary concern in the Coeur d'Alene River basin because of its toxicity to aquatic ecosystems, can exist in its native mineral form (largely as sphalerite, or zinc sulfide [ZnS], also known as zincblende or zinc ore), in other mineral forms often altered from sphalerite (such as smithsonite, or zinc carbonate [$ZnCO_3$], which is also a zinc ore), in reduced sediments (as authigenic ZnS),[2] in solution in a com-

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[1]Box et al. (in press) described the size ranges of jig tailing grain sizes from eight impoundments of jig tailings in the Prichard and Beaver Creek drainages as follows: >8 millimeter (mm), 16%; 4-8 mm, 9%; 2-4 mm, 11%; 1-2 mm, 12%; 0.5-1.0 mm, 10%; 0.25-0.5 mm, 15%; 0.125-0.25 mm, 13%; 0.063-0.125 mm, 8%; and <0.063 mm, 6%. Tailings from the flotation process are typically 80% by weight finer than 0.25 mm.

[2]Authigenic ZnS can be formed when Zn^{2+} interacts with hydrogen sulfide (H_2S) that is produced during sulfate reduction in sediments containing organic matter. Authigenic ZnS forms in oxygen-depleted wetlands, marshy areas, and lake sediments of the Coeur d'Alene basin.

pletely dissociated ionic state (Zn^{2+}), or in a dissolved form complexed with other inorganic or organic solutes. Speciation of metals is driven by a variety of biotic and abiotic processes. Solid compounds can dissolve in water to the ionic form. This process occurs rapidly for solids that are soluble but slowly for those that are insoluble.

Weathering (commonly oxidation) can convert relatively insoluble forms of minerals into more readily soluble ones (such as the conversion of sphalerite to smithsonite or hydrozincite [$Zn_5(CO_3)_2(OH)_6$]). Weathering occurs on surfaces, so more rapidly in minerals with increased surface area (for example, in finely ground rock compared with large pieces). Once in solution, ionic zinc is a reactive molecule and undergoes a variety of interactions with other ions or with dissolved organic matter. These interactions affect the solubility of the compound. For example, the formation of authigenic ZnS will remove zinc from solution while zinc complexed to dissolved organic matter likely will remain in solution. These are dynamic and reversible processes, driven by a multitude of ever-changing biologic and environmental variables (pH, oxic state, temperature, and moisture). Thus, the potentially toxic metals exist as multiple chemical species in the environment whose behavior and toxicity can be markedly different.

Several groups (EPA 2003, 2004a; NRC 2003) recently have pointed out the importance of speciation in making metals bioavailable (in a form capable of exerting toxicological effects). To exert toxicity, a metal must be present as a species that is capable of interacting with a target site, the target site must be accessible to the chemical, and the target site must be available to interact with the metal. To illustrate, zinc exerts toxicity to fish by interacting with receptors on their gills. It is expected that zinc must be in its dissolved state to interact with these sites. If zinc is adsorbed to, for example, ferric oxyhydroxide,[3] it will not be available to interact with the sites of toxic action. Accessibility (or exposure) of the sites of toxic action is not a constraint, because gills are in intimate contact with the water and have an extremely high surface area to facilitate oxygen exchange between the water and the fish's blood. However, these sites may already be occupied by other nontoxic metals with similar chemical properties, particularly calcium and magnesium, the commonly dissolved cations that constitute the "hardness" of water. Because these other cations also can react with the receptor site, the toxicity of zinc depends on the concentrations of these competitive species. Thus, the toxicity of zinc to fish is also highly dependent on the hardness of the water.

In humans, the same types of interactions are important, but the organism and the environment (terrestrial instead of aquatic) are fundamentally

[3]Also referred to as hydrous ferric oxide.

different. Here, lead is the metal of primary concern, and the factors limiting the expression of toxicity are conversion of the metal to its ionic state and uptake of the metal from the gut to the bloodstream. Except in exposures from ingestion of water, lead is present as a solid upon ingestion or inhalation. Similar to zinc, the ongoing process of oxidation/weathering in the environment can convert lead sulfide (PbS), which is relatively insoluble, to a variety of more soluble species such as lead carbonate ($PbCO_3$). This process is accelerated by large surface-areas-to-volume ratios (small particle sizes) and favorable environmental conditions.

Thus, in similar environmental conditions, finely ground flotation tailings may present a greater risk to humans and waterfowl than coarser jig tailings, even though flotation tailings contain a lower concentration of lead in them. The fine tailings have a much larger surface area per pound of material than the coarser materials, providing much more opportunity for the PbS in the tailings to be oxidized to a form that is more biologically available.[4]

For humans, there are several other reasons why the finer particles may present more risk. They are more likely to cling to children's skin, which makes them more likely to be ingested when children put their hands in their mouths or touch food without washing their hands. They are more likely to cling to children's clothes and shoes, which makes them more likely to be tracked into the house where they contribute to continuing exposure through house dust (see discussion in Chapters 5 and 6 of this report). They are also more likely to be picked up by breezes and become atmospheric dust, making them more likely to be inhaled by children playing outside or be carried into children's homes (particularly, as indicated above, in older homes that have higher air infiltration rates).

An additional reason why the finer particles may present increased risk to waterfowl is that floods are more likely to carry the finer materials into the wetlands and lateral lakes in the lower basin. The coarser metal-enriched sediments tend to settle out of the flood waters near the river channel, forming the natural levees that border the river.

Within the organism, the different lead-bearing compounds will have various tendencies to dissociate into ionic lead (Pb^{2+}). For example, PbS is poorly soluble, but other lead species such as $PbCO_3$ are substantially more

[4]However, there are a number of reasons why these opportunities may not be realized. The fine tailings and coarse tailings are often found in different environmental conditions, particularly with respect to the availability of oxygen. They are often deposited in different locations, and the density of the deposits of the fine tailings makes them less permeable, and therefore slows the infusion of oxygen. Under oxidizing conditions, fine tailings may be leached of metal content more quickly than coarse particles. Of course, dissolved metals also may reprecipitate in the environment through biotic or abiotic mechanisms as solid chemical species, with a wide range of potential solubility.

soluble. After ingestion of lead-contaminated soils, the uptake of any soluble lead will also be modified by the presence of food in an individual's stomach, with absorption of lead declining in the presence of food. Once in the bloodstream, lead is available to exert a toxic effect (see Chapter 5 for further discussion).

All these factors that affect the toxicity of the wastes discharged into the basin can be affected by environmental factors. Jig tailings initially dumped into the river usually contained relatively insoluble metal compounds that exhibit limited toxicity. However, as these materials are exposed to air and water, the chemical nature of the compounds can change, increasing their bioavailability and their potential toxicity. In addition, the mixture of metals present may also change, so that the modifying effect of such mixtures on the toxicity of individual metals may also change (La Point et al. 1984).

In some cases, the indirect effects of the contamination may be a major factor. For instance, it is not only the direct toxic effect of these contaminants to fish that is of concern, but also their effect on the stream benthic organisms. These organisms are the primary source of food for the fish and fill a number of other food-web roles including herbivorous shredders, scrapers that consume attached algae and biofilm ("aufwuchs"), filterers and gatherers that consume detritus and suspended phytoplankton, and carnivorous engulfers that consume other invertebrates (Cummins and Klug 1979). They are often highly sensitive to dissolved metals and other contaminants, and in some parts of the basin only a few species (that are metal tolerant) now exist (Stratus Consulting, Inc. 2000).

Furthermore, as indicated above, the presence of contaminants can interact with other environmental factors in a way that either increases or decreases toxic effects. For instance, in addition to being a source of contaminants, the high sediment loads in the Coeur d'Alene River and its tributaries have a variety of biologic and physical effects on aquatic systems. These effects include the destruction of spawning areas, promotion of anoxic conditions, lowering the rate of recruitment into fish and invertebrate populations, inhibition of respiration, and limitation of light (Hynes 1970). These types of changes are very important in assessing the risks that the contaminants pose and what actions need to be taken to support a return of healthy aquatic ecosystems.

Finally, the risks that these contaminants pose depend on the species and segments of the population that are exposed to them (see Box 3-2).

THE UPPER BASIN

The upper basin, which includes the upper reaches of both forks of the Coeur d'Alene River as well as all the tributaries to these forks, is where

BOX 3-2 Who's at Risk?

Metals in the Coeur d'Alene River basin pose risks that vary for different segments of the human population and species of wildlife.

For humans, young children are much more susceptible to the effects of lead poisoning than adults because lead affects the neurological development that occurs during a child's early years. Young children also may have higher exposure as a result of their tendency to play on lawns or on floors, and other surfaces that may be contaminated.

For aquatic ecosystems, some varieties of fish and benthic organisms are more sensitive than others. For example, rainbow trout are particularly susceptible to dissolved metals, including zinc and cadmium (Davies and others 1976). There are numerous reports of the sensitivity of trout in the Coeur d'Alene River to dissolved metals. Farag et al. (1998) demonstrated that trout and other biota in the Coeur d'Alene system contain elevated concentrations of metals, and, in another study, that the growth and survival of cutthroat trout were reduced when they were fed macroinvertebrates from the South Fork (Farag et al. 1999). A study on trout sensitivity to metals in Coeur d'Alene River waters indicated that trout would spend as little as 3% of the time in contaminated water when given a choice of movement and that the fish avoided zinc concentrations as low as 28 μg/L (Woodward et al. 1997). Studies also indicate that dietary exposure to zinc and cadmium affects the early developmental stages of invertebrates and fish (Farag et al. 1998). Sculpin are another fish species with high sensitivity to metals. Fish population assessments conducted in the Coeur d'Alene River basin documented that these species were absent from metal-contaminated stretches of the river where they otherwise would be expected to be found, and they were more responsive than trout to environmental contamination by metals (Maret and MacCoy 2002). Sculpin are bottom-dwelling organisms that primarily feed on aquatic invertebrates. Among the aspects of their life history that make them useful as indicators of metal contamination are a small home range, inability to move during episodic events of high metal concentrations, a close association with sediments, their propensity to lay and incubate eggs in their range, and their failure to migrate to uncontaminated reaches to spawn (Dillon and Mebane 2002; Maret and MacCoy 2002).

Among waterfowl, tundra swans are particularly susceptible because of their migratory and eating habits. Most swans in the Coeur d'Alene River basin are either en route to their northern breeding grounds in the spring or heading south during wintering periods. They feed primarily on tubers and roots of aquatic plants that grow at shallow depths in lakes and wetlands in the lower basin. In the process of searching for and consuming these foods, they ingest significant amounts of sediment, putting them at particular risk from the lead these sediments contain.

much of the early mining occurred. The major tributaries are Canyon Creek and Ninemile Creek where the first silver and lead mines in the region were located. During the mining era, at least 21 mines and mining complexes operated along Canyon Creek, and at least nine operated along Ninemile Creek (URS Greiner, Inc. and CH2M Hill 2001c, p. 2-4; URS Greiner, Inc. and CH2M Hill 2001d, p. 2-4).

There is still one active mine in the upper basin, the Lucky Friday Mine, located slightly east of Mullan. This is an underground mine with an associated flotation mill, producing silver, lead, zinc, and a small amount of gold. Ore is processed at a rate of about 1,000 metric tons (1,100 U.S. tons) per day, and the workings are backfilled with cemented tailings (Hecla 2004). The ore concentrates are shipped to a smelter in British Columbia. The Lucky Friday complex employs about 100 people, although employment is likely to increase as a result of the company's recent decision to double its capacity by developing the Gold Hunter deposit, which lies about a mile northwest of the existing Lucky Friday workings (Hecla 2004).

Human Community

Although large communities of miners formerly lived in the upper basin valleys, currently there are only a few small settlements and scattered housing units in the tributary valleys. Most houses are quite old, and some lack basic water and sewage services. There are two small incorporated communities in the upper basin, Mullan and Wallace, both located on the South Fork of the Coeur d'Alene River. Table 3-1 shows selected demographic characteristics for these communities compared with the state of Idaho and the United States.

The populations of these communities, which decreased significantly during the 1980s after the mills and many of the mines in the basin closed, are somewhat older and poorer than is typical for Idaho. Wallace, in particular, has a high poverty rate. The housing stock is very old, with more than 80% of the housing units built before 1960, and the number of vacant units is very high, as would be expected in communities losing significant

TABLE 3-1 Demographic Characteristics of Upper Basin Communities

Demographic	U.S.	Idaho	Mullan	Wallace
Population			840	960
Median age (years)	35.3	33.2	41.4	40.6
Older than 65 (% population)	12.4	11.3	16.8	16.0
Median household income ($ thousands)	42.0	37.6	30.4	22.1
Below poverty level (% individuals)	12.4	11.8	12.1	20.1
Unemployment rate	5.8	5.8	11.6	11.5
% with bachelor's degree	24.4	21.7	10.8	17.2
% moved from out of state since 1995	8.4	15.3	14.8	21.8
% of owner-occupied units occupied by the same family for >30 years	9.7	6.9	22.7	14.8
Vacant housing units (%)	9.0	11.0	19.5	27.3
Houses older than 40 years (%)	35.0	27.7	78.6	93.3

SOURCE: U.S. Census 2004.

numbers of residents. A relatively high percentage of the residents in these communities has lived in the same house for more than 30 years. These are the households that stayed behind in spite of the economic problems that affected the basin.

However, there are also new residents moving into these communities. The percentage of residents who moved into these communities between 1995 and 2000 from out of state was as high as or higher than the average for Idaho and much higher than the average for the United States.

Geology and Fluvial Geomorphology

Bedrock Geology

This portion of the Rocky Mountains is a region of high mountain masses with steep valleys and no individually distinct mountain ranges. The bedrock of the basin (and host rock for the ore veins) is composed of argillite, slate, quartzite, and lesser amounts of impure, metamorphosed dolomite. These rocks are geologically grouped into the Belt Series, a sequence of indurated and mildly metamorphosed sedimentary rocks in northern Idaho, western Montana, and parts of British Columbia and Washington. Belt Group rocks were originally clay, silt, and fine sand layers deposited along the continental margins of a Precambrian sea between 1,500 and 1,400 million years ago (Winston 2000). The sediment layers have been indurated, folded, and faulted. In the Coeur d'Alene mining district, the rocks are intensely fractured and veined with minerals. Folding has so crumpled the layers that most dip at angles steeper than 45°.

The zone of intense shearing and faulting is along a regional structure known as the Lewis and Clark line, extending westward from central Montana to Spokane. Along this line, stream valleys such as the South Fork of the Coeur d'Alene River are guided by the zones of more easily eroded fractured rock.

The myriad fault and fracture zones along the Lewis and Clark line also contain the mineralized zones of the Coeur d'Alene mining district. The ore deposits are in veins composed primarily of quartz and siderite ($FeCO_3$). The ore veins are separated into two major types by mineralogy: (1) lead- and zinc-rich veins have argentiferous galena (PbS) and sphalerite (ZnS), and (2) silver-rich veins having argentiferous tetrahedrite [$(Cu, Ag)_{10}(Fe, Zn)_2(As, Sb)_4S_{13}$] and minor amounts of galena and sphalerite (Balistrieri et al. 2002a). Pyrite (FeS_2) is ubiquitous but variable in abundance in the ore veins. Most veins contain small amounts of chalcopyrite ($CuFeS_2$) and minor amounts of other minerals including arsenopyrite (FeAsS) and pyrrhotite ($Fe_{1-x}S$). The veins generally range from a few millimeters to 3 m in thickness, but some are up to 15 m thick (Hobbs and Fryklund 1968; URS

Greiner and CH2M-Hill 2001b, p. 3-15). In the early development of the district, oxidized ore mined from the Bunker Hill, Sullivan, Last Chance, Morning, and Standard-Mammoth deposits contained significant amounts of cerussite ($PbCO_3$), and locally massicot (earthy yellow PbO), and natural litharge (red PbO). Anglesite ($PbSO_4$) was notably absent (Ransome and Calkins 1908). Oxidized ore in the upper levels of these ore bodies was mined for the $PbCO_3$ and wire silver. However, by 1904 only one mine had a large deposit of carbonate ore remaining. The lower limit of oxidized ore in the district was very irregular, with carbonate noted in vugs and fractures to several hundred feet, but at the Bunker Hill Mine, unoxidized galena was discovered at the surface (Ransome and Calkins 1908, pp. 97, 133). The existence of $PbCO_3$ ore is important because it has greater bioavailability than sulfide ore and probably is present in the early jig tailings.

Beyond the main ore bodies, higher concentrations of sulfide minerals occur in proximity to an igneous stock and along the major faults. Zones of disseminated sulfide minerals extend tens to hundreds of meters outside of veins at the Lucky Friday Mine (White 1998). Within the stratified rocks, only the argillite and quartzite of the Pritchard Formation contain appreciable disseminated sulfide in the lower part of the formation, occurring as fine FeS_2 and/or $Fe_{1-x}S$ in the argillite (Hobbs et al. 1965; URS Greiner and CH2M Hill 2001b, p. 3-8).

Soils and Sediments

The natural hillsides have podzolic forest soils, with 10- to 19-inch- (25- to 50-cm)-thick upper, dark-brown horizons containing 2-5% organic matter. The soils are described as loamy skeletal soil, meaning mixed rock fragments with the soil fines having a clay content of 3-18% with the remainder being silt and sand. Soils are naturally acidic with a pH of 5.6-6.5, and cation exchange capacities of 15-30 milliequivalents (meq)/100 grams (g) in the upper 10 inches (25 cm) (NRCS 2003).

The thickness of soil and loose rock on hill slopes is variable. Bedrock exposures are common, but hill slope colluvial hollow and foot slope accumulations up to 10 m (33 feet) thick of mixed rock and fines are common. Differences in soil types and thickness and vegetation are expected between north- and south-facing slopes because of sun exposure and moisture retention.

The hillsides and hollows adjacent to former mining operations are covered with piles of waste rock and jig tailings. Waste rock dumps are uncrushed rock materials containing little metal removed during the active mining phase and placed just outside the mine openings. Jig tailings are the relatively coarse-grained materials left over from the inefficient jigging process that was used in the late 1800s and early 1900s to concentrate the ore. This process left tailings with relatively high metal concentrations. Some of the jig tailings were deposited in the waste rock dumps, some were placed in other repositories, but most, at least initially, were dumped into the upper basin tributaries to wash downstream (see Chapter 2). In the late 1960s, the dumping of mine tailings into surface water was stopped and tailings were collected in repositories or tailings ponds. The largest upper basin tailings pond is the 66-acre (27-hectare) Hecla-Star tailings pond at the bottom of Canyon Creek containing about 2.1 million cubic yards (1.6 million m^3) of material (URS Greiner, Inc. and CH2M Hill 2001c, p. 2-7; URS Greiner, Inc. and CH2M Hill 2001e, Appendix J, Table A-5).

Stream Channels

The stream segments in the upper basin have relatively steep gradients (>60 feet/mile [11 m/km]) and flow through narrow valleys in canyons with steep walls. Before the beginning of mining, the streams would have been typical mountain streams characterized by step-pool and plain-bed channels (Montgomery and Buffington 1997) lined predominantly with bedrock or cobble-boulder beds. Boulders, large logs, and log jams likely gave some degree of channel stability, providing hydraulic steps and pools and some sediment storage. The upper basin streams typically had little or no floodplain along their length, although some of the creeks did have discontinuous forested floodplains up to a few hundred meters (about 1,000 feet) wide (see Figure 3-3).

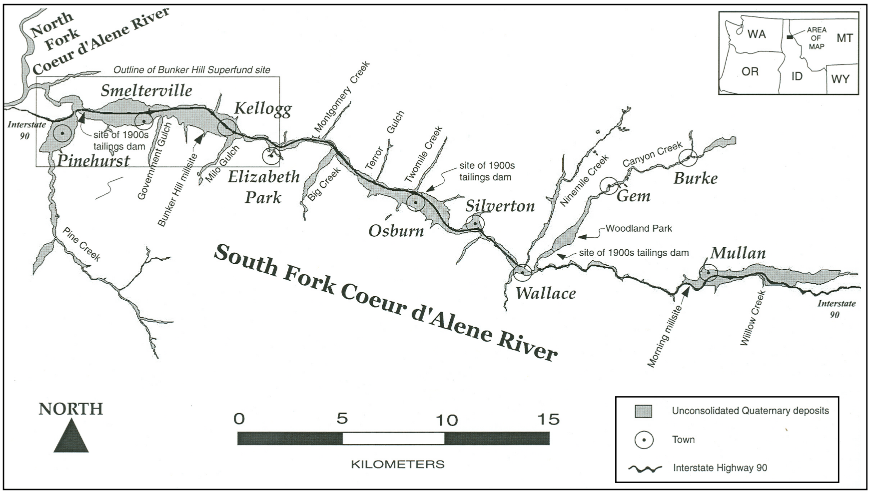


FIGURE 3-3 Upper and middle reaches of the Coeur d'Alene River showing valley fills and towns. SOURCE: Box et al. 1999.

During the early mining era massive amounts of relatively coarse jig tailings were dumped into these channels, causing them to aggrade. Since then, many reaches of these streams have been artificially channelized, and remediation projects have excavated some of the contaminated tailings and placed them in unlined and uncapped repositories out of the active channel ways (Harvey 2000; URS Greiner, Inc. and CH2M Hill 2001c, p. 3-4 to 3-14). In the more heavily mined tributaries such as Canyon and Ninemile Creek, the alluvial flats are underlain by 20-40 feet of alluvium (URS Greiner, Inc. and CH2M Hill 2001c, Fig. 2.1-1; Houck and Mink 1994, Fig. 10). The surficial layer of jig tailings in lower Canyon Creek is 2-4 feet thick (Houck and Mink 1994, p. 5).

These streams are still transferring metal-enriched sediments into the Coeur d'Alene River. Canyon Creek, for instance, is estimated to be discharging an average of 2,200 metric tons (about 2,400 U.S. tons) (equivalent to 1,360 m^3 or 1,780 cubic yards) of sediment a year to the South Fork at Wallace (URS Greiner, Inc. and CH2M Hill 2001c, Table 3.2-1). Most of this sediment is likely to be composed of native sediments mixed with tailings heavily contaminated with lead and other metals.

Hydrology

Surface Water

The upper basin streams display flow variations typical for mountain streams. Canyon Creek, for instance, has a base flow discharge estimated to be 10-15 cubic feet per second (cfs) (280-425 L/s), and the ten-year flood is estimated to have a peak flow about 100 times this base flow. The minimum discharge is less than 0.5 cfs (14 L/s) (URS Greiner, Inc. and CH2M Hill 2001c, p. 2-16). EPA's study of the upper basin tributaries (for example, Canyon and Ninemile Creeks) found that high waters overflow the banks an average of once every 1.5 years (URS Greiner, Inc. and CH2M Hill 2001c, p. 2-18; URS Greiner, Inc. and CH2M Hill 2001d, p. 2-14). However, a more recent study by the U.S. Geological Survey (USGS) finds that "the ratio of runoff to precipitation has increased, especially since the early 1960s. Some tributary streams that once ran bank-full or more about twice in 3 years now run bank-full 5 or 6 times a year. As a result, rates of erosion, sediment transport, and deposition also have increased" (Bookstrom et al. 2004a).

High water flow events carry significant amounts of sediment that are derived from erodable materials in the river bed, river banks, and floodplain (Box et al. in press). In contrast, low flows carry the highest concentrations of dissolved contaminants. The low flows are fed entirely by groundwater discharges, and the high contamination levels result from the percolation of these waters through tailings deposits.

Groundwater

In the upper basin, there are basically two types of groundwater aquifers. The first is the bedrock groundwater system, which flows through fractures in the relatively impervious bedrock. The recharge to this system occurs primarily from rainfall and snowmelt in the mountains and from stream flow and riparian aquifers losing water to bedrock in the lower reaches of streams. The underground mining operations effectively created a system of drains tapping the fracture systems and, as a result, much of the bedrock aquifer groundwater discharges into old mining operations and appears as "adit flow." The second type of aquifer is the shallow aquifer existing in the alluvium, tailings, and waste rock along the valley floor. The recharge to this system comes from seepage from the stream and discharges from the bedrock aquifer as well as from precipitation and snow melt. These surface aquifers are the source of the late summer "base flows" in the streams.

Dissolved Metals

Processes controlling the metal loading of groundwater are not known with certainty. Groundwater flow rate, water acidity, presence of carbonate minerals, fluctuating water tables, and chemical processes in the unsaturated zone are important factors that contribute to the high variability of dissolved metals in the groundwater.

The USGS sampled water draining from adits and seeping from beneath tailing piles for both total and dissolved metals (Balistrieri et al. 1998, 2002a). The investigators reported the following mean values for dissolved zinc concentrations: adits (other than the Kellogg Tunnel), 5.8 mg/L; tailings-seeps, 66 mg/L; groundwaters, 38 mg/L; and the Coeur d'Alene River, 3.4 mg/L (Balistrieri et al. 2002a). The zinc concentration is highly dependent on the pH of the water, and carbonate minerals in the soil can reduce acidity (Balistrieri et al. 2002b).

Discharges from the bedrock aquifer contain relatively low concentrations of dissolved metals. Even the adit drainages contribute few dissolved metals. Most adit drainage waters are not acidic (pH = 6.5-7.8) and, therefore, have limited capacity to dissolve metals. The few adit drainages in the upper basin that have significant concentrations of dissolved metals (Success, zinc at 50 mg/L; Gem, zinc at 16 mg/L) have low flow rates (0.02 and 0.2 cfs [0.5 and 5 L/s], respectively), which yield relatively small loads

(Balistrieri et al. 1998). The average zinc loading from all of the adits in the major upper basin mining areas (Canyon Creek, Ninemile Creek, the upper reaches of the South Fork, and Pine Creek) is about 71 pounds (lbs) per day (URS Greiner, Inc. and CH2M Hill 2001c, p. 4-106; 2001d, p. 4-77; 2001f, p. 4-68; 2001g, p. 4-44). This is about 2% of the total dissolved zinc load at the mouth of the Coeur d'Alene River.

More significant contributions of dissolved metals come from discharges from the shallow aquifers that exist in the alluvium, waste rock, and tailings deposited on the sides and bottoms of the stream valleys. Zinc concentrations in the seepage from many of these areas are in the 10-20 mg/L range but can be substantially higher (for example, the zinc concentration from a seep in the Ninemile Creek drainage was 350 mg/L) (URS Greiner, Inc. and CH2M Hill 2001d, p. 4-77). This suggests the ease of oxidation of ZnS under these conditions. However, the highest concentrations were generally associated with low flow rates. Measurements of seeps draining abandoned tailings piles have shown high concentrations of dissolved metals in Ninemile Creek and Canyon Creek (Balistrieri et al. 1998; URS Greiner, Inc. and CH2M Hill 2001c, p. 4-106; URS Greiner, Inc. and CH2M Hill 2001d, p. 4-77). Because the flow rates were low, these seeps contributed relatively little to the dissolved zinc load (an average of 11.2 lbs per day for the two seeps measured in Canyon Creek and 11.7 lbs per day for the three seeps measured in Ninemile Creek) (URS Greiner, Inc. and CH2M Hill 2001c, p. 4-106; 2001d, p. 4-77). The total contribution of these tailings and waste rock piles, however, cannot be determined from the available data because so few measurements were made, and because much of the flow through these deposits probably enters the underlying aquifer directly rather than appearing on the surface as seeps.

The other shallow aquifer discharges result from seepage of surface water into, and subsequently out of, the valley floor aquifers. A study of one of these aquifer systems showed seepage into and out of a 3.3 mile (5.3 km) stretch of alluvium underlying the downstream portion of Canyon Creek occurring at a rate of 3-5 cfs (85-140 L/s), with the return seepage flows high in dissolved zinc (650-30,000 μg/L) and other solutes (Houck and Mink 1994; Barton 2002). The estimated amount of dissolved zinc entering the stream from shallow aquifer discharges along this 3.3 mile stream segment was 150 lbs (68 kg) per day. This average load value is based on measurements during the low-flow months of September and October 1999. The contribution may be significantly higher at most other times of year when groundwater elevations are higher.

In total, however, EPA estimates that the upper basin streams contribute less than one-third of the total dissolved zinc loading measured to the Coeur d'Alene River (URS Greiner, Inc. and CH2M Hill 2001a, Figs. 5.3.5-8,9,10). Canyon Creek makes the largest contribution, 15% of the total, with Ninemile Creek next at 7%. The South Fork above Wallace and all the other tributaries contribute 2% or less (see Figure 3-4 for details on zinc loadings during water year 1999-2001).

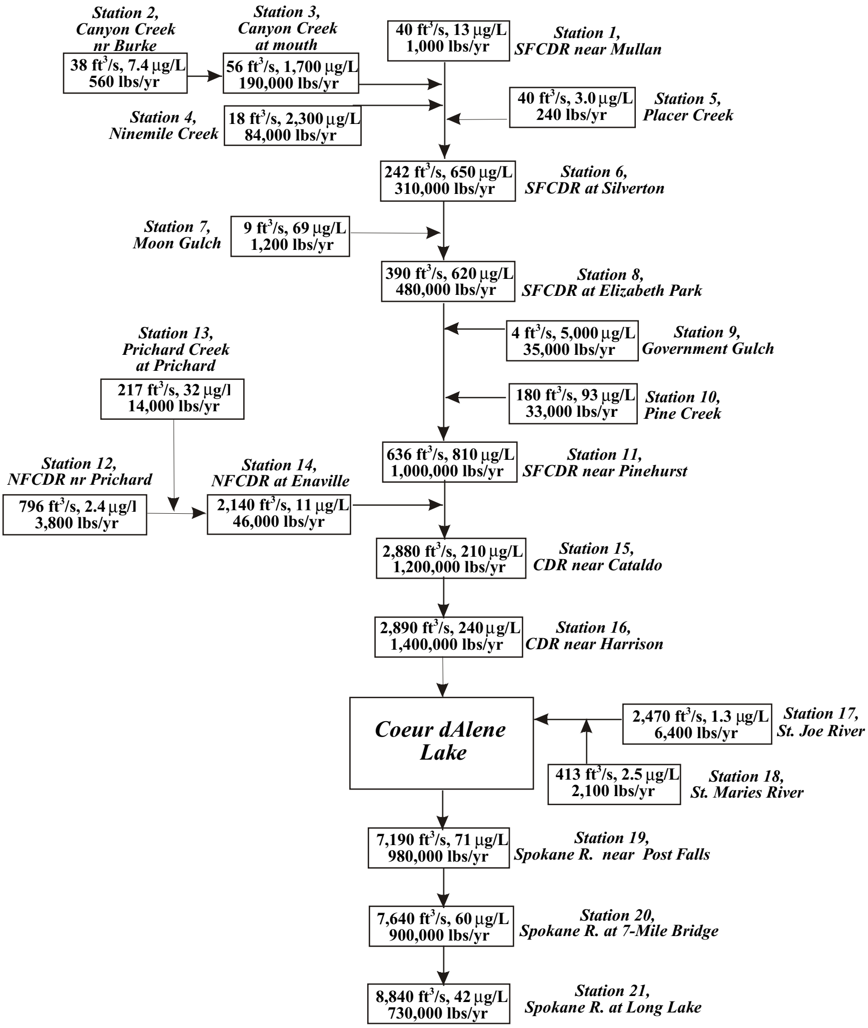


FIGURE 3-4 Sources of zinc in the Coeur d'Alene River in water years 1999-2001. Boxes for each location (station) present mean annual stream discharge, mean flow-weighted concentration, and mean annual load of total zinc. SOURCE: Clark 2003.

Ecologic Community

Before the beginning of mining, the hills and valleys of the Coeur d'Alene River basin were heavily forested. The hillsides were covered with a rich mixed-conifer forest of Douglas fir, grand fir, ponderosa pine, western larch, and western white pine, and the valleys were forested with cedar and lodgepole pine, cottonwood, and other riparian trees. Red cedar boles and large logs that fell into streams provided pool habitat for fish, sediment storage, and some degree of channel stability (Harvey 2002, p. 8).

Much of the original timber was cut down during the mining era for building construction, mine-shaft support, and fuel, or it was destroyed by fires such as that of 1910, which burned much of the basin above Kellogg (Hart and Nelson 1984; Pyne 2001). Over the past half century or longer, however, the forests have been allowed, and in some cases actively encouraged, to regenerate, and as a result the natural vegetative cover on the valley slopes is returning. The basin contains National Forest, Bureau of Land Management, state of Idaho, and private lands that can be leased out for timbering. For instance, there has been extensive timbering along the North Fork of the Coeur d'Alene River. The timbering often results in increased runoff and sediment (Idaho Panhandle National Forests, 1987, 1998, 2002; CBFWA 2001).

Although the return of the forests to most of the upper basin area has reestablished the habitat needed by the wildlife species that naturally inhabit such areas, the foresting operations and construction and maintenance of the logging roads continue to reduce the value of this habitat. Much of the basin has a very high logging-road density (greater than 4.7 linear miles of road per square mile [2.8 km/km^2]) (CBFWA 2001, p. 62).

Aquatic Habitat

Upstream of the areas affected by mining operations, the upper basin streams are relatively healthy. EPA has found that the fish, such as cutthroat trout and sculpin, and the benthic communities are diverse and healthy (CH2M-Hill and URS Corp. 2001). Abundant trout populations can even be found in some upper basin river segments affected by mining. For instance, the South Fork of the Coeur d'Alene River above Wallace has an average dissolved zinc concentration of approximately 190 μg/L, about five times the ambient water-quality criteria (AWQC), but the trout density is quite high, similar to that in morphologically similar reaches in the St. Regis River, which has not experienced serious mining impacts (Stratus Consulting, Inc. 2000). However, sculpin, which would be expected to be abundant in the South Fork and its tributaries, do not fare so well. A recent study (Maret and MacCoy 2002) demonstrated that sculpin were absent from stretches of the river where zinc concentrations exceeded the AWQC.

The quality of the aquatic and riparian habitat along many of the upper basin streams affected by mining remains severely degraded. Efforts to reestablish vegetation in the tailings deposits along the upper basin stream channels usually have been relatively unsuccessful (URS Greiner, Inc. and CH2M Hill 2001c, p. 1-1). These problems, combined with high concentrations of dissolved metal, result in the streams showing a substantial reduction (and in some segments elimination) of native fish species and a decline in the diversity and abundance of benthic macroinvertebrates (URS Greiner, Inc. and CH2M Hill 2001b, p. 3-51).

THE MIDDLE BASIN

Before the mining era, the river segments in the middle basin would have had the characteristics of braided streams, with their beds predominantly composed of gravel and having a relatively shallow depth (except during flooding). The floodplains were described as heavily forested or marshy (Box et al. 1999, p. 5).

Most of the large mining communities and large ore-processing facilities were located along the middle reach of the South Fork of the Coeur d'Alene River. These communities, with their housing, mine-processing facilities, and transportation facilities, are built on top of and, in the case of the railroad and interstate highway embankments, largely out of the vast amounts of mine tailings deposited in this reach. The original Bunker Hill Superfund site lies in the middle of this reach. This site, commonly called "the box," is a rectangular area that runs from Kellogg on the east to Pinehurst on the west and contains the Bunker Hill smelter and all the other facilities, residences, and land within its 21-square-mile (54-km^2) area. The site is composed of two OUs designated OU-1 (for populated areas) and OU-2 (for the rural and former industrial areas) and was the focus of cleanup efforts begun in the early 1990s. Although EPA has excluded the box from consideration in its plans for OU-3, it continues to be a major source of dissolved metals in the lower Coeur d'Alene River.

There are currently two active mines in the middle basin. One is the Galena Mine located 2 miles west of Wallace, and the second is the Bunker Hill Mine located in Kellogg. In addition, a group of investors is reported to be exploring the possibility of reopening the Sunshine Mine located near Kellogg[5] (Sterling Mining Company 2004).

[5]The Sunshine Mine was the richest silver mine in American history with more than 360 million ounces of production over the past century. It was also the site of the 1972 mine-fire disaster that killed 91 miners (USMRA 2004).

The Silver Valley/Galena Mine is located southwest of Silverton in the valley of Lake Creek. Silver and some copper are recovered by a flotation mill, producing a silver-rich concentrate, which is sold to third-party smelters in Canada. Flotation tailings are separated into coarse and fine fractions at the mill, and the coarse tailings pumped back into the mines to use as backfill. The fine fraction slurry is piped down Lake Creek to the South Fork valley and then to the 60-acre Osburn tailings ponds, situated at the southeast end of the Osburn Flats. The fines are settled in the impoundment and the clarified water decanted and carbon/charcoal filtered before waste water is discharged to the river (EPA 2001). The mine, which produced 165,000 tons of ore and 3.7 million ounces of silver in 2003, employs about 200 people. Development work at the mine is ongoing and production is expected to increase approximately 40% by 2006 (Coeur d'Alene Mines Corporation 2004; Gillerman and Bennett 2004).

The Bunker Hill mine is, at present, a much smaller operation. Its owner reports that he occasionally mines 18-36 metric tons (20-40 U.S. tons) of ore per day and employs nine people (Robert Hopper, Bunker Hill Mine, personal commun., April 14, 2004). If silver or zinc prices were to rise substantially, this mine might be able to return to commercial production, although it faces a number of problems related to the disposal of its mining wastes and adit drainage.

Very little development has occurred along the North Fork. Although several mining operations took place in the tributaries of the North Fork, the only settlements are Prichard at the very top of the North Fork watershed and Enaville at the junction with the South Fork. The main activity in the North Fork basin is lumbering. The dense logging roads and forestry operations are a major source of erosion and high sediment loads in the North Fork.

Human Community

From a socioeconomic standpoint, the most significant recent event in the middle basin was the closure of the Bunker Hill smelter in August 1981. The resulting loss of about 2,100 jobs caused significant declines in the populations of the basin's communities (Bennett 1994). As indicated in Table 3-2, the middle basin communities reflect these events, showing many of the same characteristics of the upper basin communities.

These communities are mostly larger than those in the upper basin. The median age of residents is older than for the rest of Idaho and the United States, but, compared with the upper basin communities, the median age is younger and a smaller proportion of the residents have been living in the same house for more than 30 years. Another major difference from the upper basin communities is that a significant portion of these

TABLE 3-2 Demographic Characteristics of Middle Basin Communities

Demographic	U.S.	Idaho	Osburn	Kellogg	Wardner	Smelterville	Pinehurst
Population			1,545	2,395	215	651	1,661
Median age (years)	35.3	33.2	44.6	37.4	41.5	39.4	41.6
Older than 65 (% of population)	12.4	11.3	20.7	18.4	13.0	18.6	19.3
Median household income ($ thousands)	42.0	37.6	29.9	25.9	25.5	21.9	27.8
Below poverty level (% individuals)	12.4	11.8	11.7	21.8	12.8	22.4	14.8
Unemployment rate (%)	5.8	5.8	8.7	11.6	7.1	14.7	8.4
% with bachelor's degree	24.4	21.7	10.9	10.6	6.6	3.1	7.4
% moved from out of state since 1995	8.4	15.3	11.0	17.8	20.9	15.8	11.7
% of owner occupied units occupied by the same family for >30 years	9.7	6.9	12.8	16.1	23.1	15.7	14.8
Vacant housing units (%)	9.0	11.0	11.1	17.4	20.7	13.2	6.9
Houses older than 40 years (%)	35.0	27.7	52.0	74.0	85.0	61.6	40.7

SOURCE: U.S. Census 2004.

residents—more than 26% in Smelterville—live in mobile homes (U.S. Census 2004).

In terms of structure, the families in these communities are more typical of state and national averages, with 5-8% of the population less than 5 years old, compared with 4-4.5% for the upper basin communities. A significant percentage of the families moved here recently, but average household incomes are low, and poverty rates are high.

Geology and Fluvial Geomorphology

The bedrock forming the valley walls in the middle basin has the same geological characteristics as that in the upper basin, and a number of major mining operations have taken place along the middle reach of the South Fork. As a result, in several areas, the hill slopes are covered with the same sorts of waste rock and tailings as are found in the upper basin. A major difference in the soil characteristics is found in the hills on the south side of the South Fork from Kellogg to Smelterville where acidic emissions from the Bunker Hill smelter substantially contaminated the soil, preventing the reestablishment of vegetation. The lack of vegetation, in turn, has made the hills subject to sloughing and erosion. Sampling of the soils on the hillsides above east Smelterville found mean concentrations of lead at approximately 9,000 mg/kg (TerraGraphics 2000, p. 6.11), making them a concern for recontamination of the remedial work completed in residential areas of the box.

The major geomorphic differences between the upper basin and the middle basin are in characteristics of the river and the valley floor. Below the confluence with Canyon Creek at Wallace the valley floor widens, the valley fill becomes thicker, and the river slope begins to gradually flatten. The valley fill beneath the floodplain increases in thickness from less than 30 feet (9 m) at Wallace to 80 feet (24 m) at Kellogg to 140 feet (43 m) at Smelterville (Dames and Moore 1991) and is largely comprised of premining depositional sediments (Figure 3-5). However, much of the floodplain is covered with jig-bearing alluvium with an average thickness of approximately 4 feet (1.3 m) (Box et al. 1999).

In its natural state, the river here would have exhibited the characteristics of a braided stream. The widening of the channel and floodplain in the middle basin would have caused a reduction in flood-water depth and velocity, resulting in the deposition of flood-entrained bedload deposits. The main channel would have switched back and forth across the floodplain, building up deposits of sand-to-cobble-sized alluvium (Box et al. 1999, p. 5).

The rate of deposition substantially accelerated after mining began, because tailings were disposed directly into streams. By 1903, tailings depo-

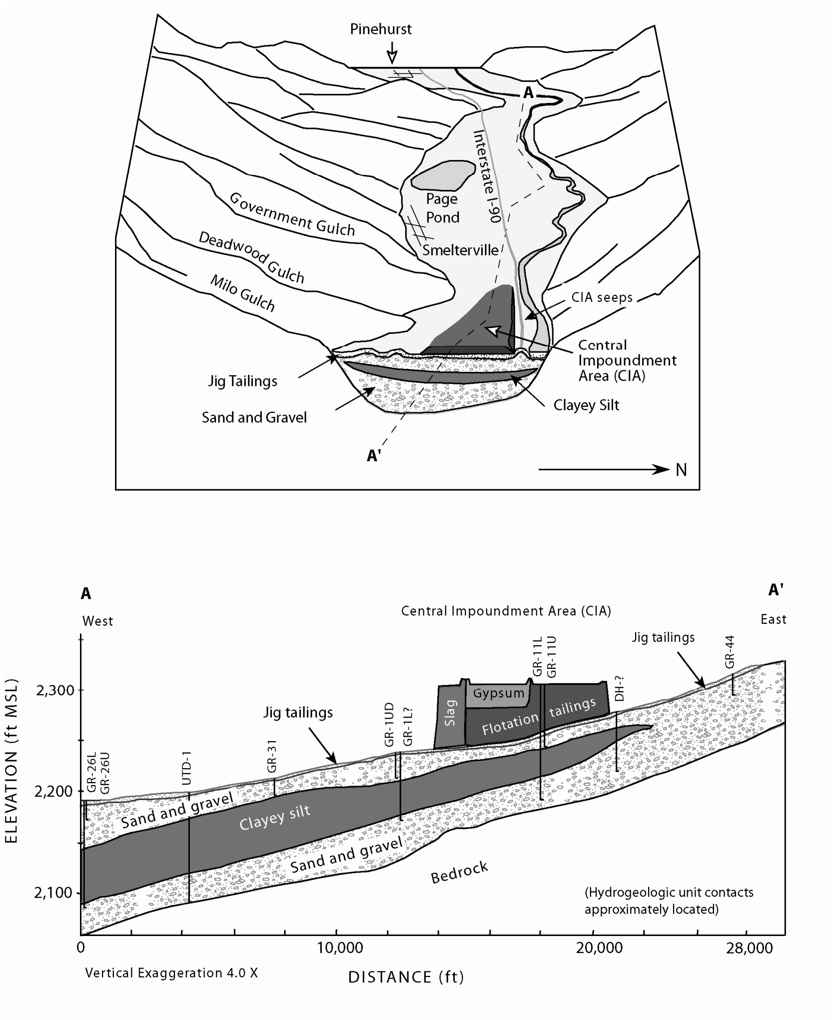


FIGURE 3-5 Diagram looking downvalley and geologic cross section of valley fill of the South Fork of the Coeur d'Alene River valley west of Kellogg showing aquifer units and wells. SOURCE: modified from Dames and Moore 1991.

sition over the broad valley floodplains at Osburn Flats and Smelterville Flats resulted in barren wastes of gray jig tailings 1-2 feet thick through which projected the dead stumps of trees (Box et al. 1999, p. 8; Bookstrom et al. 2001, p. 24).

In addition to the flood deposits, mines and mills operating along the middle reach have deposited substantial volumes of tailings and other wastes directly on the floodplains or in unlined large repositories. The largest include the central impoundment area (CIA) at Bunker Hill containing 18.5 million m^3 (24.2 million cubic yards) of various wastes, and Page Pond containing 1.6 million m^3 (2.1 million cubic yards) of tailings (URS Greiner, Inc. and CH2M Hill 2001e, Appendix J, Table A-8). The Osburn Flats tailings pond (containing about 2.7 million m^3 [3.5 million cubic yards] of material) currently receives slurried tailings from the active Galena Mill that are settled in the impoundment. A number of other large contaminated sites, ranging in size from 10 to 30 hectares (25 to 75 acres), are associated with the facilities located within the Bunker Hill complex (URS Greiner, Inc. and CH2M Hill 2001g, Table 4.1-2).

The dumping of large amounts of tailings into the stream's tributaries overwhelmed the river's ability to carry these sediment loads downstream. In an effort to address complaints from downstream farmers about their fields being covered with contaminated materials, wood-piling and cribbing dams were constructed in the channel to contain the sediments, but these were rapidly overtopped and later washed out (Box et al. 1999).

However, efforts to "stabilize" the river channel continued. As described in the remedial investigation (RI): "to accommodate the infrastructure, and to make room for storing and disposing of mining wastes in the floodplain, the channel of the South Fork Coeur d'Alene River has been moved, channelized, armored, and otherwise altered, with only a few reaches still resembling a natural river" (URS Greiner, Inc. and CH2M Hill 2001b, p. 2-11). Remediation efforts carried out pursuant to the ROD for OU-2 again moved the river channel to allow about 1.2 million cubic yards (0.91 million m^3) of mine waste to be removed from the Smelterville Flats area (EPA 2000; EPA 2004b [July 27, 2004]).

The river continues to carry large amounts of sediment downstream. From 1988 through 1998, EPA's contractors estimated that the average annual sediment load passing Pinehurst, downstream of Smelterville, amounted to almost 20,000 metric tons (22,000 U.S. tons), which is equivalent to about 12,000 m^3 (16,000 cubic yards) (URS Greiner, Inc. and CH2M Hill 2001f, p. 3-42). During 1996, a year experiencing a large flood, the load was almost 70,000 metric tons (77,000 U.S. tons). About half of this load was made up of fines (<63 μm diameter) (URS Greiner, Inc. and CH2M Hill 2001f, p. 3-42). These data emphasize the important role of heavy floods in distributing metal-contaminated sediments throughout the system.

By the time the suspended sediments reach the middle basin, the metals in the fines have had ample time to oxidize and thereby become biologically available. USGS investigators used scanning electron microscopy with x-ray detection of elements and leaching studies to characterize the speciation of lead in samples that were collected from the floodplain and the river and found that iron and manganese oxides were present and appeared to be host phases for lead, which was also present as $PbCO_3$ and $PbSO_4$. They concluded that the galena was oxidized within about 6 miles (10 km) of the original deposit (Balistrieri et al. 2002a).

The North Fork of the Coeur d'Alene River joins the South Fork at the bottom of the middle basin. The North Fork drainage basin is 3 times larger than that of the South Fork, so stream flow is usually 2.5-4 times larger from the North Fork. Mining operations were located on the Prichard and Beaver Creek tributaries of the North Fork, but these do not contribute significant mining waste. The concentrations of metals in water and sediment of the North Fork are low, usually below the EPA screening levels (URS Greiner, Inc. and CH2M Hill 2001h, p. 5-4), and the North Fork supports a good fishery for the westslope cutthroat trout (Abbott 2000). Therefore, flow and sediment transport from the North Fork dilute the South Fork metal concentrations below their confluence.

Although extensive logging activity in the basin probably has increased the magnitude of flood flows in the North Fork, at similar flows (4,000 cfs) (113 m^3/s), the South Fork transports 38 times the suspended sediment and 72 times the bedload of the North Fork (Clark and Woods 2001, Figs. 10, 18). However, because the North Fork drains a larger area, it carries more water. For instance, the peak flood flow with a recurrence interval of 2 years on the South Fork is 3,660 cfs (103 m^3/s) carrying 1,203 metric tons per day (1,327 U.S. tons per day) of sediment (including both suspended sediment and bedload). On the North Fork, the flood with a 2-year recurrence interval is almost 4 times larger (15,100 cfs [428 m^3/s]) and carries 5 times the sediment (6,590 metric tons [7,264 U.S. tons] per day) (Clark and Woods 2001, Figs. 10, 18, p. 18, 26; Berenbrock's 2002 estimates of flood recurrence). Data from 1996 (a flood with >50-year recurrence interval) and 1997 (a flood with 3-4 year recurrence interval) show that larger dilutions of metal-rich with metal-poor sediment may occur in large flood events than in the annual snowmelt flood (Box et al. 2005). In the 1996 event, lead concentrations in suspended sediment below the confluence with the North Fork were approximately 42% of the upstream concentrations while in the 1997 event, downstream lead concentrations were 73% of the upstream concentrations (Box et al. 2005).

Base flow of the North Fork is estimated to be 200-250 cfs, compared with 80-100 cfs on the South Fork, so the high concentrations of dissolved zinc that are harmful to aquatic life are diluted by the relatively uncontaminated flows from the North Fork. This dilution should result in concentrations in the main stem base flow water that are 25% to 35% of the concentrations in the South Fork water.

In the 1999-2000 water year, the South Fork delivered about 20% of the total lead load to Lake Coeur d'Alene; the remaining 80% is derived from erosion along the course of the main stem Coeur d'Alene River below the confluence of the North Fork (Clark 2003, Fig. 12). Of the approximately 850,000 metric tons of mined lead historically lost directly or indirectly to streams, Bookstrom et al. (2001, Table 15) roughly estimate that 24% (200,000 ± 100,000 metric tons) still resides as sediments in the South Fork drainage.

Hydrology

Surface Water

Several stream gauging stations in the middle reach of the South Fork provide intermittent data from 1967 to the present. The major stations are at Silverton, downstream from Wallace (which has the longest record, although it was not in service from 1988 through 1997); Elizabeth Park, upstream of Kellogg; and Pinehurst, downstream of Smelterville. At Silverton, the average flow rate was about 250 cfs (7.1 m^3/s) and the base flow was estimated to be between 50 and 60 cfs (1.4-1.7 m^3/s) (URS Greiner, Inc. and CH2M Hill 2001f, p. 2-21).

Flooding

The Coeur d'Alene River frequently experiences significant floods in late spring as a result of snow melt and, less frequently, winter floods as a result of rain-on-snow events (see Figure 3-6). Figure 3-7 shows the estimated frequency of peak flood discharges for Elizabeth Park and Pinehurst. At Elizabeth Park, the spring floods typically flow in the range of 1,000 cfs for several weeks, with peaks of 2,000-3,000 cfs (56-85 m^3/s). Heavy rainstorms in the spring can produce temporary, sharp runoff peaks on top of this continued snowmelt runoff (Box et al. in press, p. 9). Major spring floods occurred in 1893, 1894, 1917, 1948, 1956, and 1997 (S. E. Box, USGS, unpublished material, 1994, as cited in Bookstrom et al. 1999, p. 18). The largest winter floods resulting from rain-on-snow events occurred in 1933, 1974, and 1996.

These flood flows transport substantial amounts of sediment downstream (Clark and Woods 2001, Figs 10, 18). The threshold for bedload movement in the South Fork at Silverton is about 200 cfs (5.5 m^3/s), and a spring flow of 2000 cfs (56 m^3/s) transports 50 metric tons/day (55 U.S. tons/day) of bedload, and more than 300 metric tons/day (330 U.S. tons/day) of suspended sediment (Clark and Woods 2001). Measurements at Pinehurst showed a transport of 250 metric tons/day (275 U.S. tons/day) of bedload, at 1,830 cfs (52 m^3/s) and 1,500 metric tons/day (more than 1,600 U.S. tons/day) of suspended sediment in flows of 3,600 cfs (about 100 m^3/s).

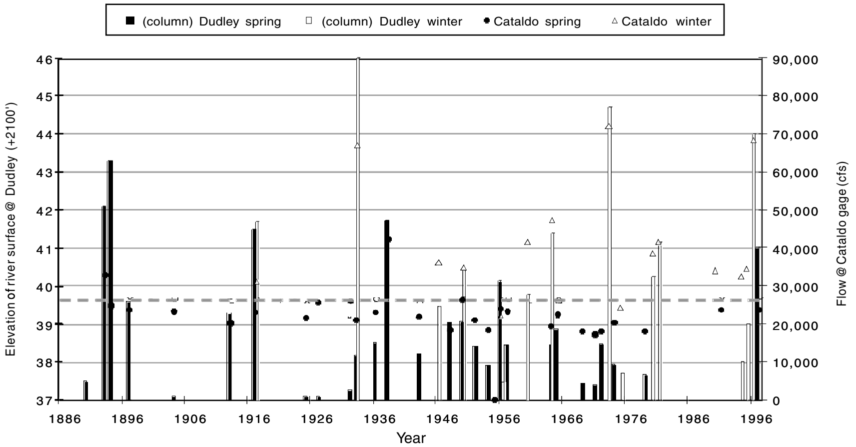
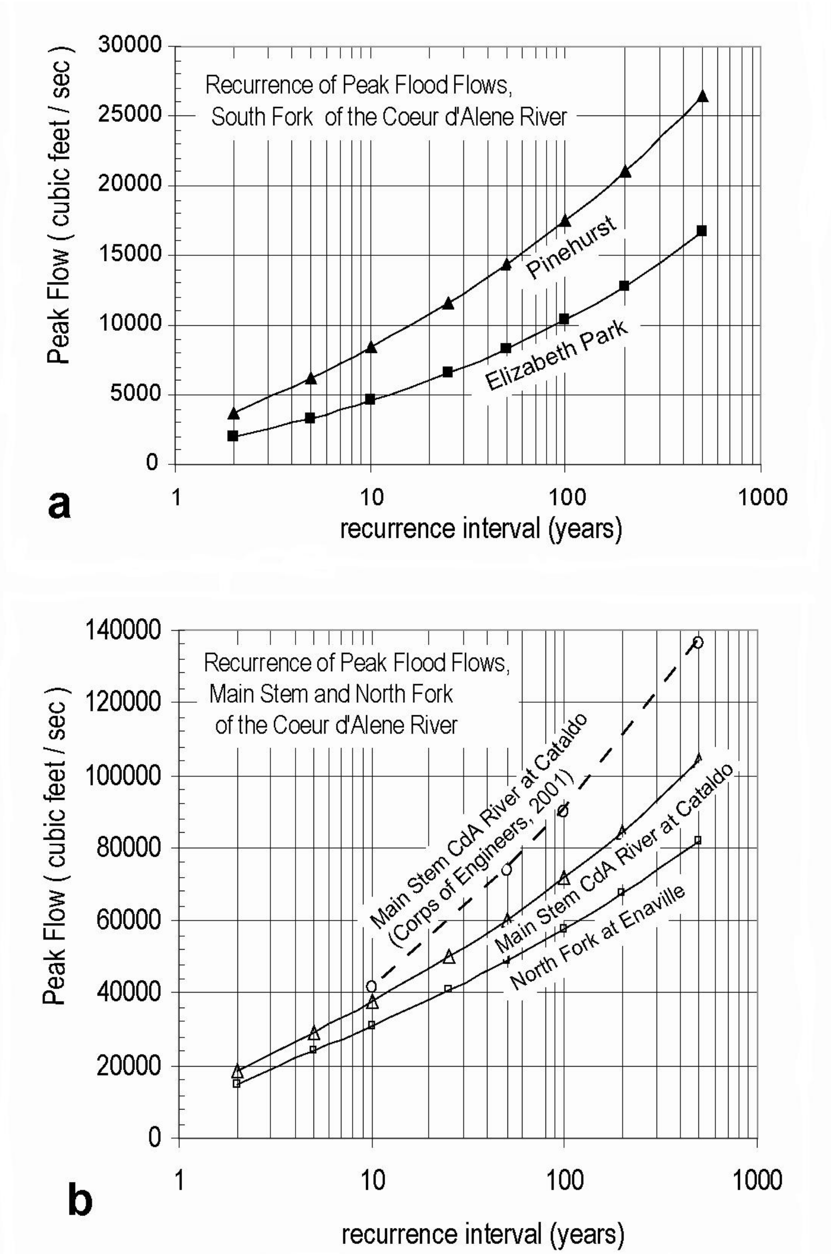


FIGURE 3-6 Coeur d'Alene River flood history, 1886-1997. Annual peak flows and water-surface elevations at Dudley and Cataldo, Idaho, during winter and spring flood events (dashed line depicts flood stage when entire floodplain is inundated). SOURCE: Bookstrom et al. 2004b.

The largest and most damaging floods, however, occur as a result of rain-on-snow storms. The first major flood after the beginning of mining resulted from such an event in December 1933. Now considered to be the 50- to 100-year flood, the peak flow at Pinehurst may have been 17,000 cfs (480 m^3/s). The floodwaters broke out of diked channels through Kellogg and severely eroded the northeast corner of the Bunker Hill tailings impoundment (Box et al. 1999, p. 5). All the Smelterville Flats north of the railroad were flooded, and tailings were deposited over the flats. However, little of the jig-tailings-aggraded floodplain above Kellogg was flooded.

Another winter flood in January 1974 exceeded that of 1933 and is considered the 100-year flood. Extensive damage occurred where tributary streams enter the South Fork valley, but little overbank flooding occurred along the South Fork. Some damage did occur to dikes, road and railroad embankments, and bridge abutments (Box et al. 1999, p. 12).

A third major winter flood occurred in February 1996. That flood had a peak flow of 11,700 cfs (330 m^3/s) at Pinehurst, slightly less than the flow of the 1974 flood (Beckwith et al. 1996) and only the floodplains in the bottom reach of the middle basin were inundated by this event (Box et al. 1999, p. 12). The USGS found suspended sediment concentrations of 410-1,900 mg/L during this flood (Beckwith et al. 1996), which indicates that the river could have transported as much as 32,000 metric tons of suspended sediment per day (equivalent to about 20,000 m^3 or 26,000 cubic yards per day).


Recurrence of Peak Flood Flows,
South Fork of the Coeur d'Alene River
Peak Flow (cubic feet / sec)
30000
25000
20000
15000
10000
5000
0
Pinehurst
Elizabeth Park
a
1
10
100
1000
recurrence interval (years)
Recurrence of Peak Flood Flows,
Main Stem and North Fork
of the Coeur d'Alene River
Peak Flow (cubic feet / sec)
140000
120000
100000
80000
60000
40000
20000
0
Main Stem CdA River at Cataldo
(Corps of Engineers, 2001)
Main Stem CdA River at Cataldo
North Fork at Enaville
b
1
10
100
1000
recurrence interval (years)

These rain-on-snow floods are of short duration. Stream discharges increase and peak sharply before they tail off over a few days. These events have produced the largest peak flows of record (1933, 1974, and 1996), reaching 9,600 cfs (270 km/s) at the Elizabeth Park gauge. Multiple-storm winter floods include those of 1917, 1933, 1961, and 1982. Single-storm winter floods include those of 1946, 1951, 1964, 1974, 1980, 1990, 1995, 1996, and 1997 (S. E. Box, USGS, unpublished material, 1994, as cited in Bookstrom et al. 1999, pp. 17-18).

Groundwater

In addition to the bedrock aquifer and the shallow aquifers found in the upper basin, the middle basin also has a deeper aquifer system within the valley fill separated from the surface aquifer by the relatively impermeable layer of silt and clay (Figure 3-5). The deeper aquifer system begins a little east of Kellogg where it is 20-50 feet (6-15 m) thick and becomes thicker in the lower river reaches. This aquifer is a source of well water for many basin residents who are not on municipal systems that obtain their water supply from up-basin surface-water sources. It is recharged by the bedrock aquifers and by seepage through the shallow aquifers. Having been formed before mining began, this aquifer is composed of relatively uncontaminated materials. There is no information about the possibility that groundwater in the aquifer is being contaminated by seepage from the more contaminated waters that lie above it.[6] This aquifer was not evaluated in the 2001 RI (URS Greiner, Inc. and CH2M Hill 2001f, pp. 2-17, 2-18).

[6]There is also apparently no information about how many people depend upon this aquifer as a source of water supply although there are a large number (thousands) of private, unregulated drinking water sources in the study area (EPA 2002, Table 6.3-3).

FIGURE 3-7 Estimated recurrence of peak flood flows for the Coeur d'Alene River. (a) South Fork at Elizabeth Park and Pinehurst; (b) main stem at Cataldo and the North Fork at Enaville. Solid lines are curves plotted from data of Berenbrock (2002), which considered basin and climatic characteristics and fit log-Pearson type III distribution to peak flow data through 1997. Berenbrock (2002) indicates a standard error of peak flow prediction from 40-70%. The dashed line is the curve plotted from data from the U.S. Army Corps of Engineers (USACE 2001), which derived the flood frequency by separating the winter rain flood and spring snow melt floods into separate flood series by cause (rain- versus snow-melt-generated floods), computing individual frequency curves for each series, and then combining the curves by the probability equation of union into a single flood-frequency curve. Analysis of flood data for the Cataldo gauge indicates that the winter rain-on-snow events dominate the combined frequency curve above the 10-year-flood level. The longest peak-flow record is from Cataldo (1911-1999), and the maximum flood of record was 79,000 cfs in January 1974.

Dissolved Metals

The bedrock aquifer historically has created some contamination problems, particularly in the adit drainage from the Bunker Hill Mine. This drainage is highly acidic (pH = 2.8), has a high concentration of dissolved metals (110 mg/L of zinc), and has a significant flow rate (3-4 cfs [85-115 L/s]) (Box et al. 1997). The Bunker Hill adit water has been treated to remove metals since the mid-1970s, eliminating what was previously the largest point source of zinc to the South Fork (about 2,000 lbs/day [1,000 kg/day]) (Box et al. 1997). Bunker Hill adit water continues to be treated using the central treatment plant (CTP), and the sludge from the CTP is disposed in an active, unlined containment pond on top of the CIA, located in the Bunker Hill box (EPA 2004b [July 27, 2004]).

Currently, the shallow aquifer systems are the major contributors to the high levels of dissolved metals found in the river, particularly during the low flow periods in late summer and fall when surface water concentrations often exceed 2 mg/L Zn (Clark 2003, Figs. 4 and 6). Infiltration and seepage through the 1-2 m of tailings-contaminated sediments distributed over the floodplain, as well as infiltration into, and seepage from impoundments and tailings ponds contribute high metal loads to the groundwater in the shallow aquifer. Many of the groundwater monitoring wells in the shallow aquifer have total metals exceeding 10 mg/L, most of which is dissolved zinc (TerraGraphics 1996, p. 34-36, 2005). Zinc levels in the Government Gulch area adjacent to the former smelter have exceeded 100 mg/L (EPA 2000, p. 4-9).

In the past, one of the most important sources has been the seepage from the CIA (Rouse 1977). One of the seep areas is so localized that it has created piping and subsidence of the bed of Interstate 90 (Dawson 1998). However, the current and likely future contributions from this source are disputed (EPA 2004c; Rust 2004). These seeps still appear to be discharging into the river under the Interstate 90 embankment (see Rust 2004), but EPA believes that it has largely corrected this problem by installing an impermeable cap on the CIA and diverting the Bunker Hill adit drainage directly to the wastewater treatment plant rather than ponding it on top of the CIA (EPA 2004c). However, water-containing sludge is still disposed into a large unlined pit on top of the CIA. The effect of remedial actions on the metal content of groundwater and metal loads entering the river was uncertain as of 2001 (Borque 2001; EPA 2000; TerraGraphics 2001). Interim studies suggest some progress in reducing metal loads; however, groundwater remains heavily contaminated in this area, and continued seepage still contributes a high load of dissolved zinc to the river.

Another major source of dissolved metal loadings is groundwater return flow to the river, most of which occurs below the surface of the river.

Typically, the river loses flow to the groundwater in reaches where the valley aquifer widens and regains groundwater return flow (generally with a significant dissolved-metal load) where the valley aquifer narrows. The USGS investigated river-flow losses and metal loading by the return flow along two reaches in the middle basin: a 4.8-mile (7.7-km) reach at Osburn Flats and a 6.5-mile (10.5-km) reach in the Kellogg-Smelterville area (Barton 2002). These measurements were made in July, near the end of the high stream flow and then during the September and October 1999 base flows. For the Osburn Flats reach, Barton (2002) estimated that seepage flow carried 218 lbs (99 kg) of dissolved zinc per day into the river. The Kellogg-Smelterville reach was estimated to contribute 730 lbs (122 kg) of dissolved zinc per day.

EPA had the study of the Kellogg-Smelterville reach reproduced in 2003 after some of the major remedial actions at Bunker Hill had been completed. The new study showed 63% less zinc (464 lbs/day) and 19% less cadmium coming from this reach (CH2M Hill 2004). However, lower groundwater levels in 2003 than in 1999 also may account for some of the difference. The higher 1999 levels could have resulted both in a greater groundwater flux and in the groundwater rising through aquifer materials that previously had substantial opportunity to oxidize, thus making the metal more soluble. It is also possible that in-stream remedial activities occurring during the 1999 study could have released additional dissolved metal into the South Fork of the Coeur d'Alene River.

Substantial additional investigation will have to be completed to obtain a thorough understanding of groundwater-movement dynamics and the incorporation of dissolved metals from the aquifer materials.

EPA estimates that 41% of the total zinc loading in the Coeur d'Alene River as it enters Lake Coeur d'Alene comes from the area included in the box (URS Greiner, Inc. and CH2M Hill 2001i). The increase in zinc loadings as the South Fork travels from Mullan to its mouth is shown in Figure 3-8 and in more detail during the 1999-2000 water year in Figure 3-4. EPA estimates that the river is carrying 23% of the total zinc load when it reaches Osburn. By the time it gets to Pinehurst, it is carrying 78%. The North Fork adds another 7% when it joins the South Fork above Cataldo. The remaining 15% is picked up, presumably from pore water of the riverbed sediments and groundwater seeping through the river banks, between Pinehurst and the mouth of the river at Harrison.

Ecologic Community

Before the mining era, the valley walls in the middle basin, like the upper basin hills, were heavily forested. Large white pine flourished in the valley bottom, and large red cedars grew in marshy areas. Grassy openings

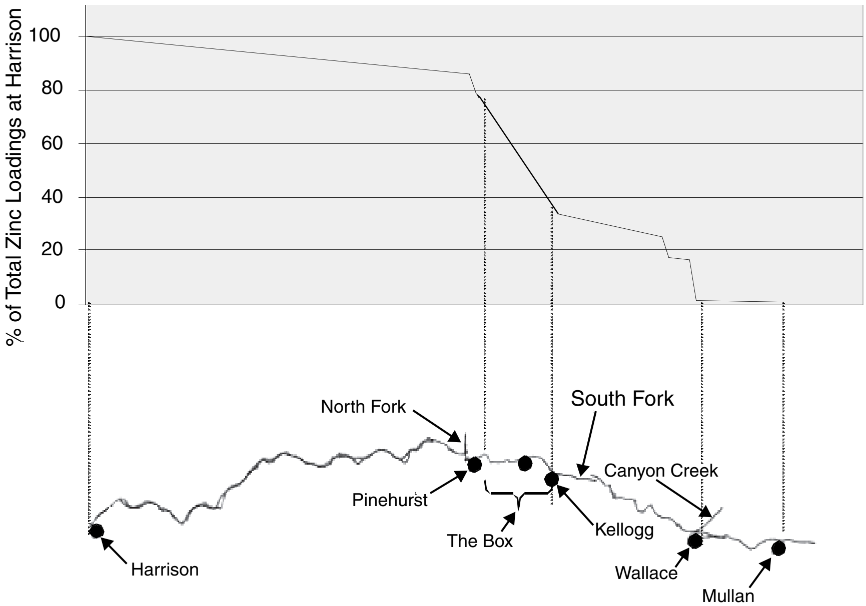


FIGURE 3-8 Zinc loadings to the Coeur d'Alene River as a percent of the total loadings at Harrison. SOURCE: Data from URS Greiner, Inc. and CH2M Hill 2001i.

were sparse (Box et al. 1999, p. 5). The riparian areas also contained alder and large cottonwoods. Wildlife was probably plentiful and diverse, and the waters would have supported large populations of native fish such as cutthroat trout, bull trout, mountain whitefish, and sculpin.

The settlement and establishment of mining activities in the basin substantially degraded all of these habitats. The hills and valleys were logged to provide timber for building structures and for fuel. The river was channelized, blocked, overwhelmed with mine tailings, and contaminated.

As in the upper basin, some of the hill forests have regenerated over the past century. However, the hillsides adjacent to Smelterville, Wardner, and Kellogg are contaminated with heavy metals from smelter emissions (TerraGraphics 2000; Sheldrake and Stifelman 2003), and an area of about 1,050 acres remained denuded of vegetation in 2000 (EPA 2000, p. 4-21). Soils on these hillsides have high acidity and lack organics and nutrients for native plant revegetation. EPA and the state of Idaho have attempted to replant, treat with lime, and hydroseed these hillsides to reestablish a natural vegetative cover. As of the first 5-year review, however, these efforts have not been successful in reestablishing ground cover (EPA 2000). Very little reforestation has occurred on the valley floor, much of which is covered by settlements, roads, former mill sites, and waste repositories that support little more than grasses.

Nor has the river channel recovered. Many of the problems created during the 20th century remain and, at least from an ecologic perspective, in some cases, have gotten worse with the increased channel stabilization that has accompanied new construction activities (such as the construction of an interstate highway through the valley), remediation efforts undertaken pursuant to the records of decision (RODs) for OU-1 and OU-2, and attempts to reduce flooding.

Although the middle basin historically has been the most affected by mining activities, fish still exist in this stretch of the river. However, fish-species richness and fish-population abundance are reduced, and sculpins (a species particularly sensitive to metals) are largely absent. No fish are present in the most heavily affected areas (CH2M-Hill and URS Corp. 2001, p. 2–23). The benthic macroinvertebrate community, particularly downstream from the box (as measured by diversity, Ephemeroptera, Plecoptera, Trichoptera [EPT] index, and abundances) has improved through the 1980s, especially after direct discharge of tailings ceased. However, the benthic community remains affected and metal-sensitive taxa (such as mayflies) remain largely absent (Stratus Consulting Inc. 2000).

THE LOWER BASIN

The lower basin differs in almost all respects from the upper and middle basin of the Coeur d'Alene River. In this reach, the river becomes deeper and takes on a meandering pattern with its bed predominantly composed of sand and silt. The river gradient is nearly flat, and during much of the year the river is essentially an arm of Lake Coeur d'Alene. In low flow, the channel is confined by natural levees bordered by broad floodplains containing wetlands, "lateral lakes," and agricultural lands. The dominant feature of this reach is extensive and rich wetland wildlife habitat, with little human settlement.

Human Community

Although housing units are scattered along the few roads in the lower basin and some settlements such as Cataldo are located there, the population is small and the U.S. census does not provide any information about communities in the lower basin. The small town of Harrison, located at the mouth of the river, actually lies predominantly outside the lower basin, along the shoreline of Lake Coeur d'Alene and is included with the lake communities. The committee lacks formal demographic data, but informal observations suggest that the lower basin is a transition zone, reflecting some of the aspects of the communities higher in the basin but also showing signs of being part of the growing recreational development, which characterizes Lake Coeur d'Alene.

Geology and Fluvial Geomorphology

The dominant geological feature in the lower basin is the change from steep valley walls to broad alluvial floodplains. The floodplains are bordered by steep hillsides, but the hills are relatively low. The lower Coeur d'Alene River valley is essentially the delta of the Coeur d'Alene River into Lake Coeur d'Alene. Here, the lake waters naturally backflood the river channel all the way to the Cataldo Mission. This arm of the post-Ice Age lake was progressively filled with sediment as the delta front (now near Harrison) migrated down-valley. The deep river channel feeding the delta front is carved into earlier fine-grained delta-front lake deposits as it extends down-valley, and the cohesive character of these deposits has inhibited significant lateral migration of the channel through time. Portions of the lake became isolated by the lengthening river channel and its levees, creating what are known as lateral lakes. These lateral lakes gradually shallow and infill with marsh deposits. At Cataldo Flats, the valley-fill sediments are about 160 feet (50 m) thick, and below Rose Lake (less than 10 miles [16 km] below Cataldo), the thickness has increased to 400 feet (120 m) (URS Greiner, Inc. and CH2M Hill 2001j, p. 2-2). The river has a typical meandering pattern in the lower reach, with point bars at the inside of meander bends. Although there are older, prehistoric meander scrolls through the lower reach (Bookstrom et al. 2004a), there has apparently been little channel migration since the mining era began (Box 2004).

The floodplains vary in width from about 1,000 feet (300 m) at Cataldo to about 3 miles (5 km) near the river's mouth. Along the lower reach, distributary streams and man-made canals diverge from the river, connecting to lateral lakes, which range to more than 600 acres (250 hectares), and thousands of acres of wetlands. The soils here are rich enough to support substantial wetlands vegetation. Approximately 9,500 acres (3,800 hectares) of floodplain along this reach have also been converted to agricultural use (CH2M-Hill and URS Corp. 2001, p. 2-29).

The metal-contaminated deposits on the floodplain of the lower segment are thinner than those along the middle stretch and generally are composed of finer materials. Metal-enriched levee silt and sand deposits extend across bank wedges and natural levees, generally thinning to 1.5 feet (0.5 m) at a distance of about 260 feet (80 m) from the channel banks (see Figure 3-9) and fining away from the river, toward lateral marshes and lakes. In these lateral marsh areas, approximately 6-17 inches (15-44 cm) of dark gray, metal-enriched silt and mud overlie the silty peat deposited before the mining era (Bookstrom et al. 2001, p. 24). The soil near the distributary streams and man-made canals carrying water to these lakes and wetlands may be covered by thicker and metal-enriched sand splays deposited by floods as they overtop the river banks. These splays fan out across the floodplain, typically cover a couple of hundred acres (about 100 hectares), and are several meters thick near the river, tapering to less than 1.5 feet (0.5 m) at their end (Bookstrom et al. 2001, p. 25, Fig. 8).

Another location with heavily contaminated sediment cover is Cataldo Flats, where the mining companies deposited contaminated materials dredged from the river channel. These dredged materials cover 2,000 acres (800 hectares) to a depth of 25-30 feet (7.5-9 m) (URS Greiner, Inc. and CH2M Hill 2001j, p. 2-6). During the first 2 years of operation, the dredge removed 1.8 million metric tons (2 million U.S. tons) of material from the channel, but each year the channel filled up again during the flood season (URS Greiner, Inc. and CH2M Hill 2001j, p. 2-6). The dredge continued operating until 1968.

The river channel has much thicker layers of contaminated sediments covering the premining materials. This contaminated channel sand is typically 9 feet (2.6 m) thick across the 260-foot (80 m)-wide channel (Bookstrom et al. 2001, p. 23). The fact that the channel deposits are substantially thicker than the floodplain deposits suggests that the premining river channel in this reach was much deeper than it is today. This is supported by a 1932 report quoting steamboat operators who remembered the channel being navigable "with 40 to 50 feet of water" (12-15 m) up to Cataldo

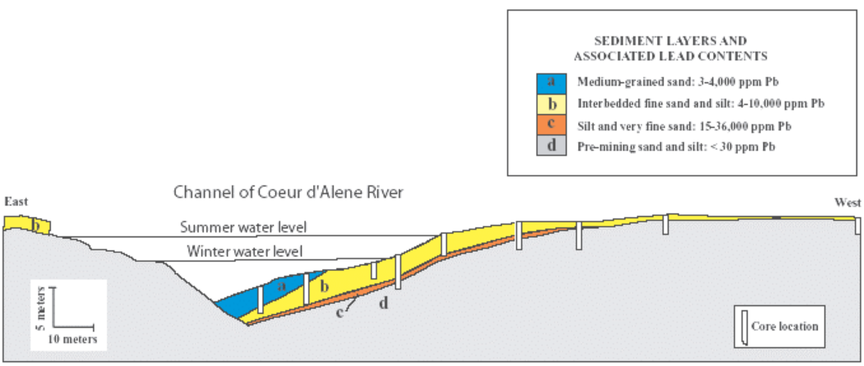


FIGURE 3-9 Cross-section of Coeur d'Alene River near Killarney Lake showing lead content of sediments in cores from the channel and floodplain. SOURCE: Balistrieri et al. 2002a.

(URS Greiner, Inc. and CH2M Hill 2001j, p. 2-6). By 1932, the river had "only 12 to 15 feet (3.5 to 4.6 meters) of water in the main channel in this region, both the channel and the main stream being obstructed here and there by large bars of mine wastes and tailings" (URS Greiner, Inc. and CH2M Hill 2001j, p. 2-5).

The USGS estimates that the river bed contains 51% of the lead in the entire lower basin (Bookstrom et al. 2001, Table 12). These channel deposits are mostly silty fine-to-medium sand (Bookstrom et al. 2004b, slide 22; Box 2004, slide 19).

Metal-enriched sand and silt also form oxidized bank-wedge deposits along the river channel, covering the premining-era levees of gray silty mud. However, the metal content of bank material at the upper end of the lower basin is relatively low (about 2,000 mg/kg) (compared with bank deposits in other reaches) as a result of the contaminated sediment carried by the South Fork being diluted by the clean sediment coming in from the North Fork (Box 2004, slide 28).[7] The volume of riverbank material is about 1.7 million cubic yards (1.4 million m^3), and it contains 4% of the lead in the lower basin (Bookstrom et al. 2001).

The remaining 45% of the lead in the lower basin is in the subaerial levees (10%), in sediments spread over the floodplain and deposited in the lateral lakes and marshes (18%), or in the dredge soils on Cataldo Flats (17%) (Bookstrom et al. 2001; Box 2004). The only wetlands and lateral lakes in the lower basin that do not receive frequent deposits of contaminated sediments are those located south of the railroad embankment, which forms a protective levee (Bookstrom et al. 2004a). In the 1999-2000 water years, approximately 80% of the lead load transported to the lake at Harrison was derived from the main stem river below the confluence of the North and South Forks (Clark 2003, Fig. 12). The peak flow in that year was about 27,000 cfs or a spring flood with a 3- to 4-year recurrence (Figure 3-7).

The preceding discussion suggests that the major source of high-metal-content sand and silt remobilized during floods is bedload scoured from the channel and that the main-stem channel, therefore, is a major source of metal-contaminated material that is delivered to the lateral lakes, marshes, and Lake Coeur d'Alene.

Complicated chemical processes occur once the sediments are deposited in the oxygen-scarce wetlands and lake bottoms. These processes tend

[7]However, by 11 km downstream of the confluence, the recent-flood-deposited bank material again has a high metal concentration (4,500 parts per million). It appears that the high-metal-content sandy bank deposited in the 1995 and 1996 flood flows in the lower main stem is derived mostly from scouring and redepositing the high-metal-content channel material (Box 2004, slide 28).

to make the metals more biologically available as described in a recent USGS report (Bookstrom et al. 2004a):

> In reducing environments of marshes and lakes, metallic oxy-hydrides, transported from oxidizing environments on levee uplands, are reduced. Reduction breaks down metallic oxy-hydrides and releases metallic ions, which combine with sulfide ions (produced by sulfate-reducing bacteria) to form authigenic sulfidic-metallic materials that are non-stoichiometric and amorphous to nano-crystalline. These materials have enormous surface area, and are much more chemically reactive than detrital grains of crystalline metallic sulfide minerals. The lead in these authigenic sulfides is therefore much more bio-reactive and bio-available than the lead in detrital grains of galena.

Hydrology

The flow of the main stem of the Coeur d'Alene River is gauged at Cataldo, where the mean annual flow for the 1911-2003 record is 2,531 cfs (72 m^3/s), with late summer flows below 500 cfs (14 m^3/s) (USGS 2004). Flow in the lower main-stem channel is nearly imperceptible for most of the summer and fall. Bank erosion during this period occurs from waves generated by wind and boat wakes. Because these low flows result primarily from groundwater discharge, they contain high levels of dissolved contaminants such as zinc.

Since 1886, 13 major floods have inundated the floodplain of the Coeur d'Alene River valley, and 26 lesser floods have flooded much of the valley floor (Figure 3-6). Since mining began, the extent and severity of overbank flooding has probably increased as a result of channel aggradation caused by sedimentation of mine wastes and reduced forest cover. During flood flow, the river breaks out into natural or artificial channels and through levee breaches to the large lateral marshes and lakes. During large floods, levees are overtopped and most of the valley floodplain is inundated. Such overtopping is relatively common, having a recurrence period of 1.5 years (URS Greiner, Inc. and CH2M Hill 2001j, p. 2-14).

Because the floodplain of the Coeur d'Alene River generally slopes away from the tops of the natural levees that flank the river, if floodwater overtops the levees or flows through low passes in the levees, it tends to cover most of the floodplain. Annual spring floods commonly inundate the lower valley, and major spring floods inundate most of the floodplain. The more severe rain-on-snow winter floods commonly occur when the lake level has been drawn down so that the hydraulic differential in the segment is unusually high. One result of this difference is that a given amount of winter flood flow is less likely to overtop the river levees than the same amount of spring flow. However, because the winter rain-on-snow floods usually move more quickly, they are likely to scour more tailings sediments from the channel and, if they do overtop the levees, deposit them on the floodplain.

Ecologic Community

Before mining began, the natural levees along the lower reach of the river would have been extensively forested with cottonwood and alder trees. These natural vegetative types continue to exist today, although probably in less abundance because of the covering of the natural levees with contaminated sediment and man-made alterations along the banks for recreational and other purposes. The wetlands and uplands vegetation in the downstream reach of the Coeur d'Alene River were not significantly affected by the mining operations. However, extensive areas have been cleared and drained for agricultural purposes (for pasture and cropland) and for urban and recreational development.

The lateral lakes and wetlands provide areas for waterfowl nesting, feeding, and other activities. Twenty-five species of waterfowl have been identified in the vicinity of the lateral lakes during spring and fall migrations, and more than 280 bird species are found throughout the Coeur d'Alene River basin (CH2M-Hill and URS Corp. 2001, p. 2-17). As described in Chapter 7, the contaminated sediments are implicated in the poisoning of many waterfowl every year and may be having negative impacts on other species of birds using these habitats (CH2M-Hill and URS Corp. 2001, p. 2-25).

Tundra swans are particularly vulnerable to lead exposure and intoxication for multiple reasons. In particular, swans that occupy the Coeur d'Alene River basin to a large degree are either en route to the northern breeding grounds during their migratory period or heading south during wintering periods. Therefore, when they arrive in the Coeur d'Alene River basin, they are searching for available habitat, particularly for food and resting areas. With their long necks, tundra swans can, as they feed, easily reach sediments beneath a meter of water. In the process of sifting through sediments, often searching for root tubers and other food products, tundra swans ingest sediment. In the Coeur d'Alene River basin—in particular the lateral lakes feeding areas—the sediment can be heavily contaminated with lead. With such feeding habits, and with their preference for the habitats of the lateral lakes which are heavily contaminated, tundra swans are at a great risk. The risk is confirmed by substantial data on swan mortality in the Coeur d'Alene Ecological Risk Assessment (CH2M-Hill and URS Corp. 2001). See further discussion in Chapter 7 of this report.

The main stem of the Coeur d'Alene River holds many species of fish, including native salmonid species and several exotic (that is, introduced) species such as rainbow trout, chinook salmon, bass, tench, northern pike, and tiger muskellunge, although apparently there is not enough information to determine the status of the fish populations (URS Greiner, Inc. and CH2M Hill 2001i, p. 2-24) or the diversity and abundance of benthic macroinvertebrates.

Numerous cold-water and warm-water fish species inhabit the lateral lakes and the Idaho Department of Fish and Game actively manages a warm-water fishery in several of these lakes. Populations of 19 nonnative fish species, such as rainbow trout, chinook salmon, bass, tench, northern pike, and tiger muskellunge, have been introduced into these lakes as well as the main stem of the river. These introductions have substantially altered the dynamics of the system (CH2M-Hill and URS Corp. 2001, p. 2-24) and have complicated the effort to protect many native species such as cutthroat trout (for example, through the introduction of predators).[8]

LAKE COEUR D'ALENE

Lake Coeur d'Alene is a large body of water approximately 25 miles (40 km) long with a width of 1-2 miles (1.6-3.2 km) along most of its length. The lake has a surface area of approximately 50 square miles (130 km^2) and 133 miles (215 km) of shoreline. The lake has become a heavily used tourist and recreational facility—for both boating and fishing—for residents throughout the Northwest.

Human Community

Most of the shoreline of Lake Coeur d'Alene is relatively unpopulated, although residential development on the shoreline is increasing. There are a few settlements at the south end of the lake, which lies within the reservation of the Coeur d'Alene tribe, but the only two communities included in the U.S. census are Harrison (at the mouth of the Coeur d'Alene River) and the city of Coeur d'Alene (at the north end of the lake). Table 3-3 summarizes some of the demographic characteristics of these communities.

Harrison shows the same high poverty rates and older population as the communities in the middle and upper basin, but the housing stock is generally newer. It is experiencing a rapid influx of new residents, a greater percentage of residents has graduated from college, and the median income is substantially higher.

[8]The Natural Resources Damages Assessment found only 11 species of native fish in the Coeur d'Alene basin compared with 19 species of nonnative fish found there (Stratus Consulting, Inc. 2000).

The city of Coeur d'Alene, however, is a relatively large community that has been growing rapidly (a 73% increase from 1980 to 2000) (Idaho Department of Commerce 2004). The median age of the population is below the national average, although almost 15% of the residents are more than 65 years old, suggesting that the community is becoming a retirement community. The median household income is substantially higher than that of other communities in the basin, although it is below the Idaho and national averages. The poverty rate and the unemployment rate were lower than those for basin communities, and, most dramatically, almost 30% of the housing units were built after 1990, and almost 80% of the residents had been living in their homes for 10 years or less in 2000. This rapid growth and change has been fueled largely by growth in tourism and recreational developments. This trend has been echoed in much of the area around the northern end of the lake with the construction of vacation homes.

The reservation for the Coeur d'Alene tribe encompasses the southern part of the lake. The U.S. census found 4,465 people living on the reservation in 2000, and about 17% of those identified themselves as American Indians (U.S. Census 2004).

Geology and Geochemistry

Lake Coeur d'Alene was created by the catastrophic glacial-outbreak floods from the Pleistocene Lake Missoula. These floods filled the lower Coeur d'Alene River Valley with coarse outwash forming a massive dam blocking the river near the city of Coeur d'Alene. The lake filled behind this

TABLE 3-3 Demographic Characteristics of Lake Coeur d'Alene Communities

Demographic	U.S.	Idaho	Harrison	Coeur d'Alene
Population			267	34,514
Median age (years)	35.3	33.2	46.1	34.8
Older than 65 (% of population)	12.4	11.3	19.5	14.8
Household income ($ thousands)	42.0	37.6	35.8	33.0
Unemployment rate (%)	5.8	5.8	7.3	7.9
Below poverty level (% individuals)	12.4	11.8	20.3	12.8
% with bachelor's degree	24.4	21.7	29.4	19.5
Moved from out of state since 1995 (%)	8.4	15.3	25.1	21.8
% of owner occupied units occupied by the same family for >30 years (%)	9.7	6.9	7.4	4.6
Vacant housing units (%)	9.0	11.0	21.0	6.3
Houses older than 40 years (%)	35.0	27.7	46.5	28.2

SOURCE: U.S. Census 2004.

natural dam, flooding the valleys of the Coeur d'Alene River and the St. Joe River (the lake's other major tributary) to the south. Except near its outlet and the mouths of its major tributary rivers, the banks of the lake are formed by the rock of the ancient valley walls, rising to low hills.

The maximum depth of the lake exceeds 200 feet (61 m), and its average depth is 70 feet (21 m). The Pleistocene lake was originally somewhat higher than it is now, extending up to about Kellogg (Box et al. 1999, p. 5). The erosion of the channel through Missoula flood gravels by the Spokane River gradually lowered the lake's surface elevation to the bedrock at Post Falls. The lake level was then raised slightly with the construction of a dam at Post Falls in 1906.

The Coeur d'Alene River has carried immense amounts of sediment—containing 300-400 thousand metric tons (350-440 thousand U.S. tons) of lead—into the lake (Bookstrom et al. 2001, Table 15). Horowitz et al. (1995a) estimated that 75 million metric tons (83 million U.S. tons) of metals-contaminated sediments had been deposited on the bottom of Lake Coeur d'Alene since the onset of mining. The coarser sediments tend to settle near the point where the river enters the lake, forming 20-foot (6-m)-thick delta-front deposits of metal-enriched sand that slope from the river-mouth bar almost a kilometer (0.6 mile) from the delta front to the bottom of the lake (Bookstrom et al. 2001). Finer sediments have been carried farther into the lake, creating a metal-enriched sediment layer up to 119 cm (3.9 feet) thick closest to the Coeur d'Alene River delta, thinning to 10-14 cm (4-5.5 inches) near the city of Coeur d'Alene[9] (Horowitz et al. 1995a).

Lake-bottom sediment samples (one sample per km^2) have a mean lead concentration of 1,900 mg/kg but range up to 7,700 mg/kg (Horowitz et al. 1993, p. 410, 1995b). Nearshore areas show much lower levels. For instance, seventeen beaches and common-use areas along Lake Coeur d'Alene tested for lead contamination showed an average lead concentration of less than 200 mg/kg for all sites except Harrison Beach, which averaged 1,250 mg/kg (URS Greiner, Inc. et al. 1999). Harrison is adjacent to the mouth of the Coeur d'Alene River where, as indicated above, the deposition of sediment from the river continues to build a large delta out into the lake.

Some of the fine contaminated sediment is carried completely across the lake and into the Spokane River. This process is particularly evident during spring floods (see Chapter 4 of this report for further discussion).

[9]Very little contaminated sediment has been found in the far southern part of the lake. However, some landowners in this area are concerned about possible contaminants leaching out of the railroad embankment and causing serious localized contamination problems (Hardy 2004). The committee's charge did not include evaluation of this issue, and the committee has not evaluated it.

EPA estimated that in water year 1999, approximately 50% of the dissolved zinc input was converted into the particulate form within the lake (URS Greiner, Inc. and CH2M Hill 2001k, p. 5-90), which presumably settles to the lake bottom. Soluble zinc within the lake will interact with biotic and abiotic components in the water column that are capable of affecting the disposition and transport of the metal. For instance, soluble zinc coming from the Coeur d'Alene River will associate with phytoplankton (and become sorbed to the organic matrix of the cell or incorporated into the silica in diatom frustules). Upon dying, the phytoplankton fall out of the water column and become incorporated in sediments. Zinc also may associate with dissolved or particulate organic matter or with inorganic species, particularly ferric oxyhydroxides. Samples taken from the lake bottom contain a ferric oxyhydroxide flocculent material that is enriched in zinc (Woods 2004).

The fate of the zinc within the sediments is complex, related to the oxic state of the sediments and the geochemical associations. The zinc can remain bound to organic or inorganic substrates, or it can become soluble after oxidation. The oxidation of organic matter in the sediment requires a terminal electron acceptor. Oxygen and nitrate, both electron receptors, become depleted near the sediment-water interface. Below this, sulfate becomes reduced to sulfide. The sulfide reacts with iron and trace metals, such as cadmium, copper, lead, and zinc (Di Toro 2001), which results in the formation of amorphous metal monosulfide precipitates, such as FeS, PbS, and ZnS, that will effectively sequester zinc.

The solubility of FeS is greater than that of CdS, CuS, PbS, and ZnS. Consequently, FeS is a reservoir that provides sulfide to react with cadmium, copper, lead, and zinc. The solubility of metals from the metal monosulfides is less than that of the metals associated with ferric oxyhydroxide or particulate organic matter. There are limited data for the lake sediments in which this speciation has been determined. Tests that have been conducted suggest that not all the zinc and lead are present as ZnS and PbS, but that some metal is contained in other forms, likely associated with ferric oxyhydroxide or particulate organic matter (Harrington et al. 1999; Horowitz et al. 1995a; see Chapter 4 for further discussion).

The geochemistry of the lake bottom is of concern because the processes occurring there determine the extent to which the metals in the contaminated sediments will become biologically available and thus a risk to the fish and benthic populations. If the metals remain in the insoluble form, these risks are reduced. Maintaining a lake environment that will keep these metals insoluble is a primary goal of a lake management plan being developed (see Chapter 8 of this report).

Hydrology

With the construction of Post Falls Dam in 1906, the control gates allowed the lake level to be raised 6-7 feet (2 m). In 1940, the dam was raised another foot (0.3 m). The dam gates are used to reduce outflow from the lake and to control lake level at a fixed elevation from about June to September. In September, the power company manipulates the gates to increase the outflow rates for power generation and cause the lake level to fall about 1.5 feet (0.5 m) per month until mid-November to provide storage capacity for spring runoff. From mid-November to May or June, the gates are fully open, and the lake seeks its natural low winter level. After spring runoff, the gates are again used to control outflow and lake level.

Lake levels are also affected by flood flows entering the lake from the Coeur d'Alene and Saint Joe Rivers (see Figure 3-10). These floods can raise the water level 12-14 feet (about 4 m). The 1933 flood raised the lake level 19 feet (5.8 m) above the winter low (Kootenai County 1998).

In 1999, USGS investigators also observed the spring flood with its suspended sediment load coursing across the surface of the lake to the Spokane River (Woods 2004). They hypothesized that this occurs because the river waters warm faster than the lake waters and, therefore, essentially float

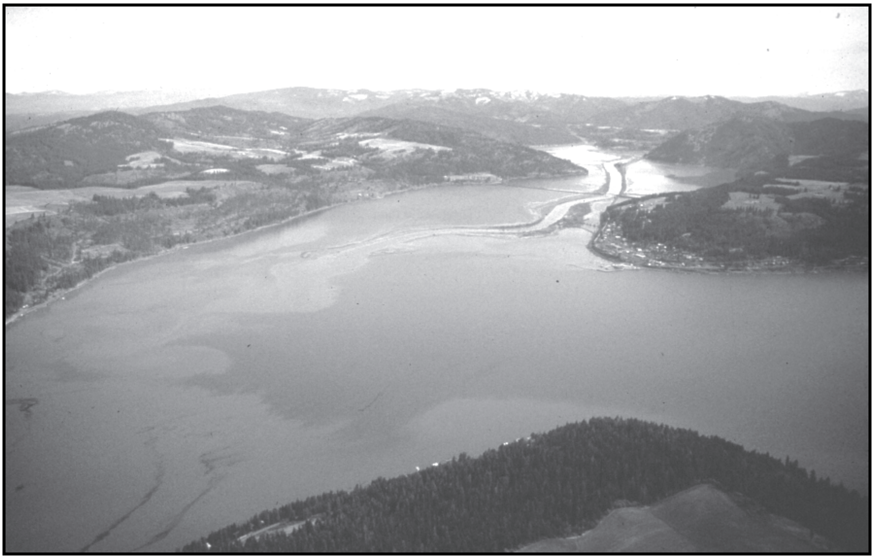


FIGURE 3-10 Coeur d'Alene River delta and inflow plume adjacent to Harrison, Idaho, on Lake Coeur d'Alene. SOURCE: Woods 2004.

across the surface of the lake. They intend to do more research to document this phenomenon. The existence of these flows would indicate that more contaminated sediment is being delivered to the Spokane River than otherwise might be expected (see Chapter 4 of this report for further discussion).

Ecologic Community

Lake Coeur d'Alene is home to a diverse mix of both cold-water and warm-water species of fish. Several of these fish, however, are exotic species that were artificially introduced there. The populations of at least some of the native species (westslope cutthroat trout, bull trout, mountain whitefish, yellow perch, northern pikeminnow) are probably being stressed by the introduced species.

The richness and abundance of the benthic community is greatest in the shallow waters and at the southern end of the lake, below the mouth of the Coeur d'Alene River. However, EPA concludes that there is no good evidence that these differences are caused by the deposition of contaminated sediments (CH2M-Hill and URS Corp. 2001, pp. 2-26 to 2-27). Some of the difference may result from the higher nutrient loads in the southern portion of the lake. Nevertheless, the contaminated sediments provide at least a potential threat to the benthic community and fish life. The extent of this threat will, as discussed above, depend significantly on geochemical reactions taking place on the lake's bottom. The responses of benthic invertebrates to the metal-contaminated Lake Coeur d'Alene sediments have been studied only minimally (Hornig et al. 1988; CH2M-Hill and URS Corp. 2001, p. 2-26) as has the relationship of benthic communities to the presence of metals within sediments. Further, although the metal flux has been investigated, there has been no study of the influence of invertebrates on the bioavailability of metals in Lake Coeur d'Alene, a potentially important factor in metals dynamics (Kennedy et al. 2003).

SPOKANE RIVER

The Spokane River drains Lake Coeur d'Alene at its northern end through the Rathrum Prairie to Post Falls, where it spills over the Post Falls Dam and cascades over a natural 40 foot (12 m) bedrock waterfall. From Lake Coeur d'Alene, the Spokane River flows at a relatively flat gradient through a 3- to 8-mile (4.8- to 12.8-km)-wide valley extending westward to the junction with the Little Spokane River. Along this route, the river flows over five more dams, four of which are within the city of Spokane (URS Greiner, Inc. and CH2M Hill 2001l, p. 2-7). At its lower end, the valley narrows, and the river is largely contained in the reservoirs behind Long Lake Dam and Grand Coulee Dam.

Human Community

The city of Spokane, with a population close to 200,000 in the year 2000, is the largest community along the Spokane River. The unincorporated area of Opportunity, Washington, with a population of 25,000, and Post Falls in Idaho, with a population of 17,000, are other large communities (Table 3-4). Post Falls demonstrates many of the same demographic characteristics as the city of Coeur d'Alene—for instance, a very rapid growth rate and a relatively young population. The population growth in Spokane and Opportunity is much lower, although these communities have also grown slightly faster than the national average.

All these communities use the Spokane aquifer as their primary source of drinking water (URS Greiner, Inc. and CH2M Hill 2001l, p. 2-6). This aquifer covers the entire valley and extends from the bedrock below the valley as much as several hundred feet up to the surface. Lake Coeur d'Alene and the upper Spokane River are primary sources of recharge to this aquifer.

The reservation for the Spokane tribe lies along the lower part of the Spokane River where it joins the Columbia River. According to the U.S. census, approximately 2,000 people lived on the reservation in 2000.

The River and Its Contamination

When not contained in a reservoir, the Spokane River above the city of Spokane is 200-400 feet (60-120 m) wide with a gravel bottom and many of the characteristics of a braided stream (URS Greiner, Inc. and CH2M Hill 2001l, p. 3-3). Because of the substantial hydraulic buffering capacity

TABLE 3-4 Demographic Characteristics of Larger Communities Along Spokane River

Demographic	U.S.	Idaho	Post Falls	Opportunity	Spokane
Population			17,247	25,065	195,629
Median age (years)	35.3	33.2	31.3	35.8	34.7
Older than 65 (%)	12.4	11.3	9.8	14.8	14.0
Household income ($1,000)	42.0	37.6	39.1	38.7	32.3
Below poverty level (% individuals)	12.4	11.8	9.4	9.0	15.9
% with bachelor's degree	24.4	21.7	15.9	20.3	25.4
Moved from out of state since 1995 (%)	8.4	15.3	26.1	9.8	10.1
Vacant housing units (%)	9.0	11.0	4.9	5.4	7.3

SOURCE: U.S. Census 2004.

of Lake Coeur d'Alene, the substantial variations in the flows experienced in the Coeur d'Alene River are not reflected in the Spokane River. Indeed, the Lake is managed by allowing the water level to fall during the winter so that it can store the spring flood flows and reduce downstream flooding. As a result, the flood with a 100-year recurrence interval is projected to carry only slightly over a third more water than the 10-year flood (URS Greiner, Inc. and CH2M Hill 2001l, Table 2.3-1).

The RI states that the Spokane River water frequently exceeds water-quality standards for zinc, lead, and cadmium (URS Greiner, Inc. and CH2M Hill 2001l, p 5-1). The major source of these metals is the outflow from Lake Coeur d'Alene.

Some of the lead is contained in fine sediment that traverses Lake Coeur d'Alene during the spring runoff. This sediment comes predominantly from the channel in the lower basin of the Coeur d'Alene River (Clark 2003). The re-suspension of previously deposited sediments is another major source (Grosbois et al. 2001; Box and Wallis 2002). The sediment is largely deposited behind the upstream dams, along shoreline beaches, and in backwaters behind channel obstructions. Concentrations of lead exceeding 2,000 mg/kg have been measured in shoreline sediment (EPA 2002, Table 7.1-21). Elevated arsenic levels, also a source of concern, are generally associated with high lead levels. Polychlorinated biphenyls, which are not derived from mining wastes, are also a contaminant problem in the Spokane River (EPA 2002).

Approximately 70% of the dissolved zinc entering Lake Coeur d'Alene flows out into the Spokane River, resulting in total annual dissolved zinc loadings ranging from 225,000 kg (496,000 lbs) to 767,000 kg (1,690,000 lbs) per year (URS Greiner, Inc. and CH2M Hill, 2001k, p. 5-4; Clark 2003). The dissolved zinc concentrations exceed ambient water-quality standards throughout most of the year and remain relatively constant through the upper part of the river down to the city of Spokane. Below Spokane, they decrease to the point where water-quality standards are not exceeded in Long Lake (URS Greiner, Inc. and CH2M Hill 2001l, p. 5-7, 5-8). Unlike the situation in Coeur d'Alene River, the zinc concentrations (as well as the zinc loadings) increase with increased discharge (URS Greiner, Inc. and CH2M Hill 2001l, p. 5-8).

Ecologic Community

The several dams along the Spokane River provide artificial lacustrine (lake) habitats with substantial fish populations but, at the same time, interfere with the migration of salmonid species. Most of the river has limited (narrow and sparsely vegetated) riparian habitat and very little palustrine (wetland) habitat (CH2M-Hill and URS Corp. 2001, p. 2-20).

However, the shorelines around some reservoirs such as Long Lake and Nine Mile Reservoir do have substantial riparian vegetation (CH2M-Hill and URS Corp. 2001, p. 2-29).

The diversity of benthic invertebrates is lower than normally would be expected for a river like the Spokane (CH2M-Hill and URS Corp. 2001, p. 2-24), but the fish community is "diverse and moderately productive" (CH2M-Hill and URS Corp. 2001, p. 2-25). More than 20 species of fish have been identified, although many of these, like the rainbow trout, have been artificially introduced into the river for recreational purposes.

LOOKING AHEAD

This chapter has focused primarily on the current conditions in the basin and the historical events that have led up to them. However, particularly for a project that will take decades, and perhaps even centuries, to implement, it is important to consider how these systems might change in the future.

If current trends were to continue, the basin would expect over a long period, to experience three major changes in the conditions described above. The first would be continued regeneration of the forests on the basin slopes and the slow recovery of some of the riparian areas. This would result in decreased runoff and erosion during precipitation events, would likely reduce the magnitude of the normal late spring floods by slowing the rate of snow melt, and could reduce the floods resulting from rain-on-snow events by partially insulating the snow from the warm air masses that accompany these events.

The second change would result from the continuing erosion of the mine tailings and other materials deposited in the upper valleys and the deposition of these materials in the middle and lower segments as well as Lake Coeur d'Alene and the upper Spokane River. Ultimately, the erosion in the upper basin and sedimentation rates in the middle basin would diminish, and channels would stabilize, particularly if the riparian areas are allowed to recover and the stream reaches return to mostly transport reaches as they were before mining.

The lower basin, however, is unlikely to return to its premining condition. Sedimentation will continue at lower rates in the lower basin floodplain and wetlands. Over millennia, the lateral lakes would become marshes, and the marshes would become floodplain grass and brush areas used for fields and pastures. Bookstrom et al. (2004a) indicate pre-mining depositional rates of about 1 mm/year in the open-water environment of Killarney Lake. The post-1980 rates are about 4 mm/year. At Medicine Lake, pre-1968 depositional rates exceeded 8 mm/year but have since declined to 4 mm/year (Bookstrom et al. 2004a). Some marsh areas with substantial accumulation of peat do not have high levels of lead, and the slow conversion of lake to marsh ultimately may cover some of the contaminated areas.

The delta would continue to build lakeward, creating new lateral lakes and marshes on the flanks of the leveed channel. The fate of the large inventory of contaminated sediments in the channel of the main stem is uncertain. The historic channel has not migrated, but it is subject to scour and remobilization of bed material. This process would be substantially influenced by the relative prevalence of serious rain-on-snow flooding events compared with the normal flooding pattern resulting from late spring snow melt. The latter results in more deposition, and the former is more likely to carry its sediment into (and across) the lake.

The third trend would be declining loadings of zinc and other dissolved metals in the downstream segment of the river as the available supplies of soluble metals diminish in the upper and middle segments.

It is also unclear what will happen in Lake Coeur d'Alene. It will continue to receive sediments, which will extend the delta of the Coeur d'Alene River farther out into the lake and increase the depth of contaminated sediments on the lake bottom. The major question is whether the lake will become more eutrophic, and, if so, what effects this will have on the lake's chemistry and biota. There is substantial concern that changes in the lake's chemistry could result, as indicated in the above description of Lake Coeur d'Alene, in the release of contaminants currently bound in the sediments coating the lake bottom. These released contaminants could be toxic to fish and other aquatic biota and, therefore, in conjunction with the other effects of eutrophication, could cause significant changes in the lake's biological systems.

The Spokane River would continue to receive some of the sediment carried down the Coeur d'Alene River, necessitating continuing cleanup of contaminated riparian recreation areas and resulting in a gradual filling in of the reservoir behind Upriver Dam.

All these processes would continue over a period of centuries, and none of the possible changes is likely to occur in the near term except, perhaps, those that might occur in Lake Coeur d'Alene. There is no reason to expect any natural perturbations that might significantly disrupt these processes, although serious forest fires in the basin could temporarily disturb them, as would a major volcanic ash fall from an eruption in the Cascades or Yellowstone.

The most significant possible perturbations are likely to result in the future, as they have in the past, from human activities. Some could occur within the project area; others are likely to occur more globally.

Local Human-Induced Perturbations

Within the basin, it is conceivable that substantial increases in metal prices could stimulate increased interest in mining opportunities. As indicated earlier, some mines have continued to operate in the basin, and plans are currently in place for expanded activities. Other mines probably could be brought back into production under extremely favorable economic conditions (or as a result of government demands such as occurred during World War II). Even if this were to occur, however, it is unlikely that any future mining activities would have as much impact on the basin as the historical mining activities did, primarily because the mines are now prohibited from disposing of their mining wastes in such an environmentally destructive manner.

One particularly remote possibility under the increased mining scenario is that metal prices would rise so high as to support the remining of the old tailings and other wastes containing low concentrations of metals. Such remining is occurring in old gold mining areas in the West (see NRC 1999) and is arguably reducing environmental risks at these sites. In the Coeur d'Alene River basin such remining activities conceivably could result in the removal of large amounts of contaminated materials from some of the stream channels as well as the tailings piles and other terrestrial deposits. This possibility, however, is diminished not only by the likely adverse economic conditions but also by the fact that the basin has been designated a Superfund site with all the liabilities associated with such a designation.

A much more likely development pattern in the basin is for it to become a center for outdoor recreational activities and leisure home developments. Lake Coeur d'Alene already has experienced substantial development of this type, and the demand for these developments continually increases with rising incomes in the United States. Both the natural beauty and the historical significance of the Coeur d'Alene basin make it an attractive location for such developments to occur.

Such recreational developments could significantly change socioeconomic conditions in the basin, bringing higher-income residents and economic stimulus for the basin's merchants and labor force. If properly controlled, such developments need not generate significant environmental damage, and their residents may be highly sensitive to the quality of the environment. There would undoubtedly be some erosion associated with the new construction, and recreational demand could also result in the construction of access roads and even the clearing of large areas for snow sports. Both could result in increased runoff and erosion, with the concomitant increase in downstream floods and sedimentation.

Although some valley residents fear that the potential for these recreational developments will be diminished by the designation of the valley as a Superfund site, the elimination of significant health risks as a result of the Superfund cleanup might make the valley more attractive to these potential residents. Support for this hypothesis is provided by the proposal announced this year for building a major recreational facility near Kellogg within the area that was designated a Superfund site in 1983 and that has since been largely cleaned up under the Superfund program (Kramer 2004).

Another economic change that could occur in the more distant future is the relogging of the forests in the basin after they have regenerated. As discussed earlier in this chapter, the intensive management of the forests in the North Fork basin is already thought to be increasing erosion and runoff there. And, considering the massive amounts of metal-contaminated sediments that can be remobilized during large floods (especially the scouring of highly contaminated and deeply buried riverbed sediments), water retention and yield from the watershed is a significant issue. Ironically, the increased transport of relatively clean sediment from the North Fork is reducing the average concentration of lead in sediments below its confluence with the South Fork.

Regional and Global Human-Induced Perturbations

One possible perturbation that could occur at the regional level is an increase in acid rain resulting from electrical power generation, increased vehicle traffic, or other sources. However, it is unlikely that this would become a significant problem in the Coeur d'Alene River basin, and the neutralizing effects of the basin's soils would largely prevent any serious effects.

At a global level, the most likely perturbations affecting the basin will be those resulting from climate change. Most scientists agree on the likelihood of climate change occurring, which is attributed directly or indirectly to human activity, and many argue that some of its effects can already be observed. Major characteristics of climate change are expected to be increased average global temperatures and an increase in the frequency and magnitude of storms (NAST 2000; Mote 2001; NRC 2001). It is very difficult to predict the impact of climate change in a particular region such as the Coeur d'Alene River basin. Some areas are likely to experience increased storms and precipitation, others a warmer dryer climate.

Climate change models focusing on the Pacific Northwest generally predict warmer temperatures and increased winter precipitation by the mid-21st century (Climate Impacts Group 2004). The modelers predict that the following changes would occur (Hamlet and Lettenmaier 1999; Mote et al. 1999, 2003; Miles et al. 2000; Climate Impacts Group 2004; Palmer et al. 2004):

- Increase the amount of winter precipitation falling as rain rather than snow.
- Increase winter stream flow.
- Increase winter flood risks in transient (rain/snow mix) basins.
- Reduce the amounts of water stored as snow, particularly in mid-elevation transient (rain/snow mix) basins.

- Induce earlier snow melt and advance peak runoff earlier into the spring.
- Decrease late spring and summer stream flows.

Other studies have suggested that the increased winter flood flows will produce greater channel scour and sediment load in rivers (Hamlet et al. 2004) and that the early snow melt and dry summers may increase the number and size of forest fires, as well as lead to drought-stressed forests subject to disease and insect infestation (Service 2004). Drier summers could reduce the basin's ability to support its current rich vegetation. One result could be increased wind erosion of contaminated sediments, increasing human health risks from their inhalation.

It is difficult, often impossible, to predict what perturbations will occur and, if they do occur, what effects they might have on the Coeur d'Alene River basin. Nevertheless, it is prudent to keep such possibilities in mind in the process of evaluating and designing remedies that are expected to protect human health and the environment in the basin for the future.

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4

Remedial Investigation Assessment

INTRODUCTION

Superfund activities began in the Coeur d'Alene River basin in 1983 with the listing of the Bunker Hill Mining and Metallurgical Complex on the National Priorities List (NPL). This site, commonly referred to as the Bunker Hill "box," encompasses a 21-square-mile area including the historic smelter and ore-processing operations in the heart of the Coeur d'Alene River basin. The site was divided into two operable units (OUs) for which records of decision (RODs) were issued in 1991 and 1992.[1]

In 1998, the U.S. Environmental Protection Agency (EPA) extended Superfund activities and undertook a remedial investigation/feasibility study (RI/FS) of mining-related contamination in the Coeur d'Alene River basin outside the box. This is the third operable unit of the site (OU-3, commonly termed the "basin"). The geographic area includes the Coeur d'Alene River, associated tributaries, Lake Coeur d'Alene, and the Spokane River that drains from Lake Coeur d'Alene and crosses from Idaho into Washington. Within this geographic scope are residential communities; recreational areas; active and inactive mining facilities; parts of the Coeur d'Alene Indian Reservation; the Spokane Indian Reservation; parts of Kootenai, Benewah, and Shoshone counties of northern Idaho; and parts of Stevens, Lincoln,

[1]Operable unit 1 (OU-1), the "populated areas" of the box, includes the communities of Kellogg, Smelterville, and Pinehurst. Operable unit 2 (OU-2), the "non-populated areas," includes the site of the Bunker Hill smelter, ore-processing complex, and mine.

and Spokane counties in eastern Washington (see Figure 3-1 in Chapter 3 of this report).

The RI report (URS Greiner, Inc. and CH2M Hill 2001a) was prepared by contractors for EPA Region 10 based on EPA's guidance document for conducting RI/FS studies (EPA 1988) through the RI process set forth in the National Oil and Hazardous Substances Pollution Contingency Plan (NCP, 40 CFR Part 300) (URS Greiner, Inc. and CH2M Hill 2001b, p. 1-2). The information in the RI report is used to evaluate risks to human health and the environment and potential remedial alternatives.

In this chapter, the RI of the Coeur d'Alene River basin (URS Greiner, Inc. and CH2M Hill 2001a) is assessed with respect to the following:

- Adequacy and application of EPA's own Superfund guidance for RIs
- Consistency with best scientific practices
- Validity of conclusions

Additionally, this chapter evaluates the scientific and technical aspects of the following:

- EPA's determination of the geographic extent of areas contaminated by waste-site sources
- Types of data and analyses used to assess the extent of contamination
- Approaches used to collect and analyze the data that resulted in conclusions
- Considerations of contaminant chemical speciation and transport

Human health aspects of the RI are primarily evaluated in Chapter 5, "Human Health Risk Assessment in the Coeur d'Alene Basin." The Human Health Risk Assessment (HHRA), undertaken concurrent with the RI, characterizes heavy-metal contamination in relation to potential human health risks.

EPA'S RECOGNITION OF THE BASIN SYSTEMS AND THEIR INTERACTIONS

The Coeur d'Alene River basin is a large-scale, complex system with extensive anthropogenic overprints that have increased the multiple complexities and interacting processes at work throughout the basin. This vast, mountainous river system has a long history of mining, logging, fishing, trading, and tourism (see Chapters 2 and 3). The high precipitation and high-flow events, which are characteristic of the Coeur d'Alene basin, have distributed mining wastes over many miles. The size and complexity of the basin combined with the highly variable nature of the mine wastes render site characterization a formidable task.

Systems Approach and the Conceptual Site Model

One way of characterizing the Coeur d'Alene basin for the purpose of remedial planning is to use a "systems approach" (see Box 4-1). This "system" is logically defined by watershed[2] boundaries. Within the Coeur d'Alene system, relevant aspects are considered, including the geology, hydrology, ecologic communities, climate, human factors, and mining-related wastes. Under the systems approach, subwatershed boundaries are used for looking at smaller, more-manageable units while maintaining an awareness of interconnectedness between those units and the entire system.

EPA's process for investigating a Superfund site calls for the creation of a "conceptual site model" (CSM) at the beginning of the RI. This model is intended to guide the way the RI is conducted and establishes a conceptual framework for the rest of the Superfund cleanup process. The CSM developed for the basin is largely based on geographic characteristics of the stream valleys and hydrologic characteristics of water bodies and is tantamount to looking at the overall Coeur d'Alene system in terms of more manageable subwatersheds. The basin was subdivided into five CSM units that correspond with Chapter 3's description of the basin's topography.[3] The description of each CSM unit in the RI is accompanied by a complex "process model" diagram, characterizing the multifarious interactions that may take place in each unit. Figure 4-1 shows the process model for the Canyon Creek watershed.

One aspect of a systems approach only nominally considered in the development of these models is the amount of variability that exists in the basin—particularly with respect to the climatic and hydrologic systems. As evidenced by the large floods experienced in the basin and their tremendous impact on contaminant transport, these events are a critical element in the basin's hydrologic system. The conceptual models, and therefore the definition of possible remedies, seemingly are based primarily on average conditions, and the committee believes that variations in the basin's systems, particularly flood events, may have a significant impact on the effectiveness of the proposed remedies.

In addition, in carrying out assessments of the individual geographical components of the basin, the RI appears to have lost sight of the broader interactions within this complex system. Based on a systems approach, the RI should look at the watershed boundaries defining the basin system and then develop a flux-reservoir model of where each metal of importance

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[2]The watershed is also referred to as a catchment or drainage basin.

[3]These units include: CSM Unit 1, upper watersheds; CSM Unit 2, midgradient watersheds; CSM Unit 3, Lower Coeur d'Alene River; CSM Unit 4, Coeur d'Alene Lake; CSM Unit 5, Spokane River.

BOX 4-1 Systems Approach

"In the context of water resources the essential function of a *systems approach* is to provide an organized framework that supports a balanced evaluation of all relevant issues (e.g., hydrologic, geomorphic, ecologic, social, economic) at appropriate scales of space and time. Within a systems framework, multiple stressors can be identified and quantified, multiple goals can be investigated, trade-offs among competing objectives can be evaluated, potential unintended consequences can be identified, and the true costs and benefits of a project can be examined in a context that incorporates the interest of all those with any substantial stake. . . . The merits of a systems approach are broadly endorsed . . . throughout the water resources community, and in several NRC reports (NRC 1999a,b, 2000, 2001). . . . A systems framework supports a balanced consideration of all relevant aspects of water resources problems at all relevant time and space scales."

Source: NRC 2004.

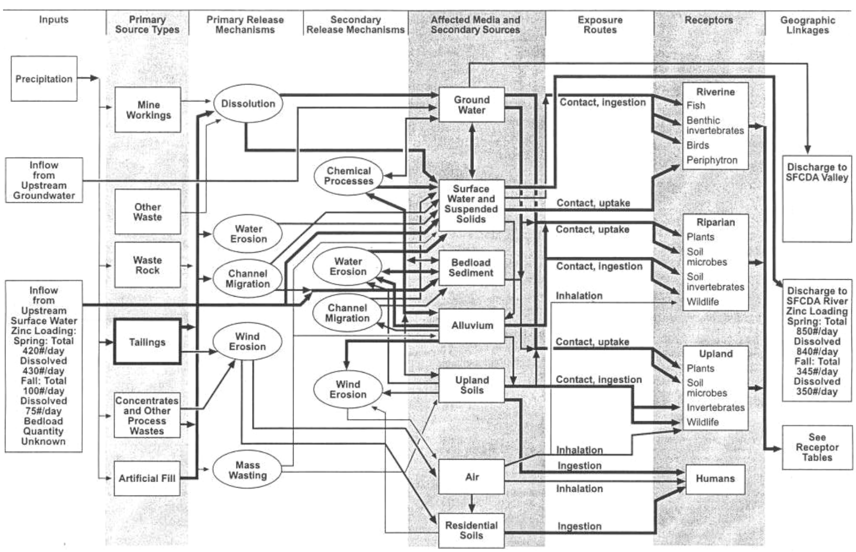

FIGURE 4-1 Process model for Canyon Creek Watershed (CSM Unit 1). ...... low importance, —— medium importance, ▬▬ high importance. SOURCE: URS Greiner, Inc. and CH2M Hill 2001b, p. 2-22.

resides and where that metal is transported at the established flux. The RI should consider the roles that geology, hydrology, geomorphology, geochemistry, forest management practices, infrastructure, etc. all play as components of the system. In fact, a similar approach was recommended in an EPA report (Hornig et al. 1988) that looked at the water quality monitoring in the Coeur d'Alene River basin:

> A whole basin environmental management approach to the Coeur d'Alene system should also address the relative importance of habitat degradation and other factors (for example, nonpoint impacts from agricultural or forestry practices) in the prevention of full potential of aquatic resources. The dynamics of cadmium and lead in the ecosystem also needs to be further addressed, including the relative importance of the contribution of present South Fork loadings of these metals to the downstream sediments and biota.

EPA made preliminary steps toward looking at the Canyon Creek watershed using a systems approach. However, this approach appeared to be less in evidence in other parts of the basin, particularly regarding the box which is excised from consideration in the basin's RI and subsequent documents. A systems approach would consider the contaminant sources and pathways within the box along with those stemming from upstream portions of the South Fork of the Coeur d'Alene River and also consider their potential to serve as contaminants in downstream areas.

Operable Unit Designation

Operable Units 1 and 2

As mentioned, OUs 1 and 2 are the populated and nonpopulated areas, respectively, of the 21-square-mile box. OU-3, the subject of this review, includes all the rest of the basin from the headwaters west into eastern Washington. In some cases, defining separate OUs may facilitate an earlier start on cleanup of a more-contaminated area. This was the situation for OU-1 and OU-2 because cleanup of these units began well before the RI for OU-3 was initiated. While this segmentation may have been appropriate at the time based on the severity of contamination in the box, it currently creates technical issues regarding implementation of remedies for protecting ecologic health downstream of the box.

These technical difficulties arise, for instance, in efforts to protect fish downstream of the box. In this stretch of the river, the major source of dissolved zinc comes from groundwater discharges to the river that occur within the box but apparently cannot be addressed in remedies considered for OU-3.[4] It is not clear whether there are cost-effective remedies for controlling these sources, but it makes no technical sense to ignore this possibility entirely. The manner in which the Superfund site was segmented has also created public perception problems. For example, privately-owned properties on different sides of the dividing line could have similar levels of contamination, but properties outside the box had to wait a decade before becoming part of the Superfund site and be considered for remediation.[5]

Operable Unit 3

EPA has substantial flexibility under the NCP in establishing what areas or actions will constitute an OU at a site.[6] However, the guidance does state that "sites should generally be remediated in operable units when ... phased analysis and response is necessary or appropriate given the size or complexity of the site, or to expedite the completion of total site cleanup." Certainly, the Coeur d'Alene River basin is such a site though the entire basin (minus the box) was considered a single OU. The committee's evaluation suggests that a different segmentation approach to OU-3 might have been preferable. There is a remarkable independence between protecting human health and protecting the environment. None of the remedies undertaken for human health protection will have any discernable impact on the protection of fish and wildlife (see Chapter 8). Similarly, EPA identifies only limited human health benefits that would result from the remedies being considered for protecting environmental resources (EPA 2002,

[4]EPA states that they intend to integrate actions selected in the ROD with those implemented in the box (EPA 2002, p. 4-6). However, exactly what EPA intends to do is not yet clear. The agency has postponed implementing any efforts to cleanup groundwater seeping through the CIA until it sees how successful the cap on this facility will be in reducing groundwater contamination. The following is provided in the 5-year review for OU-2: "For groundwater, the cleanup levels specified in the ROD for site-wide groundwater were maximum contaminant levels (MCLs) and MCL goals for arsenic, copper, lead, mercury, PCBs [polychlorinated biphenyls], selenium, silver, zinc, and nitrate as identified under the Safe Drinking Water Act. The ROD further defined contingency measures to be implemented if these cleanup goals were not capable of being met" (EPA 2000, p. 5-2).

[5]Public perception problems also stem from the fact that the agency seems to have reversed its original position, which was to deal with the environmental problems outside of the box using programs other than Superfund (see Chapters 1 and 2 for further discussion).

[6]The NCP states that "Operable units may address geographical portions of a site, specific site problems, or initial phases of an action, or may consist of any set of actions performed over time or any actions that are concurrent but located in different parts of a site" (40 CFR § 300.5[2004]).

Table 12.2-1). These remedies include limiting exposures associated with recreational activities at mine-waste sites or riverbanks.[7]

A more rational segmentation might have been to make one OU the protection of human health (or even several OUs based on subwatersheds of the basin, or addressing, for example, residential properties, public use areas, and other human health risks), and the second OU the protection of environmental resources (or perhaps several OUs based on the subwatersheds of the basin).[8] This approach would have had some clear technical advantages in allowing the agency to analyze risks more systematically and in considering remedial alternatives more effectively, because of the more manageable size and differing characteristics of the smaller OUs.

In addition, such an approach probably would reduce the pall that so many residents believe will shadow the basin for decades to come, for the human health protection remedies in the basin will be completed relatively quickly. When this occurs, the basin could be declared to be cleaned up with respect to human health, although further work would be required to protect the environmental resources. To the extent that the designation of the basin as a Superfund site affects its economic prospects, such a distinction might well have reduced these negative effects.

It is probably too late to make such a change, but the agency might consider such an approach at other large sites where some of the cleanup activities will take long periods to complete.

SAMPLING AND ANALYSIS

Samples Collected

Some 7,000 samples had been collected in the Coeur d'Alene River basin between 1991 and 1999 by the Idaho Department of Environmental Quality, the U.S. Geological Survey (USGS), mining companies, and EPA under other regulatory programs (URS Greiner, Inc. and CH2M Hill 2001b,

[7]In addition, the environmental remedies, because they should reduce the transfer of contaminants to Lake Coeur d'Alene and the Spokane River, could have some health benefits for tribal members pursuing traditional lifestyles and to recreational users along the Spokane River.

[8]It appears that this was considered by EPA. As provided by Villa (2003): "At one time, consistent with the operable unit concept, Region 10 considered dividing the Basin cleanup plan into two phases, with the human health component to be released before the ecologic component. However, the proposal provoked a public outcry, led by the State of Idaho, and EPA responded by agreeing to keep the human health and ecologic cleanup for the Basin together in one plan." Villa (2003) indicated that the "[c]oncerns by the State of Idaho included presenting the public with one plan to comment upon and allowing consideration of tradeoffs between human health and environmental protection."

p. 4-8). These historical samples, obtained from sediments, surface waters, groundwater, and soils, had been collected to support investigations with different objectives than those set forth for the RI. Nevertheless, a decision was made by the EPA to rely on data from these 7,000 historical samples already collected, although the quality assurance and quality control (QA/QC) procedures varied among the various studies, and the results from the several data sets were generated from multiple methods of analysis. Because the levels of metal contamination from these studies were large in comparison to the levels considered problematic, the EPA was less concerned with the uncertainties associated with the QA/QC and analytical methodologies used. Based on review of the data from the 7,000 historical samples, EPA made the decision to collect additional samples and developed a Draft Technical Work Plan (URS Greiner, Inc. and CH2M Hill 1998a), which considered the EPA's Data Quality Objective (DQO) process (EPA 1994). The Draft Technical Work Plan was used to develop field sampling plan addenda (FSPAs) (URS Greiner, Inc. and CH2M Hill 2001b, pp. 4-10 to 4-29), each with a specific purpose and scope, for collection of an additional 10,000 samples to characterize source areas. These samples were collected from sediments, sediment cores, adits, seeps, creek surface waters, soils, drinking water (wells, residential, and school/daycare), indoor dust, vacuum cleaner bags, lead-based paint, and groundwater. Two types of sampling were conducted: judgmental and probabilistic. Judgmental sampling (that is, nonprobabilistic) entailed sampling specific areas to confirm the existence of contamination. The committee did not assess EPA's DQO process, Draft Technical Work Plan, FSPAs, or the methodology used by EPA to review and incorporate data from the 7,000 historical samples.

The 17,000 samples, collected over the large basin area, perhaps represent less than a dozen samples per square mile (although a much higher density of samples exists in the contaminated floodplain). The Bureau of Land Management identified approximately 1,080 mining-related source areas in the basin. Source areas were identified as either primary or secondary. Primary sources, mostly present in the upper basin (that is, the area characterized by high-gradient tributaries to the South Fork Coeur d'Alene River), include mine workings, waste rock, tailings, concentrates and other process wastes, and artificial fill. Secondary sources, principally located in the lower segments of upper basin tributaries, the middle basin (Wallace to Cataldo), and the lower basin (Cataldo to Harrison), include affected media (for example, groundwater, floodplain deposits, and bottom sediments) that may act as sources of metals to other media or receptors.

EPA points out that of the approximately 1,080 sources, samples were collected from about 160 (15%) with fewer than five samples collected from most of these source areas (URS Greiner, Inc. and CH2M Hill 2001b, p. 4-36). These areas range in size from less than an acre to hundreds of acres and are listed in Appendix I of the RI. Major tailings, waste rock, and floodplain sources of metal contaminants were identified by EPA as to location and area. Sample locations and data collected were documented. Sources with an area greater than 5 acres were surface sampled; few samples were collected at a depth of greater than 1 foot. Not all sources were systematically characterized in terms of thickness. Greater effort was expended to document contamination in the floodplains of the Coeur d'Alene River. The USGS mapped, measured thickness and surface extent, and analyzed floodplain sediments in upper basin tributaries, the South Fork of the Coeur d'Alene River, and the lower basin (Box et al. 1999, 2001; Bookstrom et al. 2001, 2004; Box and Wallis 2002; Box et al. in press). It will be important to incorporate data from these analyses that was not considered in the RI in remedial planning within the basin.

In addition to collecting samples from only 15% of the sources identified by the Bureau of Land Management, the agency made no effort to characterize groundwater "source terms."[9] The committee learned from EPA's written response to submitted questions that leachability data per se, which would characterize the source term, were not available and therefore were not used in the analyses and estimates of loading (see the section "Analyzing Sample Data" for a discussion of metal loading). Very simply, localized areas of high (or low) leachability were inferred from what are considered to be sources (such as nearby floodplain tailings) and measured increases in dissolved metal loadings in streams (EPA 2004 [June 23, 2004]).

Nonetheless, the committee believes that the large number of samples collected and analyzed provides information on contaminant locations and trends related to contaminant transport and fate in the basin, especially for surface water. Much new information has become available since the ROD was issued (EPA 2002), and EPA is commended by the committee for its cooperative, scientific relationships with sister agencies and others. The agency is urged to proceed with more-thorough identification of specific sources contributing dissolved or particulate metals to surface waters before proceeding with cleanup to ensure the location, magnitude, and disposition of contaminant sources and their contribution to the system.

[9]The phrase "source term" is defined as the amount and chemical form of a contaminant released to the environment from a specific source over a certain period of time. "Source" identifies the nature and origin of the release and "term" refers to how much of a substance, or metal in the case of the Coeur d'Alene basin, is released to the environment over a specified time period. Source terms are used in risk-assessment studies.

TABLE 4-1 COPCs and Affected Media for the ERA

	Ecologic COPC		
Chemical	Soil	Sediment	Surface Water
Antimony			
Arsenic	*	*	
Cadmium	*	*	*
Copper	*	*	*
Iron			
Lead	*	*	*
Manganese			
Mercury		*	
Silver		*	
Zinc	*	*	*

SOURCE: URS Greiner, Inc. and CH2M Hill 2001b, Table 5.1-1.

Nature of Contamination

Chemicals of Potential Concern

Based on preliminary results of the ecologic risk assessment (ERA), ten chemicals of potential concern (COPCs)[10] were identified by EPA for inclusion and evaluation in the RI. These initial COPCs were evaluated, and those that met the data evaluation requirements and screening against applicable risk-based screening criteria were incorporated. Applicable risk-based screening levels were compiled from available federal numeric criteria (for example, national ambient water-quality criteria), regional preliminary remediation goals, regional background studies, and other guidance documents. Table 4-1 lists these initial ten COPCs and affected media considered for the ERA. COPCs not carried forward in the ERA were antimony, iron, and manganese, because they did not meet the applicable risk-based screening criteria (URS Greiner, Inc. and CH2M Hill 2001b, p. 5-1). Groundwater data were screened against surface-water screening levels to evaluate the potential for impacts to surface water from groundwater discharge (URS Greiner, Inc. and CH2M Hill 2001b, p. 5-2).

The two chemicals of ecologic concern (COECs) receiving the most attention from EPA for the Coeur d'Alene River basin system are lead and

[10]EPA uses the term "chemical of potential concern" (COPC) when considering all the substances (metals in the case of the Coeur d'Alene River basin) that may be of possible concern to human health and the environment. The term "chemical of potential ecologic concern" (COPEC) is used for those metals that may possibly affect ecologic receptors. "Chemical of ecologic concern" (COEC) is the term used for those metals that meet the applicable risk-based screening levels.

zinc. The environmental chemistry of these two metals is appreciably different. Lead is primarily present and transported in the basin as a particulate and is a major concern because waterfowl ingest lead-contaminated sediment (see Chapter 7) and children are exposed to lead through lead-contaminated soil or dust (see Chapters 5 and 6). Dissolved lead concentrations are low because lead is quite insoluble under the chemical conditions of the basin. Zinc is transported primarily in dissolved form (Beckwith et al. 1997, p. 6) and is a toxicant for fish and aquatic invertebrates (see Chapter 7), but zinc is also significantly transported in particulate form especially during floods (Beckwith 1996; Box et al. in press). Other COECs have been compared with total lead and dissolved zinc in the RI. EPA uses dissolved zinc concentrations as an indicator of the behavior of each dissolved chemical of concern and total lead concentrations as an indicator of the behavior of each total chemical of concern to avoid having to consider each chemical of concern separately (URS Greiner, Inc. and CH2M Hill 2001c, p. 4-11).

Of the dissolved COECs, zinc is the principal dissolved metal of concern, and EPA reports using zinc as an indicator metal for the following reasons (URS Greiner, Inc. and CH2M Hill 2001c, Section 4.2.1; URS Greiner, Inc. and CH2M Hill 2001d, p. 1-8):

- Zinc is the most ubiquitous of the metals.
- Zinc occurs at the highest measured concentrations and has the highest ratios of average measured concentration to ambient water-quality criteria or, equivalently, average measured load to total maximum daily-load loading capacities.
- Zinc is relatively mobile compared with other metals.
- Dissolved metals generally correlate with dissolved zinc.

In the South Fork of the Coeur d'Alene River, zinc accounts for about 96% of the dissolved heavy-metal load, and zinc is the main dissolved metal as the Coeur d'Alene River flows into Lake Coeur d'Alene at Harrison (Woods 2001). EPA discussed the correlation of zinc with other metals (URS Greiner, Inc. and CH2M Hill 2001c), and although cadmium appears to correlate well with dissolved zinc throughout the basin, other COEC metals (copper, mercury, silver, and arsenic) exhibit various degrees of correlation with dissolved zinc. The committee clarifies that arsenic and antimony behave similarly but these two elements should not be expected to correlate with either zinc or lead, because their chemistries are substantially different. Arsenic and antimony occur in water as oxyanions (with negative charges), whereas zinc and lead are positively charged cations. Furthermore, the aqueous mobilities of arsenic and antimony are affected by redox changes and depend on the redox conditions of the water, whereas zinc and lead undergo no redox reactions. Aqueous arsenic and antimony are derived from the oxidative weathering of sulfide minerals, such as arsenopyrite (FeAsS), enargite (Cu_3AsS_4), and tetrahedrite [$(Cu,Ag)_{10}(Fe,Zn)_2(As,Sb)_4S_{13}$], which are all found in some of the mineralized areas of the basin. Although it is reasonable to consider zinc as the principal dissolved metal of concern, care must be taken in correlating zinc with other metals.

Groundwater Considerations

Groundwater is the primary source of dissolved metals in the surface water of the basin. As stated by EPA (EPA 2004 [June 23, 2004]),

> Except under very high-flow flood events, the majority of the zinc load, and particularly the dissolved zinc load, in the CDA [Coeur d'Alene] River at Harrison is contributed by groundwater. . . . Except for direct loading from adit discharges and storm water discharges from waste piles, zinc loading to streams is from affected groundwater in the floodplains.

The committee notes that investigations documenting aqueous concentrations of dissolved metals within the basin focused primarily on monitoring surface-water concentrations. A more-limited campaign to sample groundwater was undertaken that included establishing monitoring wells in Canyon Creek (Houck and Mink 1994; MFG 1995, 1998; Ridolfi 1998; Barton 2000; URS Greiner, Inc. and CH2M Hill 2001c). In the middle basin between Wallace and Pinehurst, other studies (Dames and Moore 1991; Barton 2000, 2002; CH2M Hill 2004a,b) evaluated the complex relationship between surface water and the shallow groundwater aquifer that can lead to losses of dissolved metals to the aquifer in some reaches and dissolved metal gains from others.

The committee found there to be limited information on groundwater contamination in the main stem and lower Coeur d'Alene River (Spruill 1993; Balistrieri et al. 2000, 2002; URS Greiner, Inc. and CH2M Hill 2001e, p. 4-8). Groundwater-contaminant contribution is suspected where it discharges to the river from contaminated bank and floodplain sediments, and groundwater may be a continuing source of contaminants in the lateral lakes area. Little information is available on metal transport in groundwater around Lake Coeur d'Alene and along the upper Spokane River (URS Greiner, Inc. and CH2M Hill 2001e, p. 4-8).

Because groundwater is the primary source of dissolved zinc to the system, the committee believes that developing a more-thorough understanding of the metal concentrations, dynamics, and specific source areas and media is necessary. Understanding this dynamic undeniably will require additional characterization. The committee acknowledges that groundwater characterization studies are expensive and draw from limited funds potentially used for remediation projects (generally source removals), which attempt to directly reduce the flux of water through contaminated surficial aquifers. However, it is necessary to characterize source areas and media contributing dissolved metals to groundwater (which is later discharged to surface waters) to accurately define remedial strategies, particularly source removals, intended to curtail zinc contributions to surface water. Tracer injections and synoptic sampling can be combined to understand and quantify metal loading to stream reaches impacted by mining, an approach developed in part by the USGS Toxic Substances Hydrology Program (Kimball 1997; Kimball et al. 2002). These studies simultaneously sample metals and a tracer (for example, lithium or bromide injected upstream) in surface water to permit high-resolution determinations of metal loading along a stream. These cost-effective techniques can be used to define source areas and metal contributions from groundwater, guide future cleanup efforts, and ascertain the effectiveness of remedial actions (Kimball 1997). This approach could be used as part of a site-characterization strategy in the Coeur d'Alene River basin.

Analyzing Sample Data

EPA relies on mass loading to describe the amounts and types of contaminant constituents in surface waters and identify sources, particularly secondary sources, of contamination. Mass loading is the mass of a constituent passing a given point per unit time; in the RI, mass loading is expressed in pounds per day. To measure mass loading, stream gauging is conducted to determine stream discharge in cubic feet per second. Chemical analyses of water samples are carried out, and the constituent concentrations are expressed in micrograms per liter (μg/L). Mass loading is the product of stream discharge and constituent concentration, converted to pounds per day.[11]

Mass loading is evaluated by two different methods, although these methods are not mutually exclusive. One method calculates "point estimates of mass loading" from discrete discharge and concentration data (URS Greiner, Inc. and CH2M Hill 2001b, pp. 5-6, 5-7). That is, mass loading is determined at a single USGS gauging station at one point in time. The second method, "estimated average mass loading," uses a combined data set and a probabilistic model (URS Greiner, Inc. and CH2M Hill

[11]Mass loading (pounds per day) = stream discharge (cubic feet per second) × constituent concentration (μg/L) × 0.00538 (pounds × L × s) ÷ (ft^3 × μg × day).

2001b, p. 5-24) (1) to predict metal concentrations in the stream, (2) to predict metal loading in the stream (how much metal is flowing in the stream), and (3) to quantify the uncertainty associated with the predictions. Estimated average mass loadings are derived by taking all the historical data from all points in time at a USGS gauging station, plotting it, and obtaining from the plotted data a measure of central tendency—the "expected value." Estimated average mass loading data in the RI refer to current conditions in the basin. These data are presented in parts 2-6, section 5, of the RI (URS Greiner, Inc. and CH2M Hill 2001a) and are used to characterize dissolved metals, total lead, and sediment for the entire basin excluding Lake Coeur d'Alene.[12]

The committee found that the analyses provided for zinc and lead are useful for understanding the contributions of various tributaries and large-scale geographic areas to metal loadings in the basin by providing a central estimate at each gauging location. This "estimated average mass loading," with appropriate application of standardization methods to accommodate stream flows, as a methodology to describe dissolved constituents and sediment in surface waters provides an overall depiction of dissolved and particulate contaminants moving through the river system over time. Further, the committee found this method adequate to demonstrate that surface-water concentrations of dissolved zinc are substantially elevated compared to water-quality criteria and to show that large amounts of metals are transported through the system. It is a method for evaluating the total input of metal to the system, but the committee emphasizes that the method does not provide the location of sources or underscore the high concentrations of toxic metals that may occur in the system at any one time. The committee cautions that averaged data can be misleading in several ways:

- The highest concentrations of dissolved metals, especially zinc, occur during low-flow events. Therefore, low-flow events have the greatest impact on the aquatic ecosystem. This fact could render inadequate certain remedial decisions made with averaged mass loading data.
- The highest suspended sediment loads, which can contain particulate lead and zinc, occur during high-flow events, when the erosive ability of the river is greatest. High mass loadings of lead-containing sediment are transported during high-flow events to wetlands, marshes, and the lateral lakes inhabited by waterfowl. Use of averaged sediment mass loadings to arrive at remedial alternatives may result in unanticipated recontamination during

[12]In the FS, the probabilistic model is used to make quantitative estimates of the potential remedial performance associated with each remedial alternative selected (URS Greiner, Inc. and CH2M Hill 2001b, p. 5-24). This use of the probabilistic model and mass loading is discussed in Chapter 8.

high-flow events. This issue is discussed in greater detail in Chapter 8 of this report.

DETERMINING BACKGROUND CONCENTRATIONS

For the purpose of identifying areas within the Coeur d'Alene and Spokane River basins that are contaminated by mining wastes, EPA (URS Greiner, Inc. and CH2M Hill 2001f, p. 1-1) defines "background" as follows:

> For the purpose of the RI/FS, background is considered to be the concentration of a substance in environmental media that are not contaminated by the sources being assessed. Background concentrations are due to naturally occurring substances and other anthropogenic metal sources unrelated to mining (for example, leaded gasoline emissions from cars).

The committee considers this definition of background concentrations to be vague and open to interpretation but focused on the derivation of the values that were ultimately used. Background concentrations are determined primarily for two purposes: first, to estimate the extent of contamination (that is, where contamination levels in various media exceed background levels); and, second, to assist in the selection of remedial goals or target cleanup levels when used in conjunction with risk-based values determined through risk assessments. The process for establishing these background concentrations is described in a technical memorandum (URS Greiner, Inc. and CH2M Hill 2001f). Section 104(3)(a) of Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) provides the regulatory basis for determining background concentrations of metals (and other naturally occurring hazardous substances at CERCLA sites) and states in part that

> The President shall not provide for a removal or remediation action under this section in response to a release or threat of a release of a naturally occurring substance in its unaltered form, or altered solely through natural occurring processes and phenomena, from a location where it is naturally found.

CERCLA uses various strategies to estimate baseline metal levels at Superfund sites. CERCLA guidance for site-specific evaluation of baseline levels of metals in soils is not applicable to nonsoil media (for example, surface water), which tend to be more dynamic and are more likely to be influenced by upstream and distal sources. Assessment of background levels for nonsoil environmental media requires more complex spatial and temporal sampling strategies, analysis of releases and transport, and different ways of combining and analyzing data.

For the RI (URS Greiner, Inc. and CH2M Hill 2001a), EPA estimated background concentrations for ten COPCs (antimony, arsenic, cadmium, copper, iron, lead, manganese, mercury, silver, and zinc) for three environmental media (soils, sediments, and surface water) affected by mining activities (URS Greiner, Inc. and CH2M Hill 2001f, Table ES-1). Background concentrations were not determined for groundwater.

In view of the large geographic area and geologic diversity of the basin, EPA used a range of concentrations rather than a single-point estimate in the characterization of background for this site (URS Greiner, Inc. and CH2M Hill 2001f, p. 1-3). Because of the differing mineralization and erosion/deposition characteristics of the basin, background concentrations for the COPCs were developed separately for geographic areas: the upper basin (high-gradient tributaries to the South Fork), the middle basin (Wallace to Cataldo), the lower basin (Cataldo to Harrison), and the Spokane River Basin from the city of Coeur d'Alene to Lake Roosevelt on the Columbia River. EPA included Lake Coeur d'Alene in the lower basin, justifying this because the lake is part of the Coeur d'Alene River complex that supplies metal contaminants to downstream ecosystems. This section assesses the derivation of upper limits of background concentrations for COPCs reported by EPA (URS Greiner, Inc. and CH2M Hill 2001f) and related issues.

Background Concentrations of Metals in Coeur d'Alene and Spokane River Basin Soils and Sediments

EPA reviewed existing literature and concluded sufficient information was available to define background-concentration ranges for the COPCs in the upper and middle basin soils. However, the agency concluded that existing studies were not adequate to establish background ranges for all ten metals in sediments of the upper, middle, and lower Coeur d'Alene River basin and the Spokane River. The background ranges and summary statistics for sediments in these areas were derived by EPA from upper and lower basin sediment data collected for the RI/FS and Spokane River Basin soil data collected by the Washington State Department of Ecology (EPA 2004 [June 14, 2004]).

Background concentrations of the 10 COPCs in soils in the upper and middle basin were based on the data reported by Gott and Cathrall (1980) from 8,700 soil samples collected from approximately 300 square miles of the Coeur d'Alene mining district. From this database, the 90th percentile metal concentration was used as the background soil concentration for the ten COPCs (URS Greiner, Inc. and CH2M Hill 2001f).

Background-concentration ranges of COPCs for upper and middle basin sediments were estimated based on samples from monitoring well boreholes located largely in Canyon Creek but also included samples from Ninemile Creek and Pine Creek (URS Greiner, Inc. and CH2M Hill 2001f). EPA was aware that these upper basin tributaries are the most highly mineralized drainages and, as such, that samples from these areas may overestimate background metals concentrations in sediments for the entire upper and middle basin. Metal concentrations in sediments at various depths in the boreholes were assembled into a single database for analysis. The committee believes this database (URS Greiner, Inc. and CH2M Hill 2001f, Table C-1) is limited by the wide sampling interval in the boreholes, small number of subsurface samples, and, likely, the varying depth to background at different locations.[13] The background concentrations of metals in sediments in the upper basin were based on 12-30 sample values, depending on the COPC being considered. According to EPA, plots of sediment COPC concentrations versus depth in the core material showed a discontinuity indicative of the onset of mining impacts in the metal profile. Further, the transition in the COPC profile was confirmed by a combination of visual and statistical techniques as described in the technical memorandum (URS Greiner, Inc. and CH2M Hill 2001f, p. 3-1). Essentially, the analysis differentiates between two populations of samples, background and contaminated, and describes the background sediments as those below a certain depth. For some metals, this was 10 feet; for others it was 15 feet (URS Greiner, Inc. and CH2M Hill 2001f, Table 4-1). EPA found that the background concentrations of the COPCs generally were much higher (90th percentile comparison) in the upper/middle basin soils than in the sediments (URS Greiner, Inc. and CH2M Hill 2001f).

Background concentrations of COPCs in lower basin sediments were derived from core samples (URS Greiner, Inc. and CH2M Hill 1998b) collected for the RI/FS (URS Greiner, Inc. and CH2M Hill 2001a). Unlike the analysis for the upper and middle basin, EPA did not use concentration versus depth profiles to identify the threshold depth for background concentrations of COPCs in the lower basin. The reason given is that sediment thicknesses in this region are highly variable. The range in background concentration for each COPC was estimated by using, in the committee's opinion, a complicated and subjective ten-step process developed by EPA (URS Greiner, Inc. and CH2M Hill 2001f). Although EPA considers this approach a reliable means of estimating background concentrations, the committee believes that the subjective nature of the agency's method poten-

[13]For example, for lead, the data provided for the two deepest boreholes came from Canyon Creek. Data on one location (CC431) had samples at 5, 45, and 80 feet, and all metals concentrations were less than 15 mg/kg. Data from the other location (CC464) had samples at 5, 20, and 43 feet, and the lead concentrations dropped with increasing depth from 6,790 milligrams per kilogram (mg/kg) to 26.9 mg/kg to 7.5 mg/kg.

tially can produce results outside the range that objective methods would provide (see Appendix B for a review of this method). Nevertheless, the lower basin background concentrations derived from the ten-step method appear consistent with previous studies, and, based on the committee's review of data from various coring studies, the background concentrations for metals with limited mobility, such as lead, appear reasonable. For example, in another study, the USGS determined the background concentrations for lead in lower basin sediments as 26 mg/kg (median concentration) and 31 ± 19 mg/kg (mean ± standard deviation) (Bookstrom et al. 2004) compared with 47.3 mg/kg (90th percentile) resulting from this analysis (EPA 2002, Table 7.2-7).

However, the committee has concerns regarding the data set and sampling methodology of the study used to determine background concentrations. In this analysis, data from multiple cores were assembled into a single database from which background concentrations were mathematically derived. However, large numbers of these cores did not penetrate through the lead-enriched sediments to uncontaminated background sediments. In addition, samples taken along the length of many of the cores were widely spaced.[14] It is possible that the limitations of this data set made it necessary to compile all the data and mathematically determine a background concentration. The background technical memo (URS Greiner, Inc. and CH2M Hill 2001f) does not comment on this issue. Bookstrom et al. (2001) noted that sample intervals crossing the contact between the lead-rich sediment and the underlying lead-poor sediment will dilute and underestimate the lead content of the lead-rich segment. The use of wide sampling intervals is particularly problematic in parts of the lower basin, where the lead-rich sediments are less than 1 foot thick and EPA's sampling interval ranged up to several feet.

Coring studies are useful techniques capable of sampling historic sediments deposited before a particular event in time; in this case, the onset of mining. To define background concentrations, it is more reliable to sample cores with high vertical resolution (many samples along the length of the core) and to such a depth that the onset of premining background sediments can be defined instead of relying on mathematic and graphical techniques. Independent measures, such as time-stratigraphic markers and radioactive isotopes (for example, ^{137}Cs), should be used to determine that sediments originate from premining times.

[14]For example, core LC-102 from the Cataldo area was 23.4 feet in length and was sectioned 10 times over that length, with section lengths ranging from 0.9 to 2.5 feet; while core LC-110, also from the Cataldo area, was 13.3 feet in length but only had three sections at approximately 4.4 feet each.

EPA included Lake Coeur d'Alene with the lower basin for background estimation. Lake Coeur d'Alene receives sediments from nonmineralized drainages and EPA stated that "its inclusion with the lower Coeur d'Alene River is expected to result in the selection of background COPC concentration ranges that may be biased high." (URS Greiner, Inc. and CH2M Hill 2001f, p. 1-3).

However, in a coring study on Lake Coeur d'Alene, Horowitz et al. (1995) estimated the background concentrations of lead in sediment to be 33 mg/kg. This average is partially derived from cores taken in the St. Joe arm of Lake Coeur d'Alene. Regardless, all these concentrations are quite similar, especially in contrast to the high concentrations of metals detected in contaminated sediments, which are orders of magnitude higher.

EPA derived the background concentrations for the COPCs in the sediments of the Spokane River basin with data for 27 soil samples, collected to depths of up to 3 feet (San Juan 1994). EPA believes that sampling was designed to exclude the impacts of mining. Summary statistics for the background data were derived with the Model Toxics Control Act background computer module and were plotted as cumulative frequency distributions (CFDs) to calculate additional summary statistics as necessary (URS Greiner, Inc. and CH2M Hill 2001f). EPA recognizes that the values adopted were biased low, because the background samples were taken from areas that historically were not exposed to the Coeur d'Alene drainage (URS Greiner, Inc. and CH2M Hill 2001f, p. 5-2).

Background Concentrations of Metal in Coeur d'Alene River Basin Surface Water

EPA used existing surface-water data collected by the USGS, EPA, and Idaho Department of Environmental Quality to estimate the background concentrations for the COPCs. For this analysis, the entire basin was divided into three subareas: the tributaries to the South Fork (upper basin), the Page-Galena Mineral Belt area (corresponding to the middle basin), and the Pine Creek drainage basin. According to EPA, the background sampling locations were from unaffected upstream reaches in watersheds affected by mining and watersheds known to have relatively minor mining impacts. EPA asserts that these locations were chosen on the basis of their similarities to the contaminated areas in terms of watershed characteristics including geology, hydrology, and extent of mineralization as described in Stratus (2000). Background concentrations for surface waters in each of the three areas were determined and then pooled to get estimates for the entire upper and middle basin. According to EPA, consideration of the effects of surface expression of ore veins and the surrounding metalliferous rocks suggested that the background concentrations are biased high when applied to the Coeur d'Alene Basin as a whole (URS Greiner, Inc. and CH2M Hill 2001f,

p. 4-6). The relation of mineralogical features to each of the sampling locations was not considered by the committee. EPA accepts that the statistics reported for background concentrations of the COPCs were influenced by the large number of water samples with metal concentrations below the analytical detection limits. EPA's approach to these samples was to use one-half of the detection limit to represent the value for the metal in the sample (URS Greiner, Inc. and CH2M Hill 2001f, p. 5-3).

Background Metal Concentrations for Groundwater

Background metal concentrations were not determined for groundwater. The technical memorandum establishing background concentrations (URS Greiner, Inc. and CH2M Hill 2001f) states that

> Affected or potentially affected media types include soil, sediment, surface water, and groundwater. Of these media types, soils, sediments, and surface water are of primary concern because of the potential for exposure to human and ecologic receptors.

Are the Background Determinations Adequate for Their Use?

The committee observed that the Superfund decision documents developed for the Coeur d'Alene River basin frequently use background concentrations as a comparative measure to assess the extent of contamination in various environmental media. The ROD (EPA 2002) has numerous such uses. With the exception of the Spokane River, background determinations were not used appreciably for the second purpose, which was to assist in selecting remedial goals or target cleanup levels when used in conjunction with risk-based values. Yard remediation in the box and basin is triggered at levels well above background. The same is true for remedies intended to protect ecologic receptors. For example, the background lead concentration of soil and sediment in the lower basin is estimated to be 47.3 mg/kg (EPA 2002, p. 12-39), whereas the lead concentrations in affected areas at this location are 3,500-4,000 mg/kg, and the site-specific benchmark cleanup criterion is 530 mg/kg.

EPA addresses the background determinations in a manner consistent with the agency's established guidelines for assessing background concentration in soils and sediments at Superfund sites. The agency is commended for attempting to determine background rather than simply using national or regional numbers. For water, soils, and sediments in the tributaries of the South Fork of the Coeur d'Alene River (upper basin) and sediments in the Spokane River basin, the committee concludes that the background determinations are reasonable but limited by the issues presented in this section.

CHEMICAL SPECIATION AND TRANSPORT OF METALS

The mobilization of metals from sources; the movement of metals through environmental media (soil, sediment, and water); the changes that metals undergo in response to interactions with air, water, soil, sediment, and rock; and the transformation of metals by microorganisms are collectively referred to as "chemical speciation and transport." In the Coeur d'Alene system, metals are transported in both dissolved and particulate form. Many of the metals defined as COPCs that are present in the tailings, waste rock, water, and other materials and discharged to the waters of the Coeur d'Alene system undergo chemical and microbiological changes as they are transported downstream and encounter various environmental conditions (URS Greiner, Inc. and CH2M Hill 2001a).

The chemical speciation and transport of metals are not only central to understanding the bioavailability and toxicity of metals to receptors but are important in selecting remedies that mitigate risk.

This segment of the report summarizes and evaluates EPA's findings and conclusions, reported in the RI (URS Greiner, Inc. and CH2M Hill 2001a), on chemical speciation and transport of metals in the Coeur d'Alene River system, Lake Coeur d'Alene, and the Spokane River. This discussion focuses on EPA's studies specifically related to understanding the chemistry and movement of the metals in the Coeur d'Alene system and summarizes information on sediment transport.

EPA's Approach to Chemical Speciation and Transport Evaluation

A CSM, described earlier in this chapter, was provided in the RI to convey in abstract the sources of contamination, mechanisms of contaminant release, pathways of transport, and ways in which humans and ecologic receptors are exposed (URS Greiner, Inc. and CH2M Hill 2001b, pp. 2-1 to 2-19). The CSM developed by EPA for Canyon Creek is shown in Figure 4-1. Geochemical and hydrological conditions and mechanisms that EPA said (URS Greiner, Inc. and CH2M Hill 2001b, pp. 5-16 to 5-23) were considered in chemical speciation and transport of metals were flow events, pH, water chemistry, effect of iron concentration on metal concentrations, adsorption/dissolution/precipitation phenomena, amounts and types of atmospheric precipitation, erosion, and sediment movement.

Chemical Speciation

In response to the committee's request for information on speciation and bioavailability in basin soils and sediments, EPA indicated (EPA 2004 [June 14, 2004]) that these issues were addressed in the responsiveness summary of the ROD (EPA 2002, Part 3). This section of the ROD comments on the presumed speciation of the sediments but contains no indication that speciation was determined:

> Prior to 1968, large masses of mine-related releases were discharged to local streams or floodplain locations in predominantly lead sulfide form. However, oxidized ores were also likely released because milling and extraction practices were primarily designed to capture galena from sulfide ore. Oxidized lead minerals present in the original ores also were likely discharged to tributaries of the Coeur d'Alene River. . . . During movement and weathering, the lead in mill tailings was subject to physical and chemical transformation through abrasion, pH changes, and exposure to the atmosphere and aerobic hydrologic environments. These conditions promoted decreased particle size and increased surface area, and enhanced oxidation and the transition from lead sulfide to oxidized species.

That section of the ROD (EPA 2002) also addresses soils and states "It is unlikely that all smelter-related soil and dust lead is in an oxide form and equally unlikely that the soil and dust particles ingested by children, that originated as mining releases, are purely a sulfide form," and that the conclusion was consistent with results from other regions. Again, it is not apparent to the committee that speciation work was conducted.[15] The importance of speciation to bioavailability and toxicologic considerations is considered for humans in Chapter 5 and 6 of this report and for ecologic receptors in Chapter 7. The need for this type of information has been long understood; in 1988, EPA concluded the following:

> Research efforts should be encouraged that elucidate how the specific physical, chemical and biological characteristics of the Coeur d'Alene River and Lake system may affect the availability and toxicity of Silver Valley metal pollutants to different components of the ecosystem. (Hornig et al. 1988)

Sediment Transport

Most of the sediment transport data presented in the RI (URS Greiner, Inc. and CH2M Hill 2001a) were for water year 1999. Although the spring runoff for water year 1999 was higher than normal, the committee notes that there was no significant flood event—a phenomenon that significantly

[15]In fact EPA provided to the committee that "We note that, because of the site-specific information on bioavailability . . . understanding speciation was not necessary to evaluate health risks" (EPA 2004 [May 17, 2004]).

affects sediment transport. As a consequence, the committee believes that information in the RI likely provides an incomplete picture of sediment transport and metal mobility associated with sediment transport in the Coeur d'Alene system. Further, the committee notes that the geographic extent of various stage floods (10 year, 100 year, etc.) is not defined (EPA 2004 [June 23, 2004]), although understanding the flood regimes is essential in characterizing the system and especially in developing durable remedial strategies. Since the RI was issued in 2001, the USGS has provided a more comprehensive understanding of sediment transport in the Coeur d'Alene system (Clark 2003; Bookstrom et al. 2004; Box et al. in press). EPA is urged by the committee to consider this information in subsequent steps of the CERCLA process.

For the RI (URS Greiner, Inc. and CH2M Hill 2001a), suspended sediment and bedload samples were not analyzed for total metals. Rather, mass loading of metals in sediments was estimated from the total metal concentration of unfiltered water and the dissolved metal content of filtered surface-water samples (URS Greiner, Inc. and CH2M Hill 2001c, p. 5-33; URS Greiner, Inc. and CH2M Hill 2001g, p. 5-13). The committee believes that this methodology would be expected to exclude the metal load associated with bedload sediments, which are those particles transported along the stream bed by rolling or sliding. The amount of bedload material would be expected to be higher in high-gradient streams, such as those in the upper basin, as opposed to more sluggish streams. Also, as for suspended sediment load, bedload would be expected to be greater in high-flow events than at low flow. The bedload may contain, for example, highly enriched jig tailings, coarse particles with high surface areas, or some high-density minerals (for example, galena and cerrusite) that would tend to concentrate in the bedload. Consequently, it is unclear whether measurements made on suspended sediments accurately reflect sediment-associated metal transport even for the 1999 water year evaluated.

Surface Waters

Given the large variations in flow and metal content, EPA decided that, rather than using a mechanistic or deterministic model, transport of metals in surface waters through the system could be dealt with by using a probabilistic model. As described above, the probabilistic model is a mathematical model based on monitoring data collected for zinc, lead, and cadmium in surface waters at various sampling locations. Some tributaries or stream reaches did not have sufficient data to use the probabilistic analysis (sampling locations required a minimum of ten data points). For example, Big Creek, a tributary with historical mining activities that enters the South Fork just upstream of the box, had two data points for lead and zinc from which the loading was determined. In this case, the RI presents these two points as the lowest and highest loadings and concludes that the limited data set shows "small but significant contributions of metals from Big Creek to the South Fork" (URS Greiner Inc and CH2M Hill 2001h, p. 5-2). Other sampling locations have substantially more data, and the probabilistic model is used to determine an "expected" concentration and load. For example, Canyon Creek and Ninemile Creek were well characterized with multiple samples associated with a range of flows along the length of the tributaries. The South Fork and main stem of the Coeur d'Alene River had extensive surface-water sampling. Dissolved zinc concentrations and loading at the station near the mouth of the Coeur d'Alene River and Harrison was estimated from approximately 100 surface-water samples[16] from a wide range of flows.

EPA used the probabilistic model to predict metal concentrations and metal loadings in streams and quantify the uncertainty associated with these predictions. As discussed earlier in this chapter, the committee emphasizes that this model does not incorporate geochemical mechanisms describing chemical speciation of metals (URS Greiner, Inc. and CH2M Hill 2001b, pp. 5-24 to 5-32). The probabilistic model also does not make a distinction among metals associated with suspended load, bedload, and dissolved load, all of which may transport metals differently in the stream. The ability of the model to predict postremediation changes is addressed in Chapter 8 of this report.

Chemical Speciation and Sediment Transport in the Upper Basin (CSM Unit 1)

Chemical Speciation

The RI reported on water samples from mine adits, seeps, and surface waters in the upper basin (URS Greiner, Inc. and CH2M Hill 2001a). Data generally included temperature, pH, conductivity, dissolved oxygen, alkalinity, flow, and total acid-soluble and dissolved major and trace ion concentrations (Balistrieri et al. 1998). These data are important for understanding the sources of dissolved metals and provide some information to

[16]It is unclear to the committee how many surface-water samples were actually considered in this analysis. The RI states that 102 samples were collected and analyzed in this reach, yet data for 100 samples are presented for dissolved zinc in surface water at this location (URS Greiner and CH2M Hill 2001i, attachment 2, data summary tables). Data presented for dissolved zinc at this location (LC-60) for the probabilistic analysis show a summary statistic of N (number of samples) = 91, but only 38 data points are presented (URS and CH2M Hill 2001a, Appendix C).

ascertain chemical speciation. In Canyon Creek and Ninemile Creek, pH measurements of surface water varied from slightly acidic for some adits and seeps to slightly alkaline for in-stream measurements. Metals are anticipated to be mobilized from the minerals by the slightly acidic conditions and oxidizing environment; these processes are governed by the mineralogy of the area (for greater detail see Balistrieri et al. 1999).

A diffuse-layer model (Dzombak 1986) was used to evaluate the adsorption of dissolved metals (cadmium, lead, and zinc) onto ferric oxyhydroxides[17] (typically colloidal particles) in Canyon Creek surface waters. Results indicated minimal adsorption of zinc and cadmium at low flows suggesting that these metals are largely transported in the dissolved phase. Lead, on the other hand, is quite insoluble under these chemical conditions and is transported as a particulate or adsorbed onto ferric oxyhydroxides at high and low flows (URS Greiner, Inc. and CH2M Hill 2001c, p. 5-6). EPA used total and dissolved concentrations for each metal to evaluate the prediction of this model. However, the committee notes that these measurements are not capable of describing actual associations between the metals and iron oxyhydroxides.

EPA used an equilibrium speciation model (MINTEQA2) to estimate the precipitation and dissolution of metals in Canyon Creek and Ninemile Creek surface waters. The results of this model suggested that generally, cadmium, lead, and zinc are undersaturated in solution and not expected to precipitate (URS Greiner, Inc. and CH2M Hill 2001c, p. 5-7; URS Greiner, Inc. and CH2M Hill 2001g, p. 5-11).

Groundwater chemistry determinations in the upper basin tributaries consisted of measuring pH, salinity and specific conductance, oxidation-reduction (redox) potential, turbidity and sulfur species (namely, sulfide and sulfate). Monitoring wells in Ninemile Creek indicated freshwater conditions, near-neutral pH, low turbidity, oxidizing conditions, and sulfate concentrations ranging from 19,000 to 488,000 μg/L (URS Greiner, Inc. and CH2M Hill 2001g, p. 2-11). Similar results were reported for groundwater in Canyon Creek (URS Greiner, Inc. and CH2M Hill 2001c, p. 2-13 and 2-14), although slightly acidic (pH 4.5-6.5) groundwater was noted in the Woodland Park area. Such data provide some information from which inferences about metal chemistry and speciation can be drawn.

As discussed above, areas of tailings, waste rock, other process wastes, artificial fill, alluvium, and sediment generally greater than 5 acres were surface sampled, with few samples greater than 1 foot deep collected. Samples were analyzed for metals, but limited (if any) metal-speciation studies (for example, mineralogy) were performed on the samples collected. Surface samples are generally more oxidized, which can increase the mobil-

[17]Also referred to as hydrous ferric oxide.

ity of the metal, and therefore surface sampling does not provide a complete picture of metal locations, concentrations, speciation, or potential mobility throughout the entire source.

Sediment Transport

The USGS measured suspended and bedload sediment transport and stream discharge data for the water year 1999 at four gauging stations in the upper basin (Clark and Woods 2001). Cumulative transport curves were indirectly derived for the RI from the USGS transport curves (Clark and Woods 2001). These derived curves, presented in the RI (for example, see URS Greiner Inc. and CH2M Hill 2001c, Fig. 3.2-1) were developed from instantaneous measurements of discharge and sediment and, as such, were rating curves. EPA applied these rating curves to mean daily discharge to obtain daily sediment transport and the resultant cumulative curves. Annual loads for the upper basin tributaries appear to be derived from the cumulative transport curves. These annual loads and cumulative loads normalized to drainage area are reported in the RI (for example, see discussion for Canyon Creek in URS Greiner Inc. and CH2M Hill 2001c, pp. 3-2 to 3-4) and are tabulated in Table 4-2.

The sediment transport data for Ninemile Creek are unclear. Different values of annual sediment transport loads are reported in the RI for water

TABLE 4-2 Water Year 1999 Sediment Transport Loads for Upper Basin Watersheds

	Sediment Transported in Water Year 1999	
Watershed	Tons	Tons/Square Mile
Canyon Creek	1,440	62
Beaver Creek	No data available	No data available
Big Creek	1,400 (estimated from Canyon and Ninemile Creeks)	No data available (46, estimated from watershed area)
Moon Creek	No data available	No data available
Ninemile Creek	397, 400, 500, and 1,350 (See text for explanation)	34
Prichard Creek	No data available	No data available
Upper South Fork, Coeur d'Alene River	2,400 (estimated from Canyon and Ninemile Creeks)	48 (estimated from Canyon and Ninemile Creeks)
Pine Creek	2,900	37
North Fork, Coeur d'Alene River	25,400	28

SOURCE: URS Greiner, Inc. and CH2M Hill 2001a.

year 1999; values stated are 500 tons (URS Greiner, Inc. and CH2M Hill 2001g, p. 3-1), 400 tons or 34 tons per square mile (URS Greiner, Inc. and CH2M Hill 2001g, p. 3-13), 397 tons (URS Greiner, Inc. and CH2M Hill 2001e, p. 4-15), and 1,350 tons (URS Greiner, Inc. and CH2M Hill 2001g, p. 5-11). The 400-tons/year and 397-tons/year numbers appear to come from adding the suspended sediment and bedload values in the two cumulative transport curves (URS Greiner, Inc. and CH2M Hill 2001g, Figs. 3.2-4 and 3.2-5). The 500-tons/year estimate may have been derived by relating 1999 discharge (18.7 cubic feet per second) to the USGS rating curves; that method would have yielded daily values that had to be multiplied by 365, resulting in overestimation. The 1,350-tons/year value appears to be an error (P.F. Woods, USGS, personal comm., December 20, 2004).

No sediment transport data are available for the Upper South Fork of the Coeur d'Alene River and Big Creek because no gauging stations were located on these segments. EPA estimated annual sediment transport loads for these watersheds based on 1 year of sediment transport gauging data available for Canyon Creek and Ninemile Creek drainages (URS Greiner, Inc. and CH2M Hill 2001j, p. 3-2). Given the discrepancies of sediment load in Ninemile Creek for water year 1999, it is not clear to the committee which value EPA used to estimate sediment transport in the Upper South Fork of the Coeur d'Alene River and Big Creek. However, adequate estimates for these two tributaries probably could be made with the cumulative load normalized to drainage area for Ninemile Creek (34 tons per square mile).

The RI identified likely sources of sediment mobilization in various segments of each tributary based on reconnaissance with aerial photographs and topographic maps. It is not evident whether this was followed up by drainage walk-through evaluations. In any event, these eroding reaches would be potential candidates for bank, hillside, and/or channel stabilization to mitigate erosion and sediment transport. Examination of some historical records indicated that sediment transport in some tributaries was less in water year 1999 than in some previous years. EPA attributed this to remedial actions that have been undertaken in some watersheds (URS Greiner, Inc. and CH2M Hill 2001g, p. 3-1). However, another reason may be that in 1999 there was no notable flood event, which may have been responsible for greater sediment transport in some previous years when records were available. Characterization of sediment transport is provided by limited but useful monitoring data. While only one water year is focused on, the analyses provide useful information on sediment transport from watersheds, particularly in those watersheds where sediment data was actually collected (compared to those where sediment transport was only modeled). Analysis of historical aerial photographs to evaluate stream channel dynamics and sources of sediments is also a reasonable approach to generate information on sediment transport but cannot replace on-site evaluations for determining contributing sources.

Chemical Speciation and Sediment Transport in the Middle Basin (CSM Unit 2)

Chemical Speciation

Information on groundwater chemistry in the middle basin was based primarily on samples from monitoring wells in the box. Some historical data were used as well as data from more recent quarterly sampling of wells. Data presented for some groundwater samples included concentrations of dissolved metals, temperature, pH, conductivity, and major ions. In some cases, sufficient data were collected to evaluate chemical speciation (for example, URS Greiner, Inc. and CH2M Hill 2001k, Table 2.2-5). However, chemical speciation information for groundwater (or other media) was not reported in the RI.

Sediment Transport

Sediment transport in the middle basin (Wallace to Pinehurst) along the South Fork of the Coeur d'Alene River was measured at two USGS gauging stations, Silverton and Pinehurst, for water year 1999. Approximately 7,200 tons of sediment were transported past the Silverton gauge station and 22,000 tons at Pinehurst (URS Greiner, Inc. and CH2M Hill 2001k, p. 5-11). Suspended sediments and bedload samples were not analyzed for total metals, so mass loadings of metals were estimated from total and dissolved surface-water data (URS Greiner, Inc. and CH2M Hill 2001k, p. 5-13). The RI presented the following on sediment sources in this reach (URS Greiner, Inc. and CH2M Hill 2001k, p. 5-12):

> Based on interpretation of aerial photographs from 1984, 1991, and 1998, the majority of sediment supplied to the South [Fork] appears to be from remobilization of floodplain sediment that has entered the South Fork from tributary watersheds.

Much new information on the source of sediments, sediment transport, and deposition (Bookstrom et al. 2004; Box et al. in press) has been developed and reported since the RI was published. The interpretations of sediment sources (for example, floodplain, riverbed, or river banks) and transport based on aerial photographs should be revisited in light of more recent data.

Chemical Speciation and Sediment Transport in the Lower Basin (CSM Unit 3)

Chemical Speciation

For the RI, chemical data that could be used to assess chemical speciation and transport mechanisms in groundwater were limited for the lower basin (URS Greiner, Inc. and CH2M Hill 2001i, pp. 2-10 and 2-11). Spruill (1993), who monitored four wells on the north side of the river upstream of Killarney Lake, reported chemically reducing conditions at a neutral pH. From these data, it was proposed that reductive dissolution of iron and manganese oxyhydroxides would occur and in turn release sorbed trace metals to groundwater (URS Greiner, Inc. and CH2M Hill 2001i, p. 2-11).

Additional testing of interstitial pore water and solids from contaminated, water-saturated levees and marshes in the lower Coeur d'Alene River area (Balistrieri et al. 2000)[18] corroborated the work of Spruill (1993). The pH of pore water from all the sources tested was lower than the pH of river water, dissolved manganese and iron concentrations were elevated, and sulfate concentrations were below detection, all suggesting suboxic to anoxic conditions.

The fate of zinc under the oxidizing and reducing zones has important implications in remediation. In leaching studies that simulated dredging, Balistrieri et al. (2000) found that exposure of dredged riverbed sediments to water and air is highly likely to enhance zinc dissolution, making it necessary to consider treatment of water draining from the dredged sediment (see Chapter 8). EPA has stated (EPA 2004 [June 14, 2004]) the following:

> . . . [remedial] alternative development was based on typical conceptual designs (TCDs). The TCDs were not considered sensitive to the issue of speciation; rather speciation data developed was a level of detail more appropriate to post-ROD detailed design work.

This statement is incorrect because zinc mobilization during dredging has significant water treatment cost implications that the ROD (EPA 2002) should address (see Chapter 8 for further discussion).

After the RI was issued, Balistrieri et al. (2002) summarized findings on metal speciation and mobility of metals in the Coeur d'Alene River basin that were more extensive than the information available at the time the RI

[18]This study was conducted in support of the RI (URS Greiner, Inc. and CH2M Hill 2001b, p. 4-33), but results from the paper were not found in the RI, and a discussion was not included in seemingly appropriate sections of the RI (for example, URS Greiner, Inc. and CH2M Hill 2001i, Sections 2.2.4 and 2.2.5).

was issued. Analyses based on sequential extractions of samples were used to infer the speciation of metals in soils and sediments in the Coeur d'Alene River basin. Mineralogical analyses of samples were also reported. This study indicates that river sediments appeared to form authigenic sulfides near the surface.[19] The levee region, which alternates between wet and dry conditions, contains both oxidizing zones (an area where sulfide minerals may be converted to oxide and carbonate minerals) and reducing zones (an area where oxygen is limited and minerals begin to lose any associated oxygen). The oxidizing zones contain oxide-coated sediment grains, whereas the reducing zones contain detrital (particulate material of organic origin) and authigenic carbonate and sulfide phases. Balistrieri et al. (2000, 2002) also reported that detrital sphalerite was found in oxidized levee samples, leading them to conclude that a fraction of the sphalerite was resistant to oxidation. The mechanism for rendering this fraction of the material resistant to oxidation is unclear, although it could have occurred by armoring (coating) of the sphalerite grains with other materials.

Researchers at the University of Wisconsin-Eau Claire have also completed considerable work on chemical speciation (Morrison et al. 1999; Rowe et al. 1999; Hooper and Mahoney 2000, 2001; Thornburg and Hooper 2001; Plathe et al. 2004; Strumness et al. 2004). Because these collective works focus heavily on chemical conditions that mobilize and immobilize metals in the various environments of the lower basin, this information should play an important role in EPA's design phase for remedial action.

Sediment Transport

For water year 1999, the USGS used two gauging stations, one at Rose Lake and one at Harrison, to measure sediment transport (Clark and Woods 2001). Sand (material coarser than 65 μm) and fine (material finer than 65 μm) fractions were calculated separately and summed to determine the total suspended sediment discharge. Cumulative discharge for the year was calculated by summing the mean daily sediment discharges (URS Greiner, Inc. and CH2M Hill 2001i, p. 3-3). At the Rose Lake station, about 29,700 tons of suspended sediment (6,700 tons of sand plus 23,000 tons of fines) was transported. About 51,000 tons of suspended sediment was transported past Harrison in water year 1999, or about 34 tons per year per square mile, based on a drainage area of 1,475 square miles. Most of the sediment transport occurred during high-flow events, as would be expected. Suspended sediment and bedload samples were not analyzed for total metals,

[19]Authigenic sulfides are minerals—for example, zinc sulfide (sphalerite)—that are formed in place.

and mass loading was estimated from total and dissolved surface-water data (URS Greiner, Inc. and CH2M Hill 2001i, p. 5-8).

To estimate sediment transport in the years before water year 1999, the calculated discharges from the gauges were integrated with sediment transport relationships developed in water year 1999 (see URS Greiner, Inc. and CH2M Hill 2001i, Tables 3.2-1 and 3.2-2). For the RI, photographs of various locations throughout the main stem of the Coeur d'Alene River were used in an attempt to estimate erosion rates and sources of sediments (URS Greiner, Inc. and CH2M Hill 2001i, pp. 3-6, 5-9).

Since the release of the RI in 2001, much new information on sources, deposition, and transport of sediments and lead concentrations of the sediments on the bed, banks, natural levees, and flood basins of the main stem of the Coeur d'Alene River has been developed and documented by the USGS. This information, which has been compiled by Bookstrom et al. (2004) along with the implications for remedial design, greatly advances the understanding of sediment transport and fate in the lower basin and should serve as an excellent guide for EPA in the remedial design process.

Chemical Speciation and Metal Transport in Lake Coeur d'Alene (CSM Unit 4)

The chemical speciation and transport of soluble metals, particularly zinc, in Lake Coeur d'Alene are complex phenomena governed by multiple interactions that are not completely understood. The amount of dissolved zinc within the lake is regulated by imported and exported quantities, diffusive flux from sediments because of changes in speciation, and interactions with other biotic and abiotic components in the water column and sediments. For instance, there may be scavenging mechanisms whereby soluble zinc stemming from the Coeur d'Alene River may associate with phytoplankton (that is, become sorbed to the organic matrix of organisms' cells or incorporated into the silica of diatom frustules), which upon dying will fall out of the water column. At this point, the zinc may be sequestered (bound) in the sediments or may be liberated after the phytoplankton decompose. The liberated zinc may interact in situ with other components to sequester again, or it may migrate into the water column to undergo further interactions or be exported. The multitude of biotic and abiotic interactions includes complexation (binding) with dissolved, colloidal, or particulate organic matter, and association with other inorganic species, particularly with reactive iron and manganese oxyhydroxides. In addition, there may be seasonal or daily variations in hydrologic, geochemical, and biological interactions that drive these reactions, and the reactions may vary over the large and diverse area of the lake. Overall, the dynamics will remain difficult to monitor and predict with certitude.

Import/Export of Metals in Lake Coeur d'Alene

Although the dominance and interplay of these multiple chemical interactions will be difficult to ascertain, a more general picture of metal dynamics within the lake is accessible. Owing to a large data-collection effort conducted primarily by the USGS in conjunction with the Coeur d'Alene tribe, a tremendous amount of information exists to evaluate relative input and output of dissolved metals into Lake Coeur d'Alene. With these data, metal loads have been derived for a series of water years 1992-1997 and 1999. On an annual basis, more metals enter the lake than leave, and this attenuation is more pronounced for lead than zinc. The analysis shows that for the available years a median of 32% of dissolved zinc input was retained in the lake and 92% total lead was retained (URS Greiner, Inc. and CH2M Hill 2001l, p. 5-4). Upon considering the input and output on a monthly basis, the picture is less clear. During most of the year, zinc inputs exceed output; however, in the spring of the year, the reverse is true (Figure 4-2). It should be noted that data are used to estimate this one year; however, the dynamics could have large interyear differences, and the uncertainty of these monthly estimates is not depicted in this figure derived from the RI.

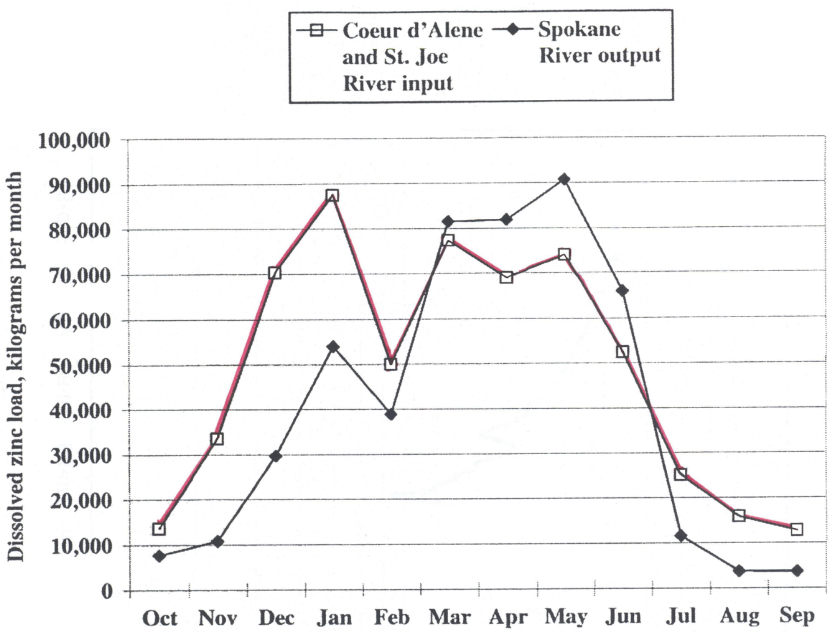

FIGURE 4-2 Measured dissolved zinc load input (Coeur d'Alene and St. Joe Rivers) and output (Spokane River) for Coeur d'Alene Lake, water year 1999. SOURCE: URS Greiner, Inc. and CH2M Hill 2001l.

Mass-Balance Modeling for Lake Coeur d'Alene

Within the RI, EPA developed a mass balance for various metals within the lake (see Figure 4-3 for dissolved zinc). The mass-balance model for dissolved zinc is based on measured and estimated components. The riverine inputs and Lake Coeur d'Alene output are derived from USGS water monitoring from water year 1999 (Woods 2001), the benthic input is estimated from USGS studies (Kuwabara et al. 2000), and the transformation percentage is essentially a factor devised to accommodate the excess 445,000 kg/year input.

The benthic flux estimate was derived from USGS work conducted to evaluate the significance of releases of metals from the sediments within the lake compared with inputs from the Coeur d'Alene River (Kuwabara et al. 2000) (essentially to determine whether the sediment is serving as a source of dissolved metals by emitting the constituent or as a sink by consuming it). Three techniques were tested, but the in situ benthic-flux chamber method was chosen as being most representative for these calculations (URS Greiner, Inc. and CH2M Hill 2001l, p. 5-19). This study used a benthic lander, a rectangular acrylic chamber that isolates 1,500 cubic centimeters of lake-bottom sediment surface and the overlying water. Deployment is for anywhere from a half day to 2 days and water is sampled throughout the deployment. Figure 4-4 presents results from this study, illustrating an increase in dissolved zinc and a decrease in dissolved oxygen over time in the benthic chamber. These results indicate that the sediments do serve as a source for zinc under the conditions tested.

The benthic lander data for zinc were obtained from two or three deployments at two locations (Table 4-3). These data were further win-

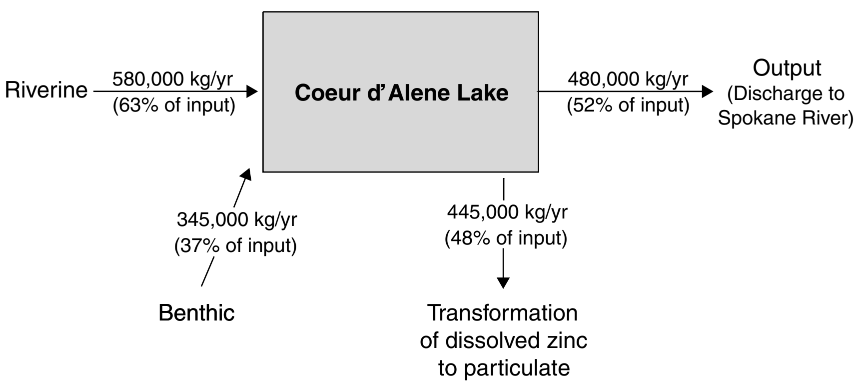

FIGURE 4-3 Dissolved zinc mass balance of Lake Coeur d'Alene. SOURCE: Adapted from URS Greiner, Inc. and CH2M Hill 2001l, Table 5.6-1.

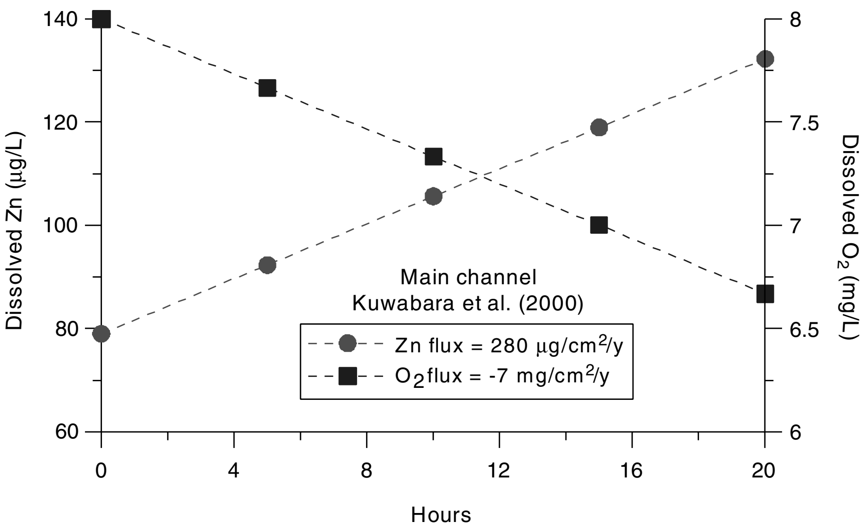

FIGURE 4-4 Zinc and oxygen concentrations over time within the benthic lander chamber. SOURCES: Balistrieri 2004; data from Kuwabara et al. 2000.

nowed in the RI where results from the two sites are averaged[20] to estimate the annual flux from the entire lake by multiplying the flux estimate by the surface area of the lake bottom. The derived number represents a major contribution of soluble zinc to the lake in the mass-balance model (37%; see Figure 4-3).

Considering the large surface area of contaminated sediments in the lake, estimated at 42 square miles (URS Greiner, Inc. and CH2M Hill 2001l, Table 5.6-3), the spatial coverage of the lander experiments is necessarily limited. The temporal component is also limited considering the experimental time period spans only a couple of days in August 1999. Zinc dynamics could have a strong seasonal component, yet this insight is not available from the benthic flux studies. Indeed, these limitations are recognized and stated by EPA (URS Greiner, Inc. and CH2M Hill 2001l, p. 5-24):

> The benthic fluxes were less robust because of their limited spatial and temporal resolution; benthic flux was measured only during mid-August of the 1999 water year and at only two locations. No information

[20]Actually, the flux data for zinc at each of the two sites are averaged, and then the average of these two averages is used as the overall benthic flux estimate.

TABLE 4-3 Benthic Flux Results

Deploy #	Station Location	Zn Flux (μg/cm²/y)
2	Mica Bay	451.0 ± 100.2
4	Mica Bay	243.4 ± 33.4
1	Main Channel	348.4 ± 71.6
2	Main Channel	198.1 ± 28.6
3	Main Channel	295.9 ± 45.3

SOURCE: Kuwabara et al. 2000.

is available on the magnitude of temporal variations of benthic flux in Coeur d'Alene Lake.

Overall, although the benthic flux work is important and interesting, it is quite limited spatially and temporally; consequently, the present database has limited utility in the evaluation of annual benthic flux for the entire lake.

Another limiting component of the mass balance presented in Figure 4-3 relates to estimates of the dissolved zinc converted to particulate form. This parameter is derived from the other components of the mass balance to compensate for the differences between the estimated input and the measured output of the lake. Figure 4-3 indicates that 48% of the estimated input is converted into particulate form. This value is estimated simply by subtracting the measured output of the lake from the estimated input (measured riverine loading + estimated benthic flux). There have been no measurements or sediment trapping studies to estimate or verify this value even though the large removal mechanism is a central component describing the lake's mass balance. Indeed, the magnitude of this mass-balance estimate is dependent on the benthic flux estimate because the benthic flux estimate represents a large portion of the lake's loading. Again, the RI states these limitations: "The removal of dissolved zinc from the water column was assumed to be due to the transformation to the particulate fraction. However, there are no sediment data to support this assumption" (URS Greiner, Inc. and CH2M Hill 2001l, p. 5-35).

To move beyond an annual mass-balance model, a model depicting zinc loads imported and exported to the lake through time was developed. However, this model has the same limitations as the annual mass balance. Here, the benthic flux input parameter, as designated in the RI, results from the same work described above and has the same limitations regarding spatial and temporal resolution. And, again, the "transformation parameter" (in this model, a "scavenging coefficient") is solely a fitting factor designed to comport the model's output with the measured zinc concentrations in Lake Coeur d'Alene discharge. In this case, the scavenging parameter varies by an order of magnitude throughout the year so that the model output will comport with measured data (URS Greiner, Inc. and CH2M Hill 2001l, p. 5-34). Although there may be biological and geochemical processes responsible for removing dissolved zinc that are of this magnitude and variability, no studies exist to support the conclusion (URS Greiner, Inc. and CH2M Hill 2001l, p. 5-35).

In general, the mass-balance models do not reflect a firm understanding of the lake's metals dynamics, considering that close to 40% of the input is derived from useful, but very limited, benthic flux data, the mechanism driving 50% of the removal has not been measured or monitored, and the removal mechanism is not understood.

Inflow Plume Routing in Lake Coeur d'Alene

Preferential routing of metals-rich discharges from the Coeur d'Alene River through the lake is another phenomenon believed to affect the disposition of metals (URS Greiner, Inc. and CH2M Hill 2001l, p. 5-9). As water enters the lake from the river, uniform mixing may not occur. Temperature differences between river and lake waters are believed to affect density to such an extent that warmer river waters will spread as a buoyant plume over the top of colder lake waters (overflow) with limited mixing, or colder river waters will move to the bottom of the warmer water in the lake (underflow) again with limited mixing. In overflow conditions with substantial flows, there may be preferential transport of dissolved and suspended constituents to the outlet of Lake Coeur d'Alene into the Spokane River, and this may explain how particulate lead reaches the Spokane River. Researchers (Brennan et al. 2000; P.F. Woods, unpublished material, USGS, 2000, as cited in URS Greiner, Inc. and CH2M Hill 2001l) from the USGS monitored this phenomenon in June 1999. Their results documented a layer of warmer water above cooler water; the warmer water contained elevated concentrations of total lead, decreased zinc concentrations, and decreased light transmission. This profile of physical and chemical constituents is similar to the presumed riverine sources. From a fate and transport perspective, the implication is that the preferential routing of overflow will carry constituents (principally particulate lead) through Lake Coeur d'Alene into the Spokane River instead of the lake serving as a settling basin where particulate-bound lead can settle from the water column. Indeed, the USGS data (Brennan et al. 2000; P.F. Woods, unpublished material, USGS, 2000, as cited in URS Greiner, Inc. and CH2M Hill 2001l) from this event indicated elevated lead concentrations transported through the lake. There is a concern in the Spokane River about accumulation of lead-enriched sediments in eddy areas and beaches where humans may recreate. Consequently, the monitoring and understanding of these events are important in comprehending the dynamics of pollutant transfer in the lake. The USGS has continued its efforts to document and understand this phenomenon through additional sampling. These efforts will remain important in understanding transport of contaminated sediments through the system.

The RI (URS Greiner, Inc. and CH2M Hill 2001l, p. 5-9) further attempts to document the overflow phenomenon by comparing water temperatures from the Coeur d'Alene and St. Joe Rivers with temperatures at the deepest point in the lake. This text states that

> overflow was the most common mode of inflow plume routing, occurring in about 60% of the comparisons. Interflow or underflow each occurred in about 20% of the comparisons. Overflow was present in all months except October, November, and December.

However, it appears difficult to make these statements about an overflow or underflow condition with the information provided. Indeed, this conclusion apparently is drawn by noting a difference of only a few degrees Celsius between the rivers and a mid-lake station located several miles away, and it does not present evidence that the upper water column is preferentially enhanced in chemical constituents derived from riverine sources. Although overflow or underflow conditions may have been occurring, and the month-by-month breakdown may make sense in terms of expected seasonal water temperatures, existence of the overflow/underflow phenomena during these months is not established by the data presented. The aforementioned USGS monitoring will be important to document the ubiquity of the overflow phenomenon.

Thermal Stratification

While the implications of overflow and underflow of the Coeur d'Alene River plume through Lake Coeur d'Alene are discussed in the RI, little discussion is provided regarding the effect of thermal stratification and turnover on metals dynamics in the lake. According to the RI (URS Greiner, Inc. and CH2M Hill 2001j, p. 5-9), Lake Coeur d'Alene is dimictic (thermal stratification breaks down and the water column undergoes mixing, or turnover, in the fall and spring). During stratification, which typically occurs in the summer months, the lower water column (hypolimnion) does not mix with upper water column (epilimnion). Constituents (for example, dissolved metals) that build up in the hypolimnion during this period of thermal stratification can be released during turnover and affect the release of metals from the lake. The RI provides data suggesting that the hypolimnion contains elevated dissolved zinc compared to the epilimnion during July and August when the lake would be expected to be stratified (URS

Greiner, Inc. and CH2M Hill 2001l, Tables 5.7-9 and 5.7-10). However, the potential for stratification and turnover to affect metals distribution and discharge is not examined.

Water-Quality Study of Lake Coeur d'Alene: Are Nutrients a Problem?

An extensive water-quality study of Lake Coeur d'Alene was initiated in 1991 by the Idaho Department of Environmental Quality (IDEQ), Coeur d'Alene tribe, and USGS with three objectives:

1. Determine the lake's ability to receive and process nutrients (phosphorus and nitrogen) to devise measures that will prevent water-quality degradation.
2. Determine the potential for releases of heavy metals from lakebed sediments into the overlying lake water.
3. Develop information to support a lake management plan that will identify actions needed to meet water-quality goals.

Woods and Beckwith (1997) report on this study and provide an evaluation of the nutrient and trace-metal balance of the lake. A nutrient load/lake response model was used to simulate Lake Coeur d'Alene's limnologic[21] responses to alterations in water and nutrient loads delivered to the lake. The empirical mathematical model simulated the following eutrophication-related variables[22]: concentrations of total phosphorus, total nitrogen, and chlorophyll a; secchi-disc transparency;[23] and hypolimnetic dissolved oxygen deficit. The model was calibrated with 1991 data. After calibration, the model's applicability to Lake Coeur d' Alene was verified with 1992 data. After calibration and verification, the model was used to simulate the lake's responses to various nutrient-management scenarios. The following two principal questions were addressed:

1. Would large increases in nutrient loads cause the lake's hypolimnion to become anoxic?

[21]Limnology is the study of relationships and productivity of freshwater biotic communities and how physical, chemical, and biological environmental parameters affect these communities.

[22]Eutrophication is a term applied to a body of water when increased minerals and organic nutrients reduce the dissolved oxygen, producing an environment that favors plant over animal life.

[23]A secchi disc is used to measure the transparency of water for lake quality studies. The secchi disc depth is a function of the absorption of light in the water column above the disc. The secchi depth is thus influenced by the absorption characteristics of water and its dissolved and particulate matter.

2. Would the lake's water quality be substantially improved by large reductions in nutrient loads?

Woods and Beckwith (1997) found that much more than a quadrupling of nutrient input would be required for the northern portion of the lake to become anoxic (devoid of oxygen)—a very unlikely event. Nutrient reduction from wastewater treatment plant discharges to the lake was predicted to produce the greatest improvement in water quality as measured by chlorophyll a and secchi-disc transparency.

The Coeur d'Alene Lake Management Plan Update (IDEQ 2004) sets forth the present status of actions for the management of the lake. The technical basis for the lake management plan largely remains based on these studies, as well as more recent water-quality monitoring conducted by USGS, the Coeur d'Alene tribe, and IDEQ.

Speciation of Metals in Lake Sediments

A number of studies have been conducted to determine zinc speciation in the lake sediments, but none is without associated possible error due to sampling, sample handling, or analysis factors. Two commonly used procedures to infer the chemical speciation of metals in aquatic sediments are the Tessier sequential fractionation procedure (Tessier et al. 1979) and the acid volatile sulfide–simultaneously extracted metal (AVS-SEM) procedure (Allen et al. 1993). The Tessier procedure is based on (1) extracting a sediment sample with extractants of increasing strength and (2) determining the metal that is released with each extractant. These releases are related to operationally defined geochemical phases. The AVS-SEM analysis is based on adding cold hydrochloric acid to a sediment sample and trapping the volatilized hydrogen sulfide. The molar amount of sulfide released is compared with the sum of the moles of trace metal, excluding iron, dissolved in the acid. If the amount of sulfide (AVS) exceeds that of the metal (SEM), it can be concluded that there is a sufficient amount of sulfide for the metals (in this case lead, zinc, cadmium, and copper) to be present as sulfides rather than as more soluble oxyhydroxides, carbonates, or sulfates. Pyrite is not detected in this procedure. The iron dissolved in the acid is not included in the comparison because it is much more soluble than are the sulfides of the trace metals. Thus, FeS (iron sulfide) acts as a reservoir to maintain sulfur for precipitation of these trace metals.

Horowitz et al. (1993) conducted an extensive sampling of the surficial sediments, which they found to be enriched in a number of trace elements. Samples were freeze-dried before analysis, resulting in the oxidation of the more labile (readily broken down) sulfide compounds. A number of the samples were subjected to a two-step procedure to partition the trace metals into an iron oxide phase and an organic/sulfide phase. The acidic first step would dissolve not only the metal oxides but also many of the metal sulfides in the same manner as the AVS-SEM procedure, which uses acid to release the metal and the sulfide. Therefore, the presence of metal sulfides in the sediment cannot be ruled out.

Harrington et al. (1998) used the Tessier sequential fractionation procedure (Tessier et al. 1979) to characterize the phase associations of trace metals in core samples taken from the lake. However, as noted by Horowitz et al. (1999), the cores were sectioned at 8 cm intervals and Harrington et al. (1998) reported the redox boundary[24] to be at approximately 2 cm. The first section would have a mixture of oxidized and reduced sediment present. At least a portion of any zinc sulfide present would have been dissolved in the acidic oxalate solution that is designed to characterize the metal oxide fraction. Harrington et al. (1999) used the AVS-SEM procedure, indicating that a substantial amount of the zinc may be present as zinc sulfide. However, the fact that the samples were from cores that were sectioned at 8 cm intervals and AVS-SEM assayed at 4 cm intervals casts doubt on the association of the metals at the water-sediment interface.

Lake Coeur d'Alene Studies: The Bottom Line

What can be easily understood from evaluating the complex phenomena in Lake Coeur d'Alene is that the better the data sets, the more thorough the understanding and ability to make informed statements about metals dynamics in the lake. The committee has found that there are large amounts of high-quality monitoring data that have been collected on the lake, particularly by the USGS, the Coeur d'Alene tribe, and IDEQ. The use of this information permits an understanding of the overall behavior of the variety of metal contaminants within the lake and, for example, elucidates the likelihood that overflow events can preferentially transport materials through the lake under certain conditions. However, data documenting metal interactions, internal cycling, and benthic flux are limited.

In future studies of metals in the sediment of the lake, more attention should be given to certain aspects of sampling, analysis, and interpretation. In particular, the depth of the oxidized layer will vary seasonally as a consequence of oxidation (breakdown) of algal detritus. The seasonality of the thickness of the oxidized layer should be evaluated along with the concurrent changes in the sulfide contained in the sediment. Coring studies on the lake provide great insight to the historical depositional pattern on

[24]Oxidation-reduction boundary differentiating between mineral species that are more chemically oxidized (for example, metal oxide or metal carbonate species) and those that are more chemically reduced (for example, metal sulfide minerals such as zinc sulfide).

metals-enriched sediments (Horowitz et al. 1995; Woods 2004). These cores could also provide a useful diagnostic on the long-term trends of the metal content and amount of deposited sediments and potentially on the effect of the basin's remedial activities on sediment transport and deposition in Lake Coeur d'Alene (A. Horowitz, USGS, Atlanta, GA, unpublished material, June 17, 2004).

To comprehend the lake dynamics and the effect of various management practices on phytoplankton production and metal fluxes from the sediment, additional experiments will be necessary. For example, water-column sedimentation trap experiments would be useful to elucidate the removal of dissolved metals from the water column by phytoplankton. The USGS is planning to develop a model that predicts the flux of metals from the sediment to the overlying water such as those discussed by Di Toro (2001). Such a model would use nutrient input to compute primary production. Appropriate sampling, including seasonal sampling of sediments for the analysis of AVS and SEM, will be needed for model calibration. The committee particularly notes the potential for nutrient management actions to affect the zinc concentrations in the lake water.

The committee recognizes that some studies are ongoing and supports further monitoring and modeling studies to understand the interplay between the hydrologic, geochemical, and biological phenomena driving the disposition of metals within the lake. The committee's understanding has benefited from the available basic information on hydrologic and chemical data (particularly metals) and suggests a continued development of such data in order to assess long-term trends.

Chemical Speciation and Sediment Transport in the Spokane River (CSM Unit 5)

Chemical Speciation

It appears that few chemical speciation studies have been conducted in the Spokane River basin. As provided by Kadlec (2000) in an ecologic risk analysis, Bailey and Saltes (1982) demonstrated that most of the zinc is in soluble form in the Spokane River. Johnson et al. (1990) reported that 73% of the zinc was in the dissolved phase (<0.45 μm diameter) in Lake Roosevelt, and Pelletier (1994) reported the ratio of dissolved to total fractions to be 69% for cadmium, 18% for lead, and 83% for zinc. Naturally occurring organic and inorganic solids did not appear to influence the bioavailability of these metals (Bailey and Saltes 1982; Kadlec 2000). Lead was reported as being mostly associated with suspended particles (Kadlec 2000).

Subsequent to publication of the RI, studies by Box and Wallis (2002) and Box et al. (in press) indicate that zinc in the Spokane River is mostly in dissolved form during low flows, but during high-flow events, zinc in the sediments is mobilized and a significant portion of the zinc load is in particulate form.

Sediment Transport

The largest sources of sediment (URS Greiner, Inc. and CH2M Hill 2001m, p. 5-9) to the Spokane River are remobilization of channel bed material, bank erosion, and tributary channels. Lake Coeur d'Alene is a source of the smallest and lightest particles, as discussed in the preceding section of this chapter. The fine-grained sediments in the Spokane River are contaminated with lead and zinc. Metal concentrations generally decrease from upstream to downstream (URS Greiner, Inc. and CH2M Hill 2001m, p. 5-1). Sediment transport is controlled by dams and reservoirs on the Spokane River, with large amounts of sediment deposited in the reservoirs; however, fine-grained sediments appear to be transported through the reservoirs (URS Greiner, Inc. and CH2M Hill 2001m, p. 5-9).

CONCLUSIONS AND RECOMMENDATIONS

Conclusion 1

The EPA did not fully consider the importance of the interacting processes of surface- and groundwater flow, metal flux, metal storage in sediments, and metal-bearing sediment transport and deposition with relevant aspects (fish habitat, forest management, climatologic variability, etc.) of the Coeur d'Alene River basin system. Because the basin has not been considered in the framework of a system and inadequate attention has been devoted to hydrologic and climatic variabilities, in particular, the CSMs seemingly are based primarily on average conditions.

Because characterization of the CSMs and the conclusions and decisions that stem from these models are based on average conditions, these decisions—for example, the definition of possible remedies—may not be fully protective of aquatic species or robust enough to withstand severe events. Extreme events are more important than averages because organisms respond to extreme events. Solid-phase contaminants are often transported during high flow (an extreme event), and concentrations of dissolved-phase contaminants are often highest during low flow (an extreme event).

Conclusion 2

The way EPA has compartmentalized the basin into OUs for remediation is inconsistent with a "systems approach" (see Box 4-1) to investigating the basin, and this compartmentalization has created some serious technical difficulties and public perception problems for EPA.

The current OU structure may have made sense in the beginning of the Superfund investigations, but it is inconsistent with the natural hydrologic and chemically linked systems operating within the basin. A systems approach based on watershed boundaries is a more appropriate means of properly characterizing contaminant sources and paths of contaminant transport. Although the committee recognizes that the OU approach was adopted by EPA to prioritize human health risks, the artificial constraints have created problems for EPA in protecting fish downstream of the box, because a large portion of the dissolved zinc (modeled at 41%) comes from sources that apparently cannot be addressed by OU-3 actions. Public perception problems arise from the fact that the agency seems to have reversed its original position, which was to deal with the environmental problems outside of the box using programs other than Superfund. This reversal undermined the public's trust and confidence.

Conclusion 3

The total number of samples collected from the entire basin area was small in relation to the large area extent of the basin and the complexity of the site, and source terms[25] were not well defined; nevertheless, trends related to contaminant transport and fate, especially for surface water, were definable from the samples that were collected.

17,000 samples were collected throughout the basin, and 1,080 mining-related source areas were identified. Approximately, 160 (15%) of these source areas were sampled with about five surface and near-surface samples collected from most tailings and sediment sources of 5 acres or more. Because the basin is such a large and chemically and hydrologically complex site—and contaminant distribution can be very heterogeneous with hot spots being less than an acre in size—this number of samples, although large, is insufficient to quantify the source terms. Leachability data were not obtained to support OU-3 decision making. Measured increases in dissolved metal loadings in streams were used to infer sources, such as nearby floodplain sediments and tailings.

[25]The phrase "source term" is defined as the amount and chemical form of a contaminant released to the environment from a specific source over a certain period of time. Source identifies the nature and origin of the release and term refers to how much of a substance, or metal in the case of the Coeur d'Alene basin, is released to the environment over a specified time period.

Conclusion 4

Estimated average mass loading of metals to the Coeur d'Alene River and Lake adequately depict an overall description of contaminants moving through the basin, but such data should not be substituted for comprehensive source characterization and remedy design for worst-case conditions.

The committee commends the agency for cooperating with other federal and state entities in conducting a variety of new studies that will provide new and improved interpretations of contamination in the basin and can be used in the next steps of the Superfund process.

Conclusion 5

Understanding the dynamics of groundwater movement, the incorporation of dissolved metals from the aquifer materials, and the complex relationship between surface water and the shallow groundwater aquifer will require comprehensive study and is necessary because groundwater is the primary source of dissolved metals into the surface water of the basin.

The investigations conducted to document concentrations of dissolved metals within the basin focused primarily on monitoring surface-water concentrations. A more limited campaign to sample groundwater was undertaken. Yet most of the zinc load in the basin is contributed by groundwater. Understanding the dynamics of groundwater movement and the incorporation of dissolved metals from the aquifer will undeniably require additional characterization.

Conclusion 6

Selecting lead and zinc as indicators of COPCs is reasonable, but caution is advised in extrapolating the behavior of these metals to other contaminants.

Zinc accounts for about 96% of the dissolved metal loading to Lake Coeur d'Alene. Lead is primarily transported as a particulate and is also a metal of major concern. Zinc, which is cationic, may have different transport characteristics from arsenic, which is anionic and undergoes redox transformations under the environmental conditions of the basin.

Conclusion 7

EPA addressed background determinations in a manner consistent with the agency's established guidelines and is commended for determining site-specific background concentrations of COPCs. The background concentrations developed for the ROD were reasonable, but these background concentrations were not used appreciably, with the exception of the Spokane

River, to select remedial goals or select target cleanup levels when used in conjunction with risk-based values. This decision is appropriate because of the disparity between the cleanup levels and the background levels.

EPA followed guidelines, as understood by the committee, for determining background concentrations for soils, sediments, and surface waters in the various basin areas. Background concentrations typically are determined to estimate the extent of contamination and to assist in selecting remedial goals or target cleanup levels. The agency compared contaminant levels with background. However, background was not used appreciably, except for the Spokane River, for the latter purpose, because under the interim cleanup, achieving background is irrelevant. There is a large disparity between the contaminant levels and background concentrations, particularly for soils and sediments. Although coring studies and techniques for background were appropriate, aspects of the sampling and background derivation methodologies were problematic. However, this has little practical effect because proposed remedial actions are not governed by background concentrations.

Conclusion 8

Owing to the complexity of metals dynamics in Lake Coeur d'Alene, additional supporting technical information is needed to develop an effective lake management plan.

The relationship between eutrophiciation and metals release is not completely understood. Zinc transport through the lake is a complex and dynamic process with seasonal variations, and the understanding of this process is continuing to evolve.

Conclusion 9

Information on chemical speciation of contaminants is limited and was not considered to any significant extent in decision making in the ROD. Recently available information on the sources, deposition, and transport of metals and sediments will be especially important in the design phase of the Superfund process.

Understanding the chemical speciation of metals is important for understanding the dissolution of metals from sources, such as tailings and floodplain sediments, and their bioavailability. Some chemical speciation studies of metals were undertaken in Canyon Creek and Ninemile Creek, and similarly important studies were conducted to estimate dissolution of zinc during dredging in the lower basin. RI sediment-transport studies were limited to water year 1999, but extensive studies by USGS have been ongoing in the lower basin and will provide much needed information for remedial design.

Recommendation 1

EPA is encouraged to incorporate in remedial planning new data that have been made available by USGS, Coeur d'Alene tribe, U.S. Fish and Wildlife Service, IDEQ, and others since issuance of the ROD. Furthermore, the agency is urged to proceed, as planned, with more-thorough source identification before proceeding with cleanup to ensure the location, magnitude, and disposition of contaminant sources.

Recommendation 2

An understanding of dissolved metals, particularly zinc, that accounts for the delivery to and from groundwater and surface waters needs to be developed. The chemical and hydrological components need to be sufficiently rigorous to permit use of the information to evaluate the consequences of alternative remedial actions to the input of dissolved metals to the basin.

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5

Human Health Risk Assessment in the Coeur d'Alene River Basin

INTRODUCTION

The objective of this chapter is to present an overview of the manner in which a human health risk assessment (HHRA) is conducted and then to describe in stepwise fashion the procedures that the U.S. Environmental Protection Agency (EPA) and its partners followed in conducting the Coeur d'Alene River basin HHRA (TerraGraphics et al. 2001). The Coeur d'Alene River basin HHRA for the area extending from Harrison to Mullan, Idaho, was jointly prepared by the Idaho Department of Health and Welfare (IDHW), the Idaho Department of Environmental Quality, and EPA Region 10. Oversight and guidance were provided by the Governor's Advisory Council on Human Health Risk Assessment, which included the Lieutenant Governor of Idaho. The five-member EPA Technical Review Workgroup for lead ultimately conducted an independent review of the document. Finally, numerous citizens, tribal representatives and community organizations provided or facilitated reviews and comments of a public draft of the document. Below, we summarize and critique the outcome of that effort. It should be noted that issues that the committee considered as the most important are emphasized in the review. A comprehensive and exhaustive review of all assumptions used in EPA's assessments and their underlying scientific basis was beyond the scope of what the committee could be expected to accomplish.

General Objectives of an HHRA

The objectives of an HHRA are two-fold: first, to estimate the level of risk to human health associated with concentrations of environmental contaminants; and second, if that risk is found to be unacceptable, to calculate media-specific cleanup levels that will protect human health.

Risks are estimated for current uses of a site as well as foreseeable future uses. All contaminated media are considered (for example, soil, water) if individuals are likely to be exposed to the media. All relevant routes of exposure are also considered, including direct contact, such as inhalation, ingestion, and dermal exposure, and indirect contact, such as exposure to vegetables that have taken up contaminants through the soil or water.

Cleanup levels are calculated based on the relationship between contaminants and risk as defined in the risk assessment and a policy decision (risk management) about the level of risk that is considered acceptable. As a result, cleanup levels for a single contaminant can vary from one site to another either because the relationship between environmental levels and risk differs or because different policy decisions have been made concerning the level of acceptable risk.

Overview of the Superfund HHRA Process

HHRA typically is described as including four steps: hazard identification, exposure assessment, toxicity assessment, and risk characterization. Early in the development of the field of risk assessment, *hazard identification* referred to determining which chemicals or compounds at a site could lead to risk. Today, the list of chemicals and compounds with associated human health risks are well known, and the first step has changed to data collection and analysis, including collecting data on the characteristics of the site and the chemicals or compounds of concern.

The second step in HHRA involves *exposure assessment*, including identifying the populations of individuals exposed to hazards at the specific site and how those exposures may occur. For example, the Coeur d'Alene River basin HHRA identifies children as the primary population of concern for lead exposure and identifies the presence of local American Indian populations. Potential pathways of exposure are defined, such as children ingesting soil and house dust contaminated with lead, and American Indian ingestion of locally grown foods contaminated with lead. At other sites, exposures could include scenarios such as inhalation and dermal exposure to volatile chemicals in groundwater while showering. In addition to identifying the potential pathways of exposure, this step may involve defining several parameters (for which there are insufficient measured data) that will govern the estimated risk from each exposure pathway. These are often referred to as assumptions, or default values, and they are assumed to be representative of a population, although they often include a conservative safety factor. These parameters include things such as time spent indoors and outdoors, which can differ as a function of climate.

The third step is *toxicity assessment*, or identifying and quantifying a chemical's or compound's intrinsic toxic properties. Again, at this point in the development of risk assessment, based on numerous controlled animal and/or human experiments and on epidemiological studies, toxicity parameters have been established by EPA and other agencies for many of the major chemicals and compounds. At times, when a great deal of information is known about a compound's toxicity, this step involves examining an EPA database for the chemical-specific cancer slope factor (SF) or reference dose. But for many compounds found at Superfund sites, much less is known, and there are myriad assumptions made that often prove very controversial.

The fourth step, *risk characterization*, combines the results of the first three steps into an estimate of risk. The estimated risk is then compared with a level of risk deemed "acceptable" according to risk management decisions (see below), and the site is thereby identified as either having acceptable risk levels or in need of remedial measures.

All the risk assessment steps described above inherently incorporate uncertainty. Each of the steps generally involves extrapolation from observations in one set of circumstances (for example, the effect of known, high doses of a chemical given to laboratory animals over a short period) to the circumstances of interest (for example, the potential effects of unknown, small doses of a mixture including the tested chemical on humans over a lifetime). Each such extrapolation introduces qualitative and quantitative uncertainties; and an adequate HHRA should describe qualitatively—and, if possible, quantitatively the sizes and types of such uncertainties.

One additional tenet of the Superfund HHRA process bears discussion, and that is EPA's preferred focus on the individual with reasonable maximum exposure (RME). A risk assessment generally includes a calculated estimate of the likely risks for an average individual—the central tendency (CT)—*and* for an individual experiencing RME conditions. EPA defines RME as the highest exposure that is reasonably expected to occur at a site. Generally, the RME risk is compared with the acceptable level of risk when determining whether remedial measures are needed.

If risks are found to be unacceptable, thus requiring remediation, then the models used in the risk assessment can also be used to determine acceptable concentrations of contaminants, equated to "cleanup levels." It is important to note that a cleanup level calculated in this way is applicable over the same geographic area that was assessed in the risk calculation and represents the same mathematical formulation used for the concentration term in the risk assessment. For example, if the chronic risk to a child exposed over several years to the average contaminant concentration in his/her yard is found to be unacceptable, then a cleanup level derived from the corresponding risk equation will represent the acceptable average concentration for soil in the yard. As a further example, if a risk calculation focused solely on a heavily used play area finds unacceptable risk, then the cleanup level calculated from that risk equation will represent the acceptable average concentration for the play area. However, the derivation of an actual cleanup level is typically controversial, partly due to the uncertainties associated with each piece of information that go into the mathematical derivation of the cleanup number.

Finally, a distinction needs to be drawn between risk assessment and risk management. Simply put, risk assessment is scientific and involves identifying pathways of exposure and some mathematical calculations; risk management involves policy and societal values. Cleanup levels are calculated on the basis of a policy decision about the level of acceptable risk as well as on the basis of the mathematical risk assessment. Further, the assessment of uncertainty in a risk assessment may lead to the development of more than one possible cleanup level or a range of cleanup levels. A risk manager will choose a cleanup level from the range after considering other site characteristics such as technical feasibility of the remediation, public desires, and so forth. As a result, a cleanup level may not be directly linked to an actual risk calculation, but it is generally expected that the cleanup level chosen during the risk management process will fall within a range developed in the course of the risk assessment.

Geographic Area Considered in the Coeur d'Alene River Basin HHRA

The Coeur d'Alene River basin HHRA considered an area that included the South Fork of the Coeur d'Alene River, its tributaries, and the main stem of the river west of its confluence with the North Fork. The region of interest spans roughly 53 miles from the Idaho-Montana border to Lake Coeur d'Alene and excluded the 21-square-mile Bunker Hill Superfund site. The towns of Mullan, Osburn, Wallace, and parts of Pinehurst, Idaho, are all included and all lie within Shoshone County.

Demographics of the Population

The demographic characteristics of the Coeur d'Alene River basin are primarily a function of its mining past and were strongly affected by the closure of the Bunker Hill smelter in 1981. Since the smelter ceased operations, the region has suffered chronically high unemployment, averaging 12.3% in the 1990s, about twice the state average. In 2001, the per capita income was just over $19,000, or 78% of the state value (Idaho Department of Commerce 2004). The lower wage base is accompanied by an increase in poverty; according to the 2000 U.S. census, 12.4% of the families and 16.4% of the individuals in rural Shoshone County lived below the poverty level during 1999. These values were higher than the statewide values of 8.3% and 11.8%, respectively. With the lack of a viable economic base, there has been a gradual out-migration of people from Shoshone County; due to limited turnover of the population, the county's age and racial profiles do not generally reflect those of the state as a whole. For example, the median age for Idaho was 33 years in 2000, but in the mining communities of the river basin, it was over 40 years. Racially, the county's population of 13,771 was predominantly white (96% white versus 93% for Idaho), with small American Indian (1.5%) and Hispanic populations (1.9%) versus 2.1% and 7.9%, respectively, statewide. The total population of the river basin areas addressed in the HHRA was 10,496 based on 1990 census data (TerraGraphics et al. 2001, Table 3-4). Children aged 0 to 4 years—a population cohort that is particularly susceptible to lead toxicity—made up 5.6% of the population (587 children).[1]

CHEMICALS OF CONCERN IN THE COEUR D'ALENE RIVER BASIN: HAZARD IDENTIFICATION

The database of environmental chemical analyses available for the HHRA process was extensive and included thousands of analyses of metals in soil, house dust, groundwater, homegrown vegetables, sediment, surface water, fish, and edible wild plants (water potatoes) in the river basin. Typically, for each sample, the precise geographic location and concentrations of up to 23 metals and other inorganic materials were ascertained. For example, 4,000 soil and sediment samples were collected within the study area and analyzed for 23 inorganic compounds. Yard soils from 1,020 homes throughout the river basin were analyzed for lead, corresponding to roughly one-quarter of the yards present in the river basin in the 1990 census. Soils from 191 residential yards were analyzed for 23 inorganic compounds. Before chemical analysis, all soil samples were sieved to obtain soil particles less than 175 micrometer (μm) in diameter. Pre-sieving is justified by the observation that fine particles preferentially adhere to hands (Duggan et al. 1985; Duggan and Inskip 1985; Sheppard and Evenden 1994; Kissel et al. 1996) and the assumption that they are therefore more likely to be ingested. Dust mats were placed and collected from 500 river basin homes, and vacuum cleaner bags were collected from 320 of those homes. Measurements of these samples allowed for estimates of both lead concentration and dust loading rates. Tap water from 100 homes

[1]The HHRA compiled population estimates from 1990 census tracts that were within or partially within the HHRA study area.

was analyzed for 23 inorganic compounds, and 425 homes had water lead analyzed. Eighty samples of water from 27 monitoring wells near Ninemile and Canyon Creeks were analyzed for 23 inorganic compounds. X-ray fluorescence measurements of lead concentrations on interior and exterior surfaces were performed in 415 homes. While this tabulation could go on, the point is that a substantial environmental database was available to the risk assessors as they sought to quantify chemicals of concern from a variety of media in the Coeur d'Alene River basin environment that might pose a risk to human health. Because of the large geographic area of the river basin, additional studies of specific areas will be required as remediation proceeds.

Not all substances present at various test sites pose a human health risk. For example, some of the numerous metals present in environmental samples from the river basin are essential nutrients, including zinc, calcium, iron, magnesium, potassium, and sodium. Yet even these, in excess, can pose health risks. Thus, EPA has developed guidelines for selecting a group of chemicals of potential concern (COPCs) based on their toxicity, concentration, and other factors (EPA 1989). Typically, applicable or relevant and appropriate requirements (ARARs) are used to compare the observed concentration of a substance in an environmental sample with some screening value, threshold, or legally defined concentration in that environmental medium. For example, the ARARs for drinking water at this site are actually the EPA maximum contaminant levels (MCLs)—concentrations of substances in drinking water above which unacceptable health risks to the public may occur. The ARARs for surface water are the MCLs as well as the ambient water-quality criteria (AWQC). The latter, used for controlling releases or discharges of pollutants, are protective of those who drink surface water, those who eat fish caught in surface water, and aquatic organisms. The only ARAR for substances in air that is relevant at this site is that for lead—the National Ambient Air Quality Criterion for lead. There are no ARARs at this site for substances in soil or sediments.

The river basin HHRA considered which COPCs might pose a human health risk for each medium of possible exposure: soil/sediment, tap water, surface water, groundwater, house dust, air, fish consumption, and homegrown vegetables. The process used was very typical of any HHRA at sites where chemical exposures might occur. In addition, it considered possible risks due to the ingestion of water potatoes, a culturally important food source for the Coeur d'Alene tribe. Because a "screening value" for substances in water potatoes is not known, cadmium and lead were evaluated as substances with possible risk, a decision consistent with the evaluation of other food substances.

As a result of these hazard-identification activities, selected metals were chosen for further evaluation of human exposure, and a list of possible sources of exposure was created for each (Table 5-1). The metals were antimony, arsenic, cadmium, iron, lead, manganese, mercury, and zinc.

In summary, the HHRA appropriately identified COPCs for each possible source of exposure. However, no effort was made to identify the particular chemical species of lead or arsenic (or other metal) in any of these sources. The absence of chemical speciation is less than ideal because the bioavailability and toxicity of particular chemical species of the same metal can vary substantially.

APPROACH USED TO ASSESS HUMAN HAZARDS: EXPOSURE ASSESSMENT

After identifying which chemicals might pose hazards to human health, the HHRA set out to characterize human exposure. Because the concentrations of metals in various media and exposure profiles in the river basin are not uniform, EPA considered it necessary to divide the region of interest into nine distinct geographical areas: lower basin, Kingston, side gulches, Osburn, Silverton, Wallace, Ninemile, Mullan, and Blackwell Island (TerraGraphics et al. 2001, Fig. 3-1a). For each of these regions, diagrams were created to conceptualize possible pathways of exposures to metals that might occur under several scenarios—for example, during residence in the home, neighborhood recreation, public recreation, occupation, and subsistence living. An example of this approach, for Silverton, Idaho, taken directly from the HHRA, is provided as Figure 5-1 (TerraGraphics et al. 2001). This portion of the HHRA was basically a paper exercise, but one that is based on a rather extensive literature that has documented that such pathways of exposure have resulted in significant chemical exposures in other circumstances. Thus, this approach represents an acceptable technique for eventually estimating potential current and future exposures.

TABLE 5-1 Possible Exposure Sources of Chemicals of Potential Concern

Possible Exposure Source	Chemicals of Potential Concern
Soil/sediment	Antimony, arsenic, cadmium, iron, lead, manganese, and zinc
Tap water	Arsenic and lead
Surface water	Arsenic, cadmium, lead, manganese, and mercury
Groundwater	Antimony, arsenic, cadmium, lead, and zinc
House dust	Antimony, arsenic, cadmium, iron, lead, manganese, and zinc
Fish	Cadmium, lead, and mercury
Homegrown vegetables	Arsenic, cadmium, and lead

SOURCE: TerraGraphics et al. 2001, Table 2-12.

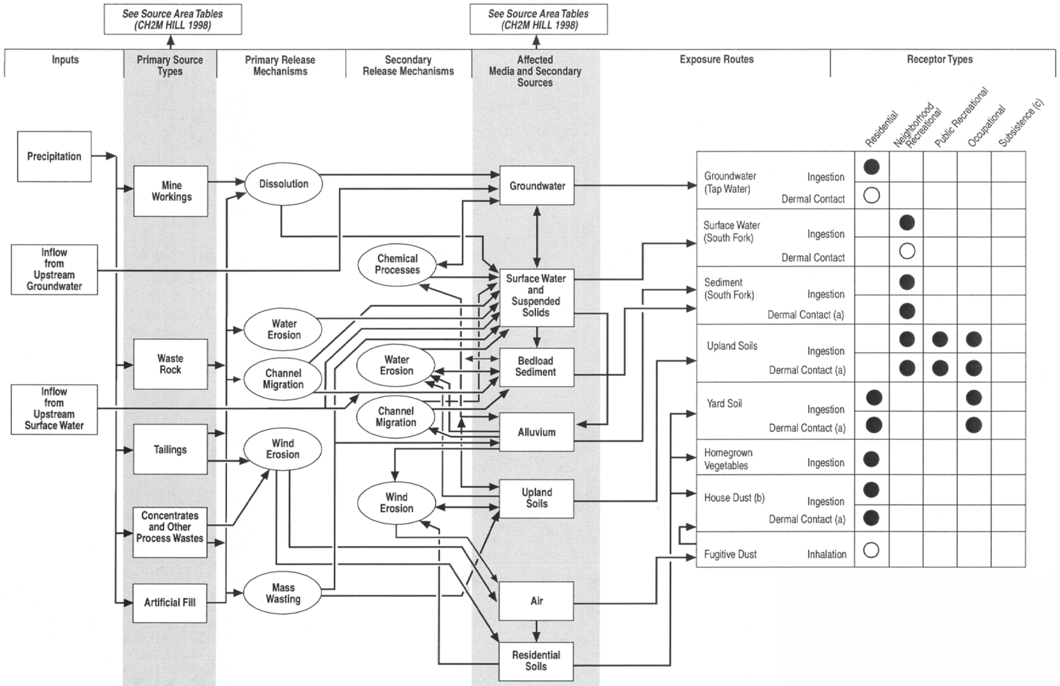

FIGURE 5-1 Example of conceptual site model diagram from HHRA. ●, Complete pathway; evaluated quantitatively in the HHRA. ○ , Pathhway potentially complete, but of minor concern; not qualified in the HHRA. A blank cell indicates that the pathway is not complete or the receptor type does not exist in the this area. (a) Quantified for arsenic and cadmium only. See text for discussion. (b) This pathway evaluated quantitatively for nonlead COPCs and quantitatively for lead. (c) Both traditional and modern subsistence tribal member exposure scenarios will be evaluated at a later time.

Ultimately, to estimate possible risks of adverse health outcomes, it is necessary to estimate the metal concentration in each environmental medium to which an individual may be exposed. EPA guidelines (EPA 1991a, 1992a) state that this concentration term (exposure point concentration [EPC]) should represent the average concentration to which one is exposed for the relevant portion of one's lifetime. Because of the obvious uncertainty in estimating the true average concentration from measurements of samples, EPA recommends using the 95% upper confidence limit (UCL95) of the mean as a conservative estimate of the EPC, because this is associated with only a 5% probability of underestimating the true average (EPA 1991a, 1992b, 1993a). In addition to the concentrations in each environmental medium, it is necessary to estimate the pathway-specific intakes from that medium to ultimately estimate exposures. In the Coeur d'Alene River basin HHRA, intakes were estimated in two ways, consistent with EPA guidelines for risk characterization (EPA 1995). A CT exposure estimate is considered to be representative of average human exposures, whereas a higher value, the RME, illustrates a high-exposure scenario that is nevertheless likely to occur.

For each of the nine geographic regions, the Coeur d'Alene River basin HHRA used this approach to estimate point concentrations and intakes of surface soil, vacuum bag dust, floor mat dust, tap water, groundwater, subsurface soil, waste piles, and sediments. A total of 49 data sets were analyzed rather than 72 (nine regions × eight sources) because not every region had potential exposure from each of these sources. In 38 of 49 cases, at least 10 measured values were available to make this estimate, and in many cases, hundreds of measurements were used, thus providing stable estimates of the true average concentration. In the remaining 11 cases, fewer than 10 measurements were available; in these cases, the maximum value was used in place of the UCL95. Because the formula used to appropriately calculate UCL95s depends on the distribution of the data, the HHRA first examined the shape of the distributions before carrying out these calculations.

Regional estimates of chemical *intakes* were subsequently made for soil, sediment, drinking water, surface water, homegrown vegetables, and fish. The exposure models utilized were straightforward and took into account a variety of behavioral and physiological factors, including exposure frequency and duration, contact rate, EPC, body weight, and averaging time. An example of one of these models, derived from the HHRA, which estimated exposure via the consumption of groundwater as a drinking source, is shown below:

$$\text{Chemical intake (mg/kg/day)} = \text{Cw} \times \text{SIFw} \times \text{CF} \quad (1)$$

and

$$SIFw = IRw \times EF \times ED/(BW \times AT), \tag{2}$$

where

Cw = chemical concentration in groundwater/tap water (µg/L);
SIFw = summary intake factor for ingestion of tap water (L/kg/day);
IRw = ingestion rate for tap water (L/day);
EF = exposure frequency (days/year);
ED = exposure duration (years);
CF = conversion factor (mg/µg);
BW = body weight (kg); and
AT = averaging time (days).

The intake parameters used to solve such equations (in this case, IRw, EF, ED, BW, and AT) for children and adults were obtained from previous EPA guidance for such calculations (EPA 1989, 1991a, 1993a). In the example presented, the intake parameters are known with a relatively high degree of certainty (for example, ingestion rate for tap water). In other equations, such as those related to exposure from homegrown vegetables or dermal exposure to surface water, intake parameters are less certain (for example, vegetable ingestion rates, and gastrointestinal and dermal absorption factors) but represent conservative estimates of the weight of current scientific evidence.

HUMAN HEALTH: TOXICITY ASSESSMENT

After identifying the chemical hazards and estimating the human exposures to each, the next step in an HHRA involves evaluating the scientific evidence from animal and human epidemiologic studies that have examined dose-response relationships for cancer and noncancer health outcomes. The fundamental tenet of toxicology is that the dose determines the effect.

For Carcinogens (Arsenic)

For cancer outcomes, the dose-response information is condensed into an SF, in units of $(\text{mg/kg-day})^{-1}$, which expresses excess cancer risk as a function of (lifetime average) daily dose. EPA maintains an online database, the Integrated Risk Information System (IRIS) (EPA 2004a), which contains SFs that are based on the current weight of toxicologic evidence. Of the metals identified as potential hazards in the river basin, only arsenic was evaluated for carcinogenic risk.[2,3] Arsenic's SF—unchanged since the early 1990s—is based largely on data from international epidemiologic studies that have been reviewed in previous National Research Council (NRC) reports (NRC 1999, 2001). Several U.S.-based studies have failed to find an association between arsenic in drinking water and cancer risk in non-smokers (Bates et al. 1995; Lewis et al. 1999; Karagas et al. 2001; Steinmaus et al. 2003), possibly suggesting that the SF may overstate the risks at low doses. In this regard, however, a recent study of arsenic and bladder cancer in New Hampshire that examined individual arsenic exposures using toenail arsenic as a biomarker of exposure found that low-level arsenic exposure was associated with a doubling of the risk for bladder cancer (Karagas et al. 2004). At the present time, a great deal of research concerning arsenic and cancer is ongoing, much of it supported by the Superfund Basic Research Program, and it seems possible that the SF may need to be reexamined in the future as a result of past and ongoing work.

For Noncarcinogens Other Than Lead

For noncancer outcomes, a chronic reference dose (RfD) is derived from the no-observed-adverse-effect level (NOAEL) or lowest-observed-adverse-effect level (LOAEL) in animals or humans.[4] RfDs are derived by dividing the NOAEL or LOAEL by an uncertainty factor that represents a combination of various sources of uncertainty associated with the database for that particular chemical. Once again, EPA's IRIS database served as a source of RfDs for the chemicals of concern in the basin, except for lead (discussed below) and iron, for which there is no IRIS RfD and for which other sources of toxicity data were used. Note that arsenic also has noncancer effects and its own IRIS RfD.

[2]EPA's HHRA for lead did not include cancer as a possible health outcome. In a recent report from the National Toxicology Program (NTP), lead and lead compounds were listed as "reasonably anticipated to be human carcinogens" (NTP 2005). The committee did not further consider the potential carcinogenicity of lead in its review of EPA's HHRA.

[3]EPA's HHRA for cadmium did not include cancer as a possible health outcome. The Ninth Report on Carcinogens (NTP 2000) listed cadmium and cadmium compounds as known human carcinogens. The HHRA, released in June 2001, states that arsenic was the only established human carcinogen and that there are no cancer SFs to conduct a quantitative evaluation of cancer risk for other metals. EPA's IRIS database does not provide a quantitative estimate of carcinogenic risk from oral exposure for cadmium and states, "There are no positive studies of orally ingested cadmium suitable for quantitation" (EPA 2004a). Further, the committee noted ATSDR's Environmental Health Assessment in the Coeur d'Alene River basin (ATSDR 2000), which reported urine cadmium analyses for 752 Coeur d'Alene River basin residents and that stated, "In contrast to the results for lead, no link between soil or dust exposures and elevated urine cadmium was found in the study population. Rather, elevated cadmium in this population appears to be related to smoking behaviors."

[4]More recently, a benchmark dose (BMD) for an appropriate end point may also be used as the starting point, rather than LOAELs or NOAELs.

For Lead

Of all the metals of potential concern, the adverse health effects of lead are best characterized in human populations. Risk assessments for lead therefore differ from those for other noncarcinogens in that they rely on observed or predicted blood lead levels (BLLs) because blood lead concentrations have been directly related to adverse outcomes in adults and children. In studies conducted around the world, population average blood lead concentrations have been found to be associated with adverse effects on average measures of cognitive and behavioral development in young children. In short, dose-response relationships between blood lead and adverse health outcomes in children are sufficiently well described that community BLLs can be used to estimate risk. Community BLLs can be determined precisely through appropriately designed surveys, or they can be estimated from environmental data through modeling techniques. The estimation of BLLs through modeling, which involves environmental rather than biological measurements, is considered in Chapter 6.

RISK CHARACTERIZATION

Risk characterization, the last step in an HHRA, strives to combine the estimates of chemical exposure with the estimates of potential human hazard (based on known dose-response relationships) to estimate the actual or potential risks to human health at the site. At the Coeur d'Alene River basin site, EPA estimated cancer and noncancer health risks for both CT and RME conditions. As mentioned above, the CT estimate represents an average level of chemical exposure, while the RME is a more conservative estimate intended to be the highest exposure that can reasonably be expected to occur. Risks were estimated separately for different segments of the population, such as children, adults, and those with occupational exposure.

For Carcinogens

The probability of developing cancer due to arsenic exposure, the only carcinogen assessed, was estimated by a standard approach that involved multiplying the arsenic SF by the estimated arsenic daily intake.

$$\text{Cancer risk} = \text{chemical intake (mg/kg-day)} \times \text{SF (mg/kg-day)}^{-1}.$$

EPA's target "acceptable" excess cancer risk is between 10^{-6} and 10^{-4} in a lifetime (EPA 1991b). In the HHRA, the method for estimating cancer risk due to estimated arsenic exposure involved multiplying estimated arsenic intakes (in different age groups within different geographic regions) by the arsenic SF. Under RME conditions, cancer risks exceeded 10^{-6} for each scenario in each of the nine geographic regions. Under RME conditions, residents of the side gulches had cancer risk estimates exceeding 10^{-4}. Under CT conditions, several of the regions also had cancer risk estimates greater than 10^{-6}. Collectively, these findings indicate that arsenic in the side gulches must be dealt with by risk managers. The analysis in the HHRA indicated that exposure to yard soils was the primary driver of arsenic cancer risk in residential scenarios, and that, in the side gulches, tap water also contributed significantly to cancer risk. It should also be noted that cancer risk for the 90th percentile background soil level of 22 mg/kg arsenic in the upper basin is associated with an estimated cancer risk greater than 10^{-6} using the risk assessment methodology employed in the basin.[5]

Modern tribal subsistence scenarios yielded cancer risk estimates similar to those for the highest nontribal residential exposures, but traditional subsistence scenarios had risks roughly 10 times higher. During visits to the river basin, the committee learned from tribal leaders that tribal members no longer practice subsistence living in the basin (CDA Resolution 42 [2001]). Nevertheless, risk managers need to address the tribe's concerns should their members engage in subsistence activities.

For Noncarcinogens Other Than Lead

Methods used for characterizing risks differ for carcinogens and noncarcinogens. For noncarcinogens other than lead, a hazard quotient (HQ) is derived by dividing the estimated total daily exposure to a chemical by the RfD. If the average daily intake exceeds the RfD (if the HQ is greater than 1), there is a potential for risk for an adverse noncancer health outcome:

$$\mathrm{HQ} = \frac{\text{chemical intake (mg/kg-day)}}{\text{RfD (mg/kg-day)}}\,.$$

The river basin HHRA estimated HQs separately for children and adults; in general, children were found to have higher HQs because they are likely to ingest more soil/dust relative to their body weight. For CT exposures to nontribal residents, the only potentially unacceptable hazards would occur if future residents of the Burke/Nine Mile area were to use groundwater as a source for drinking water. In general, however, soil rather than drinking water contributed most to the HQs. Several other estimated HQs exceeded 1

[5]Tribal exposure scenarios would have an even greater calculated cancer risk at reported background concentrations using the methodology employed in the HHRA.

and indicated possible hazards from the following sources: cadmium from homegrown vegetables and/or water potatoes, iron from soil/sediment ingestion in the lower basin, hypothetical exposure to cadmium and zinc from consumption of groundwater in the Burke/Nine Mile area, and mercury exposure from fish for the traditional subsistence scenario. Although the possible health risks associated with these scenarios should not be ignored, the committee believes that the primary area of focus for risk managers does not lie with these metals. Clearly, other than lead, arsenic is the chemical of potential concern that was consistently a risk driver for all non-lead risk assessment scenarios, with the major source being soil.

Risk assessment of non-lead COPCs appeared generally to follow EPA guidelines. Residential soil EPCs in the basin sub-areas were computed by lumping data from multiple residences—rather than on a residence-specific basis, which is probably more common. The fraction of ingested soil that a child typically obtains from areas other than his or her own yard is essentially unknown. The consequences of using area-wide rather than residence-specific EPC values will depend upon within-residence and across-residence variance in soil concentration. The committee did not have residence-specific soil arsenic data (the soil contaminant of greatest concern in this context) and did not investigate this question.

For Lead

As mentioned above, risk assessments for lead rely on observed or predicted BLLs in a community, as blood lead concentrations have been directly related to adverse outcomes in adults and children. In 1991, the U.S. Centers for Disease Control and Prevention (CDC) promulgated specific guidelines aimed at reducing BLLs in individual children (CDC 1991). These are summarized in Table 5-2.

Because vast quantities of lead have been distributed throughout the river basin due to historical mining-related activities, the HHRA devoted substantial effort to characterizing the risks of lead toxicity to the basin communities, and to children in particular. At sites like this one, EPA policies seek to protect the health of the most vulnerable populations, namely children and women of childbearing age. EPA policy (EPA 1994) strives to reduce soil lead levels so that no child would have more than a 5% chance of exceeding a BLL of 10 micrograms per deciliter (μg/dL). EPA has promoted use of the integrated exposure uptake biokinetic (IEUBK) model for estimating risks to children from lead exposure from soil and other media. The charge to this committee included several questions specifically directed at the IEUBK model. Thus, Chapter 6 is devoted to use of the IEUBK model to understand lead exposure and uptake. The use of the model in this HHRA has projected significant risks of lead toxicity throughout the Coeur d'Alene River basin.

TABLE 5-2 CDC Guidelines for Reducing Blood Lead in Children

Blood lead (µg/dL)	Action
<10	Reassess or rescreen in 1 year
10-14	Family education; follow-up testing; social services if warranted
15-19	Family education; follow-up testing; social services if warranted; if blood lead persists or rises within 3 months, proceed as below for blood lead concentrations of 20-44 µg/dL
20-44	Provide clinical management, environmental investigation, and lead hazard control
45-69	Immediately begin coordination of care, clinical management, environmental investigation, and lead hazard control
≥70	Hospitalize and treat immediately with chelating agents; environmental investigation and lead hazard control immediately

SOURCE: CDC 1991.

PLAUSIBLE HEALTH RISKS FROM LIVING IN THE COEUR D'ALENE RIVER BASIN

If we assume that the Coeur d'Alene River basin HHRA is correct and that without significant remedial actions, the populations of the basin are at risk from arsenic and lead exposures, what human health effects might be expected? What are the consequences of arsenic and lead exposure, and how strong is the evidence of toxicity? In addition to the actual risks due to exposure to chemicals, what are the psychosocial consequences of living in proximity to or in the midst of large amounts of potentially toxic materials? Moreover, how might the conclusions of the basin HHRA have been strengthened? In this section, we briefly explore these issues.

Risks from Arsenic

Ingestion of inorganic arsenic is an established cause of skin, bladder, and lung cancer (NRC 1999). Many noncancer health outcomes are also associated with arsenic exposure, including effects on the skin, cardiovascular, nervous, endocrine, hematologic, and renal systems. The primary toxicity from arsenic is oxidative toxicity to cells. A shortcoming of the HHRA is that no human exposure data were collected. Urine and/or hair arsenic levels are commonly used to quantify chronic arsenic exposure and could have been collected. The risks from arsenic in the basin were mainly determined by modeling human exposures based on arsenic concentrations in environmental samples. Although risk determinations using such modeling are appropriate in the absence of human data, a coupling with actual biological measurements would have strengthened the HHRA. Like lead, there are concerns that some forms of arsenic may not be bioavailable (Caussy 2003; Rodriguez et al. 2003; Turpeinen et al. 2003). The relatively small population size of the basin would make epidemiologic investigation of cancer risk impossible; cancer end points such as skin and bladder cancer are too infrequent to determine increased prevalence in such a small sample.

Risks from Lead

Toxic exposures to lead during early childhood and even fetal life can lead to permanent neurologic deficits. Communities near lead industries frequently have increased exposure. A full review of the epidemiologic evidence for the developmental toxicity of lead is beyond the scope of this report, but the developmental toxicity of lead is clear. Numerous studies have reported inverse associations between infants' scores on tests of neurobehavioral development and indices of fetal lead exposure such as umbilical cord blood lead concentration (Bellinger et al. 1987; Wasserman et al. 1994) or maternal blood lead during pregnancy (Dietrich et al. 1987). In some studies, associations between prenatal lead exposure and children's neurobehavioral outcomes ultimately decrease with time, although associations tend to emerge between postnatal exposures and later childhood (Bellinger et al. 1992). Canfield et al. (2003) recently reported that the inverse association between BLL and IQ at age 7 is apparent among children whose BLLs never exceeded 10 µg/day. This finding is consistent with Schwartz's (Schwartz 1994) nonparametric smoothing analyses of the 10-year follow-up data of the Boston study and with a report on cognitive effects associated with BLLs <10 µg/dL (Lanphear et al. 2000). Recent studies also suggest associations with important forms of psychosocial morbidity (Bellinger et al. 1994; Needleman et al. 1996; Wasserman et al. 1998), including juvenile delinquency (Needleman et al. 2002).

For decades, the impact of environmental lead exposure on children has been a central focus of the field of environmental health. However, there is a growing body of more recent evidence that environmental lead exposure is also associated with an important set of adverse health effects in adults. For example, bone lead levels that were related to lead in drinking water in Boston (Potula et al. 1999) were associated with the development of hypertension among participants in the Normative Aging Study (Cheng et al. 2001). In the same cohort, elevated blood and bone lead levels inversely predicted performance on the Mini-Mental Status Exam (Wright et al. 2003). Environmental lead exposure has also been linked to elevated blood pressure and proteinuria among pregnant women (Factor-Litvak 1992). Lead exposure in women of childbearing age is a hugely important issue because lead is known to freely pass the placenta to the unborn child (Graziano et al. 1990). Furthermore, there is evidence that calcium supplement, a simple and cost-effective intervention, will reduce the resorption of lead from bone to blood during pregnancy and limit fetal lead exposure (Janakiraman et al. 2003). Recent studies have also identified environmental lead exposure as a risk factor for essential tremor, one of the most common neurological diseases (Louis et al. 2003; Louis in press). Thus, while the focus of remedial activities has nearly always been due to potential risks to children, the adult population is also vulnerable to significant lead-related morbidity.

Risks from Psychosocial Stress

At the town hall meetings that occurred during the committee's two visits to the region, some residents, but certainly not all, expressed fears and concerns about possible exposures to hazardous substances. Nothing in the Superfund law (CERCLA) requires EPA to consider community stress from designation of a region as a Superfund site. Nevertheless, there is substantial evidence concerning the psychosocial consequences of living in proximity to hazardous materials at Superfund and other sites, including Love Canal, New York, Three Mile Island, Pennsylvania, and the Exxon-Valdez disaster in Alaska. Furthermore, an Agency for Toxic Substance and Disease Registry (ATSDR) expert panel report (Tucker 2002) recommended both additional research on the effects of psychosocial stress in communities impacted by toxic waste and the development of public health intervention strategies to mitigate such stress. These goals clearly have not been achieved, as the literature on the health effects of stress in Superfund communities is sparse, and no such interventions have been developed.

Exposure to toxic chemicals generally is perceived to involve "invisible" contaminants not detectable by the senses. For this reason, the presence of a toxic waste site may induce chronic stress independent of actual chemical exposure. Living near a toxic waste site is associated with health effects that can be slow in onset and insidious in nature. Often, little technical information is available to families about the likelihood of exposure and effects, leaving them uncertain about their actual risk. Helplessness and fear of the unknown are also common complaints in such communities (Kroll-Smith and Couch 1990). People who believe they have been exposed to toxic chemicals tend to develop chronic stress (Fleming et al. 1982), with symptoms including depression, a feeling of lack of control of the environment, increased family quarrels, increased health worries, and increased intrusive and avoidant thoughts (Stone and Levine 1985; Davidson et al. 1986; Gibbs 1986; Levine and Stone 1986; Edelstein 1988; Stefanko and Horowitz 1989). Trust in both government agencies and scientific experts erodes when communities perceive a failure to adequately respond to toxic contamination (Kroll-Smith and Couch 1990). Children of parents who report chronic stress from the uncertainty of toxic exposures also tend to report increased stress (Edelstein 1988). As a moderating factor, social support can help families cope with stressful events (Figley 1986; Unger et al. 1992). The existence of increased social supports predicted a reduction in symptomatology among subjects living proximal to Three Mile Island (Bromet and Dunn 1981). Unfortunately, social supports can also be eroded by residence near a toxic waste site. Members of a social network may blame the family for moving to the area. Residents may become stigmatized, even ridiculed, further isolating them and increasing their chronic stress (Edelstein 1988).

Such chronic stress from potentially hazardous sites can have multiple adverse health effects. Increased risks of heart disease, hypertension, infection, asthma, premature delivery, and diabetes have been associated with chronic elevated stress. A particular effect of stress that may be relevant to populations with elevated lead exposure is the role of chronic stress in neurodevelopment. Psychological stress results in activation of the hypothalamic-pituitary-adrenal axis. The traditional view is that the hypothalamus produces corticotropin-releasing hormone, which leads to downstream activation of the adrenal cortex to secrete corticosteroids (for example, cortisol) into the blood, which then enter the brain (Sapolsky 2000; McEwen 2001). The hippocampus is the brain region with the highest density of glucocorticoid receptors (Sousa and Almeida 2002). These receptors modulate neurologic development. The primary functional end point of chronic stress appears to be changes in the development and formation of memory. Whereas acute stress may enhance memory formation, chronic stress appears to inhibit it. Animal behavioral studies have confirmed the adverse independent effects of both prenatal and postnatal chronic stress on memory and learning (Zaharia et al. 1996; Vallee et al. 1999; Aleksandrov et al. 2001; Frisone et al. 2002). Research on children exposed to political or domestic violence suggests that a number of the domains of cognitive, social, and emotional function are adversely affected by exposure to such stressors (Golier and Yehuda 1998). With respect to "lower doses" of chronic stress, maternal anxiety both during pregnancy and postnatally, have been independently associated with a 1.5- to 2-fold increase in risk for behavioral/emotional problems in children at 4 years of age (O'Connor et al. 2002a,b).

The social stress associated with potentially hazardous sites may have adverse health effects independent of chemical exposure. As previously outlined, the development of the brain is likely affected by hormonal signals which modify neuronal-genesis and synaptic formation and synaptic pruning (LeDoux 2002). Environmental factors can promote or disrupt this process depending on whether they are positive (social supports, good nutrition) or negative (toxicants, malnutrition, trauma) (Nelson and Carver 1998). Animal research suggests that the social environment will modify the toxicity of lead and the combined effects of lead and social isolation may augment toxicity (Schneider et al. 2001; Guilarte et al. 2003; Cory-Slechta et al. 2004). In humans, poverty, psychological stress, and lead exposure are likely correlated, but the nature of the relationship (independence [additive toxicity], covariance [confounding], or synergy [effect modification]) in predicting health outcomes has not been determined. Clearly, this is an area of great research need, especially at Superfund sites.

Risks Unique to the Coeur d'Alene Tribe

Most hazardous waste sites on American Indian lands have never been evaluated for their impact on the cultural resources and practices of the tribes who inhabit them (Osedowski 2001; Harper et al. 2002). Furthermore, many American Indian lands border contaminated lands not designated as Superfund sites. These sites represent potentially important sources of plants and wildlife used in traditional diets and may be contaminated with toxic materials. With information on the real risks of contamination in their traditional lifestyles, tribes will be empowered to make decisions based on this information and can educate tribal members about uses of exposed resources and continue their traditional lifestyle without compromising their cultural identity or health (Harris and Harper 1997).

American Indian tribal members may choose to follow traditional lifestyles despite knowing that there are risks posed by environmental contamination. Maintaining a homeland where present and future generations may live in a clean, functioning ecosystem is a goal that often has not been respected by agencies and researchers who study the impact of environmental contamination on native lands. There is also substantial evidence that traditional (noncontaminated) subsistence diets among American Indians are inherently healthier than Western diets and reduce the risk of diabetes and heart disease (McDermott 1998; Lev-Ran 2001). Switching from a traditional lifestyle to a suburban American lifestyle carries significant health risks, emphasizing the importance of providing a clean environment to support traditional lifestyles. American Indian reservations are intended to provide permanent homelands for their members. When these lands are contaminated with industrial waste, environmental justice mandates that exposure assessments appreciate the value of traditional lifestyles.

Exposure scenarios designed for American "suburban lifestyles" have been reported to be unsuitable for tribal communities (Harris and Harper 1997). Harris and Harper described an approach to determining exposure assessment in subjects with a subsistence diet that included qualitative interviews and expert elicitation to determine foods consumed and practices common among tribal members (Harris and Harper 1997, 2001; Harper et al. 2002). Subsistence in this context refers not only to diet but also to cultural and religious practices, which may include medicinal and ceremonial uses of natural resources. The goal is not to increase precision regarding a single pathway of exposure (such as diet) but to increase overall understanding and community awareness about multiple pathways of exposure and the role of culture-based behaviors. All these factors may predispose American Indians to exposure and may make them a vulnerable subpopulation within a Superfund site.

New methodologies are being developed to assess exposure in tribal lands. For example, through the assistance of the tribal governments, expert elicitation of local traditional lifestyle practitioners and tribal elders can assist with environmental sampling strategy. Expert elicitation is a technique used in decision analysis to derive numeric data through interviews with acknowledged experts (Meyer and Booker 1991; Hora 1992). This technique has been used successfully in other studies of American Indian exposure scenarios (Harris and Harper 1997, 2001; Harper et al. 2002). Tribal experts can compare survey results with their knowledge of hunting and gathering practices of their tribal members. Sample locations of plants and animals identified as culturally important could be based on this process.

The Coeur d'Alene River basin HHRA acknowledged that American Indians likely have higher risks than non-American Indians living in the basin. As presented in the HHRA, "it is clear that a subsistence-based lifestyle requires environmental lead levels orders of magnitude lower than those measured throughout the floodplain of the Coeur d'Alene River" (TerraGraphics et al. 2001, p. 6-2). Further, the HHRA concludes, "Estimated lead intake rates for these scenarios are too high to predict BLLs with confidence. Predictions for BLLs associated with subsistence activities . . . would significantly exceed all health criteria for children or adults" (TerraGraphics et al. 2001, p. 6-51). Given the magnitude and extent of contamination, it is difficult to envision how the tribes could reduce exposure risks to an acceptable level if a return to subsistence lifestyle were to occur.

BLOOD LEAD STUDIES IN THE COEUR D'ALENE RIVER BASIN

The Coeur d'Alene River basin HHRA included some survey data of blood lead concentrations in children, but these were sufficiently limited that the document essentially relies on the IEUBK model to predict risks from lead exposure. The limitations of the blood lead data have their origins in an agreement between community leaders, the state of Idaho, and EPA, which affirmed that no studies would be conducted for "scientific research or academic" reasons (von Lindern 2004). Basically, blood lead screening programs do not work well when the community is not cooperative. How could the HHRA have been strengthened in this regard?

Ideal Blood Lead Screening Methodology

An ideal screening program would include all at-risk children in a highly lead-exposed geographic area. This program would not be limited to a single cross-sectional measurement but would include longitudinal measurements and an intervention program that is triggered at predetermined BLLs. Widespread participation would ensure not only that most children with high lead exposure are identified and treated but also would allow for epidemiologic assessment of exposure risks for specific sites within the geographic region. Ideal lead screening programs identify specific housing associated with lead exposure—information then used by the state or federal government to direct remediation efforts.

However, the American Academy of Pediatrics no long endorses universal screening for lead poisoning but instead recommends targeted screening in high-risk populations. Today, only 53% of pediatricians in the United States screen blood lead in all their patients before the age of 3 (AAP 1995), but this percentage is much higher in regions where lead hazards are thought to exist. The distinction between a high-risk population and a high-risk individual merits discussion. Questionnaires and risk factors for lead exposure have poor sensitivity and specificity in detecting individual children with elevated BLLs, in part because lead-exposure pathways include home dust, soil, water, and other more unique sources (for example, ceramic pottery). For that reason, the unit of measure for a lead screening program is a high-risk population and not a high-risk individual. The history of the Coeur d'Alene River basin certainly warrants evaluation of its residents as a high-risk population.

An ideal lead intervention program in the Coeur d'Alene River basin would include both primary and secondary prevention strategies for exposure reduction. Observational research has noted associations of lead poisoning with poor nutrition (iron and calcium intake in particular), elevated lead levels in home dust, and elevated lead levels in soil, making nutritional and environmental interventions logical starting points for tempering exposure to lead. As part of primary prevention, nutritional and behavioral risk reduction counseling would be offered to all families with children less than 5 years of age. Secondary prevention would consist of specific exposure-reduction interventions tailored to a specific child with elevated BLLs (>10 μg/dL). This may include home visits to develop and convey strategies for exposure reduction specific for that child's home environment. Home inspections for lead paint and soil lead assessments would seek to determine the source(s) of the lead exposure, assisting families in directing their exposure reduction efforts at the source for lead exposure and establishing that the exposure source is indeed the home and not a daycare center, relative's home, or other site where the child spends a significant amount of time.

However, it should be noted that interventions short of actual remediation of lead sources have not been found to reduce the prevalence of childhood lead poisoning in previous studies. Therefore, these counseling efforts should be adjuncts to remediation efforts in which the lead hazard is removed from the child's environment. Secondary prevention, which relies on identifying lead-poisoned children is important but should not be the primary focus of public health intervention. Given the lack of effective treatments for lead toxicity, primary prevention strategies are more likely to have a positive public health impact.

Screening Methods Used in the Coeur d'Alene River Basin

Participation is the key to any health screening program. On a national level, state health departments have used several strategies to maximize participation in childhood lead screening programs in the United States. Some states have instituted mandatory annual screening programs for children between the ages of 1 and 4 years. The Women Infants and Children supplemental nutritional program in many states requires that a hemoglobin and BLL be measured before families can participate. Before leaded gasoline was phased out, when high exposures to lead were more widespread, universal screening of all children aged 1-4 years was recommended. However, lead exposure in the general population has been greatly reduced, and more cost-efficient strategies are now appropriate.

Sampling the Coeur d'Alene River Basin Population for Lead Exposure

Data on the prevalence of elevated and mean blood lead concentrations in the Coeur d'Alene River basin between 1996 and 2004 consist primarily of screening conducted at a fixed site for a brief time in the summer months. Screening is not mandatory in Idaho, and there is no evidence that physicians widely screen children in the Coeur d'Alene River basin.[6] Therefore, these are the only data available with which to assess the prevalence of lead poisoning and to test the assumptions of the IEUBK model (see Chapter 6). With respect to the validity of the annual blood lead screening data as an accurate characterization of the population distribution of blood lead, only the 1996 data are from an attempt at population-based sampling. The results

[6]In 2003, the U.S. District Court concluded that "The State of Idaho is violating mandatory Medicaid provisions which require it to: (a) ensure that Medicaid eligible children receive medical screening that includes lead screening, lead blood testing of young children, and health education and anticipatory guidance regarding lead poisoning and lead poisoning prevention;" . . . (U.S. District Court for the District of Idaho; Consent Decree and Judgment; Case No. CIV 00-578-S-MHW; January 14, 2003; pp 13-14.)

of this assessment have been criticized as biased because the overall participation rate was only 25%. Because this study was the only recent attempt at generating representative population-based blood lead screening data, we focus our discussion on the methods used in this study.

A Coeur d'Alene River basin Environmental Health Assessment was conducted before the HHRA by the IDHW with ATSDR funding (ATSDR 2000). State health statistics did not provide a precise count of children living in the Coeur d'Alene River basin; therefore, a comprehensive census was undertaken to determine the denominator for the lead exposure survey. Informational public meetings were held before the 1996 assessment to publicize the meetings, encourage participation, and distribute information on the study. The Idaho Panhandle Health District and TerraGraphics Environmental Engineering collaborated on the project. A census of the basin was conducted in July and August of 1996 to identify all households within 1.5 miles (2.4 km) of the 100-year floodplain of the South Fork and main stem of the Coeur d'Alene River stretching from the border with Montana to Lake Coeur d'Alene. There were 1,643 homes identified.[7] Of these, 130 refused to participate in the census. Of the remaining 1,513 homes, 670 provided census data only. All homes were approached in a door-to-door survey. There were 3,651 persons identified as living in the study area. If a home was inaccessible or unoccupied during the visit, a call-back form was left at the home. A minimum of three attempts were made to contact each household; 815 households provided soil samples, 222 provided well-water samples, 156 provided vacuum dust samples, 400 provided floor mat dust, 710 provided interior paint samples, and 749 provided exterior paint samples for lead analysis. Paint lead was assessed by a portable x-ray fluorescence machine. The environmental samples were appropriately sieved to collect small particle sizes representative of those that would be found on a young child's hands after contact.

With respect to blood lead screening, 231 children aged 0-5 years[8] and 170 children aged 6-9 years were identified by the census. Of these, 47

[7]In the HHRA (TerraGraphics et al. 2001), it was estimated that there were 5,651 housing units, of which 74% were occupied. The study area considered in the HHRA (TerraGraphics et al. 2001, Figure 3-1b) represents an area substantially larger than the geographic area considered in the ATSDR study (2000) (the area within 1.5 miles of the South Fork and main stem Coeur d'Alene River floodplain); as a result, the number of housing units considered in the HHRA is greater.

[8]This population estimate is substantially smaller than the estimate provided by the HHRA of 587 children in the basin study area aged 0 to 4 (TerraGraphics et al. 2001, Table 3-4). The HHRA compiled population estimates from 1990 census tracts that were both within and partially within the HHRA study area. The geographic area considered in these census tracts is much larger than the area considered in the ATSDR Environmental Health Assessment (ATSDR 2000). As a result, the population estimates of children in the HHRA are greater than the ATSDR study.

(20.3%) children 0-5 years of age and an additional 51 of 170 (30%) children between 6 and 9 years of age participated.

Limitations of the Sampling

In general, a 70% participation rate will provide assurance that significant selection bias did not influence the results. However, epidemiologic studies, or for that matter political polls with targeted sampling strata, can be successful without meeting the goal of 70% overall participation if the selection of participants is not biased. Lead exposure does not occur stochastically, and there are known risk factors for exposure. If selection bias did occur, one would expect differences in the prevalence of such risk factors between those families who participated in the blood lead screening and those who did not. The health assessment (ATSDR 2000) summarized community member characteristics, stratified by blood lead screening participation. Most characteristics were similar between groups. Nonparticipants were more likely to be renters (16.4% versus 9.8%) and were less likely to have attended a four-year college (13.7% versus 18.4%). Both factors likely would be associated with higher BLLs among nonparticipants.

In the years following 1996, blood lead results were from fixed-site annual screenings. Participating families had to bring children to a fixed site for the sole purpose of obtaining a blood lead measurement. Bias is much more likely to have occurred from this screening program. The direction of this bias is impossible to predict as no demographic data were collected with the screening. For these reasons, the 1996 data (which are the best available) and subsequent blood lead data have serious limitations for the purpose of making policy decisions.

Shifting the design from a fixed site to a more widespread screening program utilizing the local health care community likely would increase participation. This type of screening program would provide a population of participants less likely to be biased. Such a practice could be timed to coincide with other medically indicated health care screening tests conducted by primary care physicians. For example, screening for iron deficiency anemia is routinely conducted for children 1-5 years of age. Blood lead screening could be timed to coincide with this blood draw, thereby minimizing inconvenience to the family and child. Linking the screening program to pediatric well-child visits likely will increase participation, will provide built-in follow-up for children with elevated BLLs, and will be more convenient for families.

Blood Lead Studies from the River Basin

The committee found it unusual that this HHRA presented aggregate data on childhood lead screening data for children aged 0-9 years

(TerraGraphics et al. 2001). Children less than 1 year of age are at very low risk for lead poisoning because of their relative lack of mobility. Likewise hand-to-mouth activity falls dramatically at about age 4 years. Children 5-9 years of age are *very* unlikely to have elevated lead levels. Although the data were further stratified in many cases to 0-5 years and 6-9 years, there was an inexplicable tendency to lump these age groups together.

Figure 5-2 displays geometric mean blood lead measurements for children aged 1-5 years found in annual Coeur d'Alene River basin surveys, together with nationwide results from the National Health and Nutrition Examination Survey (NHANES). Error bars represent 95% confidence intervals on the sample geometric mean (which is taken to be as an estimator of the geometric mean of an underlying population represented by the sample). As noted above, Coeur d'Alene River basin measurements do not reflect random sampling strategies and may or may not be representative of the basin population. However, available sample geometric means are statistically elevated relative to the most closely corresponding NHANES results for all years through 2004.[9] (The most recent available NHANES data were collected in 1999-2000. Results of more recent national sampling are expected to be available sometime in 2005 and, on the basis of historical trends, are likely to reflect still lower geometric mean values.) Figure 5-3 compares the same Coeur d'Alene River basin and NHANES blood lead data among 1- to 5-year-olds when expressed as percentages of the respective populations having levels ≥10 μg/dL. Slightly more than 2% of the national population displayed blood lead ≥ 10 μg/dL in 1999-2000. By this metric, the proportion of children in the Coeur d'Alene River basin with BLLs ≥ 10 μg/dL was elevated relative to national norms at least through 2001 (see Box 5-1). The available data indicate that the percentage of children sampled in the basin with BLLs ≥ 10 μg/dL has dropped over time and, in 2004, was approximately 2.8%.

In contrast to national data, the Coeur d'Alene River basin blood data show no discernible downward trend in the years 1996-2000. Between 2000 and 2001, an apparent sharp decline in geometric mean blood lead is observed. This apparent decline may be an artifact of nonrepresentative sampling. If it is real, it appears to be much more rapid than the background rate of decline occurring in the national population. One possibility is that the decline is real and attributable to remedial activities in the Coeur d'Alene River basin. Between 1997 (the inception of remedial activities) and 2000, sixty-six residences, six schools or daycare centers, and five common-use or recreational properties were remediated (TerraGraphics et al. 2001, Table 2.3-1). Remediation of that number of properties could have contributed substantially to

[9]Another issue limiting this comparison is that the basin data and national data are not demographically matched.

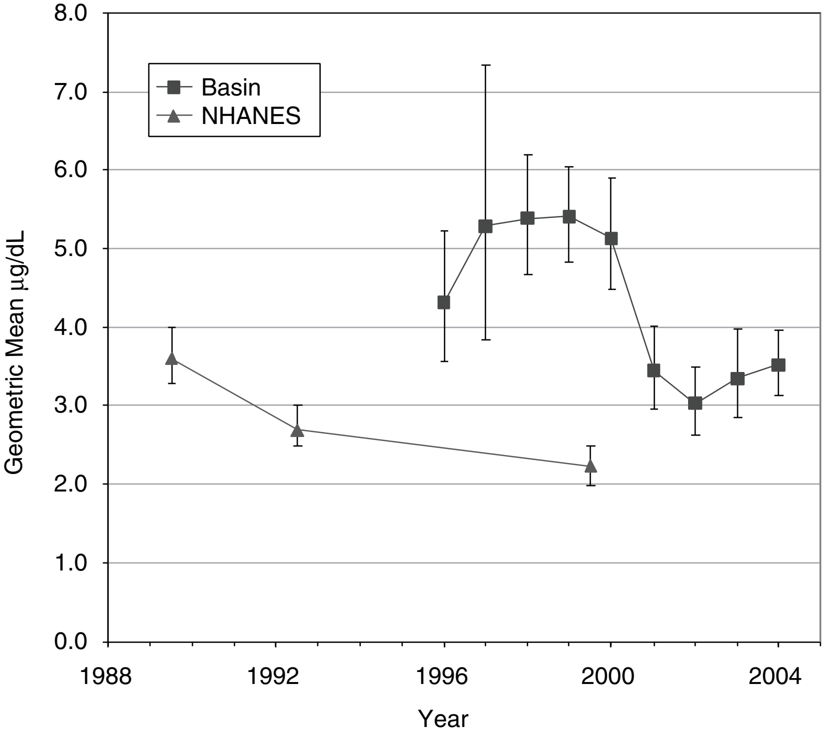

FIGURE 5-2 Geometric mean BLLs among 1- to 5-year-olds in the basin, with corresponding NHANES survey data. The estimation of basin geometric means includes the assumption that values less than the limit of detection equal half the limit of detection. Error bars represent 95% confidence intervals. Basin sample sizes in years 1996 through 2004 were 47, 12, 59, 139, 77, 98, 83, 61, and 71, respectively. It should be noted that the sampling in 1996 (ATSDR 2000) sampled individuals from a smaller area (and population) than the fixed-site sampling in subsequent years. SOURCE: Basin data, IDHW, unpublished materials 2004; NHANES data, CDC 2004.

declining blood lead, since cleanups were intended to first address sites posing the greatest apparent threats, and blood sampling was not random. In any case, this apparent improvement in the Coeur d'Alene River basin results was observed only after substantial remedial activity.

Other Information

Results of follow-up studies of 50 findings of a river basin child exhibiting a high BLL by the Panhandle Health District are reported in the HHRA

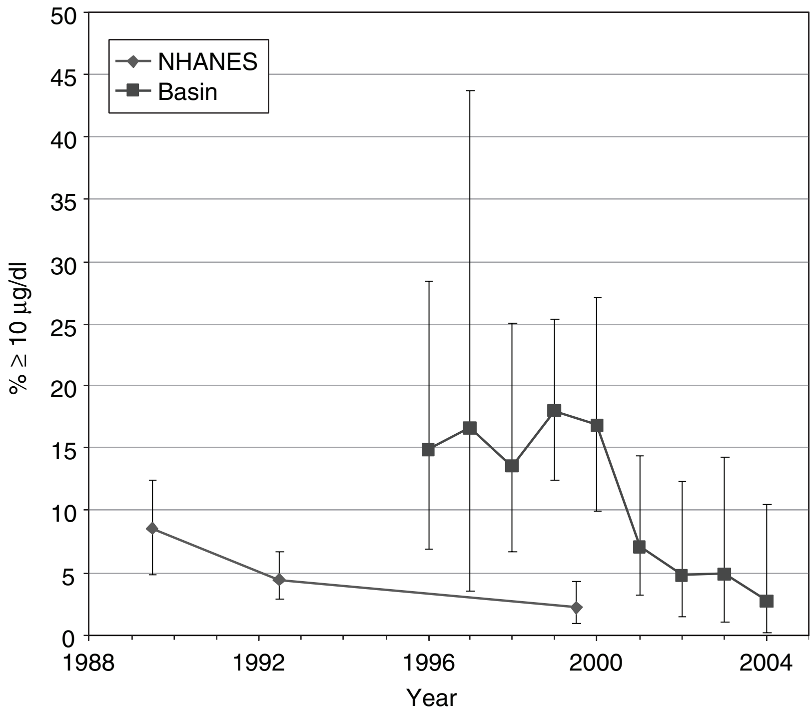

FIGURE 5-3 Comparison of fraction of blood samples among 1- to 5-year-olds from the basin with BLLs ≥ 10 µg/dL with corresponding NHANES survey data. Error bars represent 95% confidence intervals. Basin sample sizes in years 1996 through 2004 were 47, 12, 59, 139, 77, 98, 83, 61, and 71, respectively. It should be noted that the sampling in 1996 (ATSDR 2000) sampled individuals from a smaller area (and population) than the fixed-site sampling in subsequent years. SOURCE: Basin data, IDHW, unpublished materials 2004; NHANES data, CDC 2004.

BOX 5-1 BLLs in Surveys of Children in the Coeur d'Alene River Basin

Ideally, to estimate the true prevalence of elevated blood lead in a relatively small at-risk population (like that in the Coeur d'Alene River basin), all children 1 to 4 years of age would be surveyed. To estimate the prevalence at the national level, NHANES has measured a representative sample of children across the country. Some blood lead data are available for children in the Coeur d'Alene River basin, but the extent to which these data are representative of the entire population is not known. Only in 1996 was a door-to-door survey attempted, and even then only 25% of the eligible children were actually tested. Although imperfect, the Coeur d'Alene River basin blood lead data support the hypothesis that Coeur d'Alene River basin BLLs are higher than contemporaneous national BLLs.

(TerraGraphics et al. 2001). It should be noted that many potential sources of lead exposure to children are not always obvious and are difficult to detect without an extensive history of everything a child has come into contact with (for example, painted furniture, mini-blinds, keys, and key chains). However, elevated lead in residential paint was identified as a risk factor for 5 of 21 children with BLLs ≥ 15 µg/dL and for 3 of 25 children with BLLs of 10-14 µg/dL. (Some children were followed more than once.) In a much higher proportion of cases, high residential soil or dust lead or known access to other properties with high soil or dust lead or to flood-affected areas was evident. Potential risks of flooded properties were illustrated in the box by the Milo Creek flood of May 1997. In that case, a flood deposited sediments with high lead concentration, recontaminating a previously remediated area. A spike increase in elevated BLLs was observed in children in the affected zone (TerraGraphics 2000).

These observations are anecdotal and not convincing in and of themselves. However, in concert with children's known tendency to ingest soil, the demonstrated (although variable) bioavailability of lead in soil in mammalian gastrointestinal tracts, and observed BLLs in children in the Coeur d'Alene River basin, they do lend support to arguments that Coeur d'Alene River basin soils represent a lead hazard to young children.

Apportioning Risks to Humans from Multiple Contaminant Exposures

The committee was asked to assess the scientific and technical aspects of efforts regarding the following:

> Assessing and apportioning risks to humans from multiple contaminant exposures related to waste-site sources as well as other sources (for example, lead exposure via soil and house paint dust). What techniques should be used to identify contaminants of concern and estimate the human health risks attributable to waste-site sources? In this case, were risks attributable to sources other than mining and smelting activities adequately analyzed?

Two issues appear to be involved in this charge. One is whether EPA adequately identified all the exposure sources and assessed the combined risk from multiple exposures. The second is whether EPA adequately apportioned risk among the different exposures when there were multiple sources. Although the specifics of the charge relate to human health concerns, the questions presumably are also relevant with respect to environmental health concerns.

With respect to human health concerns, the agency did attempt to identify possible different sources of exposure. For lead exposures, the agency identified lead paint in older houses as a significant source of exposure, as well as the lead in yard soils and recreational and other public use areas. Another possible source of lead exposure is air deposition of lead from the exhaust of vehicles using leaded gasoline (which has been phased out) and from the emissions discharged by the Bunker Hill smelter and other ore-processing facilities in the box (eliminated in 1981). It is possible that lead from these sources still exists in the Coeur d'Alene River basin system, although the amounts would be expected to be very small in the areas covered by operable unit 3 (OU-3). The agency did not distinguish these as separate sources.

Although the agency did not identify these as distinct sources of lead exposure, it did include any exposure that still may be associated with these sources in its risk assessment. The exposure from these sources would be found in the same places as exposure from the lead in mining wastes (for example, yard soils and house dust), and the risk assessments were based on actual measurements of the amount of lead found in these exposure sources. Therefore, lead that may still exist from these nondistinguished sources would have been included in the risk assessment.

The agency did not identify any other sources of arsenic exposure, and the committee has not identified any environmental sources of arsenic that EPA may have missed. Again, the risk assessments were based on actual measurements of environmental media and, therefore, would have included arsenic from any unidentified environmental sources.

However, the residents of the area undoubtedly are exposed to other carcinogenic substances. One of these is cadmium, which has been shown to be associated with cancer in metal refinery workers who inhale cadmium fumes, but for which carcinogenicity by the oral route is equivocal. Other sources of possible exposure to carcinogens, such as smoking, pesticides, and other chemicals, are unrelated to the mining wastes. These different exposures to carcinogens may create a carcinogenic risk that is greater than that resulting from exposure to any one source. However, the consensus procedure in current risk-assessment methodology for aggregating such carcinogenic risks from multiple sources is to ignore all sources other than the one(s) of interest, treating multiple sources as exactly additive. Thus, EPA's failure to explicitly identify and assess these multiple risks reflects the current status of risk assessment procedures.

A similar line of reasoning applies to environmental exposures. Water-quality standards (for instance, for dissolved zinc) are generally established on the basis of how much of that substance alone creates unreasonable risks—although there may be modifying factors (for example, hardness of water). The fact that aquatic species are exposed simultaneously to multiple contaminants probably results in an aggregate risk greater than that posed by any of the single contaminants taken alone (although there are also examples where aggregate risks may be reduced). However, current environmental risk assessment procedures provide no guidance for aggregating such multiple risks other than by simple addition.

Thus, the answer to the first question implied in the charge is that EPA did consider risks from multiple contaminants to the extent that current risk assessment procedures provide for a basis for making such analyses. Because there is human and environmental exposure to multiple contaminants creating similar risk factors, the aggregate risk may well be greater than that estimated by EPA, but current risk assessment procedures provide no mechanism for estimating such aggregate risks.

With respect to the second question the charge appears to raise, current risk assessment procedures do not include methods for apportioning aggregate risks among multiple sources of exposure. The committee is unaware of any legal requirements that this be done or any practical use of such apportionments (except perhaps to apportion responsibility among potentially responsible parties or to obtain funds to address that portion of the total risk that cannot be remedied under Superfund).

Undertaking such an apportionment would require making a number of significantly simplifying assumptions about factors such as the shape of the dose-response curve, the amount of exposure the "typical" person has to different sources, the biological availability of contaminant in the different sources, and so forth. Given the discussion above, the only contaminant in the Coeur d'Alene River basin for which such apportionment could reasonably be attempted is human exposure to lead.

EPA did undertake a series of statistical analyses attempting to determine the relative effect of lead in mining wastes and lead in paint on BLLs (TerraGraphics et al. 2001, pp. 6-22 to 6-39). Such analyses can be considered only rough indicators because of sample weaknesses and because of the need to use surrogate measures for exposure to leaded paint.[10]

Nevertheless, these analyses, though not definitive, do strongly suggest that lead in soils was a major contributor to high BLLs. They indicated that

[10]For instance, a somewhat subjective assessment of the condition of the interior paint in houses was used as an indicator of exposure to interior leaded paint. For this variable, houses were assigned to one of three categories: category 1 if the painted surface in at least one room was considered to be in good condition, category 2 if chipping and peeling on a few surfaces in all rooms was noted, and category 3 if all paint was in chipping, peeling, and chalking condition on most surfaces. Of course, as the analyses point out, these conditions could be highly correlated with factors such as the care the resident took in cleaning the house, more care being undertaken by those who had at least one room in good condition and the least care taken by those where chipping, peeling, and chalking were observed in all rooms. If so, the correlation between this variable and BLLs could, at least to some extent, represent the resident's failure to clean the house of lead-contaminated particles tracked in from outside. In this case, the source of the lead exposure would be, at least to some extent, outside lead rather than lead paint, and attributing all of the correlation between this variable and BLLs to lead paint would be mistaken.

"although lead paint is important [as a source of exposure] for some individuals" "70% (14/20) of the children with high BLLs were not associated with an interior lead paint hazard" (TerraGraphics et al. 2001, pp. 6-29 and 6-25).[11] The analyses also include a regression model that generally supports the conclusion that lead in yard soils has a significant impact on BLLs.

Although not strictly an apportionment of risk among exposure sources, these analyses do provide support for the conclusion that lead in yard soils is a significant contributor to elevated BLLs and that reducing exposure to this source is likely to reduce the risk of elevated BLLs. The committee observes that these analyses undertaken for OU-3 go beyond normal attempts to attribute elevated BLLs to different sources of exposure and that no alternative approaches to apportioning risks would have been preferable given the information available.

STRATEGIES TO MANAGE THE RISKS TO HUMAN HEALTH

Control of Exposure by Individuals

In the face of health hazards from contaminated environmental media, a number of measures can and should be taken to reduce exposure. These protective measures include actions that can be taken at the individual level, as well as at the institutional (governmental) level. At the individual level, relatively simple interventions, such as frequent hand washing, removing of shoes before entering the home, and thoroughly washing vegetables can substantially reduce exposures to hazardous substances. Occupations associated with contact with contaminated environmental media should include practices that prevent transporting such materials into the home. The phenomenon known as "fouling one's nest" is well-known in occupational medicine.

Public notifications, such as those posted by health departments warning residents or recreators not to eat certain fish, to wash their hands, or not to drink certain water can encourage individuals to reduce their exposures to harmful substances. During the committee's visits to Coeur d'Alene River basin area, many such public warnings were found and thought to be appropriate. Yet the downside of such warnings, expressed by residents during public meetings, is that they appeared to increase psychosocial stress by making the presence of otherwise invisible hazards visible and constant. Public health departments should be aware of this and provide sufficient educational materials to residents to place the hazards in context.

[11]The Shoshone Natural Resources Coalition has raised additional potentially confounding points about this analysis (Roizen 2002). However, their critique does not undermine the basic conclusion that both lead in yard soils and lead in paint appear to have significant impacts on BLLs, with yard soils perhaps having the larger impact.

Health Programs

The HHRA states, "The Selected Remedy will include a lead health intervention program [LHIP] similar to the Bunker Hill Box LHIP, which provides personal health and hygiene information and vacuum cleaner loans to help mitigate exposure to contaminants." However, the selected remedy has few specifications of what it might involve. A comprehensive health program—one that includes health education and resources for exposure prevention—can provide more benefits to the community than just monitoring the remedy. Because soil removal (discussed below) addresses only one source of lead exposure, such a program can help address these other sources. This type of approach has been used effectively at other sites for reducing lead exposure (Kimbrough et al. 1994; Markowitz et al. 1996; Niemuth et al. 2001; Lorenzana et al. 2003). Other sites with such programs include Leadville, Colorado (EPA 1999), Butte, Montana (EPA 2005), East Helena, Montana (LCCCHD 2005), and others. Regular monitoring and intervention also help decrease the duration and magnitude of increases in blood lead. Based on current knowledge, lowering the magnitude and duration of elevated BLLs would be expected to minimize the impact.

Medical Interventions

During its visits to the Coeur d'Alene River basin, the committee heard infrequent pleas from community members who believed that medications should be administered to rid the body of potentially harmful metals. The administration of drugs to remove lead from the body, known as chelation therapy, is reserved for people with significantly elevated body burdens. The first drug ever developed for such use, calcium disodium ethylenediaminetetraacetate ($CaNa_2EDTA$), must be administered by intravenous infusion. $CaNa_2EDTA$ has been associated with improved survival in young children with lead-induced encephalopathy, a syndrome that can occur when blood lead concentrations exceed 70 µg/dL (CDC 1991). This is a level many times higher than now expected in the basin. Because use of the drug is associated with the depletion of essential minerals as well as other adverse effects, it is appropriately reserved for severe cases of lead intoxication.

The CDC currently recommends that chelation therapy be reserved for children whose blood lead concentrations are higher than 45 µg/dL (CDC 1991), who are at risk for further exposure that might lead to encephalopathy. Historically, the blood lead distribution of children in the Coeur d'Alene River basin included cases substantially higher than 45 µg/dL. However, because recent blood lead surveys no longer find children with blood lead in that range, chelation therapy does not appear to be warranted except in rare cases. Chelation therapy should never be used for prophylactic purposes, because the risks of adverse drug effects far outweigh potential benefits. Chelation in the absence of exposure reduction may be more than ineffective; it may do harm.

An oral medication with a better safety profile, dimercaptosuccinic acid (Succimer), was approved by the U.S. Food and Drug Administration in 1991 (Nightingale 1991; Graziano et al. 1992). In controlled clinical trials, Succimer has proven more effective than $CaNa_2EDTA$ in reducing blood lead concentrations and can be used on an outpatient basis (Graziano et al. 1985, 1992). Consequently, the National Institute of Environmental Health Sciences undertook a multicenter randomized, double-blind, placebo-controlled clinical trial to determine whether Succimer might be capable of improving cognitive function in children with blood lead concentrations ranging from 20-44 µg/dL (Rogan et al. 2001). The answer was no, implying that cognitive deficits associated with these levels of lead in blood are not reversible. Though there are no data concerning the impact of chelation therapy on children with lower blood lead concentrations, there is no reason to believe that the use of such drugs, which can be associated with significant adverse effects, would be effective. Thus, medical interventions with drugs that remove lead from the body do not appear to be warranted in the Coeur d'Alene River basin.

Yard Remediations: What Is the Evidence That They Are Effective?

A primary component of EPA's strategy to mitigate the effect of past lead pollution in residential areas consists of removing contaminated surface soil in residential yards and replacing it with clean soil above a geotextile membrane. The intent of the soil replacement is to reduce the amount of lead that young children take in as they ingest or inhale soil and dust. Children undoubtedly ingest some soil and dust, primarily through mouthing of objects and body parts (particularly fingers and hands), after contact of those objects or body parts with indoor dust or outdoor soil or dust. In addition, they undoubtedly inhale some dust that is raised indoors or outdoors by everyday activities.

The amount of soil and dust ingestion and inhalation in children (or in others) is not known with any great precision, although available measurements and simple calculations suggest that ingestion of dust is more significant than inhalation. Measured soil and dust ingestion clearly varies substantially among individuals and over time (van Wijnen et al. 1990; Stanek and Calabrese 1995), and its magnitude is potentially sufficient to explain elevated BLLs in the presence of lead-contaminated outdoor soil and indoor dust. Eliminating exposures to lead-contaminated dust and soil thus can be expected to result in decreases in blood lead concentrations in children.

However, it does not necessarily follow that remediation of outdoor soil will have a significant or substantial effect on children's BLLs, and the effect may vary in different circumstances. The relative contribution of indoor dust and outdoor soil to children's total soil and dust ingestion is currently a matter of conjecture rather than measurement, and their relative contributions to elevated concentrations of blood lead is also not clear. Cross-sectional epidemiological studies indicate that indoor dust is likely to be a more important contributor to elevated blood lead concentrations than outdoor soil (for example, Lanphear et al. 1998), although many such studies are of (or are heavily influenced by) residential soil contamination associated with the same residence (for example, due to lead-based paint) and not primarily due to a large external source that has contaminated or is contaminating whole neighborhoods. The relevance of such studies to a Superfund site such as the Coeur d'Alene River basin is not entirely clear, since the relationship (if any) between outdoor soil and indoor dust may be different and the dynamics of lead transport may also be different.

Typically, multiple sources of lead contribute to residential indoor dust in addition to soil just outside the residence. These include lead-based paint, wind-blown lead-contaminated dust from other locations or sources, tracked-in dust from other locations, and contaminated dust from reservoirs remaining in the household from earlier periods (for example, in attic spaces, crawl spaces, air ducts, under fitted carpets, between floorboards, and generally in nooks and crannies). Different dust sources will give rise to dusts with different characteristics (for example, particle size ranges, lead concentrations, and bioavailability of the lead when ingested or inhaled), so that equal quantities of dust from different sources, or even equal quantities of lead in dust from different sources, presumably are not equivalent in their propensity to elevate BLLs in children. Moreover, children may be exposed to lead by routes other than soil and dust and at locations other than their residence. If other exposures dominate those due to soil and dust in the residence, then reductions in residential soil concentrations may result in relatively small reductions in blood lead concentration.

In view of the uncertainties suggested here, evaluation of the likely overall effect on blood lead concentrations of various remedial actions at residences contaminated by various sources of lead currently can be adequately ascertained only by empirical studies. Realizing this, EPA and others have made efforts to perform and evaluate empirical studies of remedial actions and to evaluate observations made during remedial actions (even when the observations were not made as part of a formal study), although most such remedial actions have been directed at lead-based paint.

A 1995 EPA report (Battelle 1995) examined 16 reports evaluating the effect of remedial actions, with 12 of the reports examining children's blood lead concentrations as one end point. In ten of the reports, the principal factor evaluated was removal of exposures to lead-based paint; in five, the principal factor was cleanup of interior dust or education to encourage avoidance of dust exposures; and in one, the Boston arm of the Urban Soil Lead Demonstration Project (EPA 1993b; Weitzman et al. 1993; Aschengrau et al. 1994, 1997), the principal factor evaluated was soil removal and replacement in an urban area with no identified principal external lead source.

A 1998 update (Battelle 1998) examined 18 other reports (and in addition included further interpretation of the Boston arm of the Urban Soil Lead Demonstration Project). Five of these additional reports were of soil replacement actions—the Baltimore and Cincinnati arms of the Urban Soil Lead Demonstration Project (EPA 1993c,d) in urban areas with no identified principal external sources, and three Canadian community-wide actions, one in the South Riverdale suburb of Toronto (Langlois et al. 1996) near an operating secondary lead smelter, one in St.-Jean-sur-Richelieu in Quebec (Goulet et al. 1996) near a recently closed battery reclamation plant, and one in the Notre Dame district of Rouyn-Noranda, Quebec (Gagne 1994), around an operating copper smelter.

In a review article focused on remedial actions associated with lead contamination at locations characterized as “hazardous waste sites,” Lorenzana et al. (2003)[12] examined the outcomes of eight reports, four on actions that included soil replacement—the three Canadian actions just mentioned and the activities around Port Pirie, Australia (Calder et al. 1994), near a primary lead smelter.

During a presentation to the committee (Southerland 2004), EPA cited four additional locations, and provided some additional supporting information (EPA 2004b). At these locations (Midvale, Jasper County, Bartlesville, and Tar Creek) EPA claimed that available pre- and postremediation measurements of BLLs were supportive of EPA actions at the Coeur d’Alene River basin Superfund site. The results of cross-sectional surveys of children at the Midvale, Utah, site (the former site of a lead, zinc, and copper smelter) have been reported in the peer-reviewed literature (Lanphear et al. 2003). The Jasper County, Missouri, site is near the Eagle-Picher smelter in northwest Joplin, Missouri. An extensive report detailing the surveys of children postremediation is available (MDHSS/ATSDR 2004) and incorporates limited comparisons with an earlier survey preremediation.[13] Information available to the committee on the Bartlesville, Oklahoma, site associated with the National Zinc Company smelter is very limited. Results of surveys conducted before remediation are summarized in an ATSDR Public Health

[12]Two of the five authors are with EPA, and the other three are with a private firm that contracted with EPA for work on lead.

[13]On its Web page, this report is stated to be available only in electronic form.

Assessment (ATSDR 1995), whereas only the number of children tested and the number of those with blood lead exceeding 10 μg/dL in each year from 1995 to 2001 are documented in an EPA 5-year-review (EMC^2 and Phelps Dodge Corporation 2001). In view of this very limited information, the site is not further considered here. For the Tar Creek, Oklahoma, site, ATSDR (2004a) recently provided a Report to Congress that summarized the available studies.

ATSDR (2004b) has also recently documented the experience at Galena, Cherokee County, Kansas, where remediation included residential soil replacement, and before and after studies on BLLs are available. Louekari et al. (2004) examined BLLs around a former smelter where some soil removal actions were taken; however, the authors did not attempt to evaluate the relative contributions of multiple actions designed to reduce exposures (including closure of the smelter), so this report is not further considered. A further report on Port Pirie has been published (Maynard et al. 2003), providing updated information on BLLs and activities intended to reduce them and including further references (Heyworth et al. 1993) to published material on Port Pirie. A report (Morrison 2003) describing activities around a smelter in the Lake Macquarie area of New South Wales, Australia, was brought to the committee's attention. However, the activities described did not include soil replacement (although removal of slag was documented as was installation of landscaping covers like bark, chips, and grass), so this report is not considered further here.[14]

The EPA experience in the Bunker Hill box at the Coeur d'Alene River basin site has also been reported (Sheldrake and Stifelman 2003; von Lindern et al. 2003), where residential areas were contaminated by smelter emissions (the smelter closed in 1981) and mining waste. These studies report on 12 years of blood lead surveys that were conducted between 1988 and 2000. Participation rates over the period 1990 to 1998 averaged 50% for children aged 9 months to 9 years, and more than 4,000 blood samples were collected.

During this time frame, the site had a variety of interventions including community education programs; soil removal and replacement in yards (soil lead concentration >1,000 mg/kg), public areas, and rights-of-way; and stabilization of barren areas contributing to fugitive dusts. Actions focused on the former smelter complex included demolition of the industrial complex and removal of contaminated soils and mining wastes associated with the industrial areas.

On a site-wide basis, the geometric mean yard soil exposure metric decreased from 2,292 mg/kg in 1988 to 182 mg/kg in 1998. The geometric community soil concentration decreased from 1,528 mg/kg in 1988 to 297 mg/kg

[14]The available blood lead measurement results appear to be limited to those reported in a local newspaper.

in 1998. The geometric mean neighborhood (200 feet) soil concentration decreased from 2,119 mg/kg in 1998 to 325 mg/kg in 1998. During this period, geometric mean BLL decreased from 8.5 µg/dL in 1988 to 4.0 µg/dL in 1998 (and continued to decrease to 2.7 µg/dL in 2001).

The study concludes the following:

> Repeat measures analysis assessing year to year changes found that the remediation effort (without intervention[15]) had approximately a 7.5 µg/dL effect in reducing a 2-year-old child's mean blood lead level over the course of the last ten years. Those receiving intervention had an additional 2-15 µg/dL decrease. Structural equations models indicate that from 40 to 50% of the blood lead absorbed from soils and dusts is through house dust with approximately 30% directly from community wide soils and 30% from the home yard and immediate neighborhood.

The study also comments on the potential for other interfering effects: "The overall analysis should be viewed as a forensic exercise to learn as much as possible from this decade-long health response effort. Caution should be exercised in considering individual results, as these were not designed experiments" (von Lindern et al. 2003).

The committee agrees with the warning to interpret the results cautiously. Indeed, the lack of any control group necessarily resulted in the methodology assigning the observed decrease in blood lead concentrations to the environmental changes caused by the interventions. Moreover, even if the reductions in BLLs observed in the box were due to the interventions, extrapolation to other locations within the Coeur d'Alene River basin may not be warranted—for example because of differences in behaviors and opportunities for exposure within and outside the box.

Thus, there are 12 reports from a variety of locations that might provide some information on the effects of soil removal and replacement. We provide very short summaries of some salient information from the reports and the conclusions of the original authors in Box 5-2 at the end of this chapter. The committee located no further reports during informal searching of the published literature.

Overall, the magnitude of the effect that various remedial actions have on BLLs is not well defined. In this regard, the conclusion of Lorenzana et al. (2003) is especially appropriate when considering the effect of soil replacement:

> The outcomes of the intervention studies suggest that various approaches to intervention of the dust ingestion pathway, alone or in combination,

[15]In this quote, "intervention" indicates medical intervention.

BOX 5-2 Summary of Twelve Studies Concerning the Efficacy of Yard Remediation

We provide here very short summaries of some salient information from the reports and conclusions of the original authors.

Baltimore arm of the Urban Soil Lead Demonstration Project (EPA 1993c).

Source. No single identified source. Soil lead contamination primarily due to lead paint.

Data. Six rounds of blood lead sampling in a population of children aged 6 months to 6 years, with interventions between rounds 3 and 4. Door-to-door recruitment into the study was used. At the first round, 212 children were recruited in the study area and 196 in the control area, a total of 408. By round 3, just 270 children were tested due to attrition and additional enrollment; further attrition occurred in subsequent rounds (no further children were enrolled).

Interventions. Exterior lead paint was stabilized and contaminated soil was replaced (lead concentration > 500 mg/kg within property boundaries, with 6 inches of soil replaced and sodded or seeded). Household members were excluded from the property during these operations.

Change in surface-soil lead concentration. A reduction of 550 mg/kg ("tri-mean" measure).[a]

Results. Just before intervention, the arithmetic mean blood level in round 3 testing in the study area was 11.1 μg/dL, and in the control area it was 10.2 μg/dL (the committee estimates the corresponding geometric mean concentrations to be about 9.6 and 9.0 μg/dL respectively[b]). Similar summary statistics postremediation are not provided, although the results of extensive modeling are summarized, and a data compilation is available (EPA 1996a).

Study conclusions. "Statistical analysis of the data from the Baltimore lead in Soil Project provides no evidence that the soil abatement has a direct impact on the blood lead level of children in the study." In view of the presence of lead-based paint in both abated and control areas, it was reported that the conclusion might be more precisely stated as "in the presence of lead-based paint in the children's homes, abatement of soil lead alone provides no direct impact on the BLLs of children."

Other interfering effects. Lead-based paint was present in both abated and control areas. A smaller decrease in soil lead concentration was achieved than originally was desired in the design of the study (>1,000 mg/kg was hoped for).

Cincinnati arm of the Urban Soil Lead Demonstration Project (EPA 1993d).

Source. Soil lead contamination primarily due to lead-based paint.

Data. Three areas—A, B, and C—were examined, with nine phases of monitoring over a 2-year period, including seven phases with blood lead measurements. A total 307 children were involved, the focus being on 173 children less than 6 years of age who were in the initial recruitment.

Interventions. Soil replacement, interior dust abatement (including carpet and some upholstered furniture replacement), and exterior dust abatement were used. Between phases 1 and 2, area A was abated for soil, interior dust, and exterior dust, and area B was abated for interior dust only. Between phases 5 and 6, area B was abated for soil and exterior dust. Area C was not abated during the study (it was abated afterward, but no monitoring was performed afterward).

Change in surface-soil lead concentration. In area A, geometric mean lead concentration in the top 2 cm core-composite samples decreased from 200 mg/kg preabatement to 54 mg/kg postabatement. In area B, geometric mean lead concentration in the top 2 cm core-composite samples decreased from 161 to 60 mg/kg. The committee estimates that these correspond approximately to changes in arithmetic mean concentrations from 690 to 120 mg/kg in area A and from 410 to 90 mg/kg in area B.[c]

Results. Immediately after abatement, small but nonsignificant reductions were observed in blood lead concentrations (for example, after abatement in area A, the geometric mean blood lead decreased from 8.9 to 7.0 µg/dL), but these reductions were transient and vanished by the next phase of sampling. Moreover, similar or larger variations were observed in the control area C.

Study conclusions. "There was no evidence that blood lead levels were reduced by soil lead or dust abatement in area A. There was a slight reduction (net reduction over control area of 0.6 µg/dL in Area B that may be attributed to interior dust abatement (this difference was not statistically significant)."

Other interfering effects. Relatively small reductions in soil concentrations. The study was carried out primarily in multifamily housing units rehabilitated and lead-abated two decades earlier. However, these housing units were intermixed with nonrehabilitated units. Soil was not primarily associated with individual buildings.

Boston arm of the Urban Soil Lead Demonstration Project (EPA 1993b; Weitzman et al. 1993; Aschengrau et al. 1994, 1997).

Source. Soil lead contamination probably primarily due to lead-based paint.

Data. BLLs in 152 children initially aged up to 4 years and with BLLs from 10 to 24 µg/dL (or living in housing units containing a previously selected child with a BLL in that range), selected also according to geographical area and certain housing conditions, randomly assigned to a study group (group S, 54 children) or to comparison groups A (51 children) and B (47 children). In phase I (EPA 1993b, Weitzman et al. 1993), BLL was measured preabatement, and approximately 6 and 11 months later, the latter an average of about 9 months after abatement. In phase II (Aschengrau et al. 1994, 1997), BLLs were measured at approximately 22 months, an average of about 9 months after the second round of abatements.

Interventions. In phase I, group S homes had soil replacement, interior dust abatement, and loose interior lead-based paint stabilization; group A homes had interior dust abatement and loose interior lead-based paint stabilization; and group B homes had loose interior paint stabilization. In phase II, comparison groups A and B had soil replacement, and residential lead-based paint removal was offered to all three groups.

Change in surface-soil lead concentration. Average soil concentration in group S was reduced from approximately 2,255 to 160 mg/kg.[d]

Results.[e] In phase I, the mean BLLs of group S decreased from 13.10 to 10.65 µg/dL at 11 months, those of group A from 12.37 to 11.49 µg/dL, and those of group B decreased from 12.02 to 11.35 µg/dL (the 11-month point was considered most appropriate to minimize seasonal effects). The reduction in group S was significantly larger than in groups A and B but lower than that incorporated in the study hypothesis. Adjusting the results in various ways did not change these conclusions significantly. In phase II, the mean decline in BLLs in groups A and B was

continued on next page

BOX 5-2 continued

larger than seen in group S in phase I, and fairly complex analyses were applied to estimate the effect of soil replacement.

Study conclusions. "The combined results from both phases suggest that a soil lead reduction of 2060 mg/kg is independently associated with a 2.25 to 2.7 μg/dL decline in blood levels" implicitly, after approximately 2 years.

Other interfering effects. In phase I, paint stabilization and dust cleanup effects cannot be entirely separated from soil replacement. In phase II, seasonal effects, the secular effects of aging, and selection biases cannot be ruled out, and there was no control group. Final results depend to some extent on the modeling assumptions made.

Toronto Soil and Dust (Langlois et al. 1996).

Source. A secondary lead smelter operated throughout the period of study.

Data. BLLs collected in six cross-sectional surveys of children less than 6 years old in a study area in South Riverdale (SR), two cross-sectional surveys of a sociodemographically similar comparison area in South Riverdale (SRC), also of children less than 6 years old, and four surveys in the school-based Ontario Blood Lead Study (OBLS) (children aged 3-6) distant from the source. Surveys were carried out in 1984, 1987, 1988, 1989, 1990, and 1992 (SR); in 1988 and 1990 (SRC), and in 1984, 1988, 1990, and 1992 (OBLS). Response rates varied from 75% to 32% and decreased over time.

Interventions. Most of the 970 properties in SR with soil concentrations of lead exceeding 500 mg/kg had the top 30 cm of soil replaced in 1988. In 1989, professional housecleaning was offered to all 1,029 households in the soil testing area in SR, and 717 households agreed.

Change in surface-soil lead concentration. Not stated (soil lead concentrations were measured and were used in an analysis of variance).

Results. BLLs declined in all three areas surveyed. Mean values varied from 14 to 3.9 μg/dL in SR over the period of 1984-1992 and from 11.9 to 3.5 μg/dL in OBLS over the same period.

Study conclusions. The decrease in blood lead during the 1980s was consistent with observations from other areas, with the most-likely major responsible factors being the reduction in lead in gasoline and in canned food. Three study observations of community-level averages suggested the possibility of an effect of interventions—a reduction of BLLs in SR below extrapolated values, significant changes in time trends after 1988, and a more rapid decline after 1988 in BLLs in SR compared with SRC. However, individual data gave a different impression, because blood lead concentrations in individual children who did not experience any abatement action in their household decreased faster than blood lead concentrations for children experiencing abatement. Overall findings "were equivocal and did not strongly support or refute a beneficial abatement effect."

Other interfering effects. The concentration of air lead levels in Toronto declined over the study period and more rapidly during 1987-1988; decreased emissions from the smelter also may have played a part.

Rouyn-Noranda Soil (Gagne 1994).

Source. A 2,500-ton-per-day copper smelter operating since 1927.

Data. BLLs from three surveys in the Notre Dame district within 1 km of the smelter (in 1978, 1989, and 1991) of 2- to 5-year-olds (except in 1991, 1-year-olds were included).

Interventions. No interventions were considered necessary in 1978, because the BLL (21 μg/dL, geometric mean; 95th percentile, 29 μg/dL) was below the CDC guideline of 30 μg/dL at that time. In 1990-1991, all residential lots with soil lead concentration exceeding 500 mg/kg, including 80% of the 710 lots in the Notre Dame district, had soil replaced to a depth of 10 cm and then grassed or covered with gravel.

Change in surface-soil lead concentration. Not stated. Mean soil lead concentration in 1989 was 700 mg/kg.

Results. In 1978, geometric mean BLL was 21 μg/dL in a sample of 29 children. In 1989, geometric mean blood level was 10 μg/dL (in 117 of 124 eligible children, 94%), and this decreased to 7.3 μg/dL for 2- to 5-year-olds in 1991 (87 children 2 to 5 years old, 95% participation in 1- to 5-year-olds overall).

Study conclusions. These results were considered indirect evidence of the efficiency of soil decontamination in reducing BLLs.

Other interfering effects. Smelter emissions were declining over the period, from 850 tons/year in 1988 just before the study to 300 tons/year in 1991. In 1991, 24 of 29 children with a BLL exceeding 10 μg/dL lived in the portion of the district nearest to the smelter, with significantly more dustfall than the remainder of the district. It was hypothesized that exposure to air lead and/or actual lead dustfall on hard surfaces would explain the difference in blood lead between children living in and out of this portion of the district. Age distributions were not reported or corrected, and differences could have biased results.

St. Jean-Sur-Richelieu Soil and Dust (Goulet et al. 1996).

Source. A battery-reclamation plant, presumably emitting lead dust (the distribution of contamination corresponded to the prevailing winds).

Data. In September 1989, the BLLs of children 0-10 years of age within 600 m of the plant were measured (81.6% participation rate). A second survey in August 1991 measured the BLLs of 101 children aged 6 months to 10 years (79.2% participation rate) living within 150 m of the plant.

Interventions. Asphalting of the plant yard, contaminated soil replacement, professional home cleaning, street dust cleaning, public health education campaign.

Change in surface-soil lead concentration. Not stated. The median lead concentration of soil samples within 200 m of the site was 500 mg/kg, ranging up to 5,040 mg/kg, before replacement of soils with lead concentrations less than 500 mg/kg.

Results. For children 6 months to 5 years old, a reduction in geometric mean blood lead from 9.8 to 5.5 μg/dL; for those 6 months to 10 years old, a reduction from 9.2 to 5.0 μg/dL. Results were similar for those children measured in both surveys.

Study conclusions. The lead-poisoning prevention program reached its main objective to lower mean BLL of children to the 5-8 μg/dL range.

Other interfering effects. Other remedial actions were taken. The plant was shut down one month before the first blood lead survey. Two measured oral

continued on next page

BOX 5-2 continued

activities of children (pica and putting things in their mouths) were significantly decreased in the 1991 children compared with the 1989 children. Various demographic factors, including age distributions and differential response rates, could have biased results.

Port Pirie, South Australia, Study (Heyworth et al. 1993; Calder et al. 1994; Maynard et al. 2003).

Source. Continuing operation of the Pasminco Port Pirie, one of the largest primary lead-zinc smelters in the world.

Data. During the first 10-year lead program, beginning in 1984, school-based screening for blood lead was offered to children up to 7 years old every 6 months. Between approximately 500 and 1,000 children participated in each 6-month cycle. Since 1995 (during a second lead program) screening has been census-based and annual for children up to 5 years, with approximately 95% participation.

Interventions. In the first lead program, interior and exterior lead-based paint abatement, interior and exterior cleaning and sealing against dust ingress, soil replacement, greening, active discouragement of use of rainwater for drinking and cooking and provision of clean water, community education, and smelter environmental controls. In the second program, buying and removal from use of properties nearest to the smelter, continuing education, dust control, behavior modification, targeted residential modifications, and a continuing investigative program at the smelter to identify and control emissions.

Change in surface-soil lead concentration. Not stated.[f] Soil with lead greater than 5,000 mg/kg of lead was replaced, and assistance was provided to homeowners to cover soil measuring 1,250 to 5,000 mg/kg, with only educational advice provided for lower concentrations.

Results. Geometric mean BLLs for children up to 7 years old declined from 17.8 μg/dL in 1984-1985 to 12.5-13.0 μg/dL in 1991, and for children up to 5 years old from 13.6 μg/dL in 1993 to 10.6 μg/dL in 1999. There is considerable variation in BLLs, with children nearer the smelter having highest blood leads; the variation in 1999 was from a geometric mean of approximately 19.8 μg/dL in the highest residential location to 8.2 μg/dL in the lowest tested area. The largest reductions have occurred in the least-affected areas.

Study conclusions. House decontamination (removal of dust, abatement of paint, repairs to reduce dust entry, soil replacement) "produced a transient reduction in blood lead, levels subsequently increased again after 6-12 months" (Heyworth et al. 1993, as cited in Maynard et al. 2003).

Other interfering effects. The many other efforts to reduce exposure cited above, together with apparently continuing efforts to reduce smelter emissions. Analysis is complicated by the voluntary nature of the testing, the lack of preintervention data, and the lack of a control group (Heyworth et al. 1993).

Midvale, Utah (Lanphear et al. 2003).

Source. A former smelter (closed 1958) and milling operation (closed 1971) and the associated tailings piles with high concentrations of lead and arsenic. Contamination was by wind, through transport on workers' clothing, and through use of tailings on residential properties.

Data. Two cross-sectional surveys of children aged 6-72 months in Midvale; in 1989 a random sample (112 children, 90% participation), and in 1998 a full population sample (215 children, 70% participation).

Interventions. The tailings were covered with a clay cap in 1993. From 1993 to 1996 soil replacement was performed in yards with soil concentrations exceeding 500 mg/kg lead.

Change in surface-soil lead concentration. The decline in average "foundation soil lead" was 488 mg/kg in the intervention group (542-54 mg/kg, a significant reduction), whereas in the control group it was 49 mg/kg (from 144 to 95 mg/kg, not significant).

Results. In 1989, the geometric mean blood lead of the 73 children in homes with average soil concentrations greater than 500 mg/kg that were subsequently abated was 5.6 µg/dL, and in the 39 children in homes with mean soil concentration less than 500 mg/kg that were not subsequently abated, it was 3.9 µg/dL. In 1998, the geometric mean blood lead of the 167 children in homes that were abated was 3.0 µg/dL, and that of the 31 children in homes that were not abated was 2.6 µg/dL. Socioeconomic status differed between abated and nonabated homes both in 1989 and 1998, but mouthing behaviors did not. Adjustment for age, mouthing behavior, economic status, and year of study suggested a 2.3 µg/dL decline in blood lead concentration associated with soil abatement, but this decline was not statistically significant. A similar analysis for children aged 6-36 months gave a statistically significant decrease of 2.5 µg/dL, equivalent to 3.5 µg/dL per 1,000 mg/kg reduction in soil lead.

Study conclusions. "Soil abatement was associated with a significantly greater reduction in blood lead concentration than expected among children ages 6 to 36 months who had not been exposed to lead-contaminated yards in early childhood. In contrast, soil abatement was not associated with a greater reduction in blood lead concentrations than expected for children ages 36 to 72 months."

Other interfering effects. The study assigns the entire effect to soil abatement, but there is no discussion of any assessment of whether the capping of the tailings pile had an independent effect (for example, through reduction of the effect of windblown dust[g]). The possible effect of interior and exterior lead-based paint was also not apparently modeled—tabular data presented show significant differences between remediated and unremediated groups in an index of both interior and exterior lead paint, and significant declines in both indices between 1989 and 1998; no mention is made of the meaning of these indices or of their potential importance.

Jasper County, Missouri (EPA 2002; MDHSS/ATSDR 2004).

Source. A primary lead smelter (the Eagle-Picher smelter in Joplin, Missouri) that operated into the 1970s, together with mining and milling wastes.[h]

Data. Two cross-sectional surveys, in 1991 and 2000, of BLLs in the same geographical areas. Random samples of children (213 in 2000, 225 in 1991) aged 6-72 months were obtained, but with low response rates (36% in 1991, 34% in 2000).

Interventions. Yard soil replacement, educational efforts, bottled water in some locations.

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BOX 5-2 continued

Change in surface-soil lead concentration. Mean soil concentration was 599 mg/kg in 1991 and 519 mg/kg in 2000. These results are not comparable, because of different sampling methods and different sampling frames (all homes in 2000, whereas in 1991 only a random sample together with children with blood lead exceeding 10 μg/dL).

Results. Arithmetic mean BLL in 1991 was 6.24 μg/dL, and in 2000, it was 3.82 mg/dL. Geometric mean values are not given, but the committee estimated them from the information provided as 4.9 μg/dL in 1991 and 3.3 μg/dL in 2000.[i]

Study conclusions. "Although it is not possible to determine the individual contribution of the soil remediation compared with the health education and paint stabilization, it is reasonable to conclude that the substantial soil remediation actions contributed significantly to the reduction in numbers of children with elevated BLLs."

Other interfering effects. Several other lead-related environmental indicators were substantially changed between surveys, including measures of indoor and outdoor paint levels, and there appeared to be a substantial rebuilding rate, with approximately one-third of the houses less than 10 years old in both 1991 and 2000. Lead water-pipe use declined from 9.1% to 1.9%. Data on a somewhat augmented sample of children in 2000 (including an area outside that sampled in 1991) indicate that fewer than one-third of the homes in which the surveyed children live had soil remediation. There were no 1991 to 2000 comparative analyses that attempted to take account of any of these environmental changes. Sampling strategies differed somewhat between 1991 and 2000 (in 1991, two children were sampled from 33 homes, whereas in 2000 only one child was sampled from each home); modifying the 1991 sample by randomly selecting only one child per home reduced the arithmetic mean 1991 blood level from 6.24 to 5.85 μg/dL (geometric mean approximately 4.8 μg/dL).

The Bunker Hill Box at the Coeur d'Alene River Basin Superfund Site (TerraGraphics 2000; Sheldrake and Stifelman 2003; von Lindern et al. 2003).

Source. Smelter emissions (the smelter closed in 1981) and mining waste.

Data. More than 4,000 blood samples in children aged 9 months to 9 years over a period of approximately 12 years from 1988, obtained by door-to-door survey with $20 payment for participation. Estimated participation rate ranged from 42% to 58% from 1990 to 1998 (average 50%).

Interventions. Community education programs. Soil removal and replacement in yards (soil lead concentration >1,000 mg/kg), public areas, and rights-of-way. Stabilization of barren areas contributing to fugitive dusts. Final demolition of the industrial complex. Removal of contaminated soils and mining wastes associated with the industrial areas.

Change in surface-soil lead concentration. Multiple measures of soil concentration have been tracked and changed in different ways in different communities in the site. On a site-wide basis, the geometric mean yard-soil exposure metric decreased from 2,292 mg/kg in 1988 to 182 mg/kg in 1998. The geometric community soil concentration decreased from 1,528 mg/kg in 1988 to 297 mg/kg in 1998. The geometric mean neighborhood (200 feet) soil concentration decreased from 2,119 mg/kg in 1998 to 325 mg/kg in 1998.

Results. Geometric mean BLL decreased from 8.5 μg/dL in 1988 to 4.0 μg/dL in 1998 (and continued to decrease to 2.7 μg/dL in 2001).

Study conclusions. "Repeat measures analysis assessing year to year changes found that the remediation effort (without intervention)[j] had approximately a 7.5 μg/dL effect in reducing a 2-year-old child's mean blood lead level over the course of the last ten years. Those receiving intervention had an additional 2-15 μg/dL decrease. Structural equations models indicate that from 40 to 50% of the blood lead absorbed from soils and dusts is through house dust with approximately 30% directly from communitywide soils and 30% from the home yard and immediate neighborhood."

Other interfering effects. "The overall analysis should be viewed as a forensic exercise to learn as much as possible from this decade-long health response effort. Caution should be exercised in considering individual results, as these were not designed experiments" (von Lindern et al. 2003). Indeed, the lack of any control group necessarily resulted in the methodology assigning the observed decrease in BLLs to the environmental changes caused by the interventions.

Galena, Cherokee County, Kansas (EPA 1996b, 2000a; ATSDR 2004b).

Source. Primarily smelter emissions (one or more smelters operated in the town from 1890 through 1960 [Breggin et al. 1999]), with possibly some import of mining wastes for construction, fill, and landscaping material.

Data. In 1991, BLLs for 52 of 63 children aged 6 or younger and environmental sample results (soil, dust, paint) for their 52 homes. Also in 1991, environmental samples from a random sample of homes of children of all ages, and blood lead values for 128 children aged 6 or younger from a control area. In 2000, BLLs of 100 children aged 6 months to 6 years and environmental samples from their 72 homes, 31 of which had been remediated and 41 not. The estimated response rates of eligible children were 26% (for all 63 children) in 1991 and 33% in 2000.

Interventions. Excavation of residential soils exceeding 800 mg/kg lead or 75 mg/kg cadmium to a depth of 1 foot or until the soil concentration does not exceed 500 mg/kg lead or 25 mg/kg cadmium; or of garden soil exceeding 500 mg/kg lead or 75 mg/kg cadmium. In addition, health education, institutional controls, and an operation and maintenance program were part of the intended interventions.[k]

Change in surface-soil lead concentration. For remediated homes, soil lead concentrations declined from 1,660 mg/kg to 345 mg/kg (n = 30), while for non-remediated homes, soil lead concentration was not significantly different (448 mg/kg in 1991 and 491 mg/kg in 2000, n = 30). It was not specified whether these were arithmetic or geometric means, although the committee infers that geometric means were used.

Results. In 1991, the 52 children from Galena had a geometric mean BLL of 4.13 μg/dL, higher than the 3.13 μg/dL for the 128 children in the control area. In 2000, the 100 children from Galena had a geometric mean blood level of 2.29 μg/dL (there was no comparison group in 2000).

Study conclusions. ". . . both blood and soil lead levels have significantly decreased" and "There was no significant difference in mean BLLs in children living

continued on next page

BOX 5-2 continued

in either remediated or non-remediated homes in 2000. The reduction in BLLs from 1991 to 2000 in Galena was better than that expected, based on the U.S. population. These results suggest the effectiveness of the remediation and education effort in reducing BLLs in children in Galena."

Other interfering effects. Other interventions are mentioned above. The low response rate could have biased results. The lack of any comparison group in 2000 makes interpretation difficult. There was no attempt to estimate the effect of soil removal independent of other actions.

Tar Creek Superfund Site, Ottawa County, Oklahoma (EPA 2000b; ATSDR 2004a).

Source. Extensive chat (mining waste) piles in residential areas and use of chat as a construction product and lead-based paint.

Data. A combination of blood lead results obtained between 1995 and 2004 from the Oklahoma Child Lead Poisoning Prevention Program, Tribal Efforts Against Lead surveys, and the Ottawa Lead Poisoning Prevention Program.

Interventions. Residential and play-area soil removal and replacement, community and healthcare provider education, blood lead screening efforts, distribution of HEPA vacuums to households with children having elevated BLLs.

Change in surface-soil lead concentration. Not evaluated.

Results. Declines in measures of BLLs, including geometric mean blood levels (from about 4.8/6.7 μg/dL in 1995/1996 to 2.7/3.0 μg/dL in 2002/2003), and the fraction of children with blood lead level over 10 μg/dL.

Study conclusions. No evaluation was made of the effectiveness of soil removal/remediation efforts; it was assumed that remedial efforts had been effective in producing the observed decline in BLLs and that "Existing programs should be evaluated to determine how they may have contributed to this decline."

> contributed to declines in blood lead levels in children living in areas heavily contaminated with lead. . . . However, the effects of confounding factors and the lack of control . . . made it difficult to assess the magnitude of the contribution of intervention and the relative contributions of the various interventional approaches.

At best, the evidence available that soil replacement contributes to declines in BLLs is suggestive, as may be seen from the 12 reports discussed in Box 5-2 at the end of this chapter. Theoretically, because of the practical certainty of some ingestion of soil and dust, removal of soil should have some effect on BLLs. However, the magnitude of that effect is clearly small enough to be difficult to measure and may well be substantially smaller than would be predicted

Other interfering effects. Other interventions mentioned above. No formal study was conducted, rather a survey of available information, so the effect of various potential biases cannot be evaluated.

[a]This is the value given in the Executive Summary; however, Table 7-3 indicates a reduction of 407 mg/kg in the average tri-mean measure. The committee has not investigated the discrepancy.

[b]Assuming lognormal distributions, combined study and control group concentration distributions are plotted and appear to be consistent with lognormal.

[c]Assuming lognormal distributions of concentrations.

[d]Soil concentrations were characterized by the median of an average of eight samples for each housing unit. Different publications on this study give slightly different statistics, presumably because they have slightly different selection criteria for inclusion in the various averages.

[e]The committee cannot estimate geometric mean BLLs (as used in the other reports) by assuming lognormality of blood lead distributions, because the selection of subjects by BLL probably distorted the distribution away from the usual lognormal shape generally seen in population samples. Approximate calculations, and examination of the medians of the distributions, suggest that the changes in geometric mean blood level would be similar to the changes in mean BLL reported.

[f]This information may have been published in material not examined by the committee.

[g]Floor dust lead and arsenic loadings and lead, but not arsenic concentrations, decreased significantly in the unremediated houses, although soil lead and arsenic concentrations did not. No mention is made, for example, of the proximity of the houses to the tailings pile.

[h]The smelter was dismantled in 1982 (Eagle-Picher 2002).

[i]These values are also approximately the medians of the distributions shown in MDHSS/ATSDR 2004. They were obtained from the reported means and standard deviations by assuming lognormal distributions of BLLs. For 1991, an identical value is obtained by digitally extracting the distribution shown in MDHSS/ATSDR 2004, (Figure 1) and numerically integrating its log transform. Numerical integration of this curve untransformed reproduces the reported mean and standard deviation. Lack of certain technical information prevented use of the same procedure for the 2000 curve.

[j]Here “intervention” indicates medical intervention. What we have called interventions correspond to the “remediation effort.”

[k]The committee does not know the achieved extent of these programs.

by models such as the IEUBK as usually used to estimate the effects of soil and dust lead from the types of measurements usually made on soil and dust.

The experience with lead from gasoline,[16] the observations around operating smelters summarized in Box 5-2, and the observation of large changes in blood lead in response to fluctuations in smelter emissions (Hilts 2003; Morrison 2003)[17] suggest that more attention should be paid specifically to

[16]The effect of air lead, primarily from gasoline, on BLLs was two to three times larger than would be expected from inhalation alone (EPA 1986) but without concomitant changes in measured soil concentrations.

[17]The effect of possible reduction of emissions from the Rouyn-Noranda smelter due to the recent (2002-2003) strike might be observable in BLLs of the adjacent population.

the surface films of dust with which we come in contact rather than the larger samples generally obtained by soil sampling or vacuuming.

Institutional Controls

Institutional control programs[18] are critical for the long-term protection of human health. Once yards, recreational sites, and other properties have been remediated, opportunities for disruption of protective remedial barriers still exist. In 1995, the Idaho Panhandle Health District (PHD) was given the authority by the legislature to issue rules governing the management of contaminants. Public outreach and education play an important role offering protection to individuals. However, institutional control programs, such as the one coordinated by the PHD, can ensure that building, construction, renovation, and soil excavation activities do not lead to human exposure to soil contaminants. Those programs include important components that will need to be maintained over time.

Contractor Licensing

The PHD licenses all contractors involved in soil excavations, building renovations, and other comparable activities that might disrupt existing barriers to human exposures. Contractors are provided education and must pass a test that involves questions about methods of contaminant management and the reasons that they are important.

Large Work Permits

The PHD issues work permits for a variety of activities, including planned developments, land-clearing activities, excavations, and property improvements, all of which might expose contaminated materials. PHD work permits are required before municipal work permits can be approved.

Interior Work Permits

The PHD issues interior work permits, which are required for activities that include ceiling or attic work that might lead to exposure to contaminated dust, work in crawl spaces that contain contaminated soils, installation or removal of insulation, air conditioning or furnace duct work, and excavation of contaminated soil from an interior site.

[18]Institutional controls are actions, such as legal controls, that help minimize the potential for human exposure to contamination by ensuring appropriate land or resource use.

Inspections

The PHD also carries out inspections of work performed under interior or large work projects. Inspection of approvals and reasons for disapprovals are recorded into a database tracking system.

Collectively, this institutional control program, with its "cradle-to-grave" approach, has outstanding characteristics and components that have been designed to work synergistically. The approach gives the PHD the capability to provide incentives and enforcement to commercial and residential activities that potentially might lead to recontamination and human exposure to hazardous materials. Prolonged funding of this program will be a critical component of the long-term success of any remedial efforts.

ADHERENCE OF THE PROPOSED ACTIONS TO SUPERFUND GUIDANCE

Summary of the Guidance

The Coeur d'Alene River basin was designated as a Superfund site and listed on the National Priorities List in 1983; thus, all assessments and decisions made pertaining to the site fall under the authority of Superfund. HHRA is a key part of Superfund site cleanup. Baseline risk to humans under the status quo at the site, as well as under potential remedial actions, is estimated in order to establish remedies that will protect public health in the present and into the future. Risk assessments constitute one source of information that enters the remedial decision-making process, also known as risk management. They identify how much cleanup is desirable, and then a feasibility study is conducted to assess the likely effectiveness and cost of alternative methods for reducing these risks. The agency selects a preferred alternative on the basis of these assessments, and then, after public comment, formalizes its final risk-management decision in the record of decision (ROD). These processes are described extensively in Chapter 8 of this report.

EPA Superfund risk assessments and the level of protectiveness conferred by decisions are characterized by the following (in keeping with federal guidance). Decisions assume "reasonable maximum exposures," rather than worst-case bounds on exposure. Site-specific information forms the basis for assessments where available; however, it is necessary to rely on default assumptions about values for which data are scarce or nonexistent.

Adherence to the Guidance

Regarding human health protection and compliance with ARARs, the HHRA for the Coeur d'Alene River basin generally satisfies the guidance laid out under Superfund.

1. Baseline human health risks attributable to lead and arsenic were adequately established in the HHRA for the Coeur d'Alene River basin, including both waste site sources and other sources.

2. ARARs and other factors to be considered (TBCs) were considered in establishing this health risk. There are no ARARs relating to BLLs or the use of the IEUBK; however, the following were identified as TBC:

   a. BLLs were compared with CDC criteria (specifically, a blood lead level of 10 μg/dL) in making this assessment.

   b. The IEUBK model was used to predict BLLs as is required by Superfund Guidance.

   c. The results of the IEUBK model indicated that for young children living in the basin, there was often a greater than 5% likelihood of their BLLs exceeding the CDC criterion.

Although generally satisfying Superfund guidelines, available site-specific information about subsistence lifestyle exposures, such as consumption of the water potato by Coeur d'Alene tribe members was not adequately addressed. Further, exposures from sources outside the residential environment, such as during recreational swimming, during water sports, and from consumption of local produce, were not fully addressed in the assessment. The existence of additional routes of exposure may account for the finding of higher than predicted BLLs in children in the lower Coeur d'Alene River basin.

CONCLUSIONS AND RECOMMENDATIONS

This committee was charged with examining the assessment and apportionment of risks to humans from multiple contaminant exposures related to waste site sources as well as other sources (for example, lead exposure via soil and house paint dust). Other relevant components of the charge included the following: "What techniques should be used to identify contaminants of concern and estimate the human health risks attributable to waste-site sources? In this case, were risks attributable to sources other than mining and smelting activities adequately analyzed?"

The committee has reached several conclusions and recommendations.

Conclusion 1

Human health risk estimates generated for the basin were developed following EPA guidance. Intakes of lead to which current and future populations of children might be exposed were estimated within a reasonable degree of uncertainty. Consequently, the HHRA is correct in concluding that environmental lead exposure poses elevated risk to the health of some Coeur d'Alene River basin residents.

Conclusion 2

EPA followed guidance for determining human health risk from exposure to metals. Arsenic-related excess cancer risks potentially exceed one in a million throughout the Coeur d'Alene River basin. One subpopulation had estimated arsenic-related excess cancer risk exceeding 1 in 10,000. Following EPA guidance, risk estimates for metals other than arsenic and lead (antimony, cadmium, iron, manganese, mercury, and zinc) considered individually were sufficiently low to be excluded from subsequent analysis.

Use of risk estimates derived from modeling techniques is appropriate in the absence of human data. However, given the magnitude and costs of the remedial activities driven by these model-based risk estimates, the availability of biological indicators of actual human exposure to arsenic would substantially strengthen the justification for arsenic remediation.

Recommendation

The risks of arsenic in the Coeur d'Alene River basin were determined by estimating human exposures based on arsenic concentrations in environmental samples. The committee recommends that EPA continue to support research on biomarkers of human arsenic exposure as these could strengthen exposure evaluations in future HHRAs.

Conclusion 3

EPA's analyses consider aggregate risks from multiple contaminant exposures in a manner consistent with current risk assessment practices.

The agency has also analyzed how the risks of elevated BLLs are associated with exposures from waste materials and leaded paint to a greater extent than is normally done for such a site. Currently accepted risk assessment methods do not include procedures for such apportionment of risks, and EPA has not attempted such a quantitative apportionment. However, their analyses do provide support for the conclusion that lead associated with mining wastes is a significant source of increased BLLs, although lead paint is also a significant source for children likely to be exposed to that source.

Conclusion 4

There are logical reasons to believe that yard remediations decrease exposure to lead, but the scientific evidence supporting substantial beneficial effects is currently weak. Similarly, there is suggestive evidence of efficacy within the Bunker Hill box and river basin. Thus, the strategy for yard remediation is supportable. However, the long-term effectiveness of this remedy in the Coeur d'Alene River basin is questionable because of the possibility, even likelihood, of recontamination.

Recommendation

Long-term support of institutional controls programs should be provided to avoid undue human health risks from recontamination. Moreover, an evaluation of the efficacy of yard remediation should be supported by ongoing environmental and blood lead monitoring efforts.

Conclusion 5

Universal blood lead screening of children aged 1-4 years is indicated for this community given the prevalence of high levels of environmental lead. The current practice of annual fixed-site screening is suboptimal and produces results with too much potential for selection bias to evaluate public health intervention strategies used in the basin.

Shifting the design from a fixed site to a more widespread screening program utilizing the local health care community likely would increase participation. This type of screening program would provide a participant population that is less likely to be biased. Such a practice could be timed to coincide with other medically indicated health care screening tests conducted by primary care physicians. For example, screening for iron deficiency anemia commonly is conducted for children 1-5 years of age by performing a complete blood count. Blood lead screening could be timed to coincide with this blood draw, thereby minimizing inconvenience to the family and child. Linking the screening program to pediatric well-child visits likely will increase participation, provide built-in follow up for children with elevated BLLs, and be more convenient for families. These health surveillance activities could be conducted or sponsored by local, state, or federal (for example, ATSDR) entities.

Recommendation

The committee recommends that annual blood lead screening of all children aged 1-4 years living in the basin be initiated in conjunction with local health care providers. Results should be used to evaluate the efficacy of the environmental interventions.

Conclusion 6

American Indians who practice traditional lifestyles likely would have higher risks than other residents of the Coeur d'Alene River basin. The contamination itself likely interferes with the ability of tribal members to live subsistence lifestyles.

The committee agrees with relevant statements in the HHRA—for example, that "it is clear that a subsistence-based lifestyle requires environmental lead levels orders of magnitude lower than those measured throughout the floodplain of the Coeur d'Alene River," and the conclusion that "Estimated lead intake rates for these scenarios are too high to predict BLLs with confidence. Predictions for BLLs associated with subsistence activities . . . would significantly exceed all health criteria for children or adults."

Conclusion 7

There is strong scientific evidence that living in or near a Superfund site is associated with increased psychological stress. Chronic psychological stress may have health effects in addition to those related to chemical exposures.

Recommendation

Health interventions that address chronic stress may have significant community benefits. These should be implemented before, or concurrent with, cleanup efforts.

Conclusion 8

Children of aged 1-4 years are the group at highest risk for lead exposure. The committee found it inappropriate that the HHRA presented aggregate data on childhood lead screening for children aged 0-9 years of age.

Children less than 1 year old are at very low risk for lead poisoning because of their relative lack of mobility. Likewise, hand-to-mouth activity falls dramatically at about 4 years of age. Children 5-9 years of age are less likely to have elevated lead levels. Although in many cases the data in the HHRA were further stratified to 0-5 years and 6-9 years, there was an inexplicable tendency to lump these age groups together. We strongly discourage such a practice because it is misleading and tends to underestimate the risk among the correct target group.

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6

Human Health Risk Assessment: Lead Exposure and Uptake—Use of the IEUBK Model

MODEL DEVELOPMENT BACKGROUND

Childhood Lead Exposure and Model Development Needs

Lead exhibits a broad range of toxic effects on animal systems, organs, and cellular biochemical and metabolic processes. A National Research Council report (NRC 1993) titled *Measuring Lead Exposure in Infants, Children, and Other Sensitive Populations* concluded that "lead causes nonspecific, decremental loss of tissue and organ function, with no important pathognomonic manifestations of toxicity." Furthermore, exposure to lead occurs by multiple pathways and routes. Because many environmental reservoirs are contaminated with lead, it is seldom possible to identify a sole significant source of lead exposure.

A primary human exposure pathway to lead is through soil and dust, which children are assumed to incidentally or deliberately ingest. Empirical evidence for this assumption comes from reports of excess amounts of soil tracer elements, especially silicon and aluminum, in the feces of children (Wong et al. 1988; Calabrese et al. 1989; Davis et al. 1990). However, because of the inherent difficulties associated with sampling feces from many children over long periods, available data are limited. As a consequence, actual rates of soil ingestion are somewhat uncertain. Quantitative evidence of hand-to-mouth activity in children has been produced by videography (Zartarian et al. 1997; Reed et al. 1999; Freeman et al. 2001). It is also well established that some fraction of the lead found in soils is absorbable in mammalian gastrointestinal tracts (Casteel et al. 1996a-d, 1997a,b, 1998a-e). Studies generally are consistent in demonstrating that a nonnegligible fraction of lead in soil can be absorbed but that the efficiency of absorption depends on multiple factors including chemical speciation of lead, other dietary components, and particle size of soil ingested. Typically paint-derived lead is relatively available for absorption, whereas lead associated with sulfide minerals is relatively unavailable.

Under the environmental health paradigm, preventing injury is the first choice (see Box 6-1). As discussed in Chapter 5, the primary threat presented by lead relates to its ability to cause developmental deficits in children. Although chelation therapy can be applied to reduce body burdens of lead, available information suggests that chelation is not effective in restoring neurological function (Rogan et al. 2001). Hence a "monitor and react" strategy, even if conducted well, cannot prevent injury. The primary prevention strategy (Campbell and Osterhoudt 2000; Rosen and Mushak 2001) is widely recognized as the only truly effective method for eliminating pediatric lead poisoning; this requires a degree of predictive capability for both risk assessment and risk management.

The U.S. Environmental Protection Agency (EPA) has adopted a strategy that entails modeling lead exposure rather than biomonitoring as the first line of defense. Existing epidemiological evidence for health effects of lead exposure is anchored to BLLs rather than to dose rates. The relationship between dose and blood level is complicated by the fact that lead is stored in bone. This entails a greater level of modeling sophistication than the standard risk assessment guidance for Superfund (RAGS) paradigm.

A primary difference between lead risk assessment and cancer and noncancer risk assessment for other chemicals or compounds is that BLLs can be readily measured in individuals and used to "ground-truth" risk calculations. BLLs provide an integrated picture of lead exposure over the preceding months to years, depending on age and other characteristics of

BOX 6-1 Preventing Lead Exposure

Children with access to lead-contaminated soils are likely to be exposed to that lead. To establish levels of lead contamination that would not be expected to present unacceptable or unavoidable risk, it is necessary to define the relationship between magnitude of exposure and level of soil contamination.

Children exposed to lead who develop elevated blood lead levels (BLLs) may have already been irreversibly damaged by the time they have been identified in screening programs. A primary prevention strategy requires the predictive capability of models for exposure risk assessment and management activities.

exposure. In addition, a large body of research exists linking levels of lead in blood to various health effects. As a result, the toxicity and risk characterization steps of a typical risk assessment, as described in the previous chapter, are combined in lead risk assessment into a prediction of BLLs arising from associated lead exposures. Whether risk is deemed acceptable or unacceptable is assessed by comparing the predicted BLLs with target BLLs established by the Centers for Disease Control and Prevention (CDC 1991) and adopted by EPA.

EPA uses two predictive blood lead models for risk assessment purposes: the IEUBK model for children up to the age of 7 years (84 months) and the adult lead model for adolescents and adults. In this chapter, we discuss only the integrated exposure uptake biokinetic (IEUBK) model because children are the most susceptible population and residential soil lead cleanup levels generally are set on the basis of childhood lead risk.

Predictive Blood Lead Models

Lead exhibits a broad range of toxic mechanisms across a variety of target organ systems, and because it has multimedia exposure pathways, the overall dose-response relationships for lead are more complex than those of some other toxic agents. This argues for both biokinetic and pharmacokinetic methods of study to elucidate the concentration and rates of change of lead in various body reservoirs. Mathematical models are particularly useful in this regard because the impacts of lead exposure need to be established on a population-wide basis (NRC 1993). Thus, a variety of predictive blood lead models have evolved for use in lead exposure risk assessment and risk management activities.

Two kinds of model development approaches can be used for predicting blood lead values in response to environmental exposure factors. Slope factor models propose a simple linear relationship between BLL and the uptake or intake of lead from environmental media (air, water, food, soil, dust). If uptake is modeled, in contrast to lead intake, the models are sometimes referred to as biokinetic slope factor models. Examples include those developed by Carlisle and Wade (1992), Bowers et al. (1994), Stern (1994, 1996), the Ontario Ministry of Energy and Environment (OMOEE) (1994), and the Agency for Toxic Substances and Disease Registry (ATSDR 1999). The comparative functioning of several of these models and the multicompartment models described below are detailed in a review of adult lead models examined by the technical review workgroup for lead (TRW) (EPA 2001a).

Multicompartment predictive blood lead models simulate the movement and concentration of lead in several interconnected tissue compartments with blood or extracellular fluid (plasma) serving as the exchange medium. Rabinowitz (1998) reviewed the early development of this approach, illustrating the usefulness of such models after the experimental application of radioactive tracers showed the relatively short half-life of lead in blood (about 1 month) compared with a 15- to 20-year residence time in skeletal tissue. Models of this type have been developed by Rabinowitz et al. (1976), Marcus (1985), Bert et al. (1989), O'Flaherty (1993), Leggett (1993), and EPA (1994a,b). A simple depiction of a multicompartment model, similar to that of Rabinowitz et al. (1976) is shown in Figure 6-1. Biokinetic and pharmacokinetic models relate exposure dose to the lead concentration in various target tissues; they represent the mathematics of the time course of absorption, distribution, metabolism, and excretion (ADME) of the substance being followed. Biological, physiological, and physicochemical factors all influence the rate and extent of ADME.

Several mathematical approaches underlie the pharmacobiokinetic (PBK) model structures: in diffusion-limited models, such as the IEUBK model, rates of change of lead concentration in the various compartments are defined by the rates of transfer across compartment boundaries. The time parameter is represented in the diffusion rate constants. Lead transfers are typically assumed to follow first-order kinetics; exchanges are repre-

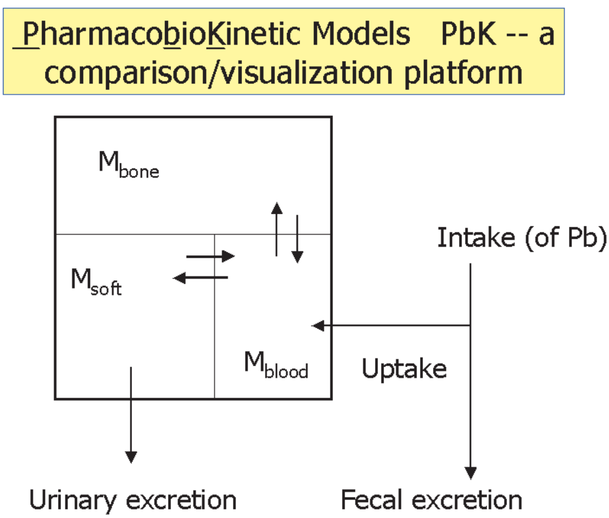

FIGURE 6-1 Simple model framework illustrating compartments and pathways of exchange for a pharmacobiokinetic model of lead in the human system. SOURCE: Rabinowitz et al. 1976. Reprinted with permission from the American Society for Clinical Investigation.

sented by first-order rate constants. However, such "constants," may take on age-specific values, an important characteristic of PBK models applied to children's lead exposure.

An alternative (O'Flaherty 1993) is a flow-limited model; this approach quantifies the mass transfer of the extracellular fluid to the tissue compartments of the model. Here, the time variable is incorporated in the flow rates of fluid between body compartments. A central feature of the O'Flaherty model is its emulation of bone growth and resorption as a mechanism for controlling plasma lead levels. "Lead is assumed to instantaneously partition between plasma and soft tissues and to achieve an equilibrium (that is, partition coefficient). Therefore the rates of change of lead masses in soft tissues are limited by the rates of delivery of lead to the tissues, given by the product of the plasma concentration of lead and the rate of plasma flow to the tissue, rather than by limiting steps in the transfer of lead across tissue boundaries" (EPA 2001a).

Predictive blood lead models generally distinguish between the intake of lead during exposure and its uptake by the body. The fraction of lead that is absorbed and enters the blood by whatever portal-of-entry compared with the total amount of lead acquired is termed the bioavailability. In the simple illustration of a PBK model (Figure 6-1), lead intake is represented as ingestion. Subsequently, a fraction of the lead present in the gastrointestinal tract is taken up into the bloodstream—a process that may vary with the age of the individual; the person's health, physiological, and/or nutritional status; and whether ingestion occurred with or without food. Bioavailability of inhaled lead may differ from that of ingested lead. By either route of entry, biokinetic or pharmacokinetic models incorporate a variable for the fraction of total lead that is actually absorbed and define it as the uptake of lead. In the 1999 EPA Guidance Document *IEUBK Model Bioavailability Variable* (EPA 1999), the following terms are defined and adopted for use in this chapter:

- *Absolute bioavailability* is the amount of a substance entering the blood via a particular route of exposure (for example, gastrointestinal) divided by the total amount administered (for example, soil lead ingested).
- *Relative bioavailability* is indexed by measuring the bioavailability of a particular substance relative to the bioavailability of a standardized reference material, such as soluble lead acetate.

Evolution of EPA's IEUBK Model

Federal agencies documented and summarized extensive research on the toxicological impact of lead exposure (McMichael et al. 1986; Bellinger et al. 1989; Bornschein et al. 1989; Needleman et al. 1990; and others) before development of the IEUBK model (ATSDR 1988; EPA 1989, 1990). As pointed out by Choudhury et al. (1992), epidemiological and behavioral research had not identified a threshold or no-observed-adverse-effect level (NOAEL) that could be used to establish a reference dose for lead—that is, a value that could be used for risk assessment in the manner discussed in Chapter 5 for other metals of concern. Empirical studies showed relationships between children's BLL and the concentration of lead in a variety of media (Barltrop et al. 1975; Yankel et al. 1977; Angle and McIntire 1982; Stark et al. 1982). These slope factor (SF) models were the foundation for the current modeling structure. The impetus for further development of such tools was to quantify the impact of lead in setting National Ambient Air Quality Standards (NAAQS) (EPA 1986) and National Primary Drinking Water Regulations. However, substantial limitations of SF models were identified, owing to the individual variability of children with respect to factors including ingestion rates and activity patterns, the influence of physiological states and nutritional factors on lead absorption, and physicochemical differences in the distribution and occurrence of lead between sites of exposure. Thus, biokinetic models were developed as an alternative approach, emphasizing the need for a predictive capability in order to implement primary prevention strategies.

In 1985, the EPA Office of Air Quality Planning and Standards (OAQPS) began a computer-simulation-model development based on the biokinetic model of Kneip et al. (1983) and Harley and Kneip (1985). These studies brought together a critical mass of biokinetic parameter information. The exposure component for model operation was developed by OAQPS. A 1989 OAQPS staff paper reviewing the NAAQS for lead contained results of model applications to point sources of air lead. Shortly thereafter, the TRW for lead was formed to advise on cleanup at Superfund and Resource Conservation and Recovery Act of 1976 (RCRA) sites; they modified the model for lead risk assessment, calling it the uptake biokinetic (UBK) model. The TRW recognized the desirability of a frequency distribution for BLLs of a population and used a geometric standard deviation based on NHANES II (1986) data.

Initial calibration and validation exercises for the developing model were based on the 1983 Helena, Montana, primary lead smelter study, as cited in the 1989 *Review of the National Ambient Air Quality Standards for Lead: Exposure Analysis Methodology and Validation* (EPA 1989). Further validation of the UBK model was reported by DeRosa et al. (1991) and by Bornschein et al. (1990); whereas the latter study used the Midvale, Utah, data set, the data source for the DeRosa study was not identified. Choudhury et al. (1992) indicated that, for the Midvale exposure data, the UBK default conditions provided an acceptable agreement between observed and calculated values for measures of central tendency but that the upper end of the distribution was not well predicted. Agreement between predictions and empirical results for Midvale data improved when an age-dependent dust/soil ingestion rate was used. The latter are the same as the current default values for the model. Subsequent to the release of the IEUBK model executable in 1994, additional evaluation of the model was conducted by EPA, including an independent validation and verification of the source code (Zaragoza and Hogan 1998) and an evaluation of predictions of BLLs in children for whom environmental levels and BLLs were measured (Hogan et al. 1998).

The EPA Clean Air Science Advisory Committee (CASAC) of the Science Advisory Board provided initial review and approval of model structure and functioning in 1989. In 1990, CASAC concluded that the model provided "an adequate scientific basis for EPA to retain or revise primary and secondary NAAQS for airborne lead." In 1992, the EPA Science Advisory Board reviewed and reported on the UBK model for lead. Suggested modifications also derived from comments on the draft 1992 Office of Solid Waste and Emergency Response (OSWER) Soil Lead Directive proposed using the UBK model in support of lead exposure risk assessments. Since 1991, the TRW has been responsible for model development. Modifications have made it suitable for evaluating exposure from all media, and the product became a stand-alone PC software package. The biokinetic model approach was deemed suitable for assessing total lead exposures and for developing cleanup levels at residential Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA)/RCRA sites. With refinements resulting from comments on early model versions, the model was released in executable form only in 1994 as the IEUBK model.

DESCRIPTION OF THE IEUBK MODEL

Model Structure and Operation

This section presents an overview of the model's structure and operation. A more detailed summary of the IEUBK model can be found in the work of White et al. (1998). The compartmental structure of the IEUBK model is slightly more complex than that shown previously for the simple PBK example and is illustrated in Figure 6-2 (EPA 1994a). Despite significantly more structure in this version of a multicompartment model, lead accumulation in various model reservoirs still has, as a fundamental control, the time-dependent difference between the uptake and the excretion pathways. When concentrations of lead in environmental media are specified, the model calculates a point estimate of a child's blood lead values over the age range of 0-84 months.

The IEUBK model is defined operationally by EPA's computer program(s). These programs have been publicly available in object code form

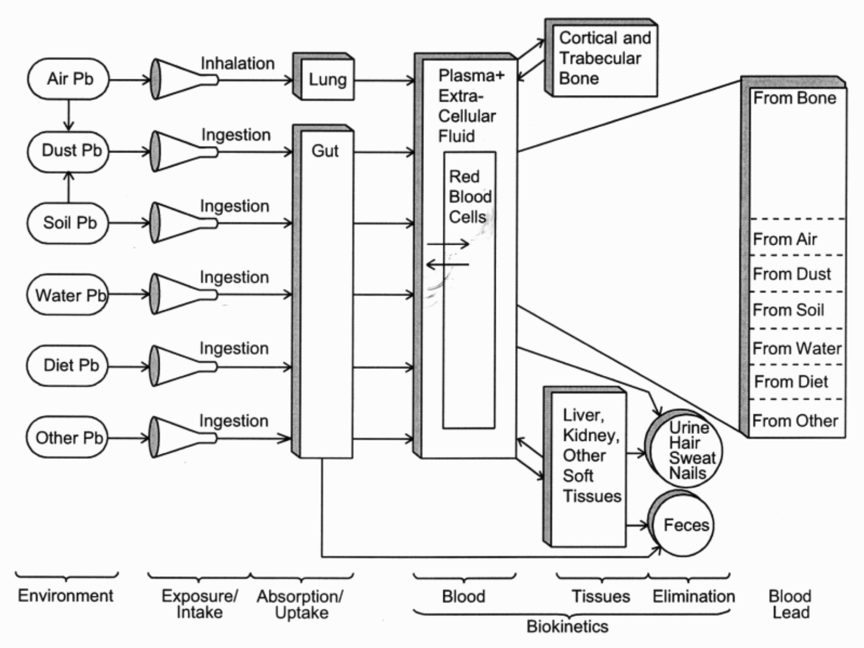

FIGURE 6-2 Compartments and functional arrangement of the IEUBK components for prediction of children's blood lead values. SOURCE: EPA 1994a.

(that is, in a form suitable for running on a computer) since 1994 and have been through multiple versions. The latest version is available from EPA's Superfund Web site (EPA 2004a),[1] and that site also contains technical documentation on the model. The source code for the IEUBK model is not linked at this or any other Web site and has never been readily available in this way; rather, it has always been necessary to specifically request it from EPA.

The primary technical source describing the model is the Technical Support Document (TSD) (EPA 1994b). Although this is explicitly for version 0.99d of the model, the model specification has not changed in any essential way in the 10 years since then. Examination of the computer code shows that the biokinetic portion of the code is identical in all relevant (and some irrelevant) respects. Notably, the current code contains the same

[1]Surprisingly, there appears to be no link to the IEUBK model information from EPA's "lead in paint, dust, and soil" (EPA 2005).

errors[2] and redundancies, as described below, that were present in the original version.

The essential parts of the IEUBK model[3] can be partitioned into four components: an intake component, an uptake component, a biokinetic component, and a probability component. These four components are strictly independent of one another, each feeding into the following one with no feedback.

Intake Component

The intake component of the model collects information on exposures to lead-contaminated media (air, dust, soil, food, water) and sums the quantities of lead that enter the body from each exposure medium. Within each medium, the intake of lead is obtained as the product of an average concentration or mass fraction[4] of lead in the medium and the average intake rate of that medium. For example, the intake of lead from soil is the product of the soil lead concentration (milligrams [mg] of lead per kilogram [kg] of soil) and the ingestion rate for soils (mg of soil ingested per day) to provide an intake rate for lead from soil.

The exposure module contains default values for environmental concentrations and ingestion rates should no site-specific information be available. Similarly, default values for absolute bioavailability are programmed for model operation but may be altered by the user. For soil and dust ingestion, default bioavailability values of 30% are assigned. That value is derived from an absolute bioavailability for soluble lead in water and diet constituents of 50%, together with a 60% relative bioavailability for soil and dust lead compared with water (EPA 1999). Table 6-1 summarizes the IEUBK default values.

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[2]As described in the subsection "Incorrect Model Specifications" below, the committee considers the computer code for the biokinetic part of the model to be in error if it does not solve, in the limit of small time step, the set of algebraic and differential equations and boundary conditions specified in the TSD (EPA 1994b) (which is taken to define the model). The committee has not examined other parts of the code and does not certify that even the examined code is free of other errors. The documentation is considered to be in error if it specifies physical impossibilities or fails to define some element of the model. These definitions are imposed because the committee believes that the model specification should be the standard of comparison (for observations, other implementations, and other models), rather than the computer code itself.

[3]The user interface is not considered here because that does not comprise an essential component of the model. The principal changes in the model over the last 10 years have been in the user interface and in the default values that are automatically present in that user interface.

[4]We do not subsequently distinguish between concentration and mass fraction, using the first term in the usual colloquial sense to represent both.

TABLE 6-1 Default Values for the EPA IEUBK Model

	0-1 y	1-2 y	2-3 y	3-4 y	4-5 y	5-6 y	6-7 y
Ventilation rate, m^3 per day	2	3	5	5	5	7	7
Diet intake, μg lead per day	5.53	5.78	6.49	6.24	6.01	6.34	7.00
Water intake, L per day	0.20	0.50	0.52	0.53	0.55	0.58	0.59
Soil/dust ingestion, total mg per day	85	135	135	135	100	90	85

Water = 4 μg of lead per L, air = 0.1 μg of lead per m^3, maternal blood lead = 2.5 μg of lead per dL.
Indoor air lead concentration = 30% of outdoor concentration.
Soil lead concentration = dust lead concentration = 200 μg lead per gram of soil/dust.
Soil = 45% of total ingestion, dust = 55% of total ingestion.
Diet and water bioavailability = 50%, soil and dust bioavailability = 30%.
NOTE: Bioavailability is not constant. The values cited apply for low lead intake rates. Absolute bioavailability decreases as lead intake increases and uptake saturation is reached.
SOURCE: EPA 1994b.

Uptake Component

The uptake part of the model contains two parts: one deals with absorption in the lung, the other with absorption in the gut. Absorption in the lung is treated as linear; some fixed fraction of the inhaled quantity of lead is assumed to be absorbed. Absorption in the gut is assumed to consist of two fractions: a linear, nonsaturable component and a nonlinear, saturable component. Details of the gastrointestical tract uptake specifications are illustrated in Box 6-2 and Figure 6-3. For each ingested medium (labeled

BOX 6-2 Lead Uptake Formulations for the IEUBK Model

Description of Model Formulation for Uptake of Lead from the Gastrointestinal Tract

Figure 6-3 illustrates the two types of uptake from the gut. Suppose the total lead ingestion intake in medium *k* is Z_k. Then defining

$$Z = \sum_k \alpha_k Z_k \qquad (0\text{-}1)$$

the linearly absorbed component U_l and nonlinearly absorbed component U_n are assumed to be given by

$$U_t = pZ \qquad (0\text{-}2)$$
$$U_n = (1 - p)Z/(1 + Z/Z_{sat})$$

with the total gut absorption given by the sum $U_l + U_n$. The value p has default value 0.2, and Z_{sat} is estimated by default as 100 μg/day at 24 months, and is scaled with body weight for other ages.

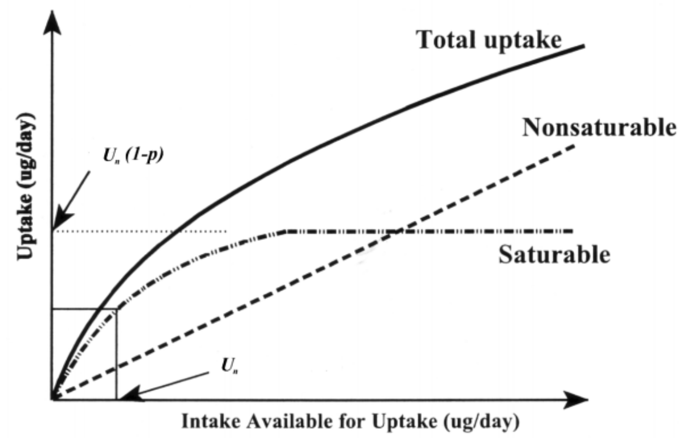

FIGURE 6-3 Mathematical treatment of the lead absorption in the IEUBK model. SOURCE: EPA 1994b.

here by index k), there is assumed to be a fixed fraction α_k (the bioavailability of lead from that medium) that could be absorbed at a low exposure level. The user can override the program default values and specify separate bioavailability values for each exposure medium.

Biokinetic Component

The biokinetic component of the IEUBK model is a compartment model with seven compartments plus three excretion-only pseudocompartments (URINE, FECES, and SNH) as named and numbered in Table 6-2.

The plasma-ECF compartment exchanges lead with all the other compartments, and excretion occurs only to the urine pseudocompartment. The only other connectivity between compartments and pseudocompartments is the excretion of lead from liver to feces and from soft tissues to skin, nails, and hair. The only connection between the uptake and biokinetic components of the model occurs through uptake in the lung and gut. These uptakes are assumed to be independent of the internal state of the body incorporated in the biokinetic component. In theory, there is some dependence—for example because of excretion of lead into the gut (from where it could be re-absorbed) in bile; however, the effect of any such dependencies is expected to be small.

Equations describing the transfer of lead between these compartments (equations of motion) are presented in Box 6-3. Transfer between these compartments is described by the time constants F_i and T_i, which denote uptake to plasma or transfer from plasma, respectively. Similarly, A_i is the

TABLE 6-2 Compartments[a] of the IEUBK Model

Compartment Name	Number	Description
PLECF	0	Plasma-ECF (extracellular fluid)
RBC	1	Erythrocytes
TRAB	2	Trabecular bone
CORT	3	Cortical bone
KIDNEY	4	Kidney
LIVER	5	Liver
SOFT	6	Other soft tissue
URINE	7	Urine
FECES	8	Feces
SNH	9	Skin, nails, and hair

[a]For the compartments, these names are abstracted from the nomenclature used in the documentation and source code of the IEUBK model (EPA 1994b). The compartment numbers are committee constructs. The equations of motion are more compact using this subscript notation.

time constant for the transfer of lead from a compartment to the plasma-ECF compartment or any pseudocompartment. These constants for the different compartments vary with age, and some depend on tissue concentration or are written in such a way as to be related to tissue concentration ratios in order to use experimental data on such ratios. For instance, lead excretion rates vary substantially during a child's early life (O'Flaherty 1993); whereas less than 70% of daily lead uptake may be excreted at age 6 months, more than 90% of daily uptake is excreted at age 24 months. Values of the parameters controlling the transfer processes play a critical role in the accuracy of model predictions. Despite an increase in model complexity (compared with the model structure shown in Figure 6-1), lead accumulation in the IEUBK model compartments is still controlled by the time-dependent difference between uptake and excretion pathways.

The tissue masses (or volumes, for red blood cells, plasma [extracellular fluid], and blood) at each age are defined by mathematical functions that have been chosen to give a good fit to experimental data on tissue masses (or volumes) as a function of age. The masses M_i are supposed to be initialized at age zero to values that give a blood lead concentration of 0.85 times the blood lead concentration of the mother. Equations 0-3 (see Box 6-3) are then integrated over age to obtain the masses of lead in each compartment at any age. Lead concentrations (or mass fractions) in each compartment at each age are obtained by dividing lead mass by tissue volume (or mass) at that age. In particular, blood concentration is obtained by summing the mass in the red blood cells and the mass in the fraction of the plasma-ECF that is in the blood and dividing by blood volume. Finally, the blood concentration value output by the current model user interface is an average over various time periods (for example, the first 6 months of age, 6-12 months, and annual averages to age 7).

BOX 6-3 Equations of Motion for the Transfer of Lead Between IEUBK Model Compartments

The equations of motion for the mass of lead in each of the compartments are as follows:

$$\frac{dM_0}{dt} = I - \sum_{i=1}^{7} \frac{M_0}{T_i} + \sum_{i=1}^{6} \frac{M_i}{F_i}$$

$$\frac{dM_i}{dt} = \frac{M_0}{T_i} - \frac{M_i}{A_i} \qquad 1 \le i \le 6$$

$$\frac{dM_7}{dt} = \frac{M_0}{T_7} \tag{0-3}$$

$$\frac{dM_i}{dt} = \frac{M_{i-3}}{T_i} \qquad i = 8, 9$$

- i compartment number (0-9), from Table 6-2,
- t age,
- I total lead intake rate (mass per unit time) into the plasma-ECF compartment (from the gut and lung),
- M_i for $0 \le i \le 6$ the mass of lead in compartment I; for $7 \le i \le 9$ the cumulative mass of lead excreted to the pseudo-compartment,
- T_i for $1 \le i \le 7$ a time constant for transfer of lead from the plasma-ECF compartment to compartment i,
- T_8 time constant for transfer of lead from the liver to feces,
- T_9 time constant for transfer of lead from soft tissue to skin, nails, and hair,
- Fi for $1 \le i \le 6$, a time constant for transfer of lead from compartment i to the plasma-ECF compartment, and
- A_i for $1 \le i \le 6$, a time constant for transfer of lead from compartment i to the plasma-ECF compartment or any pseudo-compartment.

Only the liver and soft tissue compartments excrete lead (to feces and to skin/hair/nails, respectively; excretion in urine is treated as a transfer from the plasma-ECF compartment), so for compartments 1 through 4 the only exchange is with the plasma-ECF, leading to:

$$A_i = F_i \qquad 1 \le i \le 4, \tag{0-4}$$

and for compartments 5 and 6 it is assumed that

$$1/A_i = 1/F_i + 1/T_{i+3} \qquad 5 \le i \le 6. \tag{0-5}$$

Probabilistic Component

The fourth component of the IEUBK is the probabilistic component. The deterministic estimates of blood concentrations obtained as just described are assumed to represent the median values for a lognormal distribution of values that would occur in a population that was subject to fixed lead concentrations in the input media (soil, dust, air, water) equal to those input to the model. The standard deviation (or geometric standard deviation [GSD]) of the lognormal distribution was derived based on observations of exposed populations of children. EPA (1994a) stated that the default value of the GSD is based on analyses at Midvale, Utah; Baltimore, Maryland; and Butte, Montana. The analyses are not available for review.

Issues Associated with Using the Model

The statement of task directed the committee to address whether the design, input data, and assumptions of the IEUBK model were consistent with current scientific understanding. Issues associated with IEUBK model predictions of blood lead values can be grouped into three categories: (1) the computer code implementing the mathematics of model computations, (2) the default exposure values related to ingestion rates and to bioavailability of lead, and (3) extension of a deterministic, point value for blood lead concentration to a probability distribution function for a population. Although the model has been subjected to several evaluation and critique efforts, as well as to EPA Science Advisory Board reviews, no comprehensive published account of the peer review content is available. Therefore, a variety of comments on these several categories of uncertainty seem warranted.

Incorrect Model Specifications

With regard to the first category, the TSD has contradictory claims as to the numerical method used to integrate the equations (EPA 1994b). On page 45 of the TSD, the backward Euler scheme is discussed, whereas on page A-14 there is the claim that "These differential equations are translated into difference equations employing the forward Euler solution in the series B-6.5a to B-6.5i, then to the solution algorithm for differential equations using the backward Euler method, or alternate difference equation scheme." It is not clear what this means, or whether any consistent approach was used. The equations given in the TSD agree with a backward Euler scheme except for equations B-8c and B-8d, but the difference for those equations is second order in the time step, the same as the error in any such first-order scheme.

Further, the scheme indicated in the TSD is not actually carried out in the computer program. Rather, it evaluates all age-dependent functions used in the coefficients of the differential equations (in defining the time constants) at monthly intervals and assumes that those values are constant throughout each month. The integration time step (about one-sixth of a day) is then applied to these functions that remain constant for a month at a time. The choice of a first-order integration method must also be questioned, particularly when the time step is left to the user. A better approach would be to use one of the many standard numerical integrators that allow specification of the allowable error and require the error to be trivially small. Careful review of the model implementation code reveals a number of additional inconsistencies or minor errors in the formulation of the equations documented in the TSD. These are detailed in Appendix C. Combined with the points enumerated above, however, the cumulative uncertainty in computed results is no more than a few percent. Nevertheless, the documentation should accurately reflect the programming.

Uncertainty in Key Default Parameters

Soil/dust ingestion rates and lead bioavailability are two key variables the user may specify in making blood lead value predictions with the IEUBK model. Its default age-specific ingestion rates have remained unchanged since before the 1994 release of the model (Choudhury et al. 1992). Large uncertainties exist in measures of the central tendency for these exposure media ingestion rates by children. Binkowitz and Wartenberg (2001), in their review of literature reports on the subject, showed rates between 10 and 1,000 mg per day for children, with a median value of about 100 mg/day. Little consistency has been shown in the methodological approaches used; variations exist in the media being estimated, the time period used in the observations, and the analytical chemistry techniques of the measurements. Lee and Kissel (1995) suggested a slightly narrower range at a factor of 2 and highlighted the importance of studies to refine ingestion rate values.

Lead bioavailability as a function of age is not well characterized, although there is general agreement among many investigators that bioavailability in pediatric populations is generally higher than it is for adult populations (O'Flaherty 1995; Pounds and Leggett 1998). Although the animal studies of Quarterman and Morrison (1978) supported this view, Mahaffey (1998) urged caution in this interpretation from the limited study data that exist. In the model of O'Flaherty (1993, 1995, 1998), bioavailability is estimated in the 50-60% range for children under the age of 2 years, declining to the 10-20% range by age approximately 5 years. The latter values are similar to those for adults (Maddaloni et al. 1998). The IEUBK default values for soil and dust bioavailability are 30% and are constant across age groupings of children (except see footnote *a* in Table 6-1). Uncertainty in ingestion rate and in bioavailability has a strong, direct influence on the model results.

Uncertainty in Projecting Point Estimates into Population Distributions

One of the more contentious issues associated with the predictive capability of the IEUBK model is the choice of a GSD. The IEUBK model is designed to predict one BLL for a given set of exposure conditions, and this BLL is designated as the geometric mean of a population of children who would be exposed to the specified environmental levels. The GSD is then used together with the predicted geometric mean to estimate a range of BLLs that might arise in this population. Contention arises in part because EPA's blood lead target of protecting 95% of such a population at a BLL of 10 μg/dL means that the outcome, either in predicted 95th percentile blood lead or in estimated soil lead cleanup level, is very sensitive to the value of the GSD. EPA materials (EPA 2002) state that the GSD should not be site specific because it represents variability in exposure and behavioral parameters outside of soil and dust lead variability and therefore should not change significantly, at least in large populations, from site to site. Although EPA's *IEUBK Guidance Manual* (1994a) specifies a default value for the GSD and states that it is based on calculations at three sites, material documenting these calculations is not in the public domain and therefore cannot be examined or verified.

Although EPA argues strongly for use of the default GSD value, several EPA risk assessments (EPA 1995 [Sandy], 1998a [Palmerton]; Life Systems, Inc. 1995 [Bingham Creek]) have developed and used alternative values of the GSD, leading to the concept that the GSD may be site specific. In the Vasquez Boulevard and Interstate 70 health risk assessment (EPA 2001b), uncertainty in IEUBK model predictions was examined specifically with regard to dietary lead, soil-ingestion rate, and GSD. The report suggested that the default GSD of 1.6 might be too high for this site. Accurate calculation of a site-specific GSD value is a complex procedure (Griffin et al. 1999) involving significantly more effort than a simple analysis of blood lead results; this perhaps underscores EPA's approach to the use of alternative GSD values in IEUBK applications.[5] However, the apparent disparity between stated policy at the federal level and (some) implementations at the regional level can lead to confusion on the part of risk assessors/managers as well as the general public. The economic consequences associated with an inaccurate GSD used for setting cleanup levels can be substantial and a more objective, scientifically comprehensive policy needs to be articulated. A fully probabilistic version of the IEUBK model, such as was demonstrated at EPA's 1999 workshop[6] (see Box 6-4), would estimate the variability in

[5]EPA states, "Model users should not substitute alternate values for the default GSD without detailed site-specific studies designed to document the difference that would justify changing the default value" (EPA 2002).

[6]This version did not incorporate any variability in the biokinetic portion of the model, although it is unclear whether there is any substantial variation in this component at lead intakes corresponding to blood levels of concern at Superfund sites. It is technically straightforward to incorporate such biokinetic variability, although obtaining experimental data for any but the simplest estimates of its size may be infeasible.

BOX 6-4 EPA IEUBK Workshops

EPA has held three workshops focusing on the development and use of the IEUBK model. These workshops include Lead Model Validation (1996), Modeling Lead Exposure and Bioavailability (1998), and Probabilistic Risk Assessment and Biokinetic Modeling (1999). Publications based on presentations at the first workshop are in a supplement to Environmental Health Perspectives (Vol. 106, Supplement 6, December 1998), including a preface by Grant and others stating that the key outcome of the workshop was the establishment of requirements and procedures for model validation.

Although manuscripts were collected from the presenters at the two subsequent workshops in 1998 and 1999, no proceedings have ever been published. The 1998 workshop focused on exposure parameters and produced general consensus among attendees that regulators and industry scientists should work together to reduce uncertainties in the model to improve the accuracy of BLL predictions. Recommendations formed at the workshop included the need to analyze soil and dust samples in multiple ways to better understand bioavailability, the need to develop an improved methodology for differentiating exposure to soil versus dust, and the need to conduct detailed adult soil-ingestion studies.

The 1999 workshop focused on efforts by several groups, including EPA, in developing a fully probabilistic blood lead prediction model. General consensus among attendees was that a fully probabilistic model would aid in understanding how the variability in exposure affects the range of BLLs. EPA presented early work toward developing an "all ages" model. From all appearances, there has been little to no follow up on the work or recommendations regarding the development of a fully probabilistic blood lead prediction model.

BLLs as a function of the variability in all exposure and environmental parameters and would obviate the need for such an ad hoc approach as tacking on a GSD at the end of the calculation in the current version of the model. A fully probabilistic version of the IEUBK model would also end the debate about the extent to which the GSD may be site specific because it could be estimated mathematically for each site.

Model Performance Assessments

Comparison with Other Model Structures

Part of the committee's statement of task was to address whether alternative tools were appropriately used to assess and interpret the model results. The committee found little evidence in the human health risk assessment (HHRA) or in the record of decision (ROD) for the Coeur d'Alene River basin that alternative tools were used to interpret and assess model results. In the absence of this analysis, we examined the Agency for Toxic

Substances and Disease Registry (ATSDR) OU-3 Public Health Assessment (ATSDR 2004, public comment version) and the Heath Consultation (ATSDR 2000a) that did incorporate an analysis of different methodologies.

The ATSDR (2000a) Health Consultation evaluated lead-exposure risks for children living in the Coeur d'Alene River basin (operable unit 3 [OU-3]) based on the environmental lead sampling carried out at residential locations within the basin as targeted by Field Sampling Plan Addendum 6 (FSPA06) conducted in support of the remedial investigation (URS Greiner, Inc. and CH2M Hill 2001). ATSDR used three screening methodologies to predict exposure risk as displayed by blood lead distributions, assuming the exposure environments sampled to be representative of those occupied by children basin-wide. These included the biokinetic SF model of the OMOEE (1994, 1996), the multiple linear regression SF model of ATSDR (1999), and the multicompartment IEUBK model of EPA.

The results from the ATSDR (2000a) comparison of these models indicated that between 22.5% (ATSDR model) and 79% (OMOEE model) of the basin homes sampled have environmental lead concentrations high enough that children in the 1- to 2-year age group would have lead exposures expected to produce BLLs greater than 10 μg/dL. As employed in the ATSDR Health Consultation (2000a), the IEUBK model predicted an intermediate result; 40% of children[7] would be expected to have blood lead exceeding the CDC guideline. In reviewing this study, the committee recognized that the exposure parameters were not standardized between models in this analysis. To address this shortcoming and make further comparisons between these models, additional analyses were conducted on the FSPA06 data set (see Appendix D). First, results using the model input parameters from the original ATSDR (2000a) study were generated. Then, the results were recalculated after input parameters to the different models were standardized to provide similar exposure regimes. Additionally, the models were run using the input parameters from the "box" model used in OU-1 and OU-3 of the Coeur d'Alene River basin. The comparisons were further extended by including predictions from the physiologically based pharmacokinetic model of O'Flaherty (1993, 1995, 1998). These analyses were conducted on 75 homes from the FSPA 06 data set that had both soil and dust lead measures. Details of the methodology comparison are presented in Appendix D.

[7]An important difference in the results from the comparison of models presented in the ATSDR (2000a) study is that IEUBK model output was apparently generated for children 7-84 months of age, not 1- to 2-year-olds as is presented for the ATSDR and OMOEE models. Further comparisons conducted by the committee (presented below) generate output for children of approximately the same age.

Table 6-3 summarizes the results of the model estimates derived from this work. It presents the percentages of children in the 1- to 2-year age group who would exhibit blood lead values below the CDC (1991) level of concern—10 μg/dL—as predicted by the four models using the seventy-five homes' data as residential environments. Its purpose is to compare model results based on realistic environmental lead-exposure potential. Column 1 shows the recalculated results for the 75 homes' data, utilizing the model parameters originally used in the ATSDR Health Consultation (ATSDR 2000a). Column 2 contains results where the OMOEE model ingestion rates were adjusted to match those of the IEUBK default values, recommended ATSDR regression model uncertainties were applied, and IEUBK predictions were targeted for the 12- to 24-month age class. Column 3 entries were computations based on the Bunker Hill Superfund site box model conditions for the IEUBK model detailed above.

The results indicate that the original computations (column 1) were biased by the high ingestion rates applied to the OMOEE model computations. When column 2 results are compared, the range of predictions is substantially reduced. Here, the IEUBK default model predictions are the most conservative (predict the highest BLLs in children).

As noted earlier, SF models, such as the ATSDR and OMOEE models, have significant limitations in their applicability. Multicompartment models in which exposure and biokinetic parameters can be adjusted for site-specific conditions overcome many of these limitations. Very close agreement is achieved for predictions by the two multicompartment biokinetic models (the IEUBK and O'Flaherty model; see Box 6-5). Although this may be expected owing to the common or similar data sets used in model calibrations, the two models used very different computation strategies. The small differences between the IEUBK and the O'Flaherty model results in column 3 are related to the shapes of the bioresponse curves. The O'Flaherty model predicts blood lead for a 2-year-old that is slightly higher than that predicted by the IEUBK model, but it predicts lower values than the IEUBK model for children ages 3-7 years. When averaged by 12-month age classes, the two models agree within less than 5%.

APPLICATION OF IEUBK TO OU-3 (COEUR D'ALENE RIVER BASIN)

Use of the IEUBK Model in a Regulatory Context

The IEUBK model has two uses. The first is to estimate BLLs arising from site-specific environmental lead levels, taking into consideration any relevant site-specific information such as soil lead bioavailability or altered exposure parameters. If those BLLs are found to be elevated above acceptable levels, the second function of the model is to calculate a soil lead

TABLE 6-3 Blood Lead Values for Children in the 1- to 2-Year-Old Group

Model	Column 1 Original ATSDR Health Consultation Input Parameters, Recalculated for 75 RI/FS Homes (% of individuals with BLLs < 10 μg/dL)	Column 2 Adjusted for IEUBK Default Ingestion Rates and 1-2 Year Age Class (GM [GSD] in μg/dL) (% of individuals with BLLs < 10 μg/dL)	Column 3 Same as Column 2, Except Adjusted to BHSS Box Model Conditions (GM [GSD] in μg/dL) (% of individuals with BLLs < 10 μg/dL)
ATSDR[a]	73% < 10.0 μg/dL	9.79 (1.8) 56% < 10.0 μg/dL	8.90 (1.6)[b] 63% < 10.0 μg/dL
OMOEE[c]	20% < 10.0 μg/dL	9.70 (2.0) 53% < 10.0 μg/dL	5.29 (1.8)[b] 89% < 10.0 μg/dL
O'Flaherty		9.84 (1.5)[d] 56% < 10.0 μg/dL	8.40 (1.5)[e] 71% < 10.0 μg/dL
IEUBK	60% < 10.0 μg/dL	11.9 (1.6)[f] 37% < 10.0 μg/dL	7.93 (1.5)[g] 73% < 10.0 μg/dL

Abbreviations: BLLs, blood lead levels; GM, geometric mean; GSD, geometric standard deviation; RI/FS, remedial investigation/feasibility study.

NOTE: Predictions by ATSDR (1999), OMOEE (1994), O'Flaherty (1998), and IEUBK models used paired soil and dust environmental lead data from 75 RI/FS homes (in FSPA06) (see Appendix D). Models included EPA default lead intake values from diet and inhalation (air), and water lead at 4 μg/L except where higher values were measured.

[a]The ATSDR regression model calculates a maximum blood lead value using an uncertainty of the soil and dust SF. In the Health Consultation, the uncertainty was specified as ±1 standard deviation. In columns 2 and 3 of this table, an uncertainty of ±3 standard deviations is used to correspond with the original ATSDR regression model description.

[b]Soil and dust concentrations were set at 60% of the box model values to compensate for reduction in bioavailability to 18%.

[c]The Ontario Ministry of Energy and Environment (OMOEE) model calculates an intake of concern (IOC), not a blood lead value, but this tabulation can be expressed as a percentage of predicted blood lead levels < 10.0 μg/dL. The (estimated) BLLs assumed two times the IOC is equivalent to 10.0 μg/dL.

[d]Soil and dust ingestion rates are fixed program functions; they peak at about 135 mg/day at age 2 but decline subsequently more rapidly than those of the IEUBK model. The integrated soil plus dust ingestion rate is about 65 mg/day over the interval 0-84 months of age.

[e]Model parameters were adjusted to reflect the 60% soil to 40% dust ingestion ratio and the weighted soil concentrations of the box model.

[f]Batch mode IEUBK runs were specified for age 20 months. This produces a blood lead value equivalent to the normal mode blood lead concentration tabulated for the 1-2 year age class.

[g]Results for IEUBK and O'Flaherty models (column 3) do not have statistically different geometric mean values at the 95% confidence level.

BOX 6-5 Multicompartment Biokinetic Models Compared Well

Under the conditions of this comparison, cleanup levels determined by the two multicompartment models would be the same. This supports the veracity of IEUBK biokinetic computations as used in this case. It does not, however, provide a validation of the exposure/bioavailability assumptions used in the operation of these models.

cleanup level that will be adequately protective of young children in the community, such that BLLs will not exceed the established acceptable levels.

Calculation of the soil lead cleanup level requires two items, one mathematical and the other involving policy. The IEUBK model provides the mathematical relationship between environmental lead levels and BLLs that form the basis for the soil lead cleanup level. However, the level of lead in blood that is considered acceptable is equally critical to the calculation of a soil lead cleanup level, and this is a policy decision.

The 5% Criterion

EPA's current policy concerning acceptable BLLs is best articulated in its 1998 Office of Solid Waste and Emergency Response (OSWER) directive (EPA 1998b, see additional discussion in next section). EPA's policy is one of protecting the individual child and states that no child should have greater than a 5% probability of having a BLL above 10 μg/dL. (Note that this target is sometimes referred to as a "probabilistic" target. This is distinct from the IEUBK model itself, which, in its current form, is not probabilistic.) A careful reading of previous OSWER directives (1994 and 1992) and draft directives on this topic suggests that the current policy has always been EPA's policy; however, poor articulation of the statement combined with a lack of understanding on the part of many responsible parties and EPA project managers have led previous applications of the IEUBK model to calculate a soil lead cleanup level consistent with a target of having no more than 5% of the community with BLLs above 10 μg/dL. Indeed, in the Coeur d'Alene River basin, this may be particularly true as the remedial action objective of the cleanup in the box was explicitly stated as 5% of the population.

These targets are sometimes described as "community" and "individual" protection targets, where the community target requires that 95% of children in the community have BLLs below 10 μg/dL, and the individual target requires that each individual child have a 95% probability of having a BLL below 10 μg/dL. Again, although the community protection target has been adopted at some sites, EPA's policy is to use the individual protection target. One reason to debate the appropriate target of protection is that the choice can have a large impact on the soil lead cleanup level. A community level target will yield a higher soil lead cleanup level for any given site because it is necessary to ensure only that 95% of the community would be expected to have BLLs below 10 μg/dL. Some of these 95% of children with BLLs below 10 μg/dL would be living on yards contaminated just at or below the soil lead cleanup level, whereas (many) others would be living on yards with lower soil lead levels. The individual protection target is stricter than the community protection target in that it requires that 95% of children who live where they are exposed to maximum levels of lead in soil (at the soil lead cleanup level) will have BLLs below 10 μg/dL. The entire 95% of children with BLLs below 10 μg/dL would be equally exposed to yards contaminated just at or below the soil lead cleanup level. Again, this distinction is one of policy, and neither target is scientifically correct or incorrect.

Application of the Geometric Standard Deviation

One of the most critical parameters required in calculating the soil lead cleanup level is the individual blood lead GSD. The individual GSD expresses the range of BLLs that can arise due to all factors other than a narrow range of environmental lead concentrations.[8] These factors include behavioral components, such as soil ingestion rates, biokinetic differences between individuals, and ranges of lead intake from sources other than the site, such as food. The value of the individual GSD is necessarily less than the value of a community GSD, derived from the range of BLLs seen in a community. The community GSD must be higher because, in addition to all the components that contribute to the individual GSD, the community GSD also includes a component of variability due to variable environmental concentrations. The IEUBK includes a recommended default individual GSD,[9] although site-specific blood lead data have been used at some sites to alter its value (EPA 1995, 1998a; Life Systems 1995). The individual GSD is also used to estimate the percent of BLLs greater than 10 μg/dL in an

[8]EPA states that the GSD is not intended to address variability "in blood lead concentrations where different individuals are exposed to substantially different media concentrations of lead" (EPA 1994a).

[9]A fully probabilistic version of the IEUBK model, such as the ISE model, would calculate a site-specific individual GSD a priori. Such a probabilistic approach would reduce uncertainty associated with the default recommendation for the GSD and would obviate the need for large amounts of site-specific blood lead data to calculate a site-specific GSD using the current model approach.

IEUBK model prediction. If this percent agrees well with observation (considering all the limitations of such comparisons discussed below), then this is an indication that the GSD value may be appropriate for the community.

Once an adequately predictive model of the relationship between environmental lead and blood lead in a community has been developed, including the GSD, and the target level of protection has been chosen, the IEUBK model can be used to calculate the soil lead cleanup level. This is done as follows: if we assume that no individual child should have more than a 5% probability of a BLL exceeding 10 µg/dL, and we use the individual GSD model-recommended value of 1.6, we can then calculate that this requires a geometric mean (GM) BLL of 4.62 µg/dL from the following relationship:

$$10\ \mu g/dL = GM \times \exp(1.645 \times \ln([GSD]).$$

The IEUBK model is then run to find the soil lead concentration that yields a predicted geometric mean blood lead of 4.62 µg/dL. Note the overall conservativeness of this approach—EPA's target requires a predicted geometric mean BLL of 4.6 µg/dL for children living on the highest soil lead concentration left unremediated. This is the reason that communities are sometimes identified for lead remediation when no children have BLLs above 10 µg/dL. This level of protection stems from policy decisions; as such, they are not under the purview of this committee considering scientific and technical aspects.

Interpretation of the OSWER Directives

EPA issued an OSWER directive in 1998 (EPA 1998b) that specifies use of the IEUBK model for lead risk assessment for young children and describes EPA's policy concerning acceptable BLLs and the relationship of modeling to blood lead studies. This OSWER directive is an update of an earlier directive issued in 1994.

The 1998 OSWER directive articulates EPA's policy of protecting an individual child from having more than a 5% probability of a BLL elevated above 10 µg/dL (see discussion above). The 1998 OSWER directive also makes clear that EPA views blood lead data alone as insufficient for performing a risk assessment, stating "that predictive tools should be used to evaluate the risk of lead exposure, and that cleanup actions should be designed to address both current and potential future risk." The insufficiency of blood lead observations alone is linked to the policy of protecting individual children, because blood lead information without accompanying environmental lead levels cannot adequately assess the exposure potential that exists, and information about today's blood lead concentrations is insufficient to address what BLLs might occur for other current and future children exposed to the same environmental lead concentrations. The 1998 OSWER directive stresses the interpretive utility of comprehensive blood lead studies, which include an exposure assessment component, over simple blood lead screening or monitoring program observations (see discussion in Chapter 5). Nevertheless, “OSWER recommends that blood-lead studies not be used to determine future long-term risk where exposure conditions are expected to change over time.”

Unfortunately, the OSWER directive’s stated preference for IEUBK-calculated BLLs over actual observation for risk assessment purposes has been misinterpreted by the public, which does not always understand the need for risk assessment or remediation in the face of community BLLs that do not appear to be substantially elevated, and by some EPA project managers who, as a result, ignore or downgrade the importance of valid blood lead information. There is almost never a situation in which model predictions are more accurate than a representative set of observations. EPA should clarify that the IEUBK model is preferred because it does two things that blood lead information alone does not do: it mathematically describes the relationship between environmental lead levels and BLLs, and, because of that description, it allows the calculation of a soil lead cleanup level that will be sufficiently protective.

It should also be made more clear that blood lead observations can be very useful and should not be discarded during the risk assessment process. The OSWER directive acknowledges this with the following:

> Blood-lead data and IEUBK model predictions are expected to show a general concordance for most sites. However, some deviations between measured and predicted levels are expected. On some occasions, declines in blood-lead levels have been observed in association with lead-exposure reduction and health education. However, long-term cleanup goals should be protective in the absence of changes in community behavior as there is little evidence of the sustained effectiveness of these education/intervention programs over long periods of time. …Where actual blood-lead data varies significantly from the IEUBK Model predictions, the model parameters should not automatically be changed. In such a case, the issue should be raised to the TRW to further identify the source of those differences.

However, little guidance is available about what to do if IEUBK model predictions and blood lead data do not match other than to consult the TRW. It is clear that the blood lead observations should not be ignored in such a case, provided a representative sample of children has been surveyed. It is particularly important that a protocol for comparison between observed and predicted results should be standardized for risk assessment purposes to prevent further confusion being added to the interpretive process. Hogan et al. (1998) presented two types of comparisons that appear useful.

Development of Risk-Based Exposure Media Concentrations

In the statement of task, the committee is asked to examine whether the model has been appropriately applied given the local and regional characteristics of the Coeur d'Alene River basin. The committee has undertaken an analysis of environmental lead measurements specifically to determine whether EPA's work has been adequate in this regard.

Lead in Soil and Dust

The IEUBK model calculates the intake of lead derived from the incidental ingestion of contaminated outdoor soil and indoor dust as the weighted intakes of the respective soil/dust particles and the concentrations of lead in those exposure media. Although this formulation is straightforward, the underlying processes controlling children's exposures to environmental lead are complex. One of the primary links in the transfer of lead in soil and dust to the gastrointestinal tract is the hand-to-mouth behavior of children. Some of the soil and dust that hands come in contact with ends up adhering to them, and subsequent activity transfers hand-adhering dust to the mouth. Two important properties of lead-bearing dust and soil must be addressed to determine the appropriate concentrations for use in the IEUBK model and the associated sampling protocols. The first is the particle-size dependence of concentration of lead in surficial dusts and soils and the other is the contribution of outdoor soil lead to indoor lead in household dust. Both of these influence the parameter values used in the IEUBK model applications to represent the source of the exposure. Model default values appear to show percentages of time that a child is in contact with soil or dust, but, in fact, they simply establish an exposure weighting for these two sources.

Investigators have shown that fine particles, especially those less than 100 micrometeres (mm) in diameter adhere more strongly to hands (Duggan et al. 1985; Duggan and Inskip 1985; Sheppard and Evenden 1994; Kissel et al. 1996) and that, as particle size increases, adherence to skin decreases. According to EPA (2000), the upper bound of the size fraction adhering to skin is 250 µm, based on a review of several studies dealing with dermal contact with soil. The so-called "fine" fraction of a dust and soil sample (defined as particles less than 250 µm) is also likely to be enriched in lead compared with lead in the bulk soil sample. EPA's guidance for the sampling and analysis of lead-contaminated soils recommends that the maximum sieve size for such soil is 250 µm (a No. 60 sieve) (EPA 2000).

However, the guidance also states that other sieve sizes may be used but warns that lead enrichment is likely to increase with smaller sieve sizes. Soil and dust sampling programs in the Coeur d'Alene River basin that are the source of the data used in IEUBK model runs, in contrast, have relied on a standard 175 μm sieve size (a No. 80 sieve). The rationale for this particular sieve size includes compatibility with earlier soil sampling protocols in the Coeur d'Alene River basin and consistency with soil adherence data for dermal exposures (see EPA [2001a] for additional discussion). Although enhanced lead enrichment would be expected in soils processed with the 175 μm sieve instead of the 250 μm sieve, the real issue from a human exposure assessment standpoint is not lead enrichment, but rather the accurate characterization of lead in the particles that play the dominant role in the soil/dust-to-hand-to-mouth pathway. In fact, Gulson et al. (1995) contended that a 100 μm cut point would be preferable for determining concentrations of lead in both soils and dusts.

The transport potential of lead-contaminated soils to the indoor environment by foot traffic and pets is also a function of the size distribution of soil particles and the associated concentration of lead in the various fractions. Specifically, footwear normally would be expected to carry fine-mode particles indoors except under wet conditions; consequently, concentrations of lead and other metals associated with this fraction would be the most closely related to the indoor levels. Once soil-derived particles are tracked into the house environment, a variety of redistribution and dilution processes occur that collectively produce indoor dust. For example, the tracked-in soil mixes with a variety of organic-rich indoor sources such as lint, exfoliated skin, carpet fibers, and dried food particles. Concentrations of organic matter in house dust can exceed 30% by weight (see Fergusson and Kim 1991; Molhave et al. 2000). Consequently, the concentrations of outdoor-derived soil contaminants are lower in indoor dust, provided that there are no indoor sources of the soil contaminants. Dust is distributed throughout a house by foot traffic and by the resuspension of floor particles into household air by walking and by particulate emissions from vacuuming. The airborne particles are then deposited onto floor and nonfloor surfaces and exhausted to outdoor air via normal air exchange processes that also transport outdoor air particles through the building shell into the indoor environment (Schneider et al. 1999).

Epidemiological studies investigating the relationships between blood lead and environmental/socioeconomic parameters have shown that children's contact with lead-bearing household dust (represented by lead loading on floor surfaces, rather than by lead concentration in the dust) is a key determinant of BLLs (see Lanphear et al. 1998). Studies of data specific to the Coeur d'Alene River basin involving blood lead and environmental lead measurements have also supported the important role that indoor dust plays as an exposure medium for blood lead. Note, however, that the significance of the indoor dust in this context is related to the location of the exposure. Many young children spend more time indoors than outdoors, and outdoor soils may be a major source of indoor lead because of transport of soil particles on footwear and by pets. The Idaho Department of Health and Welfare (IDHW) environmental health assessment conducted for ATSDR (ATSDR 2000b) found that the logarithm of the lead loading rate (in mg/m^2/day) on entryway sampling mats explained 46% of the variance in log-transformed blood lead concentrations in children 9 years old and younger. Although this analysis did not control for such confounding factors as lead paint, the results are similar to those of the HHRA (TerraGraphics et al. 2001), which found that lead loading per unit mat area per day was the most important variable in determining blood lead in multiparameter regressions (other parameters included children's age, yard soil lead, and lead paint metrics). Statistical analyses presented in the HHRA (TerraGraphics et al. 2001, Table 6-20) of the relationship between the concentrations of lead in mat dust and other environmental lead measurements indicated that 42% of the variation in mat lead was due to yard soil lead. Other contributors were lead in community soils and interior paint condition.

An important finding of the IDHW environmental health assessment (ATSDR 2000b, Table 4) was that the average lead concentration in mat dust (n = 400, 1,416 µg/g) was nearly a factor of two greater than the average concentration of lead in the yard soils of the houses studied (n = 815, 738 µg/g). Sampling data for entryway sampling mats in houses and yard soils in Coeur d'Alene River basin communities (TerraGraphics et al. 2001, Table 6-11a-j) showed similar results—that is, significant enrichment in mat dust lead compared with lead in yard soils. If outdoor soil is the principal source of lead in indoor dust (and the key environmental medium targeted for remediation), then why is the concentration of lead in entryway dust (as sampled by mats that intercept soils tracked in by residents) significantly higher than that in the outdoor soils?

The answer to this question could result in a better quantitative characterization of the relationship between the concentrations of lead in soil and dusts and associated exposure simulations in the IEUBK model. The committee has analyzed environmental lead, iron, and manganese measurements available from one of the remedial investigation/feasibility study (RI/FS) data sets to further explore the significance of this question, as detailed in Appendix E. Key findings summarizing the significance of additional analyses for source apportionment are as follows:

- Particle size fractionation processes are the most-likely explanation for the average differences in lead observed for soils, entryway mats, and vacuum cleaner dusts. This emphasizes the significance of evaluating lead concentrations across different size fractions of environmental media in the lead exposure assessment, measurements that EPA did not undertake. Future studies should also address the possibility that perimeter soils containing paint-derived lead represent an additional source of lead in indoor dusts.

- However, the foresight to carry out the bulk analysis for the crustal elements iron and manganese made possible additional evaluations in support of exposure assessment, demonstrating their value for inclusion in the RI/FS investigations.
- The results underscore the significance of soils in the exposure pathway by virtue of their major contribution to indoor dust, providing support for the site-specific exposure parameters used in the IEUBK model runs.

Air Monitoring Data

Exposures to airborne lead can occur by the inhalation of particulate lead in indoor and outdoor air as well as indirectly by hand-to-mouth contact with lead on indoor surfaces that is derived from the deposition of airborne lead that has penetrated the building shell. In general, the inhalation exposure pathway for environmental lead plays a minor role compared with the ingestion of lead in soils and dusts. The IEUBK model includes two default methods for relating concentrations of lead in outdoor air to related levels of lead in indoor air and household dust. The first default is an outdoor level of lead in ambient air of 0.10 μg/m^3 and an indoor conversion factor of 30% (the indoor air concentration of lead is 30% of the outdoor level). The second default uses a fixed ratio of the concentration of lead in dust to the concentration of lead in outdoor air of 100 μg of lead/g of dust per μg of lead/m^3 in outdoor air. The second default option was not used in the HHRA simulations because direct measurements of lead in residential dusts were used as inputs. In the HHRA, the default value of 0.1 μg/m^3 air lead concentration was used. Although this value is greater than the expected air concentrations in the basin, the overall contribution of this pathway to absorbed blood leads is just a few percent of the lead intake (EPA 2001b).

Nevertheless, failure to determine the magnitude of airborne inputs to residences can potentially distort the relative importance of alternative transport pathways for the migration of soil-derived lead to the indoor environment and potential sources of variability in BLLs.

As a means of investigating the nature and magnitude of exposures to airborne lead in the Coeur d'Alene River basin, we reviewed historic data on measurements of airborne lead from a monitoring station in Kellogg. Air monitoring for lead started in 1982 and continued until mid-2002 when the station was shut down. Since smelter emissions ended in 1981, ambient levels of lead have steadily declined (Figure 6-4). The concentration of lead in airborne particles is determined by collecting total suspended particulates on a filter and then analyzing the lead content of the collected particles. The product of the total suspended particulate (TSP) concentration (g/m^3) and the lead concentration in collected particles ($\mu g/g$) gives the ambient lead concentration in air in units of $\mu g/m^3$. So, with data on both TSP and ambient lead levels (the reportable air quality measurements), it is then possible to determine the concentration of lead in suspended particles. Figure 6-5 presents the TSP levels and associated concentrations of lead in ambient particles for the years 1982 to 2001. The most significant features of the graph are the dramatic decline in the lead concentrations in suspended particulate matter and the gradual reduction in TSP.

It is important to point out that, after the end of smelter emissions, the principal source of ambient lead in the atmosphere would be the resus-

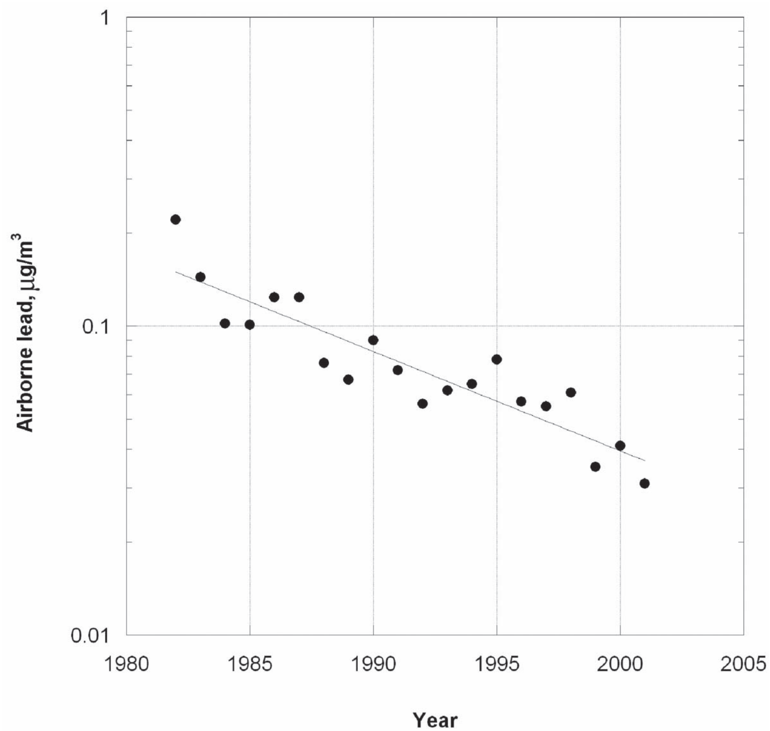

FIGURE 6-4 Concentrations of airborne lead measured at a monitoring station in Kellogg, Idaho, during the years 1982 to 2001. Monitoring ceased in 2002. SOURCE: Idaho Department of Environmental Quality, unpublished material, 2004.

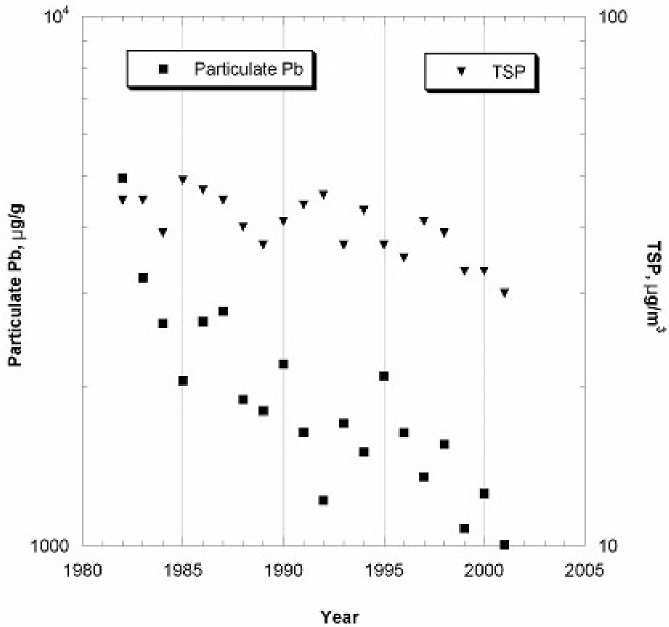

FIGURE 6-5 Long-term trends in the concentrations of lead in suspended airborne particles and mass loading of particles in air. Data are for an air monitoring station in Kellogg, Idaho. SOURCE: Idaho Department of Environmental Quality, unpublished material, 2004.

pension of lead previously deposited from the atmosphere along with wind-driven emissions of dust from surficial soils containing lead derived from previous mining operations. The decline in the concentrations of particulate lead at the Kellogg monitoring site is probably a function of both soil remediation efforts and natural soil weathering processes. But, according to von Lindern et al. (2003a) the major yard remediation work did not begin within the box until 1998; consequently, the substantial declines observed in particulate lead levels before that time depicted in Figures 6-4 and 6-5 undoubtedly are associated with weathering of soil lead.

The phenomena of contaminant weathering of surficial soil contaminants and related declines in airborne loadings has been of particular interest to researchers studying the transport and fate of radionuclides deposited onto the land surface (see Anspaugh et al. 2002). One simple approach for estimating the concentration of a soil contaminant in ambient air is to multiply the TSP level times the concentration of the contaminant in soil and an enhancement factor, which is defined as the ratio of the concentration of the contaminant in airborne particles to the concentration in soil. A recommended default value for the enhancement factor is 0.7 for soils that are weathered (NCRP 1999). On the basis of this resuspension model, the levels of lead in suspended particles at the Kellogg monitoring site are exhibiting substantial enrichment. In 1995, for example, the community-wide concentration of lead in Kellogg soils was about 1,000 μg/g (von Lindern et al. 2003a), but the airborne particles contained lead at about 2,094 μg/g (see Figure 6-5) or about a factor of 2 higher. Moreover, even though the cleanup goal for yards of 350 μg/g was achieved for residences in Kellogg by 1998, the lead concentration in soil-derived suspended particles for 2001 (about 1,000 μg/g) was nearly a factor of 3 greater! The elevated concentration of lead in airborne particulate matter compared to the levels of lead in bulk soils processed with a 175 μm sieve size provides additional evidence that lead may be preferentially concentrated on fine soil particles—due to previous atmospheric inputs as well as other geochemical weathering processes of mining wastes mixed with Coeur d'Alene River basin soils.

Lead in Drinking Water

The default concentration for lead in drinking water used in the IEUBK model is 4 μg/L; for comparison, the national drinking water action level for lead is 15 μg/L (EPA 2004b). Measured values of lead in drinking water for Coeur d'Alene River basin communities are given in HHRA Tables 6-11a-j (TerraGraphics et al. 2001). Most of the reported concentrations for lead in "first draw" water from taps and private well waters were between 2 and 4 μg/L, although some of the maximum values reported exceeded the action level for lead in drinking water. Concentrations of lead in "purged" samples of tap water were substantially lower than the first-draw samples. For example, in Wallace, the geometric mean concentration of lead in purged water was 0.65 μg/L, compared with 3.19 μg/L for the first-draw samples. Although IEUBK guidance recommends averaging the lead concentrations in the first-draw and purged samples, the HHRA used only the purged values for the batch-mode runs of the IEUBK model. No rationale was given for that decision; however, the consequences are expected to be minor given the relatively small contribution that drinking water provides to overall lead intake. In another example of potential bias, the HHRA notes that levels of lead in well waters are overestimated because the original water analyses taken in 1996 did not report concentrations below the then-current lead drinking water source standard of 5 μg/L. In fact, 183 of 222 wells sampled in 1996 had censored results—that is, values at or below 5 μg/L. Later studies indicated that the geometric mean value for well waters is 0.75 μg/L (TerraGraphics et al. 2001). So, use of the existing concentration values for lead in well waters for the batch mode

IEUBK model would have overestimated drinking water exposures to lead. But again, the consequences are not likely to be significant because of the minor role this pathway plays in the overall intake of lead.

Lead in Local Food Supplies

Dietary intakes of lead were simulated in the HHRA using baseline and incremental exposure scenarios. In the baseline scenario, children consume lead derived from a typical "market basket" of foods, and therefore the default input parameters for dietary lead were adopted. However the default dietary lead intakes in the IEUBK model are based on older data and are higher than would be suggested by more-recent dietary information (Bolger et al. 1996). Therefore, dietary exposure to lead is probably overestimated in the baseline scenario. To estimate dietary intakes for the incremental exposure scenario—designed to represent exposures associated with a limited subset of the population—information is required on both the concentrations of lead in selected foods and related intakes. Residual lead in Coeur d'Alene River basin soils and surface waters can produce elevated dietary exposures to lead for children in households that rely on homegrown produce or locally caught fish for a portion of their regular diets.

Based on sampling conducted as part of the HHRA, the median concentration of lead in fish was 0.12 µg/g wet weight, and the 95th percentile concentration was 0.68 µg/g wet weight. With a fish fillet intake rate of 5.4 g/day (TerraGraphics et al. 2001, Table 6-39), the respective lead intakes for the central tendency (CT) and reasonable maximum exposure (RME) intakes for children were 1 and 4 µg/day. These intakes represent a small increment above the baseline lead intakes that range from 30 µg/day for the lower basin to 99 µg/day for Wallace.

The median concentration of twenty-four samples of homegrown vegetables collected from Coeur d'Alene River basin communities was 3.2 µg/g wet weight (TerraGraphics et al. 2001, Table 6-40a); with an intake of 7.4 g/day of garden vegetables (based on a 15 kg child; TerraGraphics et al. 2001, Table 6-39), the associated CT lead intake is 24 µg/day. At the 95th percentile concentration in garden vegetables (24 µg/g wet weight), the lead intake becomes 178 µg/d (representing the RME estimate). In contrast, the default dietary intake for children ages 1-5 years is approximately 6 µg/day. Although the levels of lead in homegrown produce vary according to the levels of lead in soil, the HHRA uses the same median and 95th percentile intakes for all communities in the incremental exposures used in the IEUBK model (TerraGraphics et al. 2001, Figures 6-21a-h).

The estimated lead intake for the CT exposure case seems plausible; however, the RME intake is not entirely consistent with blood lead measurements. According to Table 6-55b of HHRA, the geometric means of the

BLLs predicted by the IEUBK model for the RME case would exceed 20 μg/dL for both the EPA default and box model implementations. But in Table 6-2 of the HHRA, there were only 12 instances in which BLLs exceeded 20 μg/dL out of 524 measurements made during the years 1996-1999 (about 2% exceedance). It is not possible to determine whether the consumption of homegrown produce was a contributing factor to those exceedences, because household-specific information on dietary practices was not reported. Nevertheless, the available information from several studies suggests that the consumption of homegrown vegetables is unlikely to play a dominant role in causing elevated BLLs. For example, uptake ratios for arsenic and lead into vegetables have been found to be low (Glass and SAIC 1992; EPA/SRC 2001), and biomonitoring data from many sites including the Basin (ATSDR 2000b) have not indicated that ingestion of homegrown vegetables contributes to elevated lead and arsenic exposure in residents (Polissar 1987; Polissar et al. 1990; Bornschein et al. 1991; ATSDR/CDOH 1992; BSBDH and University of Cincinnati 1992; Hwang et al. 1997).

Configuration and Use of the Bunker Hill Superfund Site Box Model

The HHRA utilizes the IEUBK model in four modes. Assumptions are either "default" or "box" and operation is either "community" or "batch." The regression analyses for examining the relationships between environmental lead and blood lead values (TerraGraphics et al. 2001; von Lindern et al. 2003a) provided a basis for the structural equation modeling (SEM) source apportionment. These results indicated that, for the Coeur d'Alene River basin, site-specific deviations from the IEUBK default proportions of soil and dust ingestion should be used. Soil was shown to be the major contributor to the combined exposure medium and should be weighted more heavily than the nonsoil lead contained in house dust. A 60% soil and 40% dust division is supported by the soil tracer element analysis described in Appendix E. The SEM also highlighted the apparent role of community-wide soil lead concentrations in the exposure dynamic. A reduction in lead uptake was indicated by the SEM analysis, and the box model implemented this by reducing the bioavailability values used by the model; default soil/dust ingestion rates were maintained. These adjustments from IEUBK default configurations provided a better fit, for the several possibilities considered, between observed and predicted blood lead values and are contrasted in Table 6-4.

When interpreting the fractions of soil/dust ingestion summarized below, the proportions reflect the source of the materials to which the child is ultimately exposed and not the proportion of time that a child spends in each of these environments. The IEUBK model does not separate the soil and dust ingestion regime with respect to time spent indoors or outdoors. It

TABLE 6-4 Default and Box Assumptions Used in the HHRA

	Fraction (%) of Soil/Dust Lead Ingestion Attributed to			Bioavailability of Lead in Soil
Model	House Dust	Yard Soil	Neighborhood Soil	(%)
Default	55	45	0	30
Box	40	30	30	18

SOURCE: TerraGraphics et al. 2001.

models the combined exposure dynamic using the concentration of lead in the two media and the fraction each contributes, either directly or indirectly, to the daily lead ingestion intake. Soil is very clearly an important constituent of household dust. Details of soil and dust transport as well as children's activity patterns will vary greatly among locations considered, and these inputs to the IEUBK model represent the average way in which the exposure parameters affect the model predictions.

Application of the IEUBK model in batch mode permits limiting simulations to those households for which both environmental and matched blood lead data are available. Evaluation of batch mode IEUBK results, therefore, avoids questions about the representative nature of the overall basin blood lead data set. Batch mode IEUBK predictions (both "default" and "box" versions) and corresponding observations are presented in Figure 6-6a (percent of blood lead ≥ 10 μg/dL) and Figure 6-6b (geometric mean blood lead in μg/dL) for each of the eight study areas. Study areas are placed on the x-axis for these figures in roughly geographical order running from west to east in the basin. Model results both underpredict and overpredict observed values depending on model version and study area.

To facilitate interpretation of data in Figure 6-6a, absolute differences between the predicted and observed sample fraction (expressed as percent) exceeding 10 μg/dL for the default model and the box model are presented in Figure 6-7a and 6-7b, respectively. Bars falling below the x-axis in Figures 3 and 4 reflect underprediction by the IEUBK model and bars falling above the x-axis reflect overprediction. In Figure 6-7a, it can be seen that the default model overpredicts in six of eight study areas and underpredicts in two. In all cases, the magnitude of deviance is greater than 5% of the observations. In contrast, Figure 6-7b shows that the box model tends to better predict the fraction exceeding 10 μg/dL in those areas in which the default model overpredicts but produces greater underprediction in the two most westerly (downstream) study areas.

Examination of differences between predicted and observed geometric mean BLLs as shown in Figure 6-8a and 6-8b reveals a very similar pattern. The default model overpredicts in the upper basin and underpredicts,

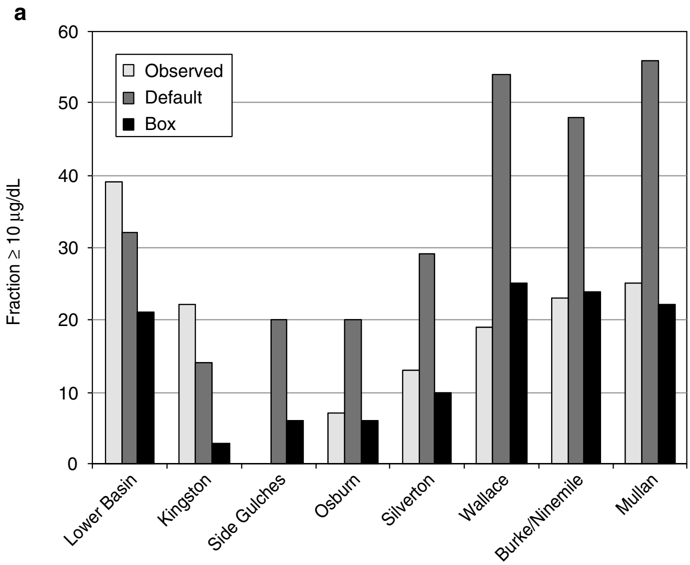

FIGURE 6-6a Fraction exceeding 10 µg/dL by study area for children 1-5 as observed and predicted using IEUBK default and box models in batch mode.

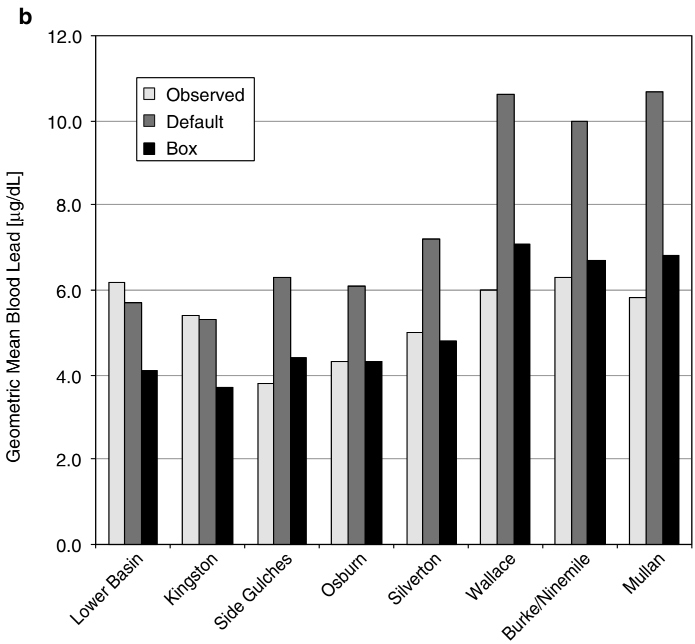

FIGURE 6-6b Geometric mean blood lead (µg/dL) by study area for children aged 1-5 as observed and predicted using IEUBK default and box models in batch mode.

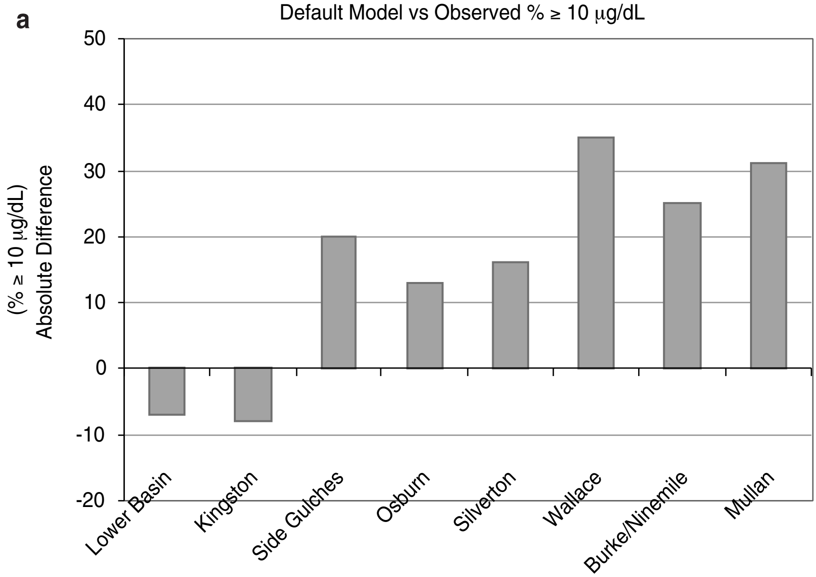

FIGURE 6-7a Absolute differences between batch mode IEUBK default model prediction and observed fraction exceeding 10 µg/dL by study area for children 1-5.

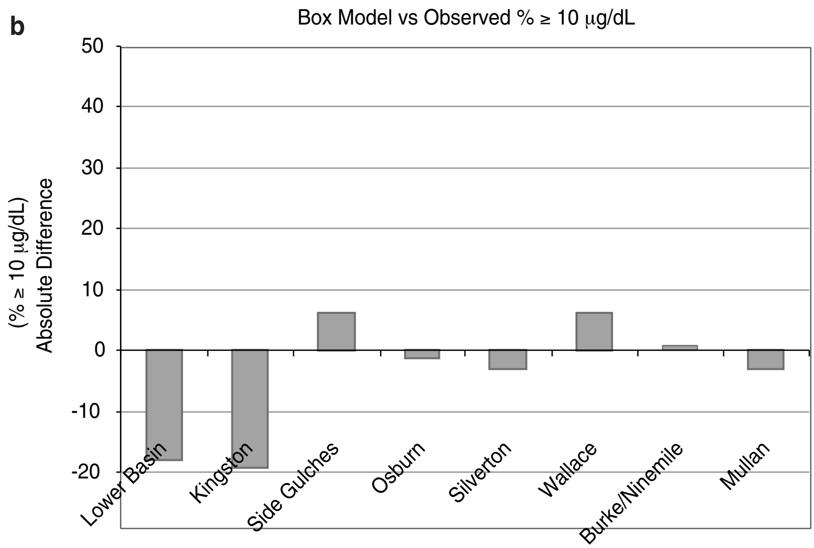

FIGURE 6-7b Absolute differences between batch mode IEUBK box model prediction and observed fraction exceeding 10 µg/dL by study area for children aged 1-5.

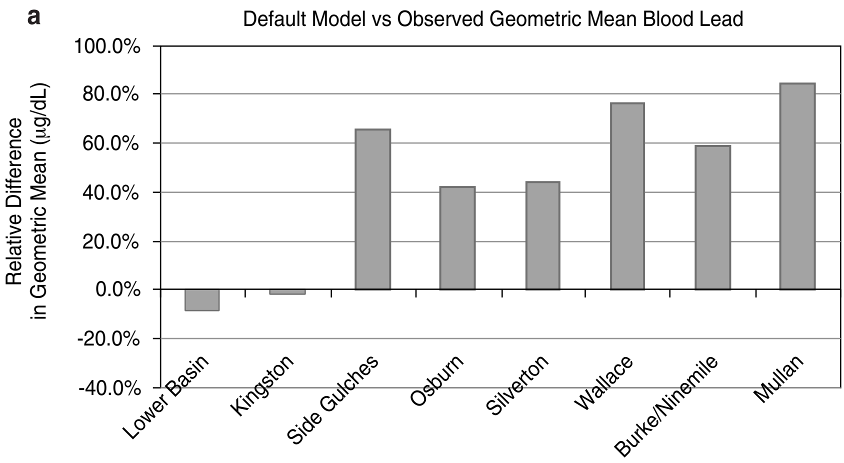

FIGURE 6-8a Relative difference (as percent) between batch mode IEUBK default model prediction and observed geometric mean blood lead (μg/dL) by study area for children 1-5.

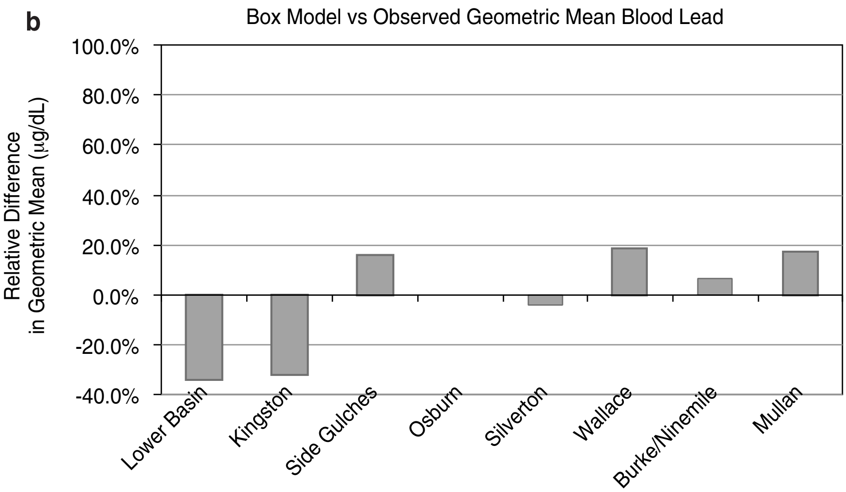

FIGURE 6-8b Relative differences (as percent) between batch mode IEUBK box model prediction and observed geometric mean blood lead (μg/dL) by study area for children 1-5.

slightly, in the lower basin. The box model does a better job of predicting upper basin geometric means but more severely underestimates lower basin values. (It should be noted that relative differences between observations and box model geometric mean predictions in the upper basin are all less than 20%, a relatively small deviation given the current state of modeling of human exposure to environmental contaminants.) The differences between default and box inputs were described previously and are presented in Table 6-4. The box model assumes lower bioavailability and greater contribution of neighborhood soil (as opposed to residential soil and dust) to exposure. Adjustment of bioavailability downward from the default value of 30% is plausible for the upper basin given the observation that the bioavailability of lead from galena is lower than the bioavailability of lead from other minerals in swine feeding trials and that a significant fraction of lead in upper basin soils may be present as unaltered galena (see Table 6-5). However, proportional adjustment of IEUBK results could also be achieved by modifying assumed soil ingesion rates and interpretation of improved model performance acknowledges this uncertainty (von Lindern et al. 2003b) (see Box 6-6 for additional discussion).

It is logical to assume that children may be exposed to lead away from their own residences, but accurate selection of a precise fractional source contribution should not be presumed. Disparate model performance in the lower basin may be related to differing exposure profiles. For example, shoreline recreation in the lower basin may lead to significant exposure to exposed materials with high lead content and bioavailability. Neighborhood soils therefore may be a poor surrogate in the lower basin, leading to box model underprediction. As described in the OU-3 HHRA, follow-up studies of children with high levels of lead in their blood in the lower basin suggest strongly that riverbank material may be an important source of lead exposure (TerraGraphics et al. 2001). The Coeur d'Alene River basin might also exhibit spatial variation in soil lead bioavailability. Smaller particles are transported farther downstream in watersheds and generally exhibit higher lead bioavailability (Mushak 1991) than larger particles.

Adherence/Adequacy of Actions to Superfund Guidelines

Weighting of Biomonitoring Data Versus Model Results

EPA includes two types of IEUBK model calculations in the HHRA, referred to as "community mode" and "batch mode" calculations. Soil lead cleanup levels typically are based on batch mode results, and those results are discussed first here. Batch mode results are a set of predicted BLLs for each individual child in the database for whom "paired data" (soil, dust, and blood lead) are known. At this site, the IEUBK model batch mode results for the paired data set, using the box model assumptions compare reasonably well with measured BLLs (TerraGraphics et al. 2001, Tables 6-49 and 6-50). For the purposes of the discussion below, the batch mode operation with a paired data set is referred to as step 1. Ideally, the paired data set would be composed of environmental lead levels that are representative of the community; often, it is composed of a biased set of environmental lead levels that do not represent the community at large. In the latter situation, it is clear that if the data set is limited to geographic areas where environmental levels are expected to be high, then the paired BLLs may also be high, and not representative of the community as a whole. However, this is not important for this step because the objective is to explore and understand the relationship between environmental lead and BLLs. To do this, the observed BLLs must be representative of levels that typically would arise upon exposure to these environmental conditions. Good agreement between observation and model predictions is one indication that the observed BLLs are typical of the environmental conditions.

Because the batch mode predictions of BLLs based on environmental lead levels for the paired data set are reasonably good, the next step in the HHRA is to apply the batch mode calculation to all residences and yards in the community for which environmental lead concentrations are available. This is referred to below for purposes of this discussion as step 2. This calculation is done regardless of whether a BLL has been obtained for any child living in the residence. This step produces a predicted distribution of BLLs for the community. If BLLs have been measured for a truly representative cross section of the community (with regard to environmental lead), then the predicted and observed BLLs may be comparable. However, if the measured BLLs (from step 1) are not representative of the distribution of environmental lead levels in the community, it is not appropriate to compare this predicted distribution of BLLs with the observed distribution of BLLs in the community. If the comparison is done and the results are favorable, this suggests that the observed BLLs are a representative cross section of those in the community. However, if the comparison yields unfavorable results, it could be either because the IEUBK model does not work well in this situation or because the observed BLLs are not representative of the community. For example, if the original paired data set used in step 1 included only children who lived in the residences with the highest environmental levels of lead, then when the IEUBK model batch mode is applied to all residences, including those with lower environmental levels, we would expect the overall predicted distribution of BLLs to be lower than the observed distribution. This discussion is presented to demonstrate that blood lead data need not always be “representative” to be useful. However, blood lead data without accompanying environmental lead levels are rarely useful in the modeling context.

TABLE 6-5 EPA Region 8 In Vivo (Juvenile Swine) Studies of Lead Bioavailability in Various Contaminated Soils and Mine Waste Residuals

Site	Sample	Lead (ppm)
New Jersey zinc, Palmerton, PA	Site soil location 2	3,230
	Site soil location 4	2,150
Smuggler Mountain, Aspen, CO	Berm soil	14,200
	Residential soil	3,870
Oronogo-Duenweg mining belt, Jasper County, MO	Near-smelter high-lead soil	10,800
	Near-mill high-lead soil	6,940
	Low-level yard soil	4,050
Murray smelter, Murray City, UT	Slag	11,500
	Surface soil	3,200
Kennecott, Salt Lake City, UT	Residential soil	1,590
	Creek channel material	6,330
Silver Bow/Butte area, Butte, MT	Waste rock dump soils	8,600
Midvale slag, Midvale, UT	Slag	7,895
California gulch, Leadville, CO	Residential soil	7,510
	Trailer park soil	4,320
	Smelter slag	10,600
	Tailings	1,270
N/A	Unweathered crystalline galena in low-lead CO soil	11,200
N/A	NIST powdered leaded indoor paint in low-lead CO soil	8,350

Note: Data shown are lead concentration in material fed, percent of lead mass derived from the most abundant lead mineral and from galena (lead sulfide), particle size range, and the resulting estimated absolute bioavailability (ABA) of lead.
SOURCE: Casteel et al. 1996a-d, 1997a,b, 1998a-e.

BOX 6-6 Are the Assumptions of the Box Model Necessarily Correct?

- The IEUBK box model configuration provides appropriate soil cleanup levels for the Coeur d'Alene River basin OU-3 as a whole.
- Adjusting some of the IEUBK model default values to box model conditions provided a better fit between observed and predicted blood lead values for some but not all geographic subregions of OU-3. Adjustments were based on empirical results, not on knowledge of which parameters more accurately reflect the true state of nature.
- Although such agreement could have been accomplished by reducing the soil/dust ingestion rates, or by lowering specifications for soil/dust bioavailability, the latter option has a more plausible connection to possible geographic differences within the basin. Ingestion rates would not be expected to show patterns of spatial variability.

Mineralogy (as lead mass)	Particle Size (μm)	Suggested ABA (%)
66% manganese-lead oxide, 0% lead sulfide	≤250	34
66% manganese-lead oxide, 0% lead sulfide	≤250	27
62% lead carbonate, 12% lead sulfide	≤250	30
64% lead carbonate, 17% lead sulfide	≤150	31
32% lead carbonate, 0% lead sulfide	≤250	29
57% lead carbonate, 3% lead sulfide	≤250	40
81% lead carbonate, 8% lead sulfide	≤250	40
69% lead oxide, 9% lead sulfide	≤250	27
29% lead-arsenic oxide, 20% lead sulfide	≤250	36
50% lead phosphate, 0% lead sulfide	≤150	15
59% lead or iron-lead sulfate, 9% lead sulfide	≤150	14
36% lead sulfate, 13% lead sulfide	≤250	10
33% lead-arsenic oxide, 6% lead sulfide	≤250	8
≈ 30% lead phosphate, <5% lead sulfide	≤250	37
>70% manganese-lead oxide, 0% lead sulfide	≤250	45
>50% iron-lead oxide, <5% lead sulfide	≤150	9
100% lead sulfide	≤50	3
100% lead sulfide	≤100	<0.5
55% lead carbonate, 0% lead sulfide	N/R	40

Soil lead cleanup levels are derived on the basis of the IEUBK model used for both steps 1 and 2 above. Note that this is actually the same model in steps 1, and 2; in step 1, it is applied only to residences where a child with a blood lead measurement lives, whereas in step 2 it is applied to all residences where environmental measurements have been made. EPA's general approach to calculating a soil lead cleanup level does not need step 2. Rather, it uses the model as applied in step 1 and calculates the highest soil lead concentration that is still consistent with a BLL that, combined with the blood lead GSD, will produce no more than a 5% probability of being above 10 μg/dL. The Coeur d'Alene HHRA takes a somewhat broader, although nearly equivalent, approach, selecting a possible soil lead cleanup level, rerunning the step 2 batch mode run, and considering the predicted blood lead exceedance rate for the residences with soil lead levels within 200 mg/kg of the possible soil lead cleanup level. This approach is somewhat less conservative than the typical approach (it will yield a higher soil lead cleanup level) because the distribution of BLLs predicted for the highest soil lead yards (within 200 mg/kg of the cleanup level) will be slightly lower than predicted for the highest soil lead yard alone. However, the resulting soil lead cleanup level is very similar.

Lack of Site-Specific Bioavailability Assessments

It is well established that some fraction of lead found in soils is absorbable in mammalian gastrointestinal tracts. The absorption of lead from soils from contaminated locales has been studied in juvenile swine by EPA personnel and collaborators (Casteel et al. 1996a-d, 1997a,b, 1998a-e). Findings from these studies are summarized in Table 6-5. Absorption of lead from soil has also been studied in rats (Freeman et al. 1992, 1994, 1996; Dieter et al. 1993). Rats are considered an inferior surrogate for humans, but those data do support trends observed in the swine studies with respect to dependence of availability on speciation. Simulated gastric dissolution of lead-bearing materials has also been conducted in vitro. The results of these studies are generally consistent in demonstrating that a nonnegligible fraction of lead in soil can be absorbed but that efficiency of absorption depends on multiple factors including chemical speciation of lead, dietary factors, and the particle size of soil ingested. Typically, paint-derived lead (lead oxides, basic carbonates) is relatively bioavailable, whereas lead associated with sulfide minerals is relatively unavailable. One study was conducted on soil from a residence within the Bunker Hill box (but not the basin) in human volunteers using a stable isotope approach (Maddaloni et al. 1998). These experiments demonstrated 26% bioavailability of lead in soil to fasted individuals and 2.5% in individuals who consumed lead contaminated soil just after eating.

Given the rather large range of absolute bioavailability (in swine) for soils and residues at site affected by mine waste (Table 6-5), the lack of any such study results applicable to the Coeur d'Alene River basin Superfund site represents a deficiency in the HHRA and the subsequent ROD. A variety of in vivo assays (Freeman et al. 1994; Casteel et al. 1997b) could have been applied; alternatively, an in vitro physiologically based extraction test (Ruby et al. 1996) would have been useful. As demonstrated by Watt et al. (1993), with actual hand wipes from children, the physicochemical form of environmental lead is extremely important in the exposure dynamic. Furthermore, these properties can change over time (Johnson and Hunt 1995), and, because particle size is also important for bioavailability, at a minimum the RI/FS and HHRA ought to have included information on the concentration of lead in different size fractions of basin soils, although EPA guidance does not currently require this. EPA should require that the

IEUBK model used for determining cleanup levels be supported by site-specific measures of bioavailability.

Evaluation/Improvement of Actions Taken for the ROD

Although the committee did not find technical or policy issues with respect to the actions taken for the ROD, in a number of instances science and policy might be considered as conflicting. This is partially a result of the size and complexity of the Coeur d'Alene River basin Superfund site and partly due to advances in scientific knowledge that have not been incorporated into the use of the IEUBK model. We outline a number of examples below.

IEUBK Model Execution Modes

The community-mode IEUBK model runs are not useful because they predict BLLs for the entire community on the basis of mean and range of soil lead levels, but they can be compared only with the subset of BLLs that were measured. If the measured BLLs correspond to children who represent a cross section of environmental lead levels in the community, then this comparison may be adequate. The comparison is shown in Table 6-47 of the HHRA, with mixed results, suggesting that the measured BLLs were not representative of the community (the range of environmental lead conditions used in the model), as discussed in Chapter 5. An alternative explanation is that the IEUBK model does not work well in this situation, possibly because bioavailability may vary from one community to another.

So, what defines a blood lead data set that is useful with the IEUBK model? The HHRA also presents calculations of soil lead cleanup levels following the community mode approach. However, EPA generally does not use this approach in setting soil lead cleanup levels, and it is not consistent with EPA's target for blood lead protection (a target that an individual child have no more than a 5% probability of a blood lead exceeding 10 μg/dL). If this approach were used as a matter of EPA policy to set the soil lead cleanup level, then the representative nature of the BLLs for the community would be a much more important concern. When the batch mode approach is used, as it generally is, and when EPA's individual target for blood lead protection is used, as it typically is, then the blood lead data need not be representative of the community but rather must be representative of the exposures that arise for the observed environmental lead levels. This concept is not articulated in any EPA guidance documents, and clarification is needed; the usefulness of nonrepresentative epidemiological blood lead data may be counterintuitive for scientists and community members alike.

Protection of the Community or of the Individual Child

It appears that it has always been EPA's policy to focus protection on the individual child, but this policy either was not applied or was incorrectly applied at past sites when the community blood lead protection goal was used instead. The community goal, which focuses on keeping 95% of children in a community with BLLs below 10 μg/dL, effectively abandons the 5% of children with BLLs above 10 μg/dL.

Adequacy of the Blood Lead Data

It is the case here that the blood lead screening rate for the community (<30%) is less than EPA often requires at other sites to feel comfortable that a representative cross section of a community has been obtained. EPA makes no decisions based on the predicted or actual average blood lead in the community. So for EPA's purposes, the question is, are the data representative of the BLLs that typically would arise in this community in children who live in houses with the observed environmental conditions (soil and dust lead levels)? This question is key because the IEUBK model and EPA's approach rely on developing an understanding of the relationship between lead in soil and dust and lead in blood. There is no way to answer this question, but there is also no reason to suspect either a systematic high or systematic low bias to the BLLs for these children exposed to their particular environmental conditions. It is possible that there was a community bias in the blood lead sampling toward children with higher BLLs. Presumably, these children live in conditions where they are exposed to higher levels of lead in soil and dust; nutritionally deprived children may be more likely to reside in housing with contamination. However, the soil lead cleanup level is based not on the number or percent of children with elevated BLLs, but only on the relationship between lead in soil and dust and blood lead. Therefore, this community bias, if it exists, does not affect calculation of the soil lead cleanup level.

Compilation of the Blood Lead Data Set

The blood lead data used for comparison in the IEUBK model contained more than one measurement for some children. This has the potential to bias community statistics or the mean and range of blood lead in the community. However, EPA does not use these community statistics in calculating the soil lead cleanup level, so this bias has no effect on selection of the soil lead cleanup level. To the extent that the soil lead cleanup level is based on the results of the HHRA, it is based on the results of the IEUBK model batch mode runs. The batch mode run of the model yields blood lead predictions for each entry in the data set that is complete or that (at least) contains environmental and blood lead information. If a child is entered in the data set twice by virtue of having a repeat blood lead measurement, then the same environmental lead levels will be used in multiple predictions for this child, and one of the predictions will be closer to observation than the others. Thus comparison or calibration of the IEUBK to site-specific conditions relies on the children sampled being representative of the relationship between blood lead and environmental lead (not on their BLLs being representative of the community). If a child is entered in the data set twice by virtue of having a repeat blood lead measurement, then the same environmental lead levels will be used in multiple predictions for this child, and one of the predictions will be closer to observation than the others. A further bias in comparison or calibration could therefore arise if the children entered in the dataset twice are not representative of the site-specific relationship between blood lead and environmental lead. There is no evidence for either type of such non-representativeness, and any such biases appear likely to be relatively small.

Improvements to Lead Source Apportionment

In Appendix E, it is noted that in perhaps half the houses studied comparing lead, iron and manganese, internal sources for lead in the vacuum cleaner dusts were indicated. Additional studies are needed to confirm this result using other crustal soil tracers and sieve sizes to more accurately characterize the indoor and outdoor sources of lead. Although this analysis was exploratory in nature, it does indicate that there is a value in designing future sampling and analysis programs so that they explicitly address crustal elements concurrently with lead to provide diagnostic information for interpreting the sampling results for lead.

Fortunately, existing sampling protocols involving entryway mats and vacuum bags can provide the analytical results needed to quantify the indoor and outdoor sources of lead in house dust. Specifically, the concentration of lead in indoor dust that is attributable to nonsoil, indoor sources (denoted here as C_{in}) can be estimated by subtracting the concentration of soil-derived lead (C_{sd}) in house dust from the concentration of lead in bulk house dust (C_{bhd}) collected from a vacuum bag. The value of C_{sd} is simply calculated as the product of the dilution ratio and the concentration of tracked-in lead in mat dust (C_t). For example, if the values of C_t and C_{bhd} at a residence are 1,000 and 550 mg/kg, respectively, and the dilution ratio is 0.5 (determined by crustal tracer measurements), then the value of C_{sd} is 500 mg/kg, and therefore C_{in} equals 50 mg/kg. In this particular case, lead from outdoor soil dominates the lead content of the indoor dust. With sufficient samples, the IEUBK model can be run with estimated indoor-outdoor source concentrations for lead in house dust (C_{sd} and C_{in}) to examine how nonsoil sources of lead in dust contribute to BLLs in children. The HHRA, in contrast, conducted statistical analyses between measured BLLs in children and measurements of lead in soils/dusts as well as x-ray fluorescence measurements of lead paint in the children's houses. Those analyses did detect an effect of lead paint on BLLs, but it was small. Unfortunately, x-ray fluorescence measurements are only a surrogate for potential paint-derived lead in house dust, and so it is unclear how representative the results truly are.

Limitations in the Use of Lead Dust Concentrations

An important point to emphasize here is that human activities are the primary source of the dilution effect on substances derived from outdoor soil that have no significant indoor sources. Accordingly, there will be an additional source of variability in the concentrations of lead and other soil-derived substances in dust beyond the variability in outdoor levels. Moreover, the loading of dust on floor surfaces that children come in contact with via hand-to-mouth behaviors is also a function of human activities including the number of household residents, and cleaning frequency, and so forth. Numerous studies have shown that dust lead loading correlates more strongly with blood lead than does dust lead concentration (Aschengrau et al. 1998; Lanphear et al. 1998; Kranz et al. 2004). The IEUBK model, however, determines intakes only as the product of the concentration of lead in soil/dust and an age-adjusted soil/dust ingestion rate prorated for the respective contact media. In essence, the fixed soil/dust ingestion rate used in the IEUBK model is an aggregate parameter that does not take into account variations in house dust loadings that contribute to ingestion exposures. Thus, according to the IEUBK exposure formulation, children in two different houses that have the same concentrations of lead in dust will also have identical lead ingestion exposures, even though the loadings of dust and lead on indoor surfaces of the houses could vary substantially.

Atmospheric Lead Contributions to Indoor Dust Exposure

To assess the potential significance of airborne lead levels on surface loadings indoors, we prepared a screening-level analysis of the inputs of lead to floor surfaces from footwear tracking and deposition of suspended particles derived from the infiltration of outdoor particles through a building shell. Table 6-17 of the HHRA provides data on the fluxes of lead into houses situated in several Coeur d'Alene River basin communities. The geometric mean values range from 0.48 mg/m^2/day (in the lower basin/Caltaldo) to 4.28 mg/m^2/day (for Burke/Ninemile). These flux values, however, are only for the entryway mats—not floors in the interior of the houses sampled. Equivalent floor loading rates due to lead redistribution by foot traffic can be estimated by multiplying mat loading rates by the mat area (0.318 m^2; von Lindern et al. 2003b) to obtain a whole-house mass-loading rate (in mg/day) that is then divided by an effective house floor area. For a lead mat loading rate of 1 $mg/m^2/day$ and an assumed floor area of 100 m^2 (about 1,000 square feet), the resulting lead floor loading rate is about 3 $\mu g/m^2/day$.

The atmospheric deposition rate onto floor surfaces can be calculated as the product of a particle settling velocity and an indoor air concentration of lead. With a reference outdoor lead concentration of 0.10 $\mu g/m^3$ and an indoor level 0.03 $\mu g/m^3$ (based on the IEUBK default indoor/outdoor value of 0.3), the associated loading rates would be 0.18 and 1.5 $\mu g/m^2/day$, respectively, for gravitational settling velocities of 0.25 and 2.1 m/hour, based on outdoor-derived particles 1 and 3 m in diameter (Milford and Davidson 1985) and a density of 2 g/cm^3. These values would represent between 5% and 32% of the total flux from both foot traffic and surface deposition. The composite concentration of lead in dust resulting from tracked-in soils on floors and deposition will vary according to the amount of particulate matter introduced by the various transport processes and indoor sources as well as other indoor lead sources. Given that the levels of lead in ambient air would have been much higher within the box when the box version of the IEUBK model was initially being developed, it is conceivable that the community soil parameter is actually a surrogate parameter that represents airborne lead derived from soil resuspension.

The IEUBK model predicts that 10 µg of lead per gram of dust would be attributable to atmospheric lead at its default concentration (0.1 $\mu g/m^3$)—based on a simple ratio of the concentration of lead indoor dust to the level in outdoor air. Unfortunately, house dust is associated with many indoor surfaces, including nonfloor horizontal surfaces such as sofas, chairs, tables, beds, and the concentrations of lead in the associated dust loadings will vary, as will ingestion exposures related to hand-to-mouth contacts with those surfaces. In essence, the IEUBK exposure module is really an over-simplification of the transport and fate processes that control indoor lead, and it is time that more mechanistically based approaches are adopted so that the exposure component of the IEUBK model is commensurate with the lead biokinetic module.

CONCLUSIONS AND RECOMMENDATIONS

In this section, the committee provides several conclusions and recommendations regarding the application of the IEUBK model in the basin and general comments on model use, function, and associated EPA guidance. This section is intended to facilitate the development of the model as a scientific tool for more accurately assessing expected children's blood lead concentrations and support the model's future application at sites with lead contaminated soil. As provided in the statement of task (Appendix A), "the committee will strive to provide guidance to facilitate scientifically based and timely decision making for this site in the future." As such, the conclusions and recommendations herein are intended to guide future decision making and not to elicit a reconsideration of the ROD for the Coeur d'Alene River basin.

Conclusion 1

Multicompartment predictive blood lead models are powerful tools for pediatric lead-exposure risk assessments, for exploring lead risk management options, and for crafting remediation strategies. Their application to Superfund sites with environmental lead contamination is an important part of the CERCLA regulatory process.

Conclusion 2

Design and functioning of the IEUBK blood lead prediction model are consistent with current scientific knowledge, but improvements could be made. Specifically, substantial unaddressed uncertainty exists in three areas: model computations, input parameter values, and application of model computations to populations of individuals.

These uncertainties are discussed in this chapter and are summarized as follows: (a) Errors and inconsistencies exist in the documentation and computer code used for model implementation, as defined in this chapter and detailed in Appendix C. (b) Uncertainties in the input parameters of bioavailability and soil/dust ingestion rate can lead to significant variations in model predictions, as illustrated in Table 6-3. Although site-specific measures of bioavailability can be made, measuring ingestion rate parameters is far more difficult and there is little agreement on their measures of CT. Difficulty in making ingestion rate measurements suggests that many (if not most) model users will employ the model default values; these have not been reevaluated for more than 12 years. (c) Point estimates are projected to population distributions by making assumptions; application of a default probability density function parameter to a point estimate is not a proper way to define a population. Probabilistic exposure modules interfaced with the IEUBK biokinetic computations have been produced (for example, integrated stochastic exposure; [SRC 2003]) and could be subjected to the same validation and verification used for the IEUBK. These approaches would provide a more scientifically sound basis to project risk calculations for populations of individuals.

Recommendation 1

After correcting errors, EPA should recompile the IEUBK model source code using state-of-the-art algorithms for integration. Cornerstones of this program should be open access to the source code for the IEUBK model and any subsequent probabilistic exposure model implementation versions of it and a peer review process to ensure its accuracy.

Recommendation 2

EPA should undertake a significant effort to improve the knowledge base for soil/dust ingestion rates. Effort in this area will bring benefits for many other contaminant-exposure risk assessments for which soil ingestion is a significant exposure pathway.

Recommendation 3

EPA should proceed with implementating a probabilistic, stochastic exposure model version of the IEUBK and initiate the verification and validation process for it. This would substantially end the debate about application of default or site-specific GSD values for model use in establishing cleanup levels. In the interim, the agency should establish a comprehensive, uniform policy for use of site-specific GSD values to be utilized in model computations and should promulgate guidelines for its determination.

Conclusion 3

The IEUBK model was adequately and appropriately used in the Coeur d'Alene River basin, although the optimum application was not undertaken. Most importantly, site-specific bioavailability would have improved the application of the model, and better characterization of the physicochemical properties of the exposure materials would have enhanced the credibility of the results.

Conducting IEUBK model evaluations using solely default parameters, without their justification, has little utility because risk assessments should not be based on default parameters. The box model incorporated in both the HHRA and the ROD used a deviation from the IEUBK model default values for bioavailability. Given the wide range of values reported at other sites affected by mining (Table 6-5), it would seem that measurements of bioavailability in the Coeur d'Alene River basin should have been carried out. Furthermore, since natural soil processes can lead to alteration of mineral forms and conceivably either increased or decreased bioavailability over time, the likelihood and consequences of such changes should have been discussed.

At the very least, estimates of the lead-exposure impact would have been improved by determination of the lead concentrations in various soil particle size fractions. Such results would have improved interpretation of soil transport from outdoor to indoor environments. If the EPA had used their bulk analyses for the crustal elements iron and manganese as the committee did, a better justification would have evolved for the structure of the box model extension to the rest of the Coeur d'Alene River basin.

Recommendation

EPA should require that IEUBK model use for determining cleanup levels be supported by site-specific measures of bioavailability and that particle-size-range lead concentration determinations be undertaken. Increased emphasis should be placed on acquiring analytical metrics that quantify the strength of the lead-based paint source(s). In addition, EPA should emphasize the interpretive benefits for source attribution that derive from additional soil and dust bulk chemical measures (for example, aluminum, silicon, iron, manganese, and calcium) and encourage acquisition of such data where feasible. EPA should consider that ingestion rates might be site specific and undertake fundamental research aimed at addressing this hypothesis.

Conclusion 4

Alternative tools for assessing the validity of model predictions were underutilized in interpretations of model results. For example, other models were not used in the assessment. The committee's analysis of alternative models suggests that at this site the outcome of additional analyses would not have affected remedial decisions, but, had they been used as part of the HHRA for inclusion in the ROD, the scientific credibility of the decisions reached would have been enhanced.

Not using alternative analyses resulted in the loss of opportunity for expanding the scientific knowledge associated with application of predictive models to real world situations. Although some alternative interpretive tools were used in the development of an IEUBK model prediction regime, such as the structural equation modeling for the regression analyses in the HHRA, use of additional techniques would have helped solidify application of the box model as it was eventually constructed. For instance, the collection of mat dust lead (and other metal) concentrations and loading rates proved to be valuable additions to the RI/FS protocols. Appropriate analysis of the iron and manganese data would have provided additional supporting evidence upon which to base a soil contribution of 60% for indoor dusts. Similarly, a comparison of box model predictions by the IEUBK and the O'Flaherty models, showing identical cleanup-level determinations, would have highlighted the critical importance of uncertainty in bioavailability and ingestion rate parameters.

Recommendation

EPA should promote use and development of both deterministic and probabilistic multipathway uptake and pharmacokinetic models for lead as research tools and provide scientific maintenance for their continued development and improvement. This could substantially improve their application as regulatory instruments.

Conclusion 5

The committee finds that EPA guidance concerning specific use of the IEUBK model and additional use of blood lead studies is incomplete. The inherent uncertainties associated with model predictions coupled with the high value placed on the need for predictive capability in the protection of both present and future populations requires a more clear and comprehensive articulation of IEUBK model-use policy.

The 1998 OSWER directive fails, as described in this chapter, to give adequate guidance about what to do when BLLs and IEUBK model results disagree by a substantial margin. It states without clear justification that model results are to take precedence in these situations. Significant emphasis in the directive suggests that, where such disagreement exists, the blood lead study may be suspect. It is clear that blood lead observations may not always be representative of the population, may have been conducted at the wrong time of year, or may have been influenced by significant knowledge of lead hazards within a population. However, uncertainties may also exist in the IEUBK model results, where the relationship between soil and dust may not be well understood, the bioavailability of soil and/or dust may be unknown, or where factors, such as lead in paint, may be inadequately addressed in the model input parameter characterizations. Additional information for addressing such uncertainties could be provided by assays of soil and dust bioavailability, determining the presence or absence of lead-based paint, which can serve as a confounder in the model, and by analyses of additional metals such as arsenic, cadmium, and zinc as these metals may co-occur with lead and can improve the estimate of soil transfer to dust.

Recommendation

EPA's guidance on use of blood lead studies in conjunction with the IEUBK model needs clarification, especially on protocols for reconciling differences between modeled and observed blood lead values and for objectively considering the uncertainties associated with each. The guidance/policy should address the following points:

- Where blood lead observations are available, a systematic protocol for comparison of predicted and observed BLLs should be used for all risk assessments, and an acceptable level of variability between such results should be established to define "significant" differences.
- Criteria should be established upon which to judge whether or not the extant blood lead observations are representative of the community concerned, covering the full range of lead-exposure potential. If "significant" differences exist between observed and predicted blood lead values, such criteria would establish whether an additional blood lead study effort was required.
- Definitive guidelines for the conduct of blood lead studies should be established. The focus should be on the coherence of the joint data set covering the full range of lead exposure risks and the collection of blood lead data associated with that range of exposure.
- When model results and acceptable blood lead study observations do not agree, and when default IEUBK exposure values have been used for some or all of the modeling exercise, additional information should be collected to examine uncertainty in model inputs and to ensure that all exposure sources and lead uptake/intake rates have been adequately established for the specific site in question.
- Before development of a fully probabilistic IEUBK model, uncertainty in the GSD should be explored with the ISE, lead risk model, or another similar model to understand how it may depart from the default for a particular site.

Conclusion 6

The IEUBK model results should not be the sole criterion for establishing health-protective soil concentrations at mining megasites such as OU-3 of the Coeur d'Alene River basin, because model uncertainty and site complexity may interact in unexpected or unknown ways.

This chapter details a variety of specific challenges associated with IEUBK application to OU-3. The geographic area defined as OU-3 exhibits a great diversity of topography, land use practice, bedrock geology, ecologic community structure, and hydrologic regime. Consequently, one would expect the nature and extent of natural geochemical mineral alteration, soil digenetic processes, and sediment transport and deposition dynamics to vary accordingly. Such variations are manifest in the IEUBK box model predictions, which suggest regional differences between the upper and lower basin in lead bioavailability and possibly in other model operation parameters as well. By extension, it is likely that similar problems will arise at other sites where ecologic, geomorphological, and sociodemographic complexity of this nature exists. A comprehensive revision of the 1998 OSWER directive on model use, incorporating those issues just outlined, is needed to adequately address issues associated with geographic variability at large geographically heterogeneous sites.

Recommendation

Incorporate the IEUBK model in a negotiated and carefully communicated HHRA/ROD structure for which the primary prevention paradigm contains the four fundamental elements of

- Predictive capability (IEUBK or successors)
- Empirical results (blood lead study results)
- Economic feasibility
- Sustainable remediation (long-term remedy maintenance)

Each of these key elements is necessary for successful remediation, but the way they are weighted for the mutual satisfaction of all stakeholders may be different across the variety of contiguous spatial elements defined for the OU. Both risk assessment and risk management activities should be structured according to natural environmental system boundaries; they should not represent the aggregation of apparently applicable policies previously found to be successful for smaller, simpler systems.

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7

Ecologic Risk Assessment

INTRODUCTION

The ecologic risk assessment (ERA) for the Coeur d'Alene River basin (CH2M-Hill and URS Corp. 2001) was prepared under contract for the U.S. Environmental Protection Agency (EPA) Region X. The ERA is intended to support the remedial investigation/feasibility study (RI/FS) under the Comprehensive Environmental Response Compensation, and Liability Act (CERCLA) regulatory framework. The purpose of an ERA under CERCLA is to describe the likelihood, nature, and severity of adverse effects to plants and animals resulting from exposure to hazardous substances. In the case of the Coeur d'Alene River basin, the hazardous substances in question represent historic and continuing releases of dissolved and particulate materials from mining operations that have been distributed from the upper and middle basin throughout the study area. The study area addressed in the ERA includes the Coeur d'Alene River and associated tributaries, Lake Coeur d'Alene, and the Spokane River downstream to the Spokane arm of Lake Roosevelt. Although performed under the direction of EPA, the ERA included stakeholder input through the Coeur d'Alene Basin Ecological Risk Assessment Work Group.

EPA used the results of the ERA as inputs to the RI/FS report and the record of decision (ROD) (EPA 2002) for the basin. The ERA addressed risks to plant and animal species exposed to contaminated surface water, sediment, and soil throughout the basin. For contaminated media that were found to pose significant risks, the ERA proposed preliminary remediation goals (PRGs)[1] for use in making remedial decisions at the site. Many of the actions included in the proposed remedy (as documented in the ROD) were specifically intended to reduce or eliminate risks to ecologic resources in the basin.

In the statement of task, the committee is directed to assess the adequacy and application of EPA's Superfund guidance in terms of currently available scientific and technical knowledge and best practices. Specifically, with regard to the Coeur d'Alene River basin site, the committee is to consider the scientific and technical aspects of the following:

- Assessing the ecologic risk from waste-site contaminants in the context of multiple stressors.
- The necessary data and appropriate analyses to estimate the ecologic risks attributable to waste-site contaminants—specifically, how well these analyses were applied to estimate the risks, including the effects of lead on migratory fowl.
- Whether risks attributable to sources other than mining and smelting activities were adequately analyzed.

In addressing the charge, this chapter reviews the Coeur d'Alene River basin ERA with respect to the following criteria:

- Consistency with agency guidance for ERAs
- Consistency with best scientific practice in ERA
- Validity of conclusions

In addition, the chapter addresses the extent to which the proposed remedy is consistent with the conclusions of the ERA and the likelihood that the selected remedy will significantly improve ecologic conditions in the Coeur d'Alene River basin.

In performing its review, the committee found it neither necessary nor appropriate to evaluate all of the underlying scientific studies or to identify all of the aspects of the ERA that could have been improved. The committee recognizes that at a site as large and as obviously disturbed as the Coeur d'Alene River basin, there is no limit to the number or types of data-collection activities that could have been conducted. Similarly, any ERA of the scope and complexity of the Coeur d'Alene River basin ERA could be

[1]PRGs are proposed concentrations of materials in soil, sediment, and surface water below which adverse effects are expected to be absent or within defined limits. PRGs are provided to risk managers to assist in making decisions for remedial action (CH2M-Hill and URS Corp. 2001).

improved through better data analysis techniques and more thorough documentation. In reviewing this ERA, the committee chose to limit its review to the studies and analyses that were critical to supporting the conclusions and management recommendations.

CONSISTENCY OF THE ERA WITH EPA GUIDANCE CONCERNING THE ERA PROCESS

EPA's primary guidance on ERA can be found in the following documents: *Guidelines for Ecological Risk Assessment* (EPA 1998), *Ecological Risk Assessment Guidance for Superfund* (EPA 1997), and *Ecological Risk Assessment and Risk Management Principles for Superfund Sites* (EPA 1999). The Superfund program office has also developed secondary guidance on specific components of Superfund ERAs; all of these are available online. This section of the committee's report addresses whether or not EPA followed its own guidance in performing the ERA. The technical adequacy of the data and analyses used in the ERA are addressed below ("Evaluation of the ERA in the Coeur d'Alene River Basin").

Description of the ERA Process

It must be recognized at the outset that the ERA process followed by EPA is much less explicit than the human health risk assessment process. EPA's ERA guidance focuses primarily on the process used to design the assessment, evaluate the data, draw conclusions, and communicate the conclusions to risk managers. The overall process consists of the three steps depicted in Figure 7-1.

Problem Formulation

During problem formulation, the risk assessment team synthesizes information concerning the site being investigated, including the history of activities at the site, nature and spatial scale of the contamination, the types of habitats and organisms exposed, and the fate and effects of the chemicals identified at the site. Risk managers and stakeholders are consulted to identify ecologic management goals for the site. From the management goals and the types of organisms at risk, the risk assessors, risk managers, and stakeholders develop a set of "assessment end points," which define the specific types of organisms ("entities") and characteristics ("attributes") to be addressed in the ERA. An assessment end point for a risk assessment could be a specific fish or wildlife species (for example, bull trout or tundra swan) or a valued habitat type (for example, floodplain lake). Corresponding attributes could include mortality or growth in the case of a species or

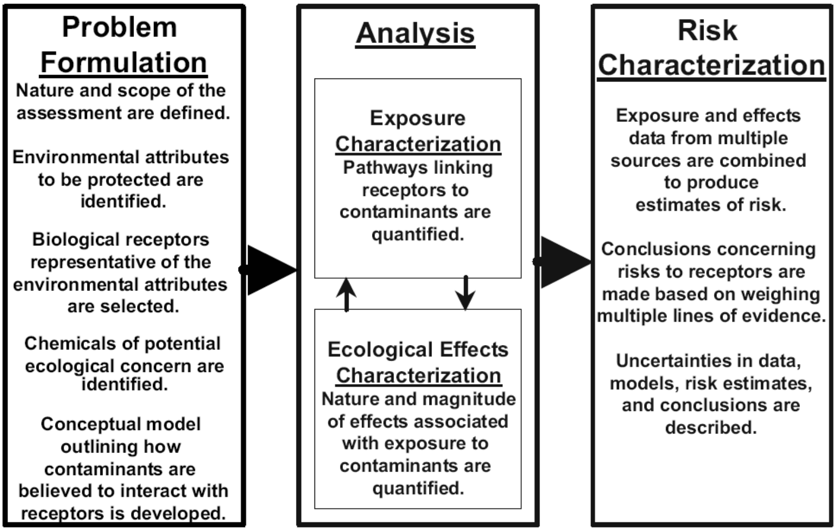

FIGURE 7-1 ERA process. SOURCE: CH2M-Hill and URS Corp. 2001.

plant community composition in the case of a habitat type. Once the assessment end points have been identified, the assessment team develops a conceptual model that shows the causal links between the hazardous substance releases and the assessment end points. A typical conceptual model would include the source of the hazardous substances that have been (or potentially could be) released, the fate and transport pathways through which the assessment end points are (or could be) exposed, and the adverse effects on those end points that are occurring (or could occur) as a result of the exposures. Once the assessment end points and conceptual model have been developed, the risk assessment team develops an analysis plan that identifies the specific types of data needed to complete the assessment and the methods that will be used to analyze the data and draw appropriate conclusions.

Analysis

During analysis, the risk assessment team implements the analysis plan developed during problem formulation. Depending on the circumstances, analysis may or may not include collection of new data. For chemical stressors, analysis typically is differentiated into separate "exposure" and "effects" components. In exposure analysis, a combination of field measurements and mathematical exposure models are used to estimate spatial and temporal patterns of exposure to the end point species and communities identified in problem formulation. In effects analysis, a combination of literature-derived toxicity information, toxicity tests performed on organisms present at the site, and field studies of the characteristics of exposed individuals, populations, and communities are used to estimate the ecologic effects of chemical exposures. Effects analysis can include development of exposure-response relationships for different types of effects and evaluation of evidence that particular types of adverse effects are caused by the stressor(s) being evaluated. EPA's guidance documents identify general categories of data and models that could be used in the analysis phase of an ERA, but do not specify which types of data or models should be used for different types of assessments. All such decisions are left to the assessment team, although the team's decisions ultimately are subject to review both inside and outside the agency.

Risk Characterization

In this process, the assessment team integrates the results of the exposure and effects analyses and draws conclusions about the magnitude and extent of risk to the end points of concern posed by the stressor(s) being evaluated. At least for chemical stressors, risk characterization includes both a quantitative and a qualitative step. In the quantitative step, termed "risk estimation," the assessment team develops numerical comparisons between exposure concentrations or doses and exposures expected to cause adverse effects. The comparisons are most often deterministic—for example, comparisons between mean or maximum exposure concentrations and single-valued toxicity benchmarks such as the lowest-observed-effect levels (LOELs). The comparison also can be probabilistic, where the exposure estimate, the effects estimate, or both are expressed as a probability distribution. Probabilistic methods are often used to estimate the fraction of an exposed population that may be exposed to a concentration or dose higher than a given toxicity benchmark. Probabilistic methods may also be used to develop risk curves that show probabilities of effects of differing magnitude.

If population- or community-level risks are being addressed, a mathematical model of population or community dynamics may be used to express the risk in terms of higher-level effects such as percent reduction in abundance, increased risk of extinction, and change in community composition. It should be noted that none of these techniques are specifically required by either the agency-wide guidelines or the Superfund guidance. The choice of which techniques will be used is left to the risk assessment team and the responsible project manager and is normally documented in a work plan prepared prior to the initiation of data collection.

The qualitative phase of risk characterization, which is termed “risk description” in the agency-wide guidelines, involves interpreting the magnitude, significance, and management implications of the quantitative risk estimates. Where multiple lines of evidence have been developed, risk description involves reconciling any inconsistencies between different types of evidence. In the case of Superfund ERAs, risk characterization also includes the development of PRGs intended to aid risk managers in designing an appropriate and effective remedy. PRGs are estimates of concentrations in environmental media that are expected to protect biota at the site from adverse effects of chemical exposure. The Superfund guidance recommends that both lower-bound and upper-bound values should be developed for each environmental medium of concern. The lower bound would be based on consistent conservative assumptions and no-observed-adverse-effects levels (NOAELs). Contaminant concentrations as low or lower than this lower bound should cause no adverse ecologic effects. The upper bound would be based on observed or predicted impacts and would be developed using less-conservative assumptions, site-specific data, lowest-observed-adverse-effects levels (LOAELs), or an impact evaluation. Contaminant concentrations as high or higher than the upper bound could cause adverse ecologic effects.

Evaluation of the ERA in the Coeur d’Alene River Basin

The following subsections evaluate EPA’s ERA for the Coeur d’Alene River basin with respect to consistency with agency guidance.

Problem Formulation

Section 2 of the ERA, which documents the problem-formulation step, begins with a statement of management objectives and then derives assessment end points from those objectives and develops a conceptual model. The management objectives were developed with input from an ERA work group consisting of representatives of the states of Idaho and Washington; the Coeur d’Alene, Spokane, and Colville tribes; the U.S. Fish and Wildlife Service; and any other governmental or nongovernmental organizations that wished to participate.

Contaminants of potential ecologic concern (COPECs) were selected using a two-step procedure. In the first step, the available data on concentrations of chemicals in soil, sediment, and surface water were subjected to a data-quality review. Resultant values were then screened against soil/sediment background levels and ambient water-quality criteria (AWQC).

The assessment end points include individual species, biological communities, and physical habitat characteristics that could be adversely affected by mining-related hazardous substances. Taxonomic groups of organisms addressed included birds, mammals, fish, amphibians, and plants. Representative species belonging to each group were identified for each Conceptual Site Model (CSM)[2] unit and habitat type. The measures of mining-related effects selected for evaluation included reductions in survival, reproduction, growth, and abundance. For migratory birds and "special status" species (that is, threatened, endangered, or culturally significant species, or state or agency species of special concern) effects of mining-related hazardous substances on the health of individual organisms were also evaluated. For migratory birds and special status species, effects were considered to be adverse if any of the attributes of interest was observed or predicted to be adversely affected. For other species, effects were considered adverse only if a 20% or greater adverse change in an attribute of interest was observed or predicted. The use of a 20% effects level as a default de minimis criterion for ecologic significance was first proposed by Suter et al. (1995), on the grounds that this value is consistent both with EPA's regulatory practices and with the practical detection limits of typical toxicity testing protocols and field survey methods.

In addition to evaluating effects of mining-related hazardous substances on individual species, the ERA also evaluated effects on aquatic and terrestrial plant and invertebrate communities, soil processes, and physical/biological characteristics. Community-level effects addressed included effects on community composition, abundance, density, species diversity, and community structure. Physical/biological characteristics evaluated included habitat suitability indices, spatial distributions of healthy riparian communities, sediment deposition rates, and turbidity. Changes in these characteristics were addressed to account for secondary effects of hazardous substance releases (for example, degradation of riparian habitat resulting from toxic effects of hazardous substances on vegetation).

Section 2 concludes with lists of COPECs and receptor species to be evaluated. Separate lists of COPECs are provided for each medium, and separate lists of receptors are provided for each of six habitat types present in the basin.

The one component that is *not* included in the ERA is an analysis plan. Such a plan would normally be developed at the conclusion of the problem-formulation phase of an ERA. Data gaps identified during the development of the analysis plan would then be filled prior to implemen-

[2]The study area was divided into five CSM units in the ERA. These roughly correspond to the high-gradient watersheds in the upper (eastern) basin (CSM 1), the mid-gradient watersheds in the middle basin (CSM 2), the expansive depositional floodplain and lateral lakes area in the lower basin (CSM 3), Lake Coeur d'Alene (CSM 4), and the Spokane River (CSM 5); see Chapters 3 and 4 of this report for further discussion.

tation of the remaining steps in the ERA. The rationale for bypassing the analysis plan (CH2M-Hill and URS Corp. 2001, pp. 1-3 to 1-4) was that a large number of investigations had already been performed within the Coeur d'Alene River basin. These investigations included sampling of environmental media and biological tissues, bioavailability tests and toxicity tests to a wide variety of biota, and numerous biological surveys. As documented in Appendix A to the ERA, EPA used a series of workshops and meetings with stakeholders to identify additional data needs. It is possible that some of the methods used in the ERA may have been selected because they were consistent with existing data rather than because they were the best approach for quantifying risks to the assessment end points. Also, because the expansion of the Superfund site vastly increased the geographic extent of the site, ecologic effects in some areas may have been incompletely described.

Although in most respects the problem formulation step of the Coeur d'Alene River ERA appears to be consistent with the requirements of guidance, the failure to develop an analysis plan may have contributed to the continued existence of data gaps (discussed later in this chapter) that limit the value of the ERA results for guiding remedy design.

Analysis

Section 3 of the ERA, which documents the analysis phase of the risk assessment, provides information on the measures of exposure and effects used in the ERA.

For the exposure analysis, Section 3 identifies, for each CSM unit and habitat type, the routes by which each receptor could be exposed to the COPECs identified in the problem-formulation step. Data on COPEC concentrations in each medium serving as a source of exposure were summarized. For aquatic biota and soil invertebrates, the media concentrations provide direct estimates of exposure. Because wildlife receptors can be exposed to COPECs via direct and indirect pathways (ingestion of soil/sediment, water, and contaminated biota), the exposure assessment for these receptors used models to quantify multimedia exposures to COPECs. The data and models used are documented in Appendices A-D of the ERA.

The effects analysis utilized available data derived from published literature on the toxicity of individual COPECs to terrestrial and aquatic biota; tests of the toxicity of soil, sediment, and water collected in the Coeur d'Alene River basin; laboratory dosing studies performed to simulate waterfowl exposures to COPECs; and field studies performed in the basin. The toxicity data were used to define, for each receptor, a range of toxicity reference values (TRVs) for comparison with the estimated exposure concentrations or doses from the exposure analysis. Data sets and procedures used to develop these TRVs are documented in Appendices E and F of the ERA.

All the data and exposure models used in the analysis phase are identified in guidance as being appropriate for use in ERA; hence, Sections 3 and 4 of the ERA also appear to be consistent with available guidance.

Risk Characterization

The risk characterization section of the ERA (Section 4) synthesizes the exposure and effects analyses documented in Section 3. Both a risk estimation and a risk description component are included. In the risk estimation step, the exposure estimates for each receptor were compared with the TRVs documented in Section 3. For birds, mammals, and aquatic biota, point estimates of exposures were compared with point estimates of effects. For amphibians, terrestrial plants, soil invertebrates, and soil processes, full distributions of exposure and effects estimates were compared, with the risk represented by the percent overlap of the two distributions. Risk estimates derived from site-specific toxicity tests and field surveys were evaluated by comparison with reference conditions. All of the techniques used are identified in the agency-wide guidelines and in the Superfund guidance as being valid risk-estimation techniques.

The risk description evaluated all the lines of evidence for each receptor group. Greater weight was given to site-specific toxicity tests and field surveys than to risk estimates based on literature-derived toxicity data. Strength of risk conclusions was considered high if multiple lines of evidence, including site-specific field surveys and toxicity tests, were available for a given receptor and all lines of evidence were in agreement. Risk conclusions were considered to be of moderate strength if the data consisted of literature-based toxicity and one other line of evidence. If only literature-based toxicity data were available, the strength of risk conclusions was rated as low.

For each habitat, the risk characterization identified the receptors at risk and the COPECs posing the greatest potential risk to each receptor. The risk description section of the ERA also includes a qualitative evaluation of secondary effects of mining-derived hazardous substances on habitat quality. Uncertainties affecting all components of the risk assessment are summarized in a separate section on uncertainty analysis.

Risk calculations are documented in Appendices G-I of the ERA. These calculations appear to be consistent both with the formal requirements of guidance and with the procedures for risk characterization documented by Suter et al. (2000).

As discussed later in this chapter, the PRGs for aquatic organisms in sediment and water provided in the ERA are lower-bound thresholds as defined in the Superfund guidance. No upper-bound thresholds are provided in the ERA. In this respect, the risk characterization component of the ERA does not conform to the Superfund guidance. In all other respects, EPA's risk characterization is consistent with agency guidance.

CONSISTENCY OF THE ERA WITH BEST SCIENTIFIC PRACTICE

EPA guidance on ERAs focuses on procedures rather than on the quality or quantity of the data and models used. Therefore, beyond considering consistency with guidance, it is also necessary to evaluate, from a technical perspective, whether the assessment was properly designed and conducted and whether the conclusions are adequately supported. This section of the committee report evaluates the consistency of the ERA with best scientific practice in ERA. The question here is not whether EPA guidance was followed but whether the site-specific studies performed to support the assessment were properly designed and conducted and whether the supporting scientific literature was properly interpreted.

Problem Formulation

Range of Stressors Evaluated

All the stressors evaluated as COPECs are mining-related metals. Section 2.4 of the ERA report discusses the data and methods used to select COPECs for the ERA. The process involved examining all data available both from historical investigations and from sampling conducted specifically to support the RI/FS. These sources are summarized in Table 2-9 of the ERA report (CH2M-Hill and URS Corp. 2001). Media evaluated included soil, sediment, water, and biological tissues. Evaluation of the data included a data-quality review, data reduction, and association of sampling locations with CSM units and habitat types. Zinc is clearly the metal with the largest ongoing discharges in the Coeur d'Alene River basin, followed by lead and cadmium. Most zinc and cadmium are released and transported as dissolved metals. Most lead is present in particulate form and is transported with sediment, especially during flood events. As a result of historical flood events, particulate lead has been deposited in streambeds, lakes, riparian zones, and floodplains throughout the lower basin, Lake Coeur d'Alene, and the Spokane River. Based on the environmental concentration data and comparisons to screening levels, as described above, the selection of COPECs was reasonable.

Non-mining-related stressors were not explicitly considered in the ERA. These types of stressors include habitat modification, infrastructure development (roads and railways), and stream channelization. Mining-related stressors besides metals, particularly sediments associated with mining and milling activities that were released to streams in vast quantities, also were not explicitly addressed in the ERA. As stated in the ERA (CH2M-Hill and URS Corp. 2001, p. 2-39),

> The EcoRA [ecologic risk assessment] does not attempt to quantify the relative effects of mining activities and other stressors. As part of the natural resource damage assessment (NRDA) process, a determination and initial quantification of mining-related injury to natural resources has been completed.

Some mention is made of the potential effects from non-mining-related stressors. Figure 2-16 in the ERA illustrates how non-mining-related stressors could affect the receptors evaluated in this ERA and identifies resource management, fire, waterborne log transport, watershed management, roads and railroads, hydraulic modification, housing and urban development, and septic/waste disposal systems as potential non-mining-related stressors. Appendix K of the ERA, which evaluates the secondary effects of mining-related hazardous substances (for example, loss of riparian habitat and stream bank stability), concludes that non-mining-related stressors (development, road building) also contribute to these secondary effects, but the relative contribution of mining-related hazardous substances (presumably metals) and other stressors cannot be quantified. According to the ERA (CH2M-Hill and URS Corp. 2001, p. 2-40), physical disturbances unrelated to mining were accounted for in the ERA by comparing site-specific information on biota and habitats from mining-affected areas with information on biota and habitats from non-mining-affected reference areas believed to be affected by the same types of non-mining-related disturbances.

The consideration of areas with similar levels of infrastructure as a reference is appropriate, especially in light of the preponderance of evidence relating to the ecologic effects of metals in the Coeur d'Alene River basin environments. Because the purpose of ERAs performed at Superfund sites is to evaluate risks associated with releases of hazardous substances, the focus on metals as stressors is reasonable. Impacts of physical disturbances, including non-mining-related disturbances, would still have to be considered during remedy selection and implementation, but they need not be explicitly addressed during the risk assessment component of the RI/FS process.

Characterization of Existing Ecologic Conditions

The Coeur d'Alene River basin is a complex ecologic zone consisting of the Coeur d'Alene River and tributaries, lateral lakes, Lake Coeur d'Alene, and the Spokane River. The question is, was a reasonable survey conducted to identify the aquatic and wildlife resources in these various habitat zones for evaluation, and was this reported in the ERA?

Section 2 of the ERA lists the groups of receptors of concern within each CSM unit and habitat type within the basin, summarizes linkages between these receptors and habitat characteristics that could indirectly be affected by hazardous substance releases, and lists representative plant and animal species and community types found within each CSM unit and habitat types.

As documented in Section 2.3 of the ERA, ecologic conditions within the upper basin were characterized based on the many ecologic investigations conducted since the 1980s. Many of these studies were performed to support a Natural Resource Damage Assessment for the Coeur d'Alene River basin (Stratus 2000). In the lower basin, extensive surveys (Audet et al. 1999) have been conducted to document waterfowl mortality. These studies, in combination with necropsy findings, have characterized the acutely toxic effect of metals-contaminated sediments on waterfowl. Far less information about the aquatic communities in the lower basin is available. As stated in the ERA (CH2M-Hill and URS Corp. 2001, p. 2-24), "Fish population assessments conducted in the main stem confirm the presence of numerous fish species. However, the information gathered is too limited to use to draw conclusions about the current status of fish populations." For macroinvertebrate communities, the ERA concludes "the current status of the macroinvertebrate community [in the main stem of the river] cannot be determined at this time." The limited data on the status of these communities preclude a complete assessment of the impact of metals from mining-derived sources. A similar situation exists for aquatic communities in Lake Coeur d'Alene. This recognition is not new; in a 1988 report (Hornig et al. 1988), EPA recommends that

> Future assessment should further document status and condition of populations, particularly of those fish that inhabit the mainstem Coeur d'Alene and lateral lakes and the salmonids that use the Coeur d'Alene River for migration to spawning areas upstream of the South Fork confluence.

The ERA could not evaluate ecologic risk to every organism within the Coeur d'Alene River basin. Receptors of high ecologic or societal value, or those that were believed to be representative of broader groups of organisms, were selected for evaluation. The receptors for the exposure analysis were chosen to represent a trophic category and particularly feeding behaviors, such as various bird feeding behaviors, that would represent different modes of exposure to the chemicals of potential concern—in particular lead—for wildlife. The following criteria from the ERA were used to select potential receptors (EPA 2002, p. 7-21):

1. The receptor utilized habitats present in the basin.

2. The receptor is considered important to the structure or function of the ecosystem of the Coeur d'Alene River basin.

3. The receptor is statutorily protected, in particular those that are identified as threatened or endangered species or migratory birds that have a higher level of statutory protection.

4. The receptor is reflective and representative of the assessment end points for the Coeur d'Alene River basin.

5. The receptor is known to be either sensitive or highly exposed to the toxic metals in the Coeur d'Alene River basin.

Section 2.3 of the ERA also identifies federally listed and state-listed or candidate species potentially present within the study area. This section also summarizes previous studies of biological conditions and metal contamination throughout the basin. This information appears to be adequate to identify representative species and communities for use in the risk assessment, although not sufficient to fully characterize risks to all of these receptors.

Management Goals, Assessment/Measurement End Points, and Conceptual Model

EPA consulted with other agencies and stakeholders in development of the following two management goals for the site:

- Maintenance (or provision) of soil, sediment, water-quality, food-source, and habitat conditions capable of supporting a "functional ecosystem" for the aquatic and terrestrial plant and animal populations in the Coeur d'Alene River basin.
- Maintenance (or provision) of soil, sediment, water-quality, food-source, and habitat conditions supportive of individuals of special status biota (including plants and animals) and migratory birds (species protected under the Migratory Bird Treaty Act) that are likely to be found in the Coeur d'Alene River basin.

The risk assessment team then developed assessment end points at the individual, population, community, and habitat/ecosystem/landscape levels intended to support these goals.

Individual-level end points included migratory bird species and threatened or endangered species covered under the second of the above goals. These types of species are protected by statute (the Migratory Bird Treaty Act and the Endangered Species Act), and detrimental effects on the health, survival, growth, or reproduction of any individual belonging to such species are considered adverse. The remaining assessment end points relate to the first goal. Population-level assessment end points included various species of birds, mammals, fish, amphibians, and plants. For these species, effects were considered adverse if key population attributes such as reproduction, survival, growth, or abundance were to be reduced by 20% or more or if greater than 20% of the individuals present in a population could be affected. Community-level end points included aquatic and terrestrial plant communities and aquatic and terrestrial invertebrate communities. For these end points, individual species were not identified. Effects were considered adverse if there was greater than a 20% reduction in key community-level attributes. Habitat/ecosystem/landscape-level end points included soil process and physical and biological landscape attributes. Effects on soil processes were considered adverse if measures of soil microbial function or other measurable soil processes were reduced by 20% or more. Effects on physical and biological characteristics were considered adverse if any measurable level of degradation of habitat structure occurred.

Specific measures of exposure defined for the site included concentrations of chemicals in sediment, soil, surface water, and biota. The types of assessment end points found in each CSM unit and habitat type were summarized (CH2M-Hill and URS Corp. 2001, Table 2-1), and a variety of specific attributes that could be adversely affected by chemical exposures were identified for each assessment end point. Indirect effects of chemicals that occur as secondary effects of alterations in physical and biological ecosystem characteristics were discussed.

A conceptual model was developed (CH2M-Hill and URS Corp. 2001, Figures 2-15 to 2-21) showing, for each CSM unit, the linkages between sources and assessment end points. Both chemical and physical effects of mining are included in these figures.

It could be argued that the extensive list of assessment end points developed for this ERA is excessively complex, given the obvious and well-documented impairment of aquatic and terrestrial biota throughout the basin. However, these end points are clearly related to the management goals and appear to be sufficient to support the subsequent analysis of ecologic exposures and effects.

Analysis

The analysis phase of an ERA includes consideration of all relevant aspects of the environmental transport, fate, and effects of a hazardous substance release, as identified in the problem-formulation section of the risk assessment. The analysis is conceptually separated into an "exposure" assessment and an "effects" assessment, although these two assessment components are necessarily closely linked. This section of the report addresses the technical adequacy of the exposure and effects analyses documented in the ERA.

Exposure Analysis

This section addresses the adequacy of the exposure assessment component of the ERA. Questions to be addressed include whether all the significant exposure pathways were identified, whether physical transport processes and environmental transformations were adequately characterized, and whether seasonal and spatial variability were adequately addressed.

Environmental Transport

The ERA was developed in tandem with the RI (URS Greiner, Inc. and CH2M Hill 2001a), and, as stated in the ERA, "some information briefly presented in the [ERA] will be presented in greater detail in the RI/FS" (CH2M-Hill and URS Corp. 2001, p. 1-1). In this case, the RI describes the magnitude and location of metals contamination in the basin and presents information about their disposition (see Chapter 4 of this report for evaluation of the RI). Extensive previous studies over a period of several decades and those conducted in support of the RI inform the characterization of contaminants and their transport through the basin. A database of metals concentrations in surface water was compiled for the RI from which expected values for metals loading through the basin were determined.[3] Metals loading diagrams are presented in the ERA and demonstrate that the original Bunker Hill Superfund site (the box) is the portion of the system contributing the largest loads of dissolved zinc, followed by Canyon and Ninemile Creeks. In contrast, the largest contributor of total lead is the broad depositional valley downstream of Cataldo.

Although this information provides a concise summary of expected loading, it is less useful for understanding the frequency, intensity, and duration of episodic extreme events (for example, flooding that mobilizes large amounts of lead-contaminated sediments or prolonged low-flow conditions containing high concentrations of dissolved metals). These events likely contribute significantly to potential toxic effects in ecologic systems in the basin. For example, Audet et al. (1999) described the impacts of severe flooding events on waterfowl:

[3]The database of environmental metals concentrations used to provide expected loading values in the RI is not the same database used to estimate exposure point concentrations in the ERA (although similar information is presented in both databases). The committee did not seek to evaluate the differences in these two data sets, except as noted below in the section "Dose Quantification."

> Large die-offs (>100 dead birds reported) occurred in 1953, 1954, 1982, 1996, and 1997. Some of these years were associated with high water events followed by low water conditions allowing for newly deposited sediments to be more readily available in waterfowl feeding areas. Beckwith (1996) reported the February 1996 flood event as the second largest flood event recorded in the Coeur d'Alene River basin based on gauge data collected from 1911 to present.

Environmental Chemistry

Speciation is a fundamental aspect of metal risk assessment for both aquatic and terrestrial systems. It is widely recognized that mobility, bioavailability, and toxicity can vary dramatically as a function of metal species. As a consequence, exposure and risks may be over- or underestimated if chemical speciation is not considered. In the Coeur d'Alene River basin, the metals arise from primary sources (such as tailings) or secondary sources (such as metals that have been redeposited) as a result of biotic or abiotic processes. In mine tailings, the zinc and lead, which are of primary concern, are largely present as sulfides. Sulfide minerals have low mobility, but mobility is greatly enhanced through oxidation of the sulfides to form secondary mineral species with much higher solubility. Changes in chemical form likely occur as metal-containing particles are eroded from tailings particles, deposited in the riverbed, and then are repeatedly resuspended and redeposited in the river channel and floodplain.

Bioavailability is discussed in Section 3.1 of the ERA, but the ERA did not address variations in bioavailability related to metal speciation.[4] For example, lead bioavailability to birds was assumed to be 50%, based on a feeding study conducted by Hoffman et al. (2000) in which contaminated wetland sediments were fed to mallard ducklings. However, the sediments used in the feeding trials, which likely would have been anoxic in situ, were dried and consequently subjected to oxidation before being used in the tests. Upon aeration, much of the sulfide and iron in the sediment would have oxidized and the lead released from its sulfidic form would have sorbed to the newly formed iron oxide. This change in speciation would have substantially enhanced the bioavailability of the lead, and therefore the bioavailability factor developed from this study would have overestimated the bioavailability of the sulfidic lead present in undisturbed wetland sediment. Overestimation of bioavailability in turn would lead to an excessively conservative estimate

[4]In fact, EPA provided to the committee that "We note that, because of the site-specific information on bioavailability (Hoffman et al. 2000 for ecologic receptors and the large body of paired blood lead and environmental data for children that was developed as part of the Bunker Hill Box residential areas cleanup), understanding speciation was not necessary to evaluate health risks" (EPA 2004).

of the remediation goal required to protect waterfowl from lead ingestion. The degree of overestimation would depend in part on the relative consumption of anoxic vs. oxidized sediment by waterfowl.

Dose Quantification

In general, EPA's exposure assessment adequately addressed exposures to aquatic biota; however, the committee still has questions about the procedures EPA used to select the data used in the ERA. Multiple studies have been conducted to document metals contamination in the Coeur d'Alene River basin and have resulted in a large database of metals concentrations in various media at various locations over time. This database from numerous sets of historical data collected by EPA, the U.S. Geological Survey (USGS), U.S. Fish and Wildlife Service, Bureau of Land Management, University of Idaho, and other investigators underwent "data qualification review and reduction protocols," described in the ERA (CH2M-Hill and URS Corp. 2001, Section 2 and Appendix A). This process essentially winnowed down a larger database into a smaller one used within the ERA and from which summary statistics for each habitat within each CSM unit could be determined. The committee could not conduct a case-by-case review of this process and the database and resulting statistics; however, it was determined that the data-reduction technique eliminated chemical data for surface water in the main body of Lake Coeur d'Alene.[5] The end product of the data-qualification process is important as these data are used in the ERA to determine risk on the basis of water concentrations of the metals (CH2M-Hill and URS Corp. 2001, p. 4-21), and it is this risk that is considered in the weight-of-evidence analysis in risk characterization (see below). As a result, this line of evidence was not available for consideration on Lake Coeur d'Alene.

[5]Table A5-4 of the ERA (CH2M-Hill and URS Corp. 2001, Appendix A) presents summary statistics for the data retained for further analysis in the ERA. Data on surface-water zinc concentrations for segment 2 (the main body of the lake) are not presented, whereas segments 1 and 3 of Lake Coeur d'Alene (representing the St. Joe River arm and Wolf Lodge Creek arm of the lake, respectively) are presented in the summary table (see CH2M-Hill and URS Corp. 2001, Figure 2-13 for a map). The arithmetic mean dissolved zinc levels in these segments are 9.93 μg/L (segment 1) and 8.07 μg/L (segment 3). Apparently, many of the data for these segments are not from the lake. For instance, data for the St. Joe segment (segment 1) are at least partially derived from the USGS sampling station in Calder, approximately 30 miles upriver from Lake Coeur d'Alene, and St. Maries, Idaho, approximately 8 miles upriver from Lake Coeur d'Alene. Data for the main body of the lake are not presented, although the lake has been the subject of numerous water-quality studies. For example, dissolved zinc data from 1999 collected by USGS are available online (USGS 2005). In contrast to the ERA, the RI does present concentration data for dissolved zinc for segment 2 of the lake (average = 174 μg/L for segment 2 [URS Greiner, Inc. and CH2M Hill 2001b, attachment 3]).

Information on exposures to fish and benthic invertebrates in Lake Coeur d'Alene is very limited, especially regarding sediment effects to benthic fauna and the bioavailability of sediment-bound metals. Although sediment metals and metal concentrations in the overlying water have been sampled, there is a paucity of data on the dynamic interaction between invertebrates, the deposited sediments, and the potential for re-entrainment into the water column. This remains a clear need for further investigation, as any management program must understand the ramifications of potential changes in the abundance and functional activity of the lake benthos.

The primary metal exposure routes for fish and benthic invertebrates in the Coeur d'Alene River and tributaries are through aqueous exposure over the gills or through dietary (food chain) uptake (see Box 7-1). The exposure

BOX 7-1 Metals in the Food Chain and Diet of Trout in the Coeur d'Alene River

In addition to exposures to metals from water, trout can also be exposed through consumption of organisms or material that has elevated metal content. In the Coeur d'Alene River system, these types of dietary exposures have been characterized.

Farag et al. (1998) observed an accumulation of metals in biofilm (algae, bacteria, and detritus attached to the substrate), invertebrates, and whole fish in mining-affected portions of the South Fork of the Coeur d'Alene River compared with reference sites. This study demonstrated that concentrations accumulated to the highest levels in biofilm and sediments, followed by invertebrates and fish, indicating that constituents of the aquatic food chain contain elevated metals concentrations, which can be passed on to trout. Mean lead concentrations were highest in samples collected from the Ninemile and Canyon Creek sediments with biofilm lead > 25,000 μg/g and 12,000 μg/g, respectively. Mean lead concentrations in whole perch collected in the lower basin were much lower than those measured in sediments, biofilm, or invertebrates; however, body burdens of lead were measured at greater than 50 μg/g. Burdens of cadmium, lead, and zinc were also elevated in trout kidney and gill compared with the reference streams.

Woodward et al. (1999) compared biota from sections of the South Fork with reference sites in the St. Regis River that were morphologically similar. They determined that "there was a significantly higher concentration of cadmium, copper, lead and zinc in the food web (water, sediment, biofilm, and benthic invertebrates) of the South Fork over that of the St. Regis River and higher concentrations in the food web components were also reflected in significant exposure of trout gill, liver, and intestine."

Farag et al. (1999) demonstrated that cutthroat trout fed metals-contaminated benthic invertebrates from the main stem and South Fork of the Coeur d'Alene River accumulated significantly greater body burdens of zinc compared with those fed a diet from the North Fork (used as a reference). The study indicated negative biochemical, histologic, and behavioral effects, and decreased growth as a result of metals in the diet. The researchers emphasize the importance of these exposures to young fish whose diet consists primarily of benthic invertebrates.

is chronic, as groundwater and surface sources continually add cadmium, lead, zinc, and other metals to the river. Exposure point concentrations in the ERA for surface water are dissolved metal concentrations, whereas exposure point concentrations for sediment are reported as total metals in sediment. Substantial databases of concentrations in these media exist for waters in the basin. Concentrations of metals in fish liver and kidney, representing "internal exposures" are also presented. A mathematical relationship was developed between sediment concentrations and concentrations in fish tissue (kidneys in rainbow, cutthroat, and brook trout) and was used to estimate metal concentrations in kidneys of trout throughout the basin. The analysis relies on data that are likely too limited to extrapolate basinwide (twenty trout total from one reference and two affected locations) and statistical issues limit the use of the regression model (for example, using individual data points [sediment concentrations] in a regression of arithmetic means to provide distributions of concentrations in individual fish), although, ultimately, it does not appear that the results of this analysis had substantial bearing on the weight-of-evidence approach used in the risk characterization.

External exposures for birds and mammals evaluated in the ERA are primarily through contact with contaminated soils and sediments. Extensive studies characterizing the concentrations in these media existed for use in the ERA, particularly for habitats in the lower basin. Where data on COPEC concentrations in tissues were available, EPA also evaluated potential effects of these internal exposures. Considerable effort was expended to develop exposure models that incorporated the feeding ecology of swans, with their potential exposure to sediment-based lead. In addition to the extensive data sets available for waterfowl, more limited surveys provided data on concentrations of cadmium and lead in livers and concentrations of lead in blood were available for minks, muskrats, deer mice, voles, and horses.

Direct quantification of relationships between soil/sediment lead concentrations and resulting doses was possible for some wildlife species; however, for most mammalian and avian wildlife receptors, doses were estimated with mathematical models similar to those used to quantify human exposures to contaminated environmental media. Wildlife can be exposed to chemicals through three routes: dermal absorption, inhalation, and ingestion. Data for estimating dermal absorption or inhalation exposures generally are not available for wildlife; therefore, ingestion was the only pathway considered in exposure modeling.

The model used to quantify doses received through ingestion considered three sources of exposure: soil/sediment, food, and water. For soil/sediment and water, doses were estimated by multiplying the concentrations of each chemical in the appropriate medium by a species-specific ingestion rate (obtained from published literature or site-specific studies) and a chemical-specific gastrointestinal absorption rate. Values for metals other than lead were derived from published studies of metal bioavailability in mammals. For lead, absorption rates were estimated from site-specific data.

Doses received from metal-contaminated food were quantified with bioaccumulation models. These models estimate the dose received from each food type consumed by a given receptor as a function of the concentration of a chemical in that food type multiplied by the consumption rate of that food type. Concentrations of metals in food organisms were estimated through a combination of site-specific data and literature-derived bioconcentration factors. The bioconcentration factors relate concentrations of metals in soil/sediment or water to concentrations in the tissues of exposed biota. Total doses of each metal were obtained by summing the contribution of each food type.

To apply the models, concentrations of metals in sediment/soil and water for all samples collected within a given CSM unit and habitat type were used to generate summary statistics. Within CSM unit 1, the data were further subdivided by watershed. The upper 95% confidence limit on the arithmetic mean concentration in each medium was used as the exposure point concentration for dose quantification. The models described above were then used to convert concentrations of metals in soil/sediment to doses received by mammalian and avian receptor species. Doses were estimated by multiplying the exposure point concentration in sediment by the species-specific sediment ingestion rate and the site-specific gastrointestinal absorption factor.

A site-specific waterfowl model was developed by using site-specific information and an adaptation of the exposure/effects model presented by Beyer et al. (2000). This model was used to generate estimates of concentrations of lead in blood and liver from incidental ingestion of sediment for tundra swans, Canada geese, mallards, and wood ducks. Previous research specific to the Coeur d'Alene River basin has indicated that exposure of waterfowl to lead is trivial in the food pathway compared with sediment ingestion (Beyer et al. 2000). Therefore, dietary exposure is assumed to be represented by sediment exposure, which is reasonable. Diet-to-blood and diet-to-liver bioaccumulation models were developed with data from studies in which waterfowl were fed diets containing sediments from the Coeur d'Alene River basin (for example, Heinz et al. 1999). Sediment-to-tissue bioaccumulation models were also developed for American dipper (cadmium, copper, mercury, lead, and zinc in liver) and for small mammals (cadmium, lead, and zinc in liver and kidney). These models were parameterized using literature-derived rather than site-specific data.

For the mammalian and avian receptor species for which deterministic exposure modeling predicted the highest risks, probabilistic exposure analysis was performed using Monte Carlo methods. The probabilistic exposure models represented the various exposure parameters as statistical distributions rather than point estimates and expressed the resulting doses as statistical distributions.

All the modeling methods used in the ERA are well-documented in the scientific literature. The parameter values that were used are fully documented in Appendices C and D to the ERA. The documentation of these values is thorough and comprehensive, and reasonable decisions appear to have been made about the use of literature-derived data when site-specific data were unavailable. However, site-specific data for validating the exposure estimates are available only for waterfowl exposures to lead. Exposure estimates for all other wildlife receptors are substantially more limited and uncertain. Even the exposure estimates for waterfowl are somewhat uncertain because of the lead-speciation concerns discussed earlier.

Effects Analysis

Various types of data can be included in an ecologic effects analysis. For the Coeur d'Alene River basin ERA, EPA evaluated data from literature-derived single-chemical toxicity tests, site-specific toxicity tests, and field surveys. Some studies were used to derive TRVs and PRGs; others were used as supporting evidence concerning the presence and magnitudes of risks. This section evaluates the technical adequacy of the effects assessment included in the ERA. Questions addressed include whether the underlying studies conform to best scientific practices, whether all the available and relevant data were considered, and whether the data were properly interpreted.

Aquatic Receptors

Metals have long been understood to be toxicants and substantial data exist in the literature on the effects of metal exposures on aquatic organisms. The ERA collected data on metal effects (adjusted for water hardness) on aquatic receptors from the national database (AQUIRE)—a database with results of aquatic toxicity tests. Site-specific tests (using Coeur d'Alene River water or sediments) on aquatic organisms were also assessed and described in the ERA, including laboratory-based lethality tests with salmonids and invertebrates and sublethal behavioral tests. In situ assays ("live box" tests) conducted with fish placed in the environment to monitor mortality are also summarized in the ERA. Surveys of populations in the field were also reviewed to document effects and to evaluate populations of benthic macroinvertebrates, trout, and sculpin.

In the Coeur d'Alene River, metals of concern for fish and benthic invertebrates include zinc, cadmium, and, to a lesser extent, lead. The ERA indicates the sensitivity of the salmonids and other aquatic organisms in a series of plots derived from the literature describing the acute and chronic toxicity of metals to aquatic organisms (CH2M-Hill and URS Corp. 2001, Figures 3-23 to 3-30). There are numerous reports of the sensitivity of trout in the Coeur d'Alene River to dissolved metals. Toxicity tests conducted for the state of Idaho indicated that, of organisms tested in a battery of bioassays conducted on field-collected fish and invertebrates (EVS 1996a), westslope cutthroat trout were the most sensitive of resident species. However, they are less sensitive to metals than hatchery-reared fish. Other tests by the same firm (EVS 1996b) determined that water samples from South Fork Coeur d'Alene River near Wallace downstream of Canyon Creek were acutely toxic to hatchery-reared rainbow trout, whereas South Fork River water collected at stations upstream from Wallace (near Mullan and near the river's headwaters) did not have a toxic effect. In a series of studies on trout sensitivities to metals in Coeur d'Alene and Clark Fork Montana River waters, Woodward and colleagues (1997, 1995) have measured the great sensitivity of trout to metals (copper, zinc, cadmium, and lead). Trout spent as little as 3% of the time in contaminated water when given a choice of movement, and the fish avoided zinc concentrations as low as 28 μg/L. Farag et al. (1998) demonstrated that trout and other biota in the Coeur d'Alene system contain elevated concentrations of metals and, in another study, that the growth and survival of cutthroat trout were reduced when they were fed macroinvertebrates from the South Fork (Farag et al. 1999). Live-box tests conducted and described by Hornig et al. (1988) along with more recent tests (for example, Woodward 1995 and Woodward et al. 1999) demonstrated the acute toxicity of water from the South Fork and main stem of the Coeur d'Alene River to unacclimated hatchery-reared trout.[6]

Field surveys for fish were conducted to support the Natural Resources Damage Assessment (Stratus 2000) and are described in the ERA. These surveys found an absence of fish in some segments of Canyon Creek and Ninemile Creek and reduced populations in the South Fork compared with

[6]These results could appear to conflict with the verbal accounts and population surveys that indicate the presence of trout in the main stem and south fork of the river. The presence of fish in these waters is not surprising though, as fish can become acclimated to elevated levels of soluble metals through biochemical changes such as metallothionein (a metal-binding protein) production and behavioral responses such as periodic movement into less contaminated areas. Resident fish can acclimate to elevated metals concentrations (or may simply be migrating through an area). As a result, it is expected that some fish could be caught in population surveys or recreational outings.

reference areas along the St. Regis River. However, upstream from Wallace—an area still affected by mining but with lower metals concentrations—the abundance and age distributions of trout populations were found to be similar to those in reference locations (CH2M-Hill and URS Corp. 2001, p. E-59). A more recent study (Maret and MacCoy 2002) corroborates these surveys but indicates the absence of sculpin from metals-affected reaches of the rivers where they otherwise would be expected to be found. Sculpin abundance and age class were found to be more sensitive than salmonid population characteristics as indicators of metal-related stress.

The approach used in the ERA, to address risks to fish in the upper and middle Coeur d'Alene River, was robust and based on a large number of high-quality laboratory and field studies. The results appear to have been properly interpreted.

Relatively limited information was available for assessing risks to benthic macroinvertebrates in the Coeur d'Alene River. Site-specific toxicity tests were performed using benthic invertebrates collected from the South Fork, but these tests addressed only the toxicity of the contaminated water and not the underlying sediment. Data were available from three independent surveys of benthic macroinvertebrate communities within the basin, but the studies used different sampling methods and could not be easily compared. Given the obvious impacts of mining-related hazardous substances on fish communities in the upper and middle Coeur d'Alene River basin, the committee believes that the existing data are sufficient to show that benthic invertebrates in the upper and middle basin are probably also at risk from exposure to mining-derived metals. However, an integrated laboratory and field study designed specifically to support the ERA could have provided a much stronger foundation for the PRGs developed in Section 5 of the ERA.

The available data for fish and invertebrates in the lower basin are substantially more limited than for the upper basin and do not appear sufficient to support any meaningful conclusions about the existence and magnitude of risks. To address risks present in Lake Coeur d'Alene, the ERA relies largely on one study by Ruud (1996), in which a qualitative survey was conducted for benthic invertebrates in the lake. No metals data were collected; hence, as the ERA states, "no definitive conclusions can be drawn from this work regarding the potential impact of metal concentrations in the lake on benthic macroinvertebrates."

Terrestrial Receptors

Although terrestrial plants and animals in the Coeur d'Alene River basin are exposed to a large number of mining-related hazardous substances, almost all of the animal studies performed within the basin have focused on lead. The adverse effects of lead in wildlife range from biochemical changes (for example, inhibition of the δ-aminolevulinic acid dehydratase enzyme involved in blood formation) to death. Waterfowl are particularly sensitive to metals-contaminated sediments that are ingested during feeding. Waterfowl are emphasized in this section and elsewhere because of the strong focus on waterfowl in the ERA and in the committee's statement of task.

The ERA considered a variety of studies from the literature on effects to terrestrial receptors to determine TRVs. A variety of site-specific laboratory studies have been conducted on waterfowl exposed to Coeur d'Alene River sediments in their diet to observe changes in biochemical parameters, growth, and other manifestations of lead toxicity. Target organ effects concentration data were derived from both site-specific observations and studies from the literature. The site-specific studies considered are described in Appendix E of the ERA. In general, a variety of biochemical and histological changes were seen in waterfowl exposed to contaminated sediments, especially when the sediments were combined with a nutritionally suboptimal diet.

Exposure of waterfowl to lead typically occurs in the wetland habitats used as feeding areas in the lower Coeur d'Alene River basin. These areas exhibit high concentrations of lead, often exceeding 4,000 milligrams per kilograms (mg/kg) (Campbell et al. 1999). Bookstrom et al. (2001, p. 18) estimated that 72% of the lower basin floodplain sediments had lead concentrations greater than 1,000 mg/kg. The ROD (EPA 2002) states that 95% of the wetland habitats in the lower basin have lead concentrations greater than 530 mg/kg. Waterfowl mortality events have been described in the lower Coeur d'Alene River basin for decades (Chupp and Dalke 1964; Audet et al. 1999); observations extending back to 1924 document exposure to and deaths from toxic materials. These mortality events tend to be greatest after winter flood events, and important routes of exposure are believed to be through ingestion of newly deposited sediments on vegetation or through consumption as grit (Audet et al. 1999).

Particularly compelling are the results from the recent field surveys combined with laboratory necropsy findings. The ERA describes a number of studies in which blood and tissues from sick and dead waterfowl collected in the lower basin were analyzed. These birds demonstrated high lead concentrations and histological indications of lead toxicosis compared with reference areas, yet had no indications of the presence of man-made lead artifacts such as lead shot or sinkers. For tundra swans (a species particularly sensitive to lead toxicosis) the ERA documents high lead concentrations in the liver that, for those animals found dead in the basin and diagnosed with lead poisoning, are consistent with levels in the literature indicative of lead toxicosis (Honda et al. 1990; Pain 1996).

Audet et al. (1999) documented animals found dead or sick in the Coeur d'Alene and St. Joe River basins between 1992 and 1997; of 682 animals found dead in the Coeur d'Alene River basin, 289 were tundra swans, 178 were Canada geese, and 55 were mallards. Lead poisoning was diagnosed in 80% of the 311 animals submitted for necropsy. Of the 250 lead-poisoned animals (elevated lead levels in the liver and histopathology indicative of lead toxicosis), approximately 92% did not have man-made lead artifacts (fishing sinkers, lead shot). This study also demonstrated a significant relationship between the sediment concentration of a feeding area and the presence of poisoned swans.

From the information presented on effects, it is apparent that wildlife are exposed to lead in the Coeur d'Alene River ecosystem. In particular, tundra swans are highly exposed and obviously quite sensitive to lead intoxication, which results in substantial poisoning and subsequent mortality. Multiple species of wildlife, in particular birds, ingest contaminated sediment, resulting in high levels of lead in their tissues. A variety of studies presented in the ERA document adverse biochemical and physiologic effects to Coeur d'Alene wildlife as well as mortality. The overall conclusion that lead exposure exceeded toxicity thresholds is supported by measurements of lead residues in blood and other tissues and by laboratory work and confirming field work. Further, lead exposure and effects were spatially consistent, in that areas with very high sediment concentrations and waterfowl utilization were also the areas with the highest observed waterfowl mortality.

Two site-specific toxicity studies on mammals have been conducted in the basin. One of these was a feeding trial on horses using grass hay grown in the area of the ore smelter (summarized in Appendix E of the ERA). This study was used to develop a lead TRV for large mammals. The other was a study of lead uptake from soil performed using volunteer human subjects. This study was used to develop a dietary absorption factor for estimating dietary uptake of lead by large mammals.

Both site-specific toxicity tests and field survey results for amphibians are summarized in Appendix E of the ERA. EPA judged the toxicity tests to be of limited value because of lack of information concerning sample locations and metal concentrations in the sediment used in the tests. A field study found decreased hatching success and overall survival as a function of increasing metal concentrations in sediment. This study was used to derive site-specific dose-response relationships for cadmium, lead, and zinc. Another field study compared amphibian communities at various sites within the basin to communities found in reference areas.

For plants, site-specific tests evaluating the phytotoxicity of metals present in site-related soils (summarized in Appendix E of the ERA) were performed using standard agricultural test plant species (alfalfa, wheat, and lettuce). These studies demonstrated negative relationships between soil metal concentrations and plant growth. In addition, a field study of plant community composition in contaminated and uncontaminated areas was performed. This study (also summarized in Appendix E) showed that a wide variety of measures of plant community composition were reduced in heavily contaminated areas.

To supplement the site-specific studies and to permit assessment of risks to a wider variety of receptor species than those for which site-specific data were available, the ERA relied on literature-derived TRVs. These TRVs are necessarily highly uncertain as applied to wildlife within the Coeur d'Alene basin.

Risk Characterization

As noted previously, EPA's approach to risk characterization involved development and evaluation of multiple lines of evidence regarding risks to each receptor group.

For birds, the following four lines of evidence were used, although not all lines of evidence were available for all species:

1. Comparisons of modeled dietary doses with literature-derived toxicity benchmarks
2. Comparisons of measured or modeled concentrations of COPECs in blood, liver, and kidney with tissue-specific toxicity benchmarks
3. Site-specific toxicity tests
4. Site-specific field surveys

For mammals, the following three lines of evidence were used:

1. Comparisons of modeled dietary doses with literature-derived toxicity benchmarks
2. Comparisons of measured concentrations of COPECs in liver or kidney tissue with tissue-specific toxicity benchmarks
3. Evaluation of the toxicity of forage contaminated by smelter emissions to horses

For fish and other aquatic organisms, the principal line of evidence used was comparison of measured concentrations of COPECs in surface water with hardness-adjusted national AWQC. This quantitative evaluation was supplemented with qualitative evaluation of results of site-specific toxicity tests and field surveys conducted in the basin.

For amphibians, the following three lines of evidence were used:

1. Comparison of concentrations of COPECs in filtered surface water with literature-derived toxicity benchmarks for embryolarval effects

2. Field-derived estimates of the influence of metal-enriched sediments on amphibian hatching success
3. Field surveys of amphibian species assemblages and relative abundance in wetlands of the lower Coeur d'Alene River basin and Lake Coeur d'Alene

For terrestrial plants, the following three lines of evidence were used:

1. Comparisons of concentrations of COPECs in soil and sediment with site-specific and literature-derived toxicity benchmarks
2. Site-specific phytotoxicity tests
3. Field surveys of plant communities in the Coeur d'Alene River basin

For terrestrial invertebrates and soil processes, the only lines of evidence used were comparisons of concentrations of COPECs in soil and sediment with literature-derived toxicity benchmarks.

This section of the committee's report evaluates the ERA with respect to whether all the available lines of evidence were considered, whether the weight-of-evidence evaluation for each receptor was appropriate, and whether all significant uncertainties were identified and discussed.

Aquatic Receptors

The risk characterization for aquatic life includes a discussion of the ameliorating effects of hardness on metal bioavailability. The ERA did not use current models, such as the biotic ligand model (Santore et al. 2001, 2002), to quantify the influence of organic and inorganic ligands on metal toxicity (see Box 7-2); however, this model may not have been sufficiently developed at the time the ERA was performed.

BOX 7-2 The Biotic Ligand Model

In the biotic ligand model (Di Toro et al. 2001), the site of toxicity is treated as a ligand (a biotic ligand) capable of reacting with the toxic metal. Other chemical species, such as protons and calcium ions, compete with the toxic metal for the reaction sites on the biotic ligand. The toxic metal can react with organic and inorganic ligands in the water, and these too will react with other chemical species, such as protons and calcium ions in the water. A computer equilibrium model is used to compute the concentrations of all chemical species in the system. Toxicity is predicted based on the accumulation of the toxic metal by the biotic ligand. Equal toxicity occurs in waters of different chemical composition when the predicted accumulation of metal is the same, regardless of differences in the total concentration of the metal in the water.

Risk characterization for metals in the Coeur d'Alene River is complicated because of habitat modifications such as channelization and dredging that can also negatively affect aquatic biota. This has resulted in habitats that are nonoptimal for trout, one of the key aquatic receptors. However, given the sensitivity of salmonids and certain benthic taxa (Ephemeroptera, Plecoptera, and Trichoptera) to metals, the emphasis on metal exceedances is warranted. The current structure of the risk characterization emphasizes that toxicity determinations using a "single-metal by single-metal" testing approach may not be appropriate. However, several site-specific ambient media toxicity tests (toxicity tests using water or sediment from the basin) were summarized for fish and macroinvertebrates and are included in the analysis. These types of assays, to the extent that the exposures represent unadulterated environmental media, necessarily account for the range of metals in the environment and other confounding factors such as bioavailability. For instance, Woodward and colleagues have shown that the combination of metals in the river water does influence trout growth and behavior (Woodward et al. 1997). Additional support is provided by population assessments that show substantially decreased populations of fish in the highly contaminated reach of the South Fork downstream from the confluence with Canyon and Ninemile Creeks (ERA, Appendix E).

In situations like the Coeur d'Alene River, where multiple influences and multiple stressors exist, the benthos can be a good overall indicator of habitat quality (La Point et al. 1984; Kiffney and Clements 1993; Griffith et al. 2004). Characterization of effects of metal contamination in the Coeur d'Alene River was too limited to support strong conclusions. Ambient media toxicity tests (ERA, Appendix E, pp. E-61 to E-62) appeared to show that the benthic invertebrates present in contaminated reaches of the river are relatively stress-tolerant. However, only very limited comparisons between benthic communities in contaminated versus reference stream reaches were possible because the surveys conducted in different areas utilized inconsistent sampling techniques.

Potential receptors in the sediments of Lake Coeur d'Alene receive very little attention in the ERA, although ample evidence exists about the extent and magnitude of sediment contamination in Lake Coeur d'Alene (Funk et al. 1975; Horowitz et al. 1993, 1995). Because the lake can serve as a conduit for metals loading to the downstream Spokane River, it is important to develop a better understanding of the role of lake benthos in metal movement. In the ERA, there was ample evidence that, at least at certain times, sufficient metals exist downstream of the lake to affect trout.

The risk characterization failed, however, to treat the river as a continuum (see discussion in Chapter 3 of this report), in which fish life history, competition, and predator behavior within the Coeur d'Alene River system is integrated with habitat and pollutant components. The individual segments of the river are treated as unique and defined, with little appreciation for the connectedness of the upper reaches, the lake, and consequences downstream in the Spokane River. There is little regard to the dependence of downstream biota on upstream events and activities. Yet, fish movement up- and downstream were noted in several reports. Fish use different habitats in different life history stages and need certain habitats at particular times.

Terrestrial Receptors

Risks to terrestrial receptors were adequately characterized where appropriate exposure and effects data were available to conduct a risk assessment. In the case of waterfowl, particularly swans, risks were appropriately characterized, integrating exposure assessment in the field, exposure modeling validated by laboratory studies, and effects assessment that included field collations and laboratory studies of lead toxicosis in waterfowl ingesting Coeur d'Alene River basin sediment.

In the case of waterfowl, all lines of evidence were considered and the weight of evidence clearly demonstrates the following:

1. Lead introduced into the Coeur d'Alene River basin from mining activities had accumulated in the environments occupied by waterfowl.
2. In those contaminated environments, waterfowl receptors are being exposed to high concentrations of lead, as validated by in vivo assessment of exposure levels.
3. Effects are occurring that include both mortality and morbidity of waterfowl in the field, as demonstrated by laboratory studies with several waterfowl species.

For other terrestrial receptors, data are adequate to demonstrate potential risks but not to document the presence of risks to the high degree of certainty that was possible for waterfowl. In the case of songbirds, for instance, inadequate data are provided to fully assess risks present in the Coeur d'Alene River basin. Lead exposures to songbirds in the Coeur d'Alene River basin were reported by Johnson et al. (1999). Livers and blood from song sparrows and American robins were collected from seven sites. Although lead concentrations found in livers of song sparrows in the assessment area were significantly greater than those in the reference sites, effects of these differences were not examined. Sediments collected from Killarney Lake were used in a 3-week feeding trial to test the bioavailability of lead from contaminated sediment in northern bobwhites (C*olinus virginianus*). No overt indications of lead poisoning were observed, and no differences in body weights were detected (Connor et al. 1994). Accumulation of lead was observed in the tissues below levels indicative of clinical lead poisoning and below the "background levels" recorded in wild populations.

Substantially fewer data were available for non-avian terrestrial receptors. This limitation was recognized by EPA in the ERA, which stated that "with the exception of receptors for which no risks were identified, the strength of risk conclusions as determined by the abundance, quality and concurrence of available lines of evidence was generally low for most mammalian receptors. This is because few lines of evidence were available for most mammals, and when multiple lines of evidence were available, there was generally little concurrence" (CH2M-Hill and URS Corp. 2001, p. 5-2).

Thus, for all terrestrial receptors other than waterfowl, there is very high uncertainty concerning the magnitude and spatial extent of risks due to lead and other metals released into the environment of the Coeur d'Alene River basin. It should be possible to address this shortcoming if additional data are collected through the Basin Environmental Monitoring Plan (URS Group. Inc. and CH2M Hill 2004).

Therefore, because of the strength of the waterfowl data and the well-established causal relationship between lead-contaminated sediment and waterfowl mortality, models predicting waterfowl risk based on sediment concentrations are appropriate to develop cleanup levels. The model use is further supported by other information including laboratory and field evidence on the response of swans to lead, and their feeding ecology, that make them highly prone to be exposed through sediment ingestion. Existing data are insufficient to develop comparable models for other wildlife receptors.

VALIDITY OF CONCLUSIONS

Aquatic Receptors

The risk assessment for aquatic receptors was largely limited to salmonids and benthic invertebrates present in the South Fork of the Coeur d'Alene River and its tributaries. Risks due to aqueous and dietary uptake of metals (particularly zinc and cadmium) were adequately characterized for the individual segments of the Coeur d'Alene River, and conclusions about these risks appear to be valid. For trout and sculpin, particularly in the upper basin, risk conclusions were based on toxicity tests that integrated in-stream exposure assessments, modeling effects validated by laboratory toxicity studies, and several behavioral effects studies (both in-stream and laboratory). For other fish and for amphibians, far fewer data were collected in field and laboratory analyses. Conclusions about these receptors are more uncertain.

Contributions to observed aquatic community degradation from habitat degradation unconnected to metal exposures, however, were not fully characterized. Fish respond sensitively to modifications of the physical habitat (for example, substrate size, flow velocities, and depth). Events upstream (mitigation, dredging) could influence downstream habitat quality; moreover, fish communities occupying an impaired habitat may not recover as expected when metal concentrations are reduced.

Terrestrial Receptors

Conclusions about risks to individual waterfowl exposed to particulate lead in wetland sediments are well supported by multiple lines of evidence. Conclusions about risks to other types of terrestrial receptors are much less certain.

The evidence for defining population- or community-level risk to terrestrial receptors is limited. Even in the case of waterfowl, it is not clear whether populations are being impaired by exposure to lead and other metals. Although EPA guidance permits risk assessments for migratory waterfowl and other special status species to be based on individual-level rather than population-level risks, the question of whether populations are being impaired is still relevant to selecting remedies and monitoring ecologic recovery within the Coeur d'Alene River basin. At present, any conclusions about population- or community-level risks must be regarded as highly uncertain.

Habitat-related stressors to wildlife are discussed only nominally in the ERA. However, in the Coeur d'Alene River basin, these stressors are of limited importance to assessment of wildlife toxicology. Moreover, habitat, particularly for waterfowl in the lower basin, is not a limiting factor.

USE OF THE ERA IN RISK MANAGEMENT

EPA's guidance for Superfund ERAs (EPA 1997) states that the risk-description component of an ERA should include, for each chemical and environmental medium considered, a range of concentrations that bound the threshold for estimated adverse ecologic effects, given the uncertainty inherent in the data and models used. The lower bound of this range should be based on conservative assumptions and NOAELs. It should be unlikely that adverse effects due to chemical exposure would occur if concentrations were reduced to this level. The upper bound of this range should be based on observed impacts or predictions that ecologic impacts could occur if this bound were exceeded. The purpose of these ranges of values is to provide risk managers with a range of target levels for selecting a preferred remedy.

In the ERA for the Coeur d'Alene River basin, these values are termed PRGs. Because the PRGs are an important output from the risk assessment, no evaluation of decision-making processes for the Coeur d'Alene River basin would be complete without an evaluation of the validity of the PRGs and the use made of the PRGs in remedy selection.

Validity of PRGs

Section 5.2 of the ERA documents PRGs for the Coeur d'Alene River basin. PRGs were developed for soil, sediment, surface water, and physical/biological habitat characteristics. The most complex set of PRGs was developed for terrestrial wildlife exposed to contaminated soil and sediment. For each of these two media and for every contaminant of concern, a range of values was provided that reflected NOAEL-based TRVs, LOAEL-based TRVs, and ED_{20} (20% effective dose) values. For each of these three PRG types, EPA used its exposure models to back-calculate soil and sediment concentrations that would produce an exposure estimate equal to the appropriate TRV or ED_{20}. The back-calculation was performed for each avian and mammalian receptor species, yielding a distribution of values for potential PRGs. The 10th percentile of this range was selected as the PRG. For soil biota (plants, invertebrates, and microbial processes combined), a separate PRG for soil-dwelling organisms was also developed from literature-derived toxicity data. The PRGs for these biota were calculated by examining the distribution of LOAELs for each chemical of concern extracted from two widely-used summaries of soil toxicity studies (Efroymson et al. 1997a,b). For each chemical, the 10th percentile of the distribution of toxicity values from the literature was chosen as the PRG. To account for the possibility that the literature-derived PRGs could be lower than regional background levels, 90th percentile soil and sediment background concentrations were also estimated. For cases in which the background concentrations were higher than the toxicity-based PRGs, background was recommended as the PRG used in risk management.

For wildlife exposed to sediment, EPA developed an additional PRG for lead by adapting the exposure/effects model of Beyer et al. (2000) to predict sediment concentrations associated with background levels of lead in the blood and liver of four waterfowl species. The 10th percentile of the resulting distribution of sediment concentrations was chosen as the PRG.

For aquatic biota exposed to contaminated sediment and water, the only PRGs provided were freshwater sediment screening values recommended by National Oceanic and Atmospheric Administration (NOAA), national AWQC, and background concentrations. For surface water, the higher of either background or the hardness-adjusted national ambient criterion was recommended as the PRG for each CSM unit. For sediment, the higher of either background or NOAA's screening value was recommended as the PRG.

The PRGs for terrestrial wildlife are well documented, although based only in part on site-specific data. They appear to be consistent with EPA guidance, although the high reliance on literature-derived TRVs for many species contributes substantial uncertainty to the calculated values. The PRGs for aquatic biota, and especially for sediment, appear more questionable and do not appear to be consistent with EPA guidance. For surface water, the AWQC are potentially applicable or relevant and appropriate requirements (ARARs) and for this reason should be included as PRGs. However, by definition, the criteria are intended to protect at least 95% of exposed aquatic species. As long as the AWQC are not exceeded, no ecologicly significant adverse effects should occur. Exceedance of the criteria, however, does not imply that adverse effects will occur. Figures 3-23 through 3-30 of the ERA compare the AWQCs for cadmium, copper, lead, and zinc with acute and chronic effects concentrations derived from various published sources. In all cases, AWQC fall near or below the lowest published effect value. Hence, although the AWQC provide a lower-bound PRG value as defined in EPA guidance, they may not be suitable as an upper bound. For sediment, the ERA does not provide a rationale for using the NOAA screening values as PRGs. All the values used are "threshold effects levels," which are estimates of the lowest values at which adverse effects might occur. These values might be suitable as lower-bound PRGs, but they clearly are inappropriate as upper-bound PRGs or as the only PRGs recommended for use in risk management.

Use of PRGs in Defining the Proposed Remedy

The ecologic PRGs are reproduced in the ROD (EPA 2002, Tables 7.2-6 to 7.2-9) and characterized as being concentrations that are "protective" of terrestrial and aquatic biota. However, with the exception of the AWQC values, it does not appear that any of these values were actually used in remedy selection. As discussed in Section 8 of the ROD, the AWQC were considered to be potential ARARs and, for this reason, were identified as long-term cleanup benchmarks. Although the ERA developed wildlife PRGs for five chemicals of concern, lead was the only chemical used in defining the remedy for soil/sediment. The value selected as the remediation benchmark, 530 mg/kg, is within the range of PRG values identified in the ERA. This value is the LOAEL from a modeling study that incorporates laboratory and field components (Beyer et al. 2000). This study developed an exposure model that described a lowest-effect level of lead as 530 mg/kg in sediments, a reasonable number based on the science to date (see Box

BOX 7-3 Relating Sediment Lead Concentrations to Waterfowl Effects—Derivation of the Cleanup Criterion in the Lower Basin

EPA heavily relied on one study in particular in decisions relating to the toxicity of metals-contaminated sediments to waterfowl and determination of a remedial goal for the protection of waterfowl.

Beyer et al. (2000) reported on studies of waterfowl experimentally fed sediments from the Coeur d'Alene River basin and compared their results with field studies conducted in the basin to relate sediment lead concentration to injury to waterfowl. The first step in their model development involved the relation of sediment lead concentration to blood concentration in mute swans (*Cygnus olor*), and these data were compared with sediment ingestion estimated from analyses of feces of tundra swans (*Olor columbianus*), migratory residents in the Coeur d'Alene River basin. With additional laboratory studies on Canada geese (*Branta canadensis*) and mallards (*Anas platyrhynchos*) fed sediment contaminated with lead, a general relation of blood lead to injury in waterfowl was developed. By integrating the exposure and injury relations, the no-effect concentration of sediment lead was estimated as 24 mg/kg, and the lowest effect level was estimated as 530 mg/kg (based on reduced δ-aminolevulinic acid dehydratase activities). Beyer et al. then combined their exposure equation with data on blood lead concentrations measured in lead-intoxicated tundra swans in the basin and estimated that some mortality would occur at a sediment lead concentration as low as 1,800 mg/kg.

EPA made a risk management decision to use the site-specific protective value lead concentration of 530 mg/kg as the benchmark cleanup criterion for the soil and sediment in the lower basin for protection of waterfowl. Although the value was not derived from the extensive analyses conducted in the ERA (and reviewed in this report), it does fall within the estimated range of sediment lead concentrations protective of aquatic birds and mammals that was determined in the ERA.

7-3). This value is supported by substantial field evaluation of lead effects on waterfowl in the Coeur d'Alene River basin, as reported by Henny et al. (2000) and in particular a report by Blus et al. (1999), reporting substantial lead toxicity in tundra swans captured in the Coeur d'Alene River basin. However, no specific justification for the use of this value rather than a NOAEL or some other value is provided in the ROD (also see Chapter 8, Ecologic Risks: Rationale for Determining Levels of Remediation). The sediment PRGs do not appear to have been used at all in remedy selection.

For surface waters, rather than relying on the PRGs, remedy selection appears to have been based on a set of "interim fishery benchmarks" (URS Greiner and CH2M Hill 2001c) that were developed outside the ERA process. These benchmarks, which are discussed in greater detail in Chapter 8 of the committee's report, identify interim remediation targets in terms of desired characteristics of the fish community in different stream reaches and metal concentrations expected to support fish communities of the desired types.

No explanation is provided in the ROD concerning why the PRGs played such a small role in the development of the proposed interim remedy. Reliance on a study performed externally to the ERA appears quite remarkable to the committee, given the extraordinary length and degree of detail concerning ecologic risks provided in the ERA report. It seems likely to the committee that a principal reason for the failure of the ROD to make greater use of the ERA in design of the remedy is that the ERA focused almost exclusively on exhaustive documentation of the presence or absence of risks. Documentation of risks due to chemical exposure and estimation of chemical concentrations that would eliminate those risks is, in fact, all that EPA guidance on ERA requires. If the ERA had been designed differently, it could have been a source of performance metrics and restoration goals for use in implementing EPA's proposed adaptive approach to remediation. Failure to provide these types of essential outputs reflects, in the committee's opinion, a failure both of EPA's guidance and of EPA's decision to rely on existing data to complete the ERA.

Importance of Habitat Impairment Relative to Chemical Toxicity

Habitat degradation occurring as a secondary effect of mining activities is discussed both in the ERA and in the ROD. Qualitative PRGs for riparian, riverine, and lacustrine habitat are recommended in the ERA. The PRGs (CH2M-Hill and URS Corp. 2001, Table 5-11) for each habitat type and physical characteristic state that the habitat should be returned either to pre-mining conditions or to a condition similar to conditions found in selected reference areas that are only affected by non-mining related disturbances. These PRGs were listed in the ROD (EPA 2002, Table 7.2-10) but were not used to define remediation benchmarks.

Despite the abundant evidence of harm caused by zinc and other dissolved metals, there is clear evidence that channel alterations also impaired fish populations in the Coeur d'Alene River (Dunham and others 2003; Wesche 2004). Wesche, using his own sampling and literature data, estimates that 40-80% of the habitat in the South Fork is degraded for trout and concludes that it is habitat limitation that precludes a healthy trout fishery in the South Fork. Substantial channel alterations have occurred in the upper South Fork for the purposes of flood control, remediation, and road building. Historically, much of the floodplain of the South Fork of the Coeur d'Alene River was forested, particularly with large cedars. The forested condition would have led to decreased stream temperatures, increased stream bank stability, and increased habitat complexity, conditions that support high-quality fish and macroinvertebrate communities. These types of habitats no longer exist along the South Fork. These alterations are clearly permanent and may well limit the recovery of aquatic communities in the river, even if all applicable AWQC are met. The conflict between the goal of returning the river to pre-mining conditions and the irreversible effects of urbanization are not discussed in either the ERA or the ROD.

CONCLUSIONS AND RECOMMENDATIONS

Conclusion 1

The ERA is generally consistent with EPA guidance concerning the ERA process, however, EPA's decision to rely on existing data limits the value of the ERA for risk management.

All except one of the components (a data analysis plan) of an ecologic risk assessment as discussed in guidance are included in the assessment. Stakeholders were appropriately involved in planning and implementing the assessment and data selection and evaluation procedures prescribed in EPA's data quality objectives guidance were followed. The results of the assessment were appropriately documented and the PRGs that were developed were consistent with the conclusions of the risk assessment. However, during the problem formulation phase of the ERA, EPA and the other stakeholders chose to bypass the development of an analysis plan and to rely on existing data to complete the ERA. If an analysis plan had been developed, some of the significant data gaps noted in this review could have been filled, and the utility of the ERA for risk management could have been substantially improved.

Conclusion 2

The ERA is generally consistent with best scientific practice in ERA. In some respects (for instance, the selection of representative species and development of literature-derived TRVs) it was more extensive and detailed than are many ERAs. However, there were some potentially significant exceptions that limit the adequacy of the ERA for supporting appropriate remedial actions.

- Assessments for birds (except waterfowl) and mammals were limited to comparisons between modeled dose estimates and literature-derived effects benchmarks. These methods are highly uncertain (although they are widely used in risk assessments).
- The evaluation of benthic invertebrates in the risk assessment included only limited measures of community structure and site-specific toxicity tests. An integrated laboratory and field study designed specifically to

support the ERA could have provided a much stronger foundation for risk management decision making.

• The risk assessment for Lake Coeur d'Alene is not supported by any defined, quantitative study linking metal concentrations in sediments or in the overlying waters to biotic communities despite ample evidence of the presence of elevated metal concentrations. The lack of data precludes an assessment.

Conclusion 3

Support for the ERA's conclusions is strongest with respect to waterfowl (lead) and fish (zinc and other dissolved metals); support for conclusions about other receptors is much more uncertain.

• The waterfowl and fish assessments are supported by multiple lines of evidence, including site-specific data that reflect effects of multiple contaminants. The conclusions concerning risk to waterfowl are especially strong because of the wealth of data on dose-response relationships developed by USGS and the U.S. Fish and Wildlife Service. Conclusions about risks to fish are also well supported, although some uncertainty exists with respect to chemical-specific values because fish within the basin are exposed to multiple chemicals.

• Conclusions about risks to other receptors are uncertain because of reliance on models and literature-derived toxicity data for single-chemical exposures.

Conclusion 4

The level of support for PRGs is highly variable among receptors.

• The range of PRGs for waterfowl is very strongly supported.

• The PRGs for fish, benthic invertebrates, small mammals, plants, amphibians, and birds other than waterfowl are uncertain, and their value for guiding remediation decisions is questionable. All these are based on regulatory criteria, literature-derived TRVs (many of which are highly conservative), and background concentrations rather than site-specific toxicity data. For fish and benthic invertebrates, only lower-bound PRGs are provided.

Conclusion 5

Despite the large number of ecologic studies performed in the basin and the complexity of the analyses provided in the ERA report, the results of the ERA had only a minimal apparent influence on the ROD.

Of the many PRGs developed in the ERA, only the national AWQC were adopted as remediation goals in the ROD. Only one remediation goal, the soil/sediment goal for lead, was based on site-specific data. Instead of basing the interim remediation goal for dissolved metals on the ERA results, the ROD relied on a set of "interim fishery benchmarks" that were developed outside the ERA process.

Recommendation 1

Further research is needed to support remedial actions intended to promote recovery of aquatic and terrestrial biota within the basin. Information is particularly lacking on effects to benthic invertebrate and fish communities in the lower basin, the magnitude and spatial extent of risks to riparian and upland communities, and the condition of benthic communities in Lake Coeur d'Alene in relation to contaminated sediments.

Recommendation 2

Further research is needed on the influence of transport and transformation processes on the fluxes and bioavailability of particulate lead and dissolved metals. Improved understanding of these processes is needed to ensure the effectiveness of remedial actions intended to reduce risks to wildlife and aquatic biota.

Recommendation 3

ERAs at large, complex sites like the Coeur d'Alene River basin should be designed to support remedy selection and not simply to document the presence or absence of risks. In particular, the ERA should be a source of performance metrics and restoration goals for use in adaptive restoration of the basin. EPA's guidance on Superfund ERAs should be modified to encourage the development of performance goals and metrics as part of ERAs for large, complex sites such as the Coeur d'Alene River basin.

Recommendation 4

In developing performance metrics and restoration goals, additional consideration should be given to development-related habitat modifications (for example, stream channelization) that may prevent a return to premining conditions. Remedial activities designed to reduce metals exposure and transport should, to the extent practicable, concomitantly strive to improve habitat for fish and wildlife.

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8

Remediation Objectives and Approaches

INTRODUCTION

The record of decision (ROD) for cleanup of the Bunker Hill Mining and Metallurgical Complex Superfund Facility Operable Unit 3 (OU-3) (EPA 2002) represents the next step in a long and contentious path for all concerned with human health and the environment in the Silver Valley of northern Idaho, Lake Coeur d'Alene, and the Spokane River down to Upriver Dam. "The Facility includes mining-contaminated areas in the Coeur d'Alene River corridor, adjacent floodplain, downstream water bodies, tributaries, and fill areas, as well as the 21-square-mile Bunker Hill 'box' located in the area surrounding the historic smelting operations" (EPA 2002, Part 1, p. 1). The facility was listed on the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) National Priorities List in 1983. It took almost 10 years for the U.S. Environmental Protection Agency (EPA) to issue RODs for remediation of the area considered to be the major source of risk to human health and the environment—a 21-square-mile area (the "box") roughly encompassing the Interstate 90 corridor from Pinehurst to Kellogg, Idaho. RODs were signed for the populated areas of the Bunker Hill box (OU-1) and the nonpopulated areas of the box (OU-2) in 1991 and 1992, respectively. In 1998, EPA extended Superfund activities outside of the box to OU-3, and the ROD for this operable unit was issued in 2002.

The Bunker Hill box has been undergoing active remediation for several years to protect residents in the area, especially children, from excessive exposure to lead and to control transport of lead and zinc downriver. Major cleanup activities by mining companies, the state of Idaho, and EPA have included regrading and/or removing mine tailings and sediment from many areas in the floodplain of the Coeur d'Alene River; constructing a central impoundment area (CIA) for the storage and isolation of mine tailings and contaminated sediments; operating the central (water) treatment plant (CTP) for treatment of acid mine drainage; remediating contaminated areas in the former smelter complex; and removing contaminated soil from yards and public areas to lower the exposure of children to lead contamination. The ROD for OU-3 was developed through the remedial investigation/feasibility study (RI/FS) process and is intended to interact with and take advantage of remedial actions taken under the RODs for OU-1 and OU-2. In essence, the ROD for OU-3 was the next step in addressing basin-wide human health and environmental issues caused by past mining operations.

As provided in the statement of task (see Appendix A), the committee is charged with assessing the scientific and technical aspects of EPA's remedial objectives and approaches set forth to address environmental contamination in OU-3 of the Coeur d'Alene River basin Superfund site.

REMEDIATION OBJECTIVES AND INCORPORATION OF CLEANUP GOALS

One of the purposes of the feasibility study (FS) (URS Greiner, Inc. and CH2M Hill 2001a), which was prepared under contract for EPA, was to develop remedial action objectives (RAOs). The RAOs are long-term goals for cleanup and recovery from historic effects of mining in the Coeur d'Alene River basin and focus on protecting human health and ecologic receptors (for example, fish and wildlife). They are intended to provide a general description of the goals of the overall cleanup (EPA 2002, p. 8-1). These objectives, described below, are inclusive of the expected sources of contaminants and routes of exposure to humans and ecologic receptors.

Human Health

RAOs for protection of human health are designed primarily to reduce human exposure to lead-contaminated soils, sediments, and house dust to protect children; reduce human exposure to contaminated soils and sediments to lower the risks of cancer; and reduce ingestion of groundwater and surface waters from private, unregulated sources that do not meet drinking water standards (EPA 2002, p. 8-1). RAOs for protecting human health that are specific to environmental media (for example, water and soil) are described in Table 8-1 (EPA 2002, Table 8.1-1) and applicable and

TABLE 8-1 RAOs for Protection of Human Health

Environmental Media	RAOs
Soils, sediments, and source materials	Reduce mechanical transportation of soil and sediments containing unacceptable levels of contaminants into residential areas and structures. Reduce human exposure to soils, including residential garden soils and sediments that have concentrations of contaminants of concern greater than selected risk-based levels for soil
House dust	Reduce human exposure to lead in house dust via tracking from areas outside the home and air pathways, exceeding health risk goals
Groundwater and surface water as drinking water	Reduce ingestion by humans of groundwater or surface water withdrawn or diverted from a private, unregulated source, used as drinking water, and containing contaminants of concern exceeding drinking water standards and risk-based levels for drinking water
Aquatic food sources	Reduce human exposure to unacceptable levels of contaminats of concern via ingestion of aquatic food sources (for example, fish and water potatoes)

SOURCE: EPA 2002.

relevant or appropriate requirements (ARARs) for drinking water are described in Table 8-2 (EPA 2002, Table 8.1-2). Cleanup actions for protection of human health were "designed to address both current and potential future risks, and . . . to limit exposure to soil lead levels such that a typical child or group of similarly exposed children would have an estimated risk of no more than 5% of exceeding a 10 μg/dL [microgram per deciliter] blood lead level" (EPA 2004a, p. 13).

Ecologic Receptors

The RAOs for ecologic protection are long-term goals used to develop ecologic remediation alternatives to protect ecologic receptors. RAOs for the protection of ecologic receptors and systems are described in Table 8-3 (EPA 2002, p. 8.6).

DESCRIPTION AND COMPARISON OF REMEDIAL ALTERNATIVES

The Superfund process requires that alternative approaches be developed to address risks to human health and the environment caused by sources of contamination and that the relative advantages of each alterna-

TABLE 8-2 ARARs for Drinking Water

Metal	MCL[a] or TT[b], μg/L
Arsenic	10
Cadmium	5
Lead	TT[c] Action Level = 15

[a]Maximum contaminant level (MCL) is the highest level of a contaminant that is allowed in drinking water. MCLs are set as close to MCL goals as feasible using the best available treatment technology and taking cost into consideration.
[b]Treatment technique (TT) is a required process intended to reduce the level of a contaminant in drinking water.
[c]Lead is regulated by a treatment technique that requires systems to control the corrosiveness of their water. If more than 10% of tap water samples exceed the action level, water systems must take additional steps.
SOURCE: EPA 2002.

tive be compared and documented. For OU-3 in the Coeur d'Alene River basin, alternatives were extensively investigated and described in the FS.

The process of identifying and developing potentially applicable cleanup methods is complex. This effort resulted in a massive, multivolume set of documents setting forth the details of each remedial alternative considered. Remedial alternatives focused on four separate but interrelated areas of risk (EPA 2002, p. 9-1):

- Protection of human health in the populated and community areas of the upper basin and lower basin
- Protection of ecologic receptors in the upper basin and lower basin
- Protection and restoration of Lake Coeur d'Alene
- Protection of human health and ecologic receptors for the Spokane River from the Idaho-Washington State line to Upriver Dam in eastern Washington

Remedial alternatives are analyzed and described only to the level needed to support development of a proposed plan for cleanup, which is then expanded after the selection of alternatives in the ROD. In this regard, EPA states: "Consistent with the NCP, the remedial alternatives have been developed to a planning level of detail, not a design level of detail. All remedial actions would require a site-specific remedial design that may include additional data collection to further define the problem and refine the action." (EPA 2001a, p. 6-1).

Consistent with the NCP, each set of alternatives must include a "no-action" alternative to provide a baseline or "do-nothing" scenario for com-

TABLE 8-3 RAOs for Protection of Ecologic Receptors

Subject	RAO
Ecosystem and physical structure and function	Remediate soil, sediment, and water quality and mitigate mining impacts in habitat areas to be capable of supporting a functional ecosystem for the aquatic and terrestrial plant and animal populations in the Coeur d'Alene River basin; maintain (or provide) soil, sediment, and water quality and mitigate mining impacts in habitat areas to be supportive of individuals of special-status biota that are protected under the Endangered Species Act and the Migratory Bird Treaty Act
Soil, sediment, and source materials	Prevent ingestion of arsenic, cadmium, copper, lead, mercury, silver, and zinc by ecologic receptors at concentrations that result in unacceptable risks; reduce loadings of cadmium, copper, lead, and zinc from soils and sediments to surface water so that loadings do not cause exceedances of potential surface water-quality ARARs; prevent transport of cadmium, copper, lead, and zinc from soils and sediments to groundwater at concentrations that exceed potential surface water-quality ARARs
Mine water, including adits, seeps, springs, and leachate	Prevent dermal contact with arsenic, cadmium, copper, lead, mercury, silver, and zinc by ecologic receptors at concentrations that result in unacceptable risks; prevent discharge of cadmium, copper, lead, and zinc in mine water, including adits, seeps, springs, and leachate to surface water at concentrations that exceed potential surface water-quality ARARs
Surface water	Prevent ingestion of cadmium, copper, lead, and zinc by ecologic receptors at concentrations that exceed potential surface water-quality ARARs; prevent dermal contact with cadmium, copper, lead, and zinc by ecologic receptors at concentrations that exceed potential surface water-quality ARARs
Groundwater	Prevent discharge of groundwater to surface water at concentrations of cadmium, copper, lead, and zinc that exceed potential surface water-quality ARARs

SOURCE: EPA 2002.

parison with alternative remedial actions. Consideration of a "no-action" alternative is necessary to ensure that there is a benefit to proposed remedial actions and that remedial actions "do no harm."

Alternatives for the protection of human health that address exposure pathways through soil, house dust, drinking water, and aquatic food sources are summarized in Box 8-1. Alternatives for the protection of the environment that mitigate ecologic risks are summarized in Box 8-2. A summary of the projected costs estimated for the various cleanup alternatives is reproduced in Table 8-4 (EPA 2001a).

BOX 8-1 Alternatives for Human Health Protection

Human health alternatives were developed to address the primary exposure pathways through soil, house dust, drinking water, and aquatic food sources. In addition to limiting direct exposure, soils remediation alternatives also address the issue of controlling the risks from eating homegrown vegetables. These alternatives are further discussed in the ROD (EPA 2002, pp. 9-2 to 9-7).

Soils

The remedial alternatives considered for controlling human health risks from lead-contaminated soils include the following: S1, no action; S2, information and intervention; S3, information and intervention and access modifications; S4, information and intervention and partial removal and barriers; and S5, information and intervention and complete removal.

All alternatives for protecting children from exposure to lead in contaminated soils involve public information and intervention, except for the no-action alternative. Other more aggressive alternatives require access modifications such as construction of fences and barriers. More complete cleanup would require either partial or complete removal of soils in residential yards and garden areas to depths of 1-4 feet and replacement with clean fill. Alternatives S4 and S5 also call for pressure washing structure exteriors when appropriate to reduce the risk of recontamination from lead-based paint. S5, the complete removal alternative, is not envisioned for recreational areas.

Drinking Water

The alternatives considered to limit human exposure to drinking water containing lead above drinking water standards include the following: W1, no action; W2, public information; W3, public information and residential treatment; W4, public information and alternative source, public utility; W5, public information and alternative source, groundwater; and W6, public information and multiple alternative sources.

Providing public information to educate citizens about the risks of consuming contaminated water was considered key to controlling these risks. However, consumer education alone was considered insufficient, and some method of making uncontaminated water readily available was considered essential. Point-of-use filtration can be very effective but requires regular filter replacement to be protective. Scheduled replacement of filters on water lines requires an extra level of public education, which would vary greatly in the general population. Hence, various approaches to providing clean water were proposed. Alternatives ranged from tapping into existing municipal water systems, to development of new water wells in uncontaminated subsurface strata, to development of multiple sources of clean drinking water—depending on the needs of communities.

House Dust

Aggressive measures are believed to be needed to protect residents, especially children, from lead-contaminated house dust in lead-contaminated areas. Alternative approaches proposed include the following: D1, no action; D2, information and intervention and vacuum loan program/dust mats; and, D3, information and intervention, vacuum loan program/dust mats, interior source removal, and contingency capping/more extensive cleaning.

A public information program to inform citizens about the risks of exposure of children to lead in house dust has been administered by the Lead Health Intervention Program in the Bunker Hill box since 1985 and throughout the basin since 1996 (von Lindern 2004). Hence, alternatives developed for house dust would include information and intervention with "pamphlet distribution, press releases, public meetings, and publicly-posted notices to inform the public of remedial actions and to provide exposure education" (EPA 2002, p. 9-5). Alternative D2 would also include a heavy-duty vacuum loan program similar to the one previously used in the Bunker Hill box, coupled with free dust mats for entryways. Monitoring would be conducted for achievement of RAOs. The most aggressive alternative, D3, in addition to features of D2, would include interior source removals such as "one-time cleaning of hard surfaces and heating and cooling systems and removal and replacement of major interior dust sources such as carpets and some soft furniture" (EPA 2002, p. 9-6). Attics and basements would be cleaned and crawl spaces beneath houses, if contaminated, would be capped with sand or covered with synthetic membrane to prevent recontamination of houses.

Aquatic Food Sources

Three alternatives were developed to protect recreational fishermen, and perhaps subsistence fishermen, from risks associated with eating fish caught in contaminated areas of the Coeur d'Alene River basin: F1, no action; F2, information and intervention; and F3, information and intervention and monitoring.

The alternatives for protection of individuals from the risks associated with the consumption of contaminated fish caught in the Coeur d'Alene River, lateral lakes, and Lake Coeur d'Alene heavily focus on educating fishermen and recreational users about the potential health risks involved. All of the public information programs to educate citizens about the dangers of lead exposure would also include warnings about consuming contaminated fish. "A well-managed signage program to educate fishermen and other water users of metal hazards would be implemented at all river/lake access sites and common use areas, including the Coeur d'Alene River Trail system corridor. Idaho Department of Fish and Game, Idaho State Parks, USFS [U.S. Forest Service], and BLM [Bureau of Land Management] field personnel who regularly contact basin fishermen and recreational users would be trained in metals risk management and supplied with appropriate pamphlets and signs" (EPA 2002, pp. 9-6 to 9-7).

The more aggressive Alternative, F3, would, in addition to the broad-based educational program in Alternative F2, include a fish-flesh sampling program to provide lake-specific recommendations and identify those areas free of metal risks so fishermen could be notified accordingly. In addition, a trained river ranger program would be developed to advise fishermen and direct them to aquatic resources with the known lowest risks.

BOX 8-2 Alternatives for Environmental Protection

Upper and Lower Basin

Six alternatives were developed to mitigate ecologic risks for waterfowl, other birds, fish, and plants in the combined upper basin and lower basin: Alternative 1, no action; Alternative 2, contain/stabilize with limited removal and treatment; Alternative 3, more extensive removal, disposal, and treatment; Alternative 4, maximum removal, disposal, and treatment; Alternative 5, state of Idaho cleanup plan; and Alternative 6, mining companies' cleanup plan.

No Action

Under the no-action alternative, the Coeur d'Alene River basin would be left to recover naturally over an undeterminably long period of time (close to a millennium for fish according to EPA estimates) assisted by the remedial work already done in the Bunker Hill box and other locations in the upper basin.

Remedial Alternatives 2, 3, and 4

Alternatives 2, 3, and 4 progress from containment and stabilization of contaminated sediments with limited removal and treatment to more extensive removal, disposal, and treatment, to maximum removal and treatment. Alternative 2, in-place and on-site containment and stabilization "would be used to control ecologic and human exposures and metal transport via erosion and leachate loading to groundwater and surface water" (EPA 2002, p. 9-8). Bioengineering, involving planting vegetation, would be used in Alternative 2 to stabilize banks and streams, control erosion, and promote natural recovery. Passive chemical treatment systems would be used to treat drainage from mine adits and groundwater collected from hydraulic isolation systems.

In Alternative 3, in addition to the contain-and-stabilize strategy proposed in Alternative 2, regional repositories would be built for disposal of contaminated materials removed from the upper basin. A regional active water treatment plant would treat contaminated groundwater, leachate, and adit drainage water. River-bed and bank sediments would be removed and stored in regional repositories. Inaccessible floodplain sediments would be subjected to hydraulic isolation.

Alternative 4 proposed the most aggressive approach for protecting ecologic receptors by maximum removal and disposal of sources of contamination, use of active water treatment, and hydraulic isolation of contaminated sediments.

State of Idaho Plan (Alternative 5)

The state's plan is most similar to Alternatives 2 and 3, which focus on containing and stabilizing the largest sources of metals loading. It includes regional repositories and passive water treatment to "achieve a balance between benefit, cost, and impact to the environment in both the long term and short term" (EPA 2002, p. 9-9). Appendix AA of the FS (URS Greiner, Inc. and CH2M Hill 2001b) outlines this plan.

Mining Companies' Plan (Alternative 6)

The mining companies' plan for remediating sources of metal contamination due to leaching of tailings to the Coeur d'Alene River basin stresses regrading and/or removing source material and stabilizing stream banks with vegetation. However, the plan does not include regional repositories. Appendix AB of the FS (URS Greiner, Inc. and CH2M Hill 2001b) outlines this plan.

Lake Coeur d'Alene

Two alternatives were developed for Lake Coeur d'Alene: no action and institutional controls. The only area evaluated that had health risks, Harrison Beach, has been remediated through Union Pacific Railroad actions; hence, institutional controls focus on developing a lake management plan to achieve water-quality goals through management of nutrients, primarily nitrogen and phosphorus. The desire to limit input of nutrients to the lake is based on the hypothesis, as yet unproven at this site, that eutrophication of the lake will increase the flux of metals from bottom sediments that eventually will reach the Spokane River. Sewers will be managed to limit nutrient input to the lake, and control of near-shore erosion will limit sediment loading to the lake. Dredging and/or capping of contaminated lake sediments was not considered because of engineering and cost considerations.

Spokane River

EPA and the state of Washington collaborated to develop five alternatives for risk management in the Spokane River between the state line and Upriver Dam: Alternative 1, no action; Alternative 2, institutional controls; Alternative 3, containment with limited removal and disposal; Alternative 4, more extensive removal, disposal, and treatment; and Alternative 5, maximum removal and disposal. Mining companies did not prepare an alternative.

Alternatives developed for the Spokane River are similar in concept to those proposed for the upper and lower basin of the Coeur d'Alene River, ranging from institutional controls, to containment and removal, to aggressive removal and disposal. Institutional controls would be limited to postings and notices to the public of potential risks and limiting vehicular traffic to reduce erosion and allow vegetation to naturally stabilize shorelines.

In Alternative 3, contaminated beach materials mostly would be left in place but covered with clean material. The physical characteristics of some areas could require limited removal and disposal or excavation and on-site consolidation. In Alternative 4, areas that would be capped in the previously described containment scenario would be excavated and disposed of off-site. Excavated areas would be backfilled with clean material. Sediments behind Upriver Dam that exceeded contaminant criteria would be capped in place.

A maximum removal and disposal option (Alternative 5) would remove and dispose off-site all contaminated sediments and beach materials, including the sediments behind Upriver Dam.

TABLE 8-4 Summary of Alternatives and Costs Developed for the Coeur d'Alene River Basin

Focus	Media/Area	Alternative designation	Description	Estimated total cost
Human health protection	Soils	S1	No Action	$0
		S2	Information and intervention	$5,410,000
		S3	Information and intervention and access modifications	$2,900,000
		S4[a]	Information and intervention and partial removal and barriers	$81,000,000
		S5[a]	Information and intervention and complete removal	$123,000,000
	House dust	D1	No action	$0
		D2	Information and intervention and vacuum loan program/ dust mats	$1,380,000
		D3	Information and intervention, vacuum loan program/dust mats, interior source removal, and capping/more extensive cleaning	$4,290,000
	Drinking water	W1	No action	$0
		W2	Public information	$428,000
		W3	Public information and residential treatment	$1,418,000
		W4	Public information and alternative source, public water utility	$10,000,000
		W5	Public information and alternative source, groundwater	$2,900,000
		W6	Public information and multiple alternative sources	$2,210,000

	Aquatic food sources	F1	No action	$0
		F2	Information and intervention	$230,000
		F3	Information and intervention and monitoring	$910,000
Ecologic protection	Coeur d'Alene River basin (including upper basin and lower basin)	1	No action	$0
		2	Contain/stabilize with limited removal and treatment	$370,000,000
		3	More extensive removal, disposal, and treatment	$1,300,000,000
		4	Maximum removal, disposal, and treatment	$2,600,000,000
		5	State of Idaho cleanup plan	$257,000,000
		6	Mining companies cleanup plan	$194,000,000
	Lake Coeur d'Alene	1	No action	$1,300,000
		2	Institutional controls	$8,800,000
	Spokane River	1	No action	$0
		2	Institutional controls	$900,000
		3	Containment with limited removal and disposal	$1,800,000
		4	More extensive removal, disposal, and treatment	$6,500,000
		5	Maximum removal and disposal	$28,000,000

[a]Based on removal, capping, and revegetation of soil with >1,000 parts per million (ppm) of lead in community areas (yards, rights-of-way) and >700 ppm of lead in common use areas in towns. Community areas between 700 and 1,000 ppm of lead would receive a vegetative barrier.
SOURCE: EPA 2001a.

EPA's Comparison of Remedial Alternatives

Remedial alternatives are compared to each other based on nine criteria described in Table 8-5. The first two criteria are requirements or "threshold" criteria: a remedy has to satisfy them to be considered unless EPA has issued a specific waiver under the second criterion. The next five are called "balancing" criteria. They are used in weighing the advantages and disadvantages of the potential remedies that satisfy the first two criteria. The last two criteria are called "modifying" criteria. If the public review of the proposed decision indicates strong opposition by the state or the community to EPA's proposal, the agency, at its discretion, can modify its decision in recognition of this opposition.

Human Health Risk in Communities

Comparative analysis of the alternatives led EPA to decide that the best balance of trade-offs would be represented by Alternative S4 for soil, D3 for house dust, W6 for drinking water, and F3 for food sources, as described above in Box 8-1.

Ecologic Receptors in Upper and Lower Basin

As described in Chapter 9 of the ROD (EPA 2002), EPA determined that Alternative 3, described above, represented the best balance of tradeoffs for a long-term cleanup approach in the upper and lower basin. This alternative entails massive removals of contaminated sediments from wetlands covering over 5,000 acres, riverbed sediments (20,600,000 cubic yards), and lower basin riverbank sediments (1,780,000 cubic yards). In addition, treatment of adit drainage, groundwater, and surface water in the upper basin would be necessary to meet ARARs. A metals load reduction of 57% was estimated at the completion of remedy implementation. The estimated cost of this alternative is $1.3 billion. It is important to note that ultimately Alternative 3 was not selected for implementation. As described below, the "selected remedy" is a subset of these actions.

Lake Coeur d'Alene

EPA selected the alternative of implementation of a multiagency lake management plan primarily to control sediment and nutrient loading to the lake.

Spokane River

EPA decided that the best balanced approach to managing metals contamination in the Spokane River would be a combination of the alternatives

TABLE 8-5 Evaluation Criteria for Superfund Remedial Alternatives

Criterion		Description
Threshold criteria	Overall protection of human health and the environment	Determines whether an alternative eliminates, reduces, or controls threats to public health and the environment through institutional controls, engineering controls, or treatment
	Compliance with ARARs	Evaluates whether the alternative meets federal, state, and tribal environmental statutes, regulations, and other requirements that pertain to the site, or whether a waiver is justified
Balancing criteria	Long-term effectiveness and permanence	Considers the ability of an alternative to maintain protection of human health and the environment over time
	Reduction of toxicity, mobility, or volume through treatment	Evaluates an alternative's use of treatment to reduce the harmful effects of principal contaminants, their ability to move in the environment, and the amount of contamination remaining after remedy implementation
	Short-term effectiveness	Considers the length of time needed to implement an alternative and the risk the alternative poses to workers, residents, and the environment during implementation
	Implementability	Considers the technical and administrative feasibility of implementing the alternative, including factors such as the availability of materials and services
	Cost	Includes estimated present worth capital and operations and maintenance (O&M) costs. O&M costs are estimated for a 30-year period using a discount rate of 7%
Modifying criteria	State/tribal acceptance	Considers whether the states and tribes agree with EPA's analyses and recommendations, as described in the RI/FS and the proposed plan
	Community acceptance	Considers whether the local community agrees with EPA's analyses and the interim action. Comments received on the proposed plan during the public comment period are an important indicator of community acceptance

SOURCE: EPA 2001a, Table 7-1.

that could include capping of contaminated sediments, riverbed sediment removal, and possibly sediment removal from Upriver Dam.

Evaluation of EPA's Comparison of Alternatives

In the statement of task, the committee was asked to assess whether EPA adequately characterized "the feasibility and potential effectiveness of the remediation plans . . ., given best engineering and risk management practices and the site specific characteristics," and whether EPA considered an "adequate set of alternatives." In answering these questions, it is helpful to distinguish between those plans focusing on protecting human health and those focusing on environmental protection.

With respect to the remedies focused on protecting human health, it is the committee's judgment that the agency considered an adequate set of alternatives and adequately characterized the feasibility and potential effectiveness of these alternatives. The feasibility and effectiveness (or lack thereof) of most of the alternatives EPA considered have been demonstrated at other sites and within the Coeur d'Alene River basin in the cleanups conducted within OU-1 and OU-2. However, as discussed in Chapter 5, the evidence regarding the effectiveness of yard remediations for decreasing blood lead levels (BLLs) in children is not firmly established. Further, more consideration needs to be given to the protection and long-term maintenance of the soil-remediation projects from flood damage and recontamination by contaminated sediment carried by these floods. Similar concerns regarding the feasibility and effectiveness of remedies exist for the selected remedy and are examined in greater detail later in this chapter.

With respect to those alternatives considered for environmental protection, questions about feasibility and effectiveness are much more germane. In particular, the committee has concerns about the accuracy of the "probabilistic model" that the agency used to predict postremediation dissolved zinc concentrations and compare remedial alternatives; whether wetland remediations will be effective in decreasing waterfowl mortality; and whether removals of contaminated floodplain materials will effectively decrease zinc concentrations in surface water. Similar concerns exist for the selected remedy for environmental protection and are examined in greater detail later in this chapter.

On the topic of whether EPA considered an adequate set of remedial alternatives, the committee is concerned that the agency has not identified any alternatives addressing the primary source of dissolved zinc loadings to the middle basin—groundwater discharges in the box (see Chapters 3 and 4). Not addressing this problem will make it much more difficult, probably impossible, to achieve water-quality standards and provide adequate protection to native fish populations. The committee also believes, similar to the case of the human health protection alternatives, that EPA has overestimated the durability of its proposed actions and should have considered alternatives that provided more protection against flood damages and the deposition of contaminated silt during flood events.

As it turns out, however, much of the effort expended by EPA to identify and evaluate alternatives for ecologic protection seems to have been for naught. None of the identified alternatives were selected, and it is unclear whether even the selected remedies will be implemented.

SELECTED REMEDY: GEOGRAPHIC AREAS, LEVELS OF REMEDIATION, AND REMEDIATION PLANS

EPA presented its "preferred alternative" in the proposed plan (EPA 2001a). This preferred alternative is an "interim action" and represents the first increment in a long-term response. For human health, "The interim action includes all of the remedy for protection of human health in the communities and residential areas of the Upper Basin and the Lower Basin." For environmental protection, "The interim action consists of the first increment of cleanup, and the remedy consists of 20 to 30 years of prioritized Ecological Alternative 3 actions" (EPA 2001a, p. 8-1). Following public and stakeholder review and input on the preferred alternative outlined in the proposed plan, a selected remedy is documented in an ROD (URS Greiner, Inc. and CH2M Hill 2001a, Part 1, p. 1-4).

The selected (interim) remedy presented in the ROD for OU-3 contains limited changes from the preferred alternative and, for human health and environmental protection in the upper, middle, and lower basin (as well as the Spokane River), the selected remedy was also the preferred alternative (EPA 2002, pp. 12-5, 12-16, 12-44). This remedy is estimated to cost approximately $360 million (see Table 8-6). The selected remedy is described in four parts in Section 12, Part 2 of the ROD (EPA 2002):

1. Protection of human health in the community and residential areas of the upper, middle, and lower basins
2. Environmental protection in the upper, middle, and lower basins
3. Lake Coeur d'Alene
4. Spokane River

There are no remedial actions for Lake Coeur d'Alene, however, because a lake management plan (Coeur d'Alene Basin Restoration Project 1996, 2002; IDEQ 2004) is proposed, which is intended to be implemented outside of the Superfund process.

This section describes the selected and interim remedies outlined in the ROD (EPA 2002) for protecting human health and the environment and evaluates them in terms of the following:

- Rationale and decisions for determining levels of remediation
- Rationale and decisions for including or excluding geographic areas
- The feasibility and effectiveness of remediation plans

TABLE 8-6 Estimated Cost of the Selected Remedy

Area	Selected Remedy	Estimated Total Cost
Human health protection in the community and residential areas of the upper basin and lower basin	Full remedy, including soil and house dust, including yards, infrastructures, repositories, rights-of-way, commercial properties, and recreation areas	$92,000,000 Including:
	Alternatives S4 (information and intervention and partial removal and barriers) and D3: (information and intervention, vacuum loan program/dust mats, interior source removal, and capping/more extensive cleaning)	$89,000,000[a]
	Drinking water: Alternative W6 (public information and multiple alternative sources)	$2,200,000
	Aquatic food sources: Alternative F3 (information and intervention and monitoring)	$910,000
Ecologic protection in the upper basin and lower basin	Approximately 30 years of prioritized actions	$250,000,000 Including:
	Upper basin tributaries	$100,000,000
	Lower basin riverbanks and bed	$71,000,000
	Lower basin floodplains	$81,000,000
Lake Coeur d'Alene	Not included in the selected remedy	
Spokane River	Combination of elements of Spokane River Alternative 3, 4, and 5	$11,000,000
Monitoring	Basin-wide monitoring	$9,000,000
Total Cost		$360,000,000

NOTE: costs are rounded to two significant figures.
[a]Includes costs for residential soil, street rights of way, commercial properties, and common areas, 31 recreational areas in the lower basin, and house dust.
SOURCE: Adapted from EPA 2002, Table 12.0-1.

Human Health Selected Remedy

The selected remedy for the protection of human health is presented in Chapter 12 of the ROD and was developed to address exposure to metals (primarily arsenic and lead) in soil, drinking water, house dust, and aquatic food sources. Soil and dust from homes, the surrounding communities, and recreational areas are considered the dominant areas of risk (EPA 2002, p. 12-4). The selected remedy does not address certain potential exposures including recreational use in areas of the basin not addressed in the ROD, subsistence lifestyles, and potential future use of groundwater. The selected remedy for human health is further discussed in Chapter 5 of this report and is summarized in Box 8-3.

Human Health Risk and Levels of Remediation

Lead and arsenic contamination of soils in yards and recreational areas constitutes the primary human health risk in the basin. Substantial effort has gone into determining the level of contamination that presents an unreasonable risk and necessitates remediation (see Chapters 5 and 6). Once it has been determined that a particular yard needs to be excavated because the soil contamination lead levels exceed 1,000 mg/kg (or 100 mg/kg for arsenic), clean soil is used to replace the excavated materials.

The approach described for soil replacement is appropriate because children are exposed to lead in a number of different sources—including drinking water, inhaled and ingested dust and soil, food, and paint—and their risk of excessive exposure is an integral of all these separate exposures, some of which the cleanup may not address at all. Cleaning up one major source of exposure to below the threshold values allows other sources to remain high without creating an unreasonable risk for all the exposures considered together.

For lawns with contamination levels between 700 and 1,000 mg/kg, a "vegetative barrier" (grass, usually applied as sod) will be used. The amount of exposure reduction resulting from such a barrier is unclear and is likely to be highly site specific depending on factors such as how well the vegetation is maintained.[1] In other areas, the barriers may take the form of asphalt pavement or a layer of clean gravel or soil. In these cases, lead concentrations should be reduced, at least initially, to well below the action level.

Soil cleanups will be supplemented by a "health intervention program" and other actions. Parts of the health intervention program, such as information about public health risks, a vacuum cleaner loan program, and voluntary BLL tests, will be available to all residents in contaminated areas. Other parts of the supplemental programs will be more focused. For instance, homes with particularly susceptible residents, such as young children and pregnant women, will be monitored while the remedy is being implemented to ensure that exposure levels decrease to acceptable levels. Where they do not, further actions such as pressure cleaning the outside of houses to remove leaded paints or even relocation of residents may be undertaken. The agency, however, has not established any clear criteria for when these discretionary supplemental activities will occur.

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[1]This approach of using less-protective remedies in areas where the contamination is lower results in an apparent anomaly that the residual risks from contaminated yard soils facing children in homes with lower initial soil contamination levels will likely end up higher than those for children living in homes with high initial levels of yard soil contamination. Such anomalies, however, are typically inherent in the types of decisions that have to be made under any cleanup program about which areas should be cleaned up and how.

BOX 8-3 Selected Remedy to Protect Human Health

Soil and house dust

- Sampling: House dust will be sampled for houses with pregnant women or young children. Yards and other areas will be sampled to determine whether the lead concentration exceeds 700 mg/kg or arsenic levels exceed 100 mg/kg.
- Remediation of residential yards: For yards having a contamination level exceeding 1,000 mg/kg or an arsenic concentration exceeding 100 mg/kg, soils will be excavated to a depth up to 12 inches and replaced with clean fill. For yards having a contamination level between 700 mg/kg and 1,000 mg/kg, some type of barrier (usually vegetation) will be installed, which will be "continuous and sustainable" and will leave no bare soil exposed.
- Remediation of gardens: For gardens having a contamination level over 700 mg/kg, soils will be removed to a depth of 2 feet and replaced with clean soil.
- Remediation of street rights-of-way: Actions taken will depend on the "location, use, and contaminant concentrations" of the right-of-way. Possible actions include "access controls, capping (barriers consistent with land use), or removal/replacement."
- Remediation of commercial properties and common use areas: Depending on the location, use, and levels of contamination in these areas, remedial actions will include soil removal and replacement, barriers (such as vegetation or a cover of clean gravel or other material), and access restrictions.
- Remediation of recreational areas: EPA has identified thirty-one "formal" recreation areas for cleanup. In most cases where soil contamination levels exceed 700 kg/kg, the cleanup action will involve installing a nonvegetative barrier such as a cap of clean soil, gravel, or asphalt. In some cases, contaminated soils may be removed.
- Dust suppression during remedial activities: This will mostly include wetting down and covering exposed contaminated soils and site cleanup.

The remedies for contaminated drinking water supplies have many of the same characteristics as those for contaminated soils. The action levels, however, have no ambiguity. Contamination levels cannot exceed drinking water ARARs (unless the mining wastes are not the source of contamination). The selected remedies (alternative sources of drinking water or, if alternative sources are lacking, point-of-use filters) are expected to provide water supplies with contamination levels well below ARARs.

Thus, the fact that remedies proposed to protect human health in most cases will result in remediation levels substantially lower than action levels is reasonable. EPA has not explicitly said that it is following this rationale, but any effort to equate remediation levels to action levels would involve some clearly irrational actions to spend additional money to increase risks.

What the agency has not done, however, is provide a clear measure of whether its strategy is successful. Its RAOs are qualitative, not quantitative.

- Disposal of contaminated materials: Contaminated materials will be disposed of in safe repositories.
- Health intervention program: This includes a wide range of activities including education, monitoring the contamination levels in house dust, loaning vacuum cleaners, and voluntary tests of BLLs.
- Remediation of interior house dust, if necessary: If homes demonstrate high lead dust levels after their yards have been remediated, further cleanups may be undertaken. These could include interior cleaning and paint abatement.
- Relocation, if necessary: In a few cases, if remediation is infeasible or recontamination is highly likely, families can be relocated to cleaner dwellings.

Drinking water

- Public information: Residents on private wells will have the opportunity to have their water tested.
- Alternative sources: Where sampling shows that the drinking water supply exceeds drinking water ARARs, EPA will connect the house to an existing water supply system, dig a well into an aquifer with clean water, or provide a point-of-use filter.

Aquatic food sources

- Information: The Idaho Department of Health will provide information to commercial and recreational fishermen and post fish advisories near the lateral lakes. The department will also monitor contamination levels in fish from Lake Coeur d'Alene and issue advisories if high contamination levels are found.

Source: EPA 2002, pp. 12-5 to 12-12.

The agency states that "The Selected Remedy is expected to reduce the residual risk from lead in soil and house dust such that a typical child has no more than a 5 percent probability of having a blood lead level above 10 μg/dL and no more than a 1 percent probability of having a blood lead level above 15 μg/dL" (EPA 2002, p. 12-14). However, there is no way of measuring these probabilities, and thus no way of determining whether the cleanup is meeting their expectation. This lack of any quantitative, measurable, indicator of success is troublesome.

Feasibility and Potential Effectiveness of Remediation Plans

Coeur d'Alene River Basin

EPA has already implemented remedies like these in the box and at other Superfund sites and has demonstrated that they are feasible. Yard remediations have been conducted in the basin for the last several years; in 2004, over 300 yards were remediated. As indicated in Chapter 5, the available evidence indicates, with some caveats, that the selected remedy for human health (Box 8-3) can also be effective. One caveat relates to a reliance on education and information. Such activities often have very limited effectiveness and probably are not sufficient when risk levels are high. EPA appears to recognize these limitations and has not relied solely on these techniques when the agency has identified high risks.

A second caveat relates to the effectiveness of residential yard remediations for decreasing BLLs. Research to date has not definitively identified a causal link between remediated yards and decreased BLLs; however, a relationship between the two is reasonably expected (see Chapter 5 for discussion on this topic).

A third caveat relates to the need to maintain the remedies that do not completely remove contaminated material and use barriers to eliminate exposure. Vegetative barriers will fail if the vegetation is not maintained; caps can be eroded by floods or their integrity can be destroyed by traffic or excavation; water filters need to be maintained and periodically replaced; and gravel or asphalt barriers on streets and rights-of-way will degrade over time.

Further, none of the remedies is permanent, and the integrity of the remedies will have to be monitored and maintained, essentially in perpetuity, all of which constitutes a considerable financial burden. This has already been demonstrated in the box where floods and other actions have either eroded the installed remedies or caused recontamination. EPA recognizes this need, and the Panhandle Health District through the Idaho Department of Health and Welfare supervised the required monitoring and repair. This program appears to have been successful in correcting the problems caused by the 1997 flooding of Kellogg and Wardner, Idaho, by Milo Creek. Remedial activities following the Milo Creek flood were funded by the Federal Emergency Management Agency (FEMA). As presented in the OU-1 5-year review, "Given the financial status of the Bunker Hill Superfund Site cities and residents, it seems unlikely that cleanup from the Milo Creek flood would have occurred so efficiently, or at all, without FEMA funds" (TerraGraphics 2000, p. 6-8).

A major uncertainty associated with the yard and common-use area remediations is that these remedies call for institutional mechanisms to monitor their effectiveness, repair any failures, and remain in place and effective for an extremely long time (at least hundreds of years). As state funding priorities change and the situation in the Coeur d'Alene River basin loses its immediacy, maintaining an effective program is likely to be difficult. Various approaches have been considered for maintaining and funding institutional controls (See NRC 2003). For instance, one approach is the creation of trust funds to finance and oversee stewardship activities (Bauer and Probst 2000).

Lake Coeur d'Alene

EPA sampled beaches and wading areas adjacent to Lake Coeur d'Alene, and, with the exception of Harrison Beach, concentrations of metals did not exceed risk-based levels for recreation (EPA 2002, p. 5-8). Lead concentrations at Harrison Beach in Harrison, Idaho, on Lake Coeur d'Alene averaged 1,250 (mg/kg) (URS Greiner and CH2M Hill 1999), and the area has been remediated. Thus, no remedies have been proposed in the OU-3 ROD to reduce exposures in Lake Coeur d'Alene. However, recontamination of Harrison Beach from deposition of flood-mobilized contaminated sediment will likely occur in the future, so the remediation must be considered interim or short term and will need to be maintained. Consumption of lake fish represents an exposure pathway to metals, but limited information was available to assess the health risks of such exposures when the human health risk assessment was initially prepared. To address this data gap, EPA funded a special study to characterize the concentrations of arsenic, cadmium, lead, mercury, and zinc in the tissues of bass, bullhead, and kokanee in Lake Coeur d'Alene (URS Greiner, Inc. 2003). Results of that study were subsequently used to prepare a fish consumption advisory (IDHW 2003) that specifies the number of meals that can safely be consumed each month for those particular fish (and species with similar dietary behaviors). The advisory targets three population cohorts: the general population and children older than 6 years, pregnant and nursing women, and children younger than 6 years. In addition, the advisory adjusts the intakes according to the section of Lake Coeur d'Alene where the fish are caught. This fish-consumption advisory is a prudent method of risk management that not only balances the nutritional value of fish consumption with the potential harm of metal toxicity for those consuming the fish but also factors in the spatial variability of metal accumulation in fish.

Spokane River

The selected remedy for cleaning up shoreline areas along the Spokane River where residents go for recreation include controlling access, capping contaminated deposits, and removing 9,000-28,000 cubic yards of contaminated material (EPA 2002, p. 12-45 and Table 12.4.1). All these actions are feasible. Access controls may have limited long-term effectiveness unless they are monitored closely. Sites that are capped or excavated have a reasonably high probability of being recontaminated. EPA recognizes this possibility but apparently has not arranged with Washington for the state to establish a special institution, like that established for the Coeur d'Alene River basin, to monitor this problem and ensure that the cleanups are properly maintained.

No remedies have been proposed for the Spokane River to address risks from possible future uses of contaminated groundwater and risks to residents who engage in subsistence lifestyles. The agency does not have sufficient information to know the extent to which there are currently, or may in the future be, residents engaging in subsistence lifestyles or how high the risks would be to people who engage in such lifestyles. Future risks from contaminated groundwater could occur if residents extracted drinking water from a contaminated near-surface aquifer. However, in a recent study, the U.S. Geological Survey (USGS) reported that, although the Spokane River does recharge the aquifer along reaches, "trace elements were below drinking-water standards and guidelines, and most were below minimum reporting levels." Dissolved zinc is detected in groundwater adjacent to the river, but it did not penetrate appreciable distances into the aquifer (Figure 8-1) (Clark et al. 2004, p. 11). Because of its limited capacity to dissolve in water and its propensity to sorb to solids, lead is even less likely than zinc to affect groundwater resources in this area.

Selected Remedy for Ecologic Protection

The selected remedy is not one of the alternatives considered in the FS (URS Greiner, Inc. and CH2M Hill 2001a) for ecologic protection, although EPA believes that the level of cleanup described in Alternative 3 of

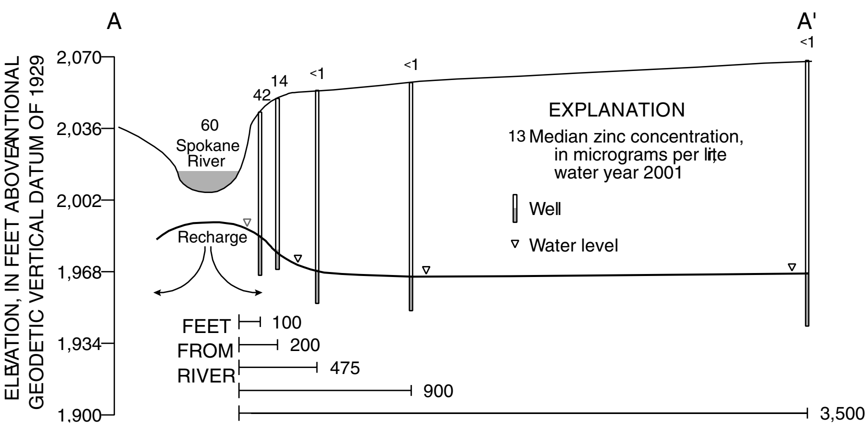

FIGURE 8-1 Water containing elevated concentrations of zinc displays limited transport from the Spokane River to the underlying Spokane Valley/Rathdrum Prairie aquifer. SOURCES: Clark et al. 2004; data from Caldwell and Bowers 2003.

the FS will be needed for protection of the environment and compliance with ARARs (EPA 2001a, p. 1-4). The selected remedy is an interim action and is generally a subset of the FS's Ecological Alternative 3 "extensive removal, disposal and treatment." The selected remedy focuses on three environmental problems in the basin: dissolved metals (principally zinc and cadmium) in rivers and streams, lead in floodplain soil and sediment, and particulate lead in surface water.

The remedy is not intended to fully address contamination within the basin, achieve ARARs, or attain the RAOs described in Table 8-3. CERCLA allows EPA to select an interim remedy, if it is part of the total remedial action that will attain all ARARs. The EPA National Remedy Review Board recommended interim remedial actions for protection of ecologic receptors in the basin, because of the magnitude of contamination to be addressed, the significant costs associated with a basin-wide remedial strategy, and the uncertainties associated with predicting the effectiveness of the basin-wide ecologic alternatives (NRRB 2001). The interim action decision for ecologic receptors gives EPA a very long time and the ability to experiment, try different remedial actions, evaluate progress, change course, and continuously seek ways to achieve the long-term goals of full environmental protection and compliance with ARARs. Interim action over 30 years is viewed by EPA to be a prioritized first increment of cleanup. However, as an interim action, it is intended to provide the best balance of tradeoffs for the following five CERCLA balancing criteria:

- Long-term effectiveness and permanence
- Reduction of toxicity, mobility, or volume through treatment
- Short-term effectiveness
- Implementability
- Cost

The long-term goals are to provide full protection of the environment as well as to return the opportunity for individuals to practice subsistence lifestyles without limits from mining contamination. EPA believes the interim approaches are consistent with these goals (EPA 2001a, p. 8-1).

The ROD (EPA 2002) recognizes that natural recovery will play a big role in improving the environmental quality of the basin. Time periods for natural recovery and achievement of ARARs are projected up to 1,000 years. Upfront aggressive cleanup activities are conceptually designed to hasten the recovery period. EPA intends to implement an incremental management approach for cleanup of the basin. Elements of this approach include the recently developed Basin Environmental Monitoring Plan (BEMP) (URS Group Inc. and CH2M Hill 2004) to measure cleanup progress, possible incorporation of innovative technologies that might be developed, prioritization of cleanup actions that may prove effective over time, and stakeholder involvement in prioritization of cleanup actions.

This section further explores factors the committee considers to be critically important in estimating the likelihood that proposed remedial actions will provide ecologic protection and includes the following:

- A brief discussion of contaminant distribution affecting ecologic receptors throughout the basin
- A consideration of the rationale and decisions for inclusion and exclusion of geographic areas for cleanup
- Assessment of EPA's cleanup actions
- An examination of EPA's use of the "adaptive management approach"

Contaminant Sources and Distribution in the Basin

Dissolved Metals

The main source areas of dissolved metals to the Coeur d'Alene River system are the upper basin (tributary streams feeding the South Fork Coeur d'Alene River) and middle basin (middle reach of the South Fork from Wallace to Cataldo). Zinc is the principal dissolved metal of concern. Woods (2001) showed that zinc represented about 99% of the total dissolved heavy metal load measured at Pinehurst in water year 1999. As discussed in Chapters 3 and 4, EPA's modeling estimates that 41% of the zinc load at Harrison (where the Coeur d'Alene River enters Lake Coeur d'Alene) stems from sources within the box. Canyon Creek contributes 15% of the zinc load at Harrison. Dissolved zinc contributions to the Coeur d'Alene River below Pinehurst account for 15% of the total zinc load at Harrison. These contributions are likely due to groundwater seeps in the Cataldo Flats area and mobilization of zinc associated with riverbanks and levees and from entrained water (pore water) in stream bed sediments (pore water concentrations of zinc in this area range from about 13,000 to 36,000 μg/L [Balistrieri et al. 2003]). Little of the dissolved metals in the river system come from discrete sources (for example, adits). An estimated 71% of the zinc load is derived from affected sediments and associated groundwater (EPA 2002, Figure 5.2-4). As described in Chapter 3 of this report, groundwater contamination by metals has been detected at locations throughout the river basin. The amounts of dissolved metal contributed by groundwater and the exact locations of groundwater influx to the river system are unknown, although EPA expects that most zinc in surface water is derived from groundwater influx (EPA 2004b [June 23, 2004]) (see discussion in Chapter 4 of this report).

Lake Coeur d'Alene exceeds water-quality standards for protection of aquatic life from dissolved cadmium and zinc. These standards are more stringent than drinking water standards. The lake retains on average about 38% of the zinc input based on the difference between metal load into the lake and load out of the lake (EPA 2002, p. 5-8). During flood events or high spring runoff from the Coeur d'Alene River, drinking water action levels for lead are exceeded in Lake Coeur d'Alene for short periods.

The water in the Spokane River meets safe drinking water standards for metals. The estimated average concentrations of total lead and dissolved zinc in surface water are 2.1 and 58 μg/L, respectively; dissolved cadmium was not detected (EPA 2002, p. 5-10). When total metals[2] were measured, 21% of the samples exceeded a cadmium screening level of 0.9 μg/L, 48% exceeded a 0.66 μg/L screening level for lead, and 68% exceeded the 30 μg/L screening level for zinc. Lead and cadmium screening levels are equal to federal ambient water-quality criteria (AWQC), and zinc is a risk-based concentration for protection of aquatic plants (EPA 2002, p. 5-10).

Particulate Metal: Tailings, Mine Wastes, and Mining-Affected Sediments

Waste rock dumps (uncrushed rock materials) and tailings piles (crushed rocks subjected to certain mineral processing steps) are located on hillsides, often very steep, and adjacent to mine adits along tributary streams in the upper basin where mining took place. In some cases, these materials are physically unstable, and sometimes they collapse into the stream. In other cases, for example at the Success Mine located adjacent to the East Fork of Ninemile Creek, groundwater interacts with the tailings, resulting in contaminated groundwater that feeds into the stream.

An estimated 62 million tons of tailings, containing about 880,000 tons of lead, were directly discharged to streams before 1968 (EPA 2002, p. 2-1). In streams and rivers, lead exists principally in the form of particles because lead minerals are relatively insoluble and any dissolved lead has a propensity to adsorb to metal oxyhydroxide particles. The present distribution of the approximately 880,000 tons of lead from released mill tailings is shown schematically in Figure 8-2, derived from analyses conducted by the USGS (Bookstrom et al. 2001; Box 2004). The lead-containing tailings mix with clean sediments throughout the length of the valley, greatly increasing the volume of streambed material that is affected. During spring runoff and flood events, streams overflow their banks, depositing metal-contaminated sediment on stream banks (Bookstrom et al. 2004).

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[2]Total metal concentrations are determined by analyzing water that has not been filtered, using chemical digestion methods.

Approximately 24% of the lead from mill tailings released to the streams resides in the tributary streams of the South Fork Coeur d'Alene River and the middle reach of the South Fork Coeur d'Alene River (Wallace to Cataldo) (Figure 8-2). In these areas, there are about 7 million cubic yards of tailings-affected sediments including an estimated 3 million cubic yards of sediment that were used as fill or otherwise located beneath Interstate 90, other roads, and residential and commercial structures. These numbers do not include deeper, less-affected sediments (EPA 2002, p. 5-6). The ROD presents average sediment concentrations at various monitoring locations in the Coeur d'Alene River. For example, in the upper basin, above Wallace, the average sediment concentration of lead is 4,060 mg/kg; in the middle basin, below Wallace but above the box, it is 3,120 mg/kg; and sediment concentrations at a site located near Pinehurst are 9,330 mg/kg (EPA 2002, Figure 5.2-2).

About 29% of the released lead is located in the lower reach of the Coeur d'Alene River (Cataldo to Lake Coeur d'Alene) (Figure 8-2). The sediments in this stream segment are stratified vertically, with sediments containing high lead concentrations buried deeper, covered by sediments with lower lead concentrations (see Figure 3-9 in Chapter 3 of this report). The potential remobilization and transport of these highly contaminated sediments is a particular concern. Severe floods, such as the one in 1996, are capable of scouring the river bottom and mobilizing these sediments. Under less severe conditions, only the upper layer of less-contaminated sediments is redistributed. EPA estimates that 1.8 million cubic yards of bank materials and 20.6 million cubic yards of bed sediments are affected (EPA 2002, Table 9.2-8). Note the vastly larger volume of affected sediment in the lower reach of the basin compared with the volume in the upper

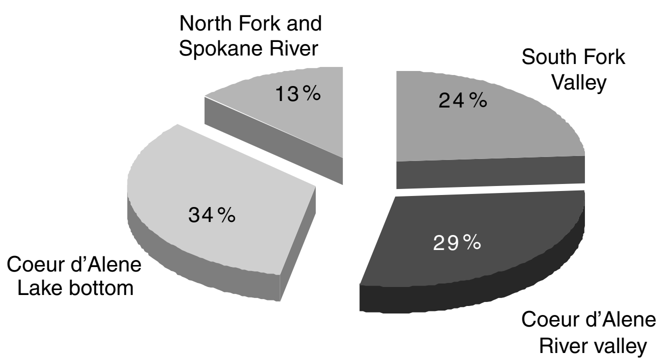

FIGURE 8-2 Distribution of approximately 880,000 tons of lead from mill tailings released to streams. SOURCES: Bookstrom et al. 2001, table 15; Box 2004.

and middle basins, collectively; yet, the percentage of distributed lead is nearly the same—29% in the lower reach compared with 24% in the upper and middle basins (Figure 8-2). The large volume of affected sediments in the lower reach of the main stem Coeur d'Alene River results from the mixing of North Fork and South Fork sediments. For example, in water year 1999, approximately 21,930 tons of sediment were discharged from the South Fork Coeur d'Alene River (URS Greiner Inc. and CH2M Hill 2001c, Figures 3.2-13 and 3.2-14) and mixed with approximately 25,400 tons of sediment from the North Fork (URS Greiner Inc. and CH2M Hill 2001d, Figures 3.2-4 and 3.2-5).

The average lead concentration in the floodplains of the lower reach of the Coeur d'Alene River is 3,100 mg/kg (EPA 2002, p. 5-7). An estimated 18,300 acres, or 95% of the 19,200 acres of floodplain habitat in the lower basin, contain more than 530 mg/kg of lead in the surface sediments. Figure 8-3, compiled by the USGS, shows lead distribution by depositional environment in the lower reach of the Coeur d'Alene River basin.

About 34% of the estimated 880,000 tons of released lead resides in the bottom of Lake Coeur d'Alene (Figure 8-2). This has resulted in an estimated 44-50 million cubic yards of contaminated sediments (EPA 2002, p. 5-8). The remaining 13% of the released lead is distributed between the North Fork of the Coeur d'Alene River and the Spokane River (Figure 8-2) (Bookstrom et al. 2001, table 15; Box 2004). The average concentration of lead in 265 sediment samples collected in the Spokane River floodway between Lake Coeur d'Alene and Long Lake is 400 mg/kg. An estimated 260,000 cubic yards of lead-contaminated sediments are present upstream of Upriver Dam (EPA 2002, p. 5-9).

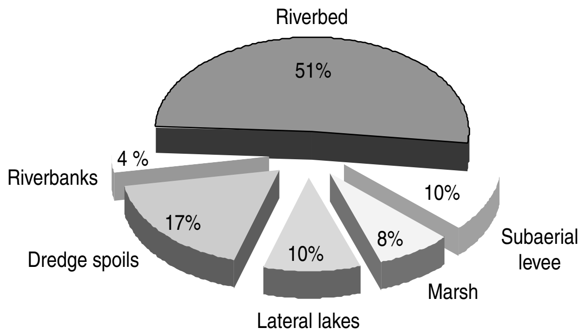

FIGURE 8-3 Distribution of lead by depositional environment in the lower reach (Cataldo to Harrison) of the Coeur d'Alene River. SOURCES: Bookstrom et al. 2001; Box 2004.

Ecologic Risks: Adaptive Management and Determining Levels of Remediation

The situation with respect to remediation levels for ecologic risks is similar to, but more complicated than, the situation with respect to human health risks. One major reason for the increased complexity is that the current ROD (EPA 2002) does not propose a final remedy; rather, the interim measures represent actions that "will neither be inconsistent with nor preclude implementation of the final remedy that will be identified in subsequent decision documents" (EPA 2002, Declaration, p. 6). EPA proposes to implement these interim remedies and conduct monitoring to determine their effectiveness. The agency refers to this approach as adaptive management. The selection of any final remedies will depend on information gained in implementing the interim remedies, some of which are admittedly experimental. The adaptive management approach and the rationale for determining remediation levels for the environment are discussed below.

Adaptive Management Approach

The ROD (EPA 2002) gives the concept of adaptive management only cursory mention. The BEMP (URS Group Inc. and CH2M Hill 2004) provides a more extensive discussion and defines adaptive management as follows:

> In general terms, adaptive management is a systematic strategy for continually learning from the ongoing monitoring results to cost-effectively improve future remediation and monitoring. It provides a purposeful feedback loop to assess evolving conditions and identify useful changes to the remedy, including long-term monitoring, as identified in the BEMP. Adaptive management is a key strategic component inherent in the BEMP. (URS Group Inc. and CH2M Hill 2004, p. 6-11)

The BEMP does not provide details on how decisions will be made to modify the remedy in response to newly available data; for this reason, the committee is not convinced that EPA fully understands or is properly implementing the principles of adaptive management. The adaptive management approach was first described by Holling (1978) and has since been widely adopted in natural resource management, especially in the Pacific Northwest (Lee 1993). It is the subject of an NRC study (NRC 2003) and similar approaches have been suggested for mining megasites (Moore and Luoma 1990).

Adaptive management is not synonymous with "trial and error." Adaptive management is a six-step process for defining and implementing management policies for environmental resources under conditions of high uncertainty concerning the outcome of management actions. A well-structured adaptive management plan contains the following interactive steps:[3]

1. Assessing the problem
2. Designing a management plan
3. Implementing the plan
4. Monitoring
5. Evaluating results obtained from monitoring
6. Adjusting the management plan in response to the monitoring results

These steps, described more definitively below and contrasted with EPA's actions, are usually conceived to be a cycle in which monitoring provides feedback for redefining the original problem, refining the management plan, and so forth. EPA's approach generally follows this process, but the separate steps and feedback mechanisms between the different steps have not been structured to maximize the effectiveness of the strategy.

Step 1: Assessing the problem. Assessing the problem begins by defining the scope of the problem, defining measurable management objectives and potential management actions, and specifying key indicators for each management objective. These indicators should be measurable attributes of the resource being managed, must be relevant to the objectives of management, and must be responsive to management actions. Multiple indicators should be identified, including indicators expected to respond in different time frames (short-term, medium-term, and long-term) and spatial scales (for example, site, watershed, and basin).

Conceptual or quantitative models are then developed and used to predict the potential effects of alternative actions on the indicators. Explicit forecasts are then made concerning the responses of the indicators to alternative management actions. Finally, key uncertainties are identified, and the implications of these uncertainties with respect to the effects of alternative management actions are described.

Under the Superfund process, the objective of the RI is to define the scope of the problem. The objectives of the FS are to define alternative management actions and develop conceptual or quantitative models to predict the potential effects of these alternative actions. As discussed further below, EPA has proposed a reasonable set of biological indicators for evaluating responses of fish communities to remedial actions intended to improve water quality but has not proposed an equivalent set of indicators for evaluating the effectiveness of sediment removal actions. Implications of

[3]This discussion is based on principles developed by the British Columbia Forest Service (BC Forest Service 1999, 2000) and the U.S. Department of Energy (DOE 1999).

uncertainties, especially uncertainties concerning the influence of groundwater sources on water quality and of flood-related transport on sediment quality, were not discussed in the ROD or in the BEMP.

Step 2: Designing a management plan. This step begins with an evaluation of the management alternatives identified in step 1. The alternatives are compared with respect to the likelihood of meeting the management objectives, cost, risk of undesired consequences, and other relevant criteria.

The literature on adaptive management distinguishes between active adaptive management and passive adaptive management. In active adaptive management, the plan is designed as a management experiment to discriminate between alternative hypotheses concerning the responses of resources to management. The actions selected are intended to maximize the power of the management experiment. In passive adaptive management, the plan is designed under the assumption that the most plausible or likely hypothesis is true, and the actions or set of actions that are forecast to have the most favorable outcomes under that assumption are selected. Although active adaptive management provides the most informative feedback to future iterations of the management cycle, it is often impractical to implement because of costs, risks, and irreversibility of actions.

After a management plan is selected, a monitoring protocol is designed. The protocol should specify the types and quantities of baseline data; the frequency, timing, and duration of monitoring; the indicators to be monitored at each interval; the appropriate spatial scales for monitoring different indicators; and the persons or organizations responsible for different aspects of monitoring. A data management and analysis plan must be specified. Finally, and most importantly, the indicator values that will trigger a change in the management actions or objectives must be specified.

Under the Superfund approach, EPA evaluates the management alternatives in the FS and selects the preferred management plan in the ROD. The agency presumes that it can accurately predict the effectiveness of the alternatives it evaluates, which supports the passive adaptive management approach, and at most Superfund sites this approach is adequate. However, several of the actions proposed for protecting fish and wildlife in the lower Coeur d'Alene River basin appear to have many of the characteristics of experiments, and an explicit active adaptive management approach might be more effective in the long run.

The agency has developed a monitoring plan (the BEMP) (URS Group Inc. and CH2M Hill 2004) but, as discussed below, has not established specific indicator values that could trigger a change in the management actions or objectives.

Step 3: Implementing the plan. Implementing the management plan is a simple matter of following the plan as specified. Circumstances requiring deviations from the original plan should be identified in advance and should be understood and agreed to by all stakeholders. Any such deviations must be clearly documented.

As indicated by formal and informal conversations with the committee, EPA has clearly begun thinking about implementation of the ROD (EPA 2002) and realizes that the proposed remedies may have to be modified, perhaps substantially, when this process is under way. The possible need to modify the remedies is also reflected in the BEMP (URS Group Inc. and CH2M Hill 2004, pp. 6-12 to 6-13) and is explicitly a component of the 5-year reviews that the agency carries out at every Superfund site. However, the circumstances and indicators that would require such deviations have not been defined, and it is not clear that EPA has discussed these possibilities with all of the stakeholders.

Step 4: Monitoring. Implementation monitoring should include three components: (1) monitoring for implementation or compliance (were the actions taken as planned?), (2) monitoring for effectiveness (did the plan meet objectives?), and (3) monitoring to validate the model parameters and relationships (which hypothesis is correct?). The monitoring protocols should have been established in step 2, designing a management plan, but were not.

Table 8-7 summarizes the RAOs, actions, benchmarks,[4] monitoring parameters, and target values for actions intended by EPA to reduce risks to aquatic receptors in the Coeur d'Alene River basin. The ROD includes forecasts of the effects of the proposed actions on the future values of these parameters. At least with respect to fisheries, these indicators appear to meet the requirements of adaptive management.

For terrestrial resources, the connection between management objectives, actions, benchmarks, and indicators is much less clear. Table 8-8 summarizes the ROD's approach to establishing performance measures for waterfowl and songbirds. For these receptors, the primary source of risk is particulate lead derived from streambed deposits and streambanks. The RAOs for these receptors are intended to prevent ingestion and dermal exposure to lead "at concentrations that result in unacceptable risks." This approach does not provide an explicit metric for unacceptable risk, in terms of a tolerable dose, an acceptable rate of mortality, or a range of acceptable population characteristics. The benchmark for feeding areas specifies an amount of clean habitat that should be provided; the benchmarks for toxicity simply specify that toxicity should be reduced.

Monitoring blood lead concentrations in waterfowl and songbirds is clearly essential for documenting whether the remedial actions are reducing

[4]Benchmarks (actions and criteria) are near-term objectives that serve as "landmarks and measures" to evaluate the progress of prioritized actions to achieve long-term goals of risk reduction (EPA 2002, 8-1 to 8-3; EPA 2001a, pp. 5-1 to 5-3).

TABLE 8-7 RAOs, Actions, Remediation Benchmarks, Monitoring Parameters, and Target Values for Actions Intended to Protect Aquatic Resources

RAOs (EPA 2002, Table 8-2)	Actions (EPA 2002, Table 12.2-1)	Remediation Benchmarks (EPA 2002, Table 12.2-1)	Monitoring Parameter(s) (URS Group Inc. and CH2M Hill 2004, Table 4-3)	Target Value (URS Group Inc. and CH2M Hill 2004)
Reduce loadings and discharges of dissolved metals to levels that satisfy surface water-quality standards	Stabilize stream beds, bank, and dumps subject to erosion; implement runoff controls; construct sediment traps (Canyon Creek, Ninemile Creek, Pine Creek, South Fork)	Reduce metals toxicity to downstream receptors	Dissolved metals concentrations	Tier 1 fishery: presence of migrating fish only (zinc concentration < 20× acute AWQC)
		Reduce dissolved metals loadings discharge from Canyon Creek by at least 50%	Fish diversity and abundance	
	Treat creek water and groundwater at mouth of Canyon Creek	Improve conditions to allow:		Tier 2 fishery: presence of resident salmonids of any species (zinc concentration between 7× and 10× chronic AWQC)
	Remove, contain, or treat significant loadings sources (various locations)	• Natural reestablishment of a salmonid fishery		
		• Natural reestablishment of migratory corridors for juvenile and adult fish		Tier 3 fishery: presence of 3 or more year classes of resident salmonids, including young of year
		• Natural increases salmonid populations and improved spawning and rearing		
		• A higher fish density		

SOURCES: EPA 2002; URS Group Inc. and CH2M Hill 2004.

TABLE 8-8 RAOs, Actions, Remediation Benchmarks, Monitoring Parameters, and Target Values for Actions Intended to Protect Waterfowl and Songbirds

RAOs (EPA 2002, Table 8-2-1)	Actions (EPA 2002, Table 12.2-1)	Remediation Benchmarks (EPA 2002, Table 12.2-1)	Monitoring Parameter(s) (URS Group Inc. and CH2M Hill 2004, Table 4-3)	Target Value (URS Group Inc. and CH2M Hill 2004)
Prevent ingestion of arsenic, cadmium, copper, lead, mercury, silver, and zinc by ecologic receptors at concentrations that result in unacceptable risks	Remove contaminated bank wedges from highly erosive areas	Increase waterfowl feeding area with lead concentration <530 mg/kg by 1,169 acres	Waterfowl population	Statistically significant increase
			Waterfowl blood lead	Statistically significant decline
	Stabilize banks and revegetate removal areas		Waterfowl mortality	Statistically significant decline
		Reduce sediment toxicity to diving ducks, dabbling ducks, and warm- and cold-water fishes	Songbird diversity and abundance	Statistically significant increase in abundance and diversity
	Construct and operate sediment traps at four splay areas			
Prevent dermal contact with arsenic, cadmium, copper, lead, mercury, silver, and zinc by ecologic receptors at concentrations that result in unacceptable risks			Songbird blood lead	Statistically significant decline
	Implement periodic removal of riverbed sediments in Dudley reach or other natural depositional areas	Reduce soil toxicity for riparian receptors		
	Reduce exposure using a combination of removals, capping, and soil amendments in areas of high waterfowl use, high lead, road access, and relatively low recontamination potential			
	Identify agricultural and other areas with lower levels of lead for cleanup to provide additional clean feeding areas			

SOURCE: EPA 2002; URS Group Inc. and CH2M Hill 2004.

lead exposures. However, the BEMP does not specify a particular target blood lead concentration that should be achieved to meet the objective of prevention of unacceptable risks to these receptors. Instead, the BEMP states the target as being a statistically significant decline in blood lead concentration. A small, but statistically detectable decline in blood lead concentrations might not substantially reduce the number of birds adversely affected by lead exposures.

Similarly, to be fully consistent with the principles of adaptive management, the BEMP should specify a target reduction in the number of waterfowl killed per year, in the fraction of the migratory population in the basin that is affected, or both. Simply monitoring for a decline in mortality will not guarantee that the objective of preventing unacceptable risks will be achieved.

The BEMP also calls for monitoring the abundance of waterfowl and the abundance and diversity of songbirds. It is not clear how either of these parameters is related to the RAOs. The use of these types of measures as monitoring parameters in the BEMP involves an implicit hypothesis that current levels of lead exposure are reducing (1) the abundance of waterfowl and (2) both the abundance and diversity of songbirds. This hypothesis was not tested in the ecologic risk assessment (ERA). The abundance of waterfowl using the basin could decline because of adverse environmental conditions occurring outside the basin, even if mortality due to lead exposure were eliminated. No evidence is provided in the ERA that songbird abundance or diversity has declined because of lead exposure (as distinct from deforestation and other habitat disturbances), and target levels of abundance and diversity that would occur if lead exposures were reduced have not been specified. Testing hypotheses concerning the causes of changes in abundance and diversity requires a substantially more complex monitoring plan than that developed by EPA. Simply measuring abundance and diversity will neither test hypotheses concerning effects of lead exposures nor determine whether the RAOs have been met. Thus, at least with respect to waterfowl and songbirds, the benchmarks and monitoring parameters clearly do not currently meet the requirements of adaptive management, at least as currently formulated.

Step 5: Evaluating results obtained from monitoring. This step involves comparing the results obtained from monitoring with the forecasts in step 1. The evaluation should explain why the results occurred and should include recommendations for future action.

EPA does not appear to have established any formal evaluation process aside from the 5-year reviews, although the agency has suggested that informal evaluations may occur more frequently. One serious weakness with the EPA approach, however, is that, because the agency did not establish any quantitative short-term indicators, the agency lacks clear measures on which to base these evaluations. The committee's confidence in EPA's approach would be much stronger if the agency had established such indicators and had more formally structured an ongoing evaluation process.

Step 6: Adjusting the management plan in response to the monitoring results. This step involves following through with the recommendations from step 5. The models used to make the initial forecasts should be updated, and the objectives of management should be reviewed and possibly adjusted. New forecasts are made, and management actions are revised as necessary. Presumably, this should occur during the 5-year reviews. In its BEMP, the agency sets forth the following questions, which are to be answered during these reviews (URS Group Inc. and CH2M Hill 2004, p. 6-12):

- Is the remedy functioning as intended by the ROD (addressed through statistical analysis of trends data for monitored parameters)?
- Does interpretation and evaluation of available data from the BEMP and other monitoring programs suggest new or refined understanding of basin processes that are relevant to the remedy (addressed qualitatively)?
- Are revisions or modifications to the BEMP warranted?
- Are exposure assumptions, toxicity data, cleanup levels, and RAOs used at the time of remedy selection still valid?
- Has any other information come to light that could call into the question the protectiveness of the remedy?

These questions address most of the items listed above. Implicit in these questions is the possibility that EPA will revise the proposed remedies (not just the BEMP). Again, the weakness is that there are no clear indicators on which to base these decisions, and some modifications probably should not wait for 5-year reviews (although, as indicated earlier, EPA staff appears to anticipate making changes informally as they observe them to be necessary or appropriate).

Adaptive management, as described above, should be unequivocally incorporated into every step of the Superfund process, beginning with the RI. EPA's approach to ecologic protection in the Coeur d'Alene River basin includes many of the components of adaptive management, but it has not been established in an explicit, structured manner that establishes unambiguous links between management objectives, management actions, performance benchmarks, and monitoring indicators. The biggest weakness is that the agency often has not established a series of quantitative indicators, particularly short-term indicators that can be monitored to unambiguously determine the success or failure of the proposed remedial actions.

Ecologic Risks: Rationale for Determining Levels of Remediation

The remedies proposed for protecting waterfowl and fish differ in terms of rationale for defining cleanup goals and the complications associated with implementing remedies that will achieve the goals.

Waterfowl. EPA made a risk management decision to use a site-specific protective lead value of 530 mg/kg as the benchmark cleanup criterion for the soil and sediment in the lower basin. This level is identical to the lowest-observed-adverse-effect level derived in a waterfowl toxicity study conducted by Beyer et al. (2000). As described in Chapter 7, this level, based on high-quality site-specific research, is consistent with field observations, and is within the range of preliminary remediation goals (PRGs) developed in the ERA. No rationale was provided, however, for selecting this specific value rather than the substantially higher or lower values provided in the ERA.[5] Given the extensive reviews and analyses used to develop the range of PRGs provided in the ERA, the committee is surprised that a more complete documentation of the decision to select 530 mg/kg as the cleanup criterion was not provided.

The selected remedy proposes to remediate about 1,200 acres of the approximately 5,800 acres of wetlands having contamination levels above 530 mg/kg using a combination of removals, capping, and soil amendments (EPA 2002, Table 12.2-1) (details of this and other actions are discussed further below). Representatives of the Fish and Wildlife Service made informal comments to the committee indicating that they hope that even this partial cleanup will result in a significant decrease in risks to the waterfowl in two ways. One way results from the fact that, even if the waterfowl move back and forth between contaminated and remediated areas to feed, their average exposure, and therefore the risks they face, will be reduced. The second way is intended to reinforce this benefit; remediated areas will be replanted with vegetation believed to be particularly attractive to the waterfowl that inhabit or migrate through the Coeur d'Alene River basin. They hope to induce the waterfowl to remain in the clean areas, thus reducing their risks further.

The other major efforts to protect waterfowl involve removing contaminated sediments from the bed and banks of the lower reach of the Coeur d'Alene River to reduce the likelihood that the cleaned up areas will become recontaminated as well as to possibly reduce the transport of con-

[5] EPA does say: "While 530 mg/kg lead in soil/sediment may not be fully protective of aquatic birds and mammals, it will address 95 percent of the habitat area. Only 5 percent of the impacted area in the Lower Basin is estimated to have lead concentrations between 530 mg/kg and background. For these reasons, EPA believes that selection of 530 mg/kg lead as the benchmark cleanup criterion for soil and sediment is technically the best alternative available at this time" (EPA 2002, p. 12-39).

taminated sediment through Lake Coeur d'Alene to the Spokane River. This appears to be a largely experimental effort and EPA has not advanced new criteria for how much of this should occur or how to determine whether it is successful.

Fish. Derivation of the final remediation levels for protecting fish is more straightforward than the derivation of remediation levels for protecting waterfowl, but the process of achieving those levels is much more complicated. The remediation levels for protecting fish are defined by Idaho's water-quality standards for protection of aquatic resources, which are presumptive ARARs for the site. According to EPA's current interpretation of the NCP, the cleanup is not complete until these standards are achieved. According to EPA, additional measures may be needed to protect threatened species (for example, bull trout) and to protect and/or enhance the potential for the Coeur d'Alene River fishery to become a "blue ribbon" trout stream.

As indicated later in this chapter, it is virtually impossible for EPA to achieve the water-quality standards by the remedy proposed in the ROD, because it does not address groundwater, which is the largest source of zinc loading to the river. EPA apparently is relying on a distinct (but currently unspecified) administrative structure to address groundwater issues.

A second complication is that contaminated water is only one of the threats facing the native species of fish—nonnative fish species and lack of habitat are other threats. For instance, nonnative fish species artificially introduced into the lateral lakes, Lake Coeur d'Alene, and the Coeur d'Alene River probably have permanently altered the fish communities of the basin and may impede or even prevent the reestablishment of viable populations of native species, even if water quality standards were achieved. Moreover, even if remediation improved water quality sufficiently to protect the health of fish, habitat restoration still would be needed to support macroinvertebrate and fish populations (see discussion in Chapter 7). A key factor relating upstream biotic communities in the Coeur d'Alene River with downstream segments is that habitats are linked in river systems (Vannote et al. 1980; Minshall et al. 1992). Good-quality riparian habitats and substrates for benthic invertebrates lead to quality trout stream fisheries. The fish, particularly salmonids, in Rocky Mountain streams are adapted to cold, clear waters (Baxter and Stone 1995). Maintaining riparian zones will optimize the biodiversity, as there are more microhabitats to exploit by benthic invertebrates and fish. Trout populations are also sensitive to sedimentation of spawning grounds and mitigation efforts will need to minimize any increase in the percentage of fine sediments as a result of, for example, bank removal or river bottom dredging practices.

Thus, in the case of fish, the ARARs represent a clear, measurable indicator of when the cleanup is successful. However, it may not be possible to achieve the ARARs, and, even if they are achieved, improved water quality alone may not be sufficient to ensure the viability of the fish populations of concern.

EPA could exempt the cleanup from meeting water-quality standards if the agency could demonstrate that fish and aquatic life can be protected without achieving these standards. In principle, such an exemption could be justified if monitoring data showed that aquatic populations and communities in the Coeur d'Alene River and its tributaries had the same characteristics as populations and communities in comparable streams unaffected by mining wastes. The approach of using biological indicators rather than chemical concentrations to evaluate water quality is well-established in the scientific literature (Karr and Chu 1999). The EPA Office of Water has published a guidance document on the development of biological indicators (also termed "biocriteria") and has advocated the use of biological indicators in state water-quality programs (Barbour et al. 1999).

Further precedent for using biological indicators in lieu of numerical water-quality standards as remediation goals is provided by the approach adopted at the Lower North Potato Creek (LNPC) site in Polk County, Tennessee, the largest and most severely degraded metal-mining site in the eastern United States (EPA 2001b; TDEC Lower North Potato Creek Voluntary Oversight and Assistance Program Order, January 4, 2001). Remediation of the LNPC site is being managed under EPA's Superfund Alternatives Program, under a Memorandum of Understanding between EPA, the Tennessee Department of Environmental Conservation (TDEC), and Glenn Springs Holdings Company (GSH). Performance goals for site closure are provided in a consent order between TDEC and GSH. According to the order, remediation will be considered complete when all on-site streams meet Tennessee's biologically based water-quality criterion for the region where the site is located. Tennessee's region-specific biocriteria, which were developed with methods documented in EPA's (1999) guidance manual, are specified in terms of aquatic community characteristics found in a suite of reference streams that are relatively unimpaired by chemical contamination or habitat disturbance. A stream is considered to be unimpaired if a standardized index of aquatic community quality measured in that stream exceeds the applicable regional value, even if Tennessee's numerical water-quality criteria (which, for metals, are the same as Idaho's criteria) are not met.

A biologically based approach to determining when sufficient protection has been achieved is consistent with EPA's approach to developing interim fishery benchmarks. The agency has defined a series of five "fishery tiers" that qualitatively describe the health of the fish communities present in the river. Methods documented in EPA guidance and other published literature could be used to develop a more rigorous set of indicators that could be used both to measure the progress of restoration and to develop quantitative closure criteria that would achieve the intent of the ARARs even if the numerical standards were not met.

Biologically based indicators of restoration success would have the additional advantage that, because they reflect both water and habitat quality, they could be used to determine the need for and the success of habitat restoration actions. Establishment of biologically based restoration goals still would require EPA and Idaho to consider the influence of introduced species and also of irreversible habitat alterations (for example, channelization, road construction) that probably will prevent the Coeur d'Alene River from ever being returned to premining conditions.

Remediation: Geographic Areas and Feasibility and Potential Effectiveness of Plans

EPA outlines remedial actions for environmental protection in the basin over the next 30 years. The committee looked at these interim actions and answered the following questions:

- What remedial actions are proposed?
- What areas of the basin were included and excluded in the remedial plans? What was the basis for the decision to include or exclude areas?
- What cleanup has already been done, and was this remediation effective?
- Are the planned remedial actions feasible?
- Will the cleanup be effective in meeting the agency's goals or benchmarks?

These questions are addressed for the following five topographical areas of the basin:

- Upper basin, which includes the high-gradient streams that flow into the South Fork Coeur d'Alene River
- Middle basin, which extends from Wallace to Cataldo
- Lower basin, which extends from Cataldo to Lake Coeur d'Alene
- Lake Coeur d'Alene
- Spokane River

EPA uses a probabilistic model to quantify the certainty that a proposed remedy could meet cleanup goals (URS Greiner, Inc. and CH2M Hill 2001e, p. 1-4). Because many of the remedial actions described in the ROD for the basin are based on the probabilistic model results, this model is assessed.

Assessing the Probabilistic Model

There were two primary functions of the probabilistic model. First, in the RI (URS Greiner and CH2M Hill 2001f), the model is used to statistically evaluate extensive data sets of surface-water dissolved zinc levels to probabilistically characterize current metal loading and concentrations in the river and provide an "expected value" or estimate of current conditions. The second function, used in the FS (URS Greiner, Inc. and CH2M Hill 2001a) and the ROD (EPA 2002), was to quantify the effect that remedial measures would have on surface-water concentrations and metal loadings and the certainty and time frame that a remedial alternative or a proposed remedy would meet cleanup goals, which may be AWQC or interim benchmarks (URS Greiner, Inc. and CH2M Hill 2001e, p. 1-3).

As described in Chapter 4, the first function (the estimated mass-loading analysis provided in the RI) provided a concise and useful tool for understanding expected contributions of zinc to surface waters at locations along the river system. However, using this model to provide estimates of postremedial effectiveness and surface-water concentrations in the future is problematic.

EPA uses the probabilistic model to estimate postremediation metal loadings at selected stream-monitoring locations. Metal loadings are estimated indirectly by using relative loading potentials (RLPs), representing metal loads per unit volume of waste material. An estimated RLP is used for each source type (for example, waste rock, floodplain material). In this analysis, it is hypothesized that postremediation loading reductions are proportional to the volume remediated (URS Greiner, Inc. and CH2M Hill 2001e, Section 2.4). Predictions of what metal load reductions might be achieved are estimated for up to 1,000 years in the future. The probabilistic model is only used by EPA to evaluate dissolved zinc. However, the results are used to predict the behavior of other dissolved metals (URS Greiner, Inc. and CH2M Hill 2001e). Figure 8-4 presents the results from the probabilistic model analysis on the impact that the various alternatives presented in the FS (see Box 8-2) will have over time. In this figure, surface-water concentrations of zinc (shown as a multiple of the AWQC) over time are modeled over 1,000 years for the various alternatives. This analysis shows, for example, that under Alternative 3 (an alternative containing substantial source removals), the surface-water zinc concentrations at Pinehurst, Idaho, would decrease below the AWQC in 400 years compared with 900 years for the no action alternative. (Note that, because of OU segmentation, this analysis does not include metals contributions from the box that, at Pinehurst, would more than double the zinc loads considered [EPA 2002, p. 5-6].[6]) Several logical and technical issues are

[6]As noted in the ROD (Figure 10.2-3): "If historic loadings from the Box were included without any future reduction, AWQC multiple would increase by a factor of approximately: Alt 1, 2.1; Alt 2, 2.6; Alt 3, 4.0; Alt 4, 5.2; Alt 5, 2.3, Alt, 6 2.2."

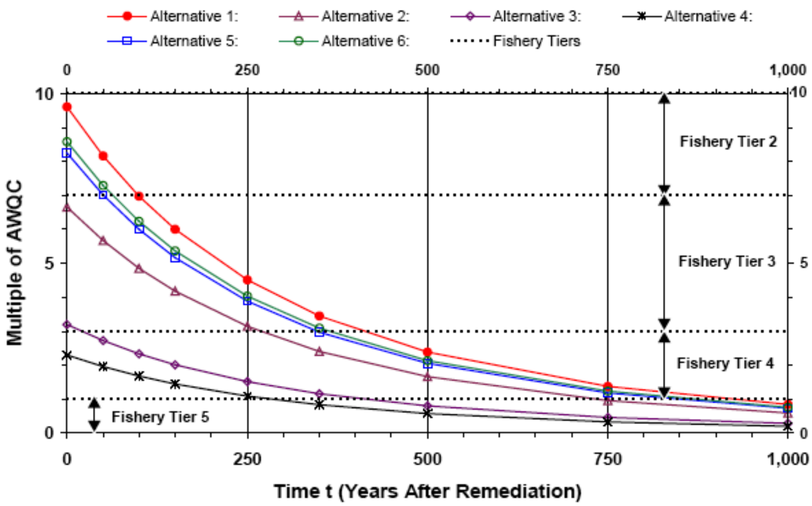

FIGURE 8-4 Comparison of the expected concentrations of dissolved zinc in surface water (presented as a ratio of the AWQC) over time at Pinehurst. Results are presented without including Bunker Hill box contributions. SOURCE: EPA 2002, Figure 10.2-3.

associated with this model and its use in extrapolating the effect of proposed remediation to 1,000 years, including the assumption that the impact of remedial strategies on the release of metals from source material to surface waters is known.

In essence, the probabilistic model estimates relative loading potentials based on estimated total volume of contaminated material, estimated concentration of available zinc, and estimated effectiveness of various remediation methodologies in reducing metal loading. The contribution of the box to dissolved metal loading is ignored, although a factor is provided that allows the box contribution to be considered.

The committee has serious doubts about the reliability of the probabilistic model to predict postremediation effectiveness. The model is based on an untested hypothesis for which no theoretical or experimental evidence is presented. For example, there are no leach test data from sediments or tailings, which would provide rates and quantities of metal release over time, allowing extrapolation of relative loading potential. Groundwater flow and metal concentrations data are not used in developing the model, although such data are available. There are no data on the effectiveness of various remediation methodologies in reducing “relative loading potential.” No formal attempt has been made to calibrate the probabilistic model in a rigorous sense other than the "calibration" that is inherent in the model's use of statistical results from historic monitoring data as the preremediation condition (EPA 2004b [July 27, 2004]) even though there have been substantial source removals and associated monitoring in the South Fork and tributaries of the Coeur d'Alene River. The overall statistical procedure and supporting technical assumptions have not been externally peer reviewed. A committee member prepared a detailed mathematical assessment of the probabilistic model for estimating metal loading and effectiveness of remedial action; it is presented in Appendix F.

Remedial Actions Proposed for Upper Basin Tributaries: Ninemile Creek, Canyon Creek, and Pine Creek

Areas slated for cleanup in the upper basin encompass Ninemile Creek, Canyon Creek, and Pine Creek. Many of the primary sources (for example, mine workings, waste rock, and tailings) of dissolved metal contamination are located in the high-gradient streams that flow into the South Fork Coeur d'Alene River. Ninemile and Canyon Creeks also have contaminated in-stream sediments and influxes of contaminated groundwater. Table 8-9 summarizes EPA's cleanup goals, planned actions, and estimated costs. Interim cleanup measures described in the ROD (EPA 2002) for Ninemile, Canyon, and Pine Creeks are anticipated to cost $85 million.

The selected remedy for environmental protection in Ninemile, Canyon, and Pine Creeks consists of cleanup actions that EPA thought could be implemented within a 30-year period and would make significant progress toward protecting the environment and ARAR compliance and that were effective, had implementability, and were cost effective—the balancing criteria for CERCLA (EPA 2002).

Ninemile Creek. Ninemile Creek has high surface-water concentrations of dissolved zinc, and the source areas of metals within this tributary are relatively well defined, with large contributions stemming from two mining areas on the East Fork. In the ROD, the probabilistic model was used to predict water-quality conditions consistent with fisheries tiers (see Table 8-7) that would result from various response actions including the installation of a pond to treat water in the East Fork before its confluence with the main stem of Ninemile Creek. The lack of available space for a regional repository for contaminant removals was also a factor in the remedial plan decision for Ninemile Creek.

Cleanup activities have been under way by the mining companies and the state of Idaho at the Success and Interstate Mill site on the East Fork of Ninemile Creek. Harvey (2000) suggests that streambed and floodplain sediment removals at the Interstate site appear to be effective in reducing zinc loading in the stream; however, EPA has commented that they are

TABLE 8-9 Goals, Remedial Actions and Costs for Upper Basin, High-Gradient Tributary Streams

Location	Remedial Goal/Benchmark	Planned Actions	Estimated Cost of Cleanup (Present Worth[a])
Ninemile Creek	Re-establish fishery above Success Mine and migration corridor below Success Mine	Evaluate success of past/current floodplain tailings removal and water treatment at Success mine; if these actions do not achieve goal, may remove and relocate tailings and cap tailings, stabilize stream banks, and install a surface-water treatment pond; potential additional actions at the Rex and Interstate mill sites	$13.5 to $36 million
Canyon Creek	Reduce, by at least 50%, dissolved and reduce particulate metal loads discharging from the creek into the South Fork	Intercept and treat creek water, possibly using passive treatment with active treatment components; stabilize mine dumps and stream banks at 11 locations	$35 million
Pine Creek	Improve conditions to allow natural increases in salmonid populations, particularly native fish, and allow spawning and rearing	Stabilize stream bank and bed; revegetate near stream; in-stream, hot spot removals	$14 million
Upper South Fork Coeur d'Alene River	Improve conditions to support a higher fish density and initial protection of riverine and riparian receptors	Stabilize and bioengineer stream channel and banks to protect riverine and riparian receptors, with associated hot-spot removals in upper floodplain; address mine/mill sites with human health exposures	$16 million

[a]The first number is the estimated cost for the cleanup; the second number is the cost with a contingency added.
SOURCE: EPA 2002.

unaware of analyses on the effectiveness of the remedial activities in Ninemile Creek (EPA 2004b [September 17, 2004]). The Silver Valley Natural Resource Trustees[7] (SVNRT) installed a pilot-scale demonstration project at the Success Mine located on the East Fork of Ninemile Creek. The objective of the project is to demonstrate the viability of a groundwater collection and passive treatment system to reduce total and dissolved cadmium, lead, and zinc (Calabretta et al. 2004). Results from this demonstration project appear to be mixed. Although good removal efficiencies have been noted, they are not consistent, and serious problems in intercepting groundwater have been encountered. The demonstration project at Success Mine exemplifies EPA's hopes that such demonstrations will lead to acceptable, passive treatment technologies for other sites. The committee encourages EPA to continue such demonstration projects and work toward improving metal-removal efficiencies and groundwater interception. These types of passive technologies are desirable for treating small or intermittent flows that come into contact with contaminated sources that cannot be excavated (for example, fill under Interstate 90 or tailings pile and adit drainages that are in remote areas with limited access).

The cleanup plan for Ninemile Creek (Table 8-9) is largely a "wait and see" plan. If the contaminant removals and groundwater treatment accomplished to date do not achieve the goals of reestablishing the fishery above the Success Mine and the migration corridor below the mine, then additional actions as outlined in Table 8-9 will be taken, including source removal and installation of a treatment pond to collect and treat creek water with the objective of removing 60-70% of the zinc load from the East Fork. The committee fully supports the agency's plan to undertake the removal of sources contributing metals to surface water, encourages stabilization actions, and endorses actions that couple fish habitat restoration with remedial actions. Without habitat restoration, achieving the goal of reestablishing a resident fishery is doubtful. Treatment of East Fork creek water would entail constructing a facility to process 10 cubic feet per second (nearly 4,500 gallons per minute). Passive treatment of this volume of water in the limited space of the canyon is expected to be difficult.

Canyon Creek. EPA considered source-by-source cleanup in Canyon Creek and concluded that this approach, which would require extensive removals, would be costly and difficult to implement in the 30-year time frame of the selected remedy. The agency also believes that the effectiveness of source-by-source removal in Canyon Creek is uncertain (EPA 2002, p. 12-27). It is unclear to the committee how EPA arrived at this conclu-

[7]The Silver Valley Natural Resource Trust Fund was formed in 1986 to administer a $4.5 million settlement between the state of Idaho and several mining companies operating within the Silver Valley.

sion, because source removal or stabilization of sources is fundamental to any remediation effort. Canyon Creek also remains a significant source of particulate lead, which continues to wash downstream during spring runoff and flood events. Until the sources of the particulate lead are removed from the floodplain or otherwise stabilized to prevent erosion, these sediments will continue to recontaminate downstream areas that have been or are proposed to be remediated. Although not explicitly stated by EPA, disposal of large volumes of source material removed from streambeds and other locations may be a serious issue given problems in finding suitable repository sites in the narrow, steep area of Canyon Creek. EPA recognizes that Canyon Creek is a major contributor of dissolved metals (about 15% of the dissolved zinc load at Harrison) to the river system and that groundwater downstream of the Hecla-Star tailings ponds contributes high concentrations of metals. It is unclear, however, how much of the groundwater contamination in lower Canyon Creek is attributable to the Hecla-Star tailings impoundments, because no definitive studies have been carried out. Erosion is observed along the side of the ponds (URS Greiner, Inc. and CH2M Hill 2001g, p. 2-7) and significant total lead and total zinc concentrations have been measured in water from seepage areas (EPA 2004b, [July 27, 2004]). Water from the Star adit is currently discharged to the Hecla-Star tailings impoundment (number 6) under a National Pollutant Discharge Elimination System permit. It is possible that this adit water is percolating through the tailings and contributing to groundwater contamination (EPA 2004b, [July 27, 2004]).

The SVNRT conducted floodplain sediment removals in Canyon Creek. One assessment by the state of Idaho (Harvey 2000) shows variable results: after removal actions, the zinc load was estimated to decline 59% under high discharges but increase 43% under low discharges. EPA considers this analysis to be based on "questionable data and fundamentally flawed analysis." EPA's analysis shows a small decrease in soluble zinc concentrations at low flow, but not high flows (EPA 2004b [July 27, 2004, and September 17, 2004]; C. Vita, URS, personal commun., September 20, 2004). As such, it is unclear if the removals conducted to date have a beneficial effect on stream-water metal concentrations. However, efforts to determine a causal relationship are confounded by limited data, a possible delay between the removal and resultant decrease in water concentrations, and the fact that the contaminated floodplain material from Woodland Park was moved to an unlined repository at the same site and apparently is serving as a source of dissolved metals to the groundwater. These issues reinforce the need for a rigorous site characterization to identify those sources contributing metals to surface water.

Stabilization of waste rock dumps and stream banks in areas around 11 mines is included in the selected remedy for Canyon Creek (Table 8-9). The committee fully endorses these actions. The ROD discusses plans to intercept groundwater and surface water near the mouth of Canyon Creek and treat this water. This plan proposes evaluating pond treatment and using passive treatment technology. The ROD anticipates that treatment of 60 cubic feet per second (about 27,000 gallons per minute) would be necessary to achieve the benchmark of 50% reduction of dissolved metal loading. In verbal discussions with EPA during the committee's tour on April 14, 2004 (EPA 2004c), agency personnel indicated they may be rethinking the idea of passive water treatment technologies for remediating surface flows from Canyon Creek and that active treatment may be used.

Treating the Canyon Creek water at the mouth of the stream will do nothing to meet EPA's overarching objective of protecting aquatic species in Canyon Creek. Moreover, the committee has considerable doubt about the efficacy of passive treatment technology for this application. Large volumes of water requiring treatment and the long retention times[8] needed demand a very large area for the passive treatment, and such an area is not available in the confines of Canyon Creek. Passive treatment systems also generate solid wastes that likely will be deemed hazardous waste, necessitating special disposal. Unprecedented innovations in passive treatment technology would have to occur over the next few years to effectively handle this situation. Active treatment technologies to treat large volumes of water are available; however, such systems also would require a large footprint,[9] generate metals-containing sludge that must be disposed, and, like passive treatment systems, are necessary in perpetuity. This remedy requires a state institutional mechanism to take full responsibility for operation and maintenance for a very long time. This issue may well be similar to the current situation at OU-2, where EPA is attempting to get the state of Idaho to enter into a Superfund state contract for operation and maintenance of the CTP located at the CIA (EPA 2004b [July 27, 2004]).

Pine Creek. Pine Creek has already experienced considerable cleanup work, particularly by the Bureau of Land Management, and the creek currently supports an adult fishery, including brook trout and native cutthroat trout. The proposed remedial action for this area focuses on habitat rehabilitation and limited removals. The committee commends EPA on efforts to restore fish habitats in upper basin tributaries. Simply removing

[8]Retention time, also called residence time or detention time, is the time that a volume of water must be in contact with the medium, or material, that removes the metal from the water. In some passive treatment technologies, the material adsorbs the metals from the water; in other technologies, microorganisms generate a product, such as hydrogen sulfide, that reacts with the dissolved metal, converting the metal into a particulate form that is filtered from the water.

[9]A footprint refers to the area required for installation of a treatment plant.

dissolved metals is insufficient to restore fisheries; to be successful, habitat restoration is critical (see Chapter 7).

Remedial Actions Proposed for the Middle Basin (Wallace to Cataldo)

The remedial benchmark for the South Fork Coeur d'Alene River between Wallace and Cataldo is to improve conditions to support a higher fish density (tier 2+ to 3 fishery) (EPA 2002, p. 12-28). EPA's expectation, as stated in the ROD, is that improvements to the South Fork will largely be due to remedial actions planned for Canyon Creek (specifically, the water treatment plant) and Ninemile Creek.

Specific cleanup plans for the South Fork over 30 years call for the removal of about 102,000 cubic yards of floodplain tailings, from what are considered hot spots from Wallace to the eastern side of the box, and some stabilization and bioengineering of the stream channel and banks at a cost of $16 million (EPA 2002, pp. 12-28 to 12-30). However, at this time, this plan is only minimally developed as the locations of the hot spots are not defined, nor are they identified by contaminant analyses, volume measurements, contaminant mobility, or other quantitative factors. Rather, EPA suggests that they will be identified by visual observation made by the Idaho Department of Environmental Quality (EPA 2004b, [July 27, 2004]). EPA dismissed more extensive floodplain sediment removal because the agency believed that this would entail excavation of deeper sediments that would be more difficult to access or that sediments with lower contaminant levels would be removed that would contribute less to achieving the remedial benchmark. The Bureau of Land Management is also planning some floodplain excavation and/or capping activities on lands owned by that agency (EPA 2002, pp. 12-28 to 12-30).

The South Fork Coeur d'Alene River has been the site of some remedial action in the past. The SVNRT conducted floodplain sediment removals at Osburn Flats, and EPA, under the ROD for OU-2, removed about 1.2 million cubic yards of mine waste from the Smelterville Flats area. No evaluations were conducted to quantify the effect of the Osburn Flats removal or the Smelterville Flats removal on water quality (EPA 2004b, [September 17, 2004]). EPA anticipates that the second 5-year review for OU-2, slated for release in September 2005, will address the effectiveness of the Smelterville Flats removals. The agency, however, offers that seeing an impact from this isolated removal may be difficult (EPA 2004b, [July 27, 2004]).

As mentioned previously, EPA concludes that groundwater influxes to the South Fork are the major sources of dissolved metals in this river. However, the committee recognizes that much of the information to implicate specific source areas contributing dissolved metals currently does not exist. As such, it is not possible to link metal loading in surface or groundwater with floodplain sediments or deeper aquifer (or alluvium) sediments, because the metal distribution between these sediments (and their relative contribution to groundwater) has not been characterized. Virtually no leach studies were conducted to assess metal dissolution rates and amounts from particular sediment types, nor has a hydrologic model that describes sources of water and their interactions been developed for the South Fork (or any other area) of the basin. Limited, but illuminating, groundwater studies by Barton (2002) point to a significant contribution of dissolved metals from groundwater influxes near Osburn in the South Fork. Tracer-injection and synoptic sampling techniques (Kimball 1997; Kimball et al. 2002) could prove useful in the middle and upper basin as tools for determining source areas contributing dissolved metals (also see Chapter 4 of this report).

Despite the significant contribution of metals from groundwater influxes, which EPA acknowledges, the agency has explicitly excluded groundwater treatment from the ROD for OU-3. The committee explored EPA's rationale for this decision and found the reasoning ambiguous (see Box 8-4).

BOX 8-4 EPA's Consideration of Groundwater in OU-3

EPA has not clearly stated its rationale for excluding groundwater in its remedial decisions for ecologic protection. The rationale outlined in the ROD can be summarized as follows:

Within the ROD, EPA recognizes that groundwater in the valley-fill aquifers of the upper and middle basin areas are the largest sources of dissolved metals loading to the river and streams (EPA 2002, p. 5.6) and indicates that groundwater will be evaluated later as the Selected Remedy is implemented (EPA 2002, p. 6-4). Conclusions in the ROD derived from the Ecological Risk Assessment (EPA 2002, p. 7-23) are that groundwater was not evaluated because it doesn't come into contact with animals. However, the agency included a groundwater RAO for the protection of ecological receptors: "Prevent discharge of groundwater to surface water at concentrations of cadmium, copper, lead, and zinc that exceed potential surface water quality ARARs" (EPA 2002, p. 8-6). Alternative 3 from the FS is outlined in the ROD and includes a regional active water treatment plant for collected groundwater (EPA 2002, p. 9-9) at Canyon Creek and Mission Flats near Cataldo (EPA 2002, Table 9.2-1). However, groundwater treatment in the South Fork (excluding the box) was eventually dismissed and not included in the Selected Remedy, because EPA concluded that treatment would do less to improve conditions than other actions (EPA 2002, p. 12-29).[a]

[a]EPA hopes that actions taken to date within the box will reduce zinc loading to the South Fork but has not ruled out future RODs, amendments to RODs, or ESDs (explanation of significant differences) if loadings are not reduced (EPA 2002, p. 12-30).

Based on removals that have been conducted up to this point, the committee has not seen evidence that suggests that removals in the basin will actually decrease surface-water concentrations of zinc, although it would be anticipated if the materials were contributing zinc to the surface water. As described above, groundwater is the primary conduit of dissolved zinc to surface water in the upper basin. Therefore, further characterization needs to be conducted to ascertain the materials and source areas contributing zinc to groundwater (which discharges to surface water) or to directly address groundwater if metal loading to the groundwater is determined to stem from subsurface materials too deep or impractical to be removed.

The committee supports the agency's plan to remediate floodplain sediments and stabilize stream banks in the South Fork Coeur d'Alene River to reduce downstream lead loading, lessen contaminated sediment transport downstream, and rehabilitate stream banks to enhance the fishery. Without removing, capping, stabilizing or treating sources, recontamination of downstream remediated sites is inevitable. The committee advocates prioritizing sources so that the most serious contributors to metal contamination are cleaned up first. It is the committee's understanding that the Basin Commission[10] will establish priorities, but the committee believes that, in some cases, this may be difficult, because of lack of data on how much contamination is contributed by source areas (also see discussion in Chapter 4 of this report).

Remedial Actions Proposed for the Lower Basin (Cataldo to Harrison)

Lower basin cleanup actions, summarized in Table 8-10, include those to address the riverbanks, riverbed, and the floodplain. The selected remedy aims to reduce particulate lead loading in the river, reduce toxicity, and reduce human exposure. Some remedial work for protecting human health is ongoing in the lower basin, including the cleanup of several boat ramp and adjacent recreational areas along the Coeur d'Alene River and lateral lakes. Some riverbank stabilization efforts have been conducted principally to minimize erosion of the banks from powerboat wave action. The targets

[10]In 2001, the Idaho Legislature established the Coeur d'Alene Basin Environmental Improvement Project Commission (Basin Commission), which is a governmental authority composed of the federal government, the Coeur d'Alene tribe, the states of Idaho and Washington, and the local counties. The Basin Commission will coordinate environmental response and natural resource restoration throughout the affected area and implement the 2002 ROD approved pursuant to the CERCLA. In August 2003, the Basin Commission issued a 5-year recommended plan outlining the scope and objectives of the proposed work for the years 2004-2008 and the lead planning agencies (Basin Commission 2003). This committee was not asked to consider the structure, development, or effectiveness of the Basin Commission and has not done so in this report.

TABLE 8-10 Summary of Remedial Actions and Benchmarks for Ecologic Protection in the Lower Basin

Area	Benchmark	Actions	Estimated Cost
Lower basin stream banks and beds (riparian and riverine)	• Reduce particulate lead load to river • Reduce soil toxicity for songbirds, small mammals, and riparian plants along 33.4 miles of river by removing 122 acres (30-ft-wide zone) • Reduce human exposure (recreational and subsistence users)	• Do complete removal of contaminated bank wedges from highly erosive areas; where complete removal is not possible, do partial removal possibly followed by capping with clean topsoil • Stabilize banks and revegetate removal areas to protect ecologic receptors and humans • Construct and operate sediment traps at four splay areas after implementing pilot study at one area • Implement periodic removal of riverbed sediments in Dudley reach or other depositional areas identified during remedial design phase	$71 million
Lower basin floodplain	• Wetlands: reduce sediment toxicity and waterfowl mortality; increase feeding areas with lead concentrations less than 530 mg/kg by 1,169 acres and possibly convert 1,500 acres of agricultural land to feeding area • Lakes: reduce sediment toxicity to duck and fishes by remediating 1,859 acres with sediment lead concentrations exceeding 530 mg/kg • Riparian: reduce soil toxicity • Reduce human exposure (recreational and subsistence users)	• Reduce exposure using combination of removals, capping, and soil amendments in areas of high waterfowl use, high lead, road access, and low recontamination potential: Lane Marsh (wetland = 213 acres) Medicine Lake (wetland = 198 acres; lake = 230 acres) Cave Lake (wetland = 190 acres; lake = 746 acres) Bare Marsh (wetland = 165 acres) Thompson Lake (wetland = 300 acres; lake = 256 acres) Thompson Marsh (wetland = 59 acres; lake = 122 acres) Anderson Lake (wetland = 44 acres; lake = 505 acres) • Identify agricultural and other areas with lower levels of lead to provide additional clean feeding areas	$81 million

SOURCE: Adapted from EPA 2002, Table 12.2-1.

cited in Table 8-10 for cleanup under the current ROD were selected by EPA for the following reasons (EPA 2002):

• The selected remedy is what EPA believed could be implemented within an approximate 30-year period and would make progress toward the five CERCLA balancing criteria; protecting human health and the environment, ARAR compliance, effectiveness, implementability, and cost-effectiveness.

• These measures are what EPA thought could achieve the benchmarks (near-term objectives).

Streambank remediation. The grounds EPA gives for cleanup of 33.4 miles of riverbanks (122 acres) along the main stem of the Coeur d'Alene River are to reduce particulate lead loading in the river; reduce soil toxicity for songbirds, small mammals, and riparian plants; and reduce human exposure. The potential exposure to humans during recreation on riverbanks is understood, but the committee questions justifications about wildlife exposure and particulate lead loading in the river for the following reasons:

• There appear to be insufficient data to assess what levels of particulate lead affect songbirds, small mammals, and riparian plants, and what, if any, benefit would be observed when the streambanks are remediated. Although research has been conducted to document exposure to lead in songbirds (for example, Johnson et al. 1999), particularly through ingestion, these results are not nearly conclusive enough to warrant the degree of remediation proposed relevant to ecologic risk in songbirds (see Chapter 7 for further discussion). The benchmarks that have been established for the ecologic receptors are also not quantitative indicators that can be readily monitored. Therefore, it will be very difficult to determine the success or failure of the proposed remedial action. This aspect is discussed in more detail earlier in this chapter.

• It is estimated that only 4% of the lead in the depositional environment of the lower basin resides in the riverbanks (Figure 8-3). Therefore, removal of this amount of lead, compared with the amount that resides in the streambed, will have minimal impact on particulate lead loading in the river. Bookstrom et al. (2004) estimate that riverbank erosion contributes only about 3% of the lead-rich sediment deposited annually on the downstream floodplain and about 3% of that deposited in Lake Coeur d'Alene.

The committee has serious doubts about the long-term efficacy of remediating the streambanks because flooding and resultant recontamination would undo any reductions in soil toxicity or human exposure. During high-flow events, the river overruns its banks, which, in addition to eroding the banks, deposits fresh lead-enriched sediment. Baseline deposition rates on riverbanks are high, averaging 6.9 ± 5.3 centimeters per decade at 3,400 ± 900 parts per million (ppm) of lead (Bookstrom et al. 2004, p. 29).

Some streambank remedial action that is ongoing entails rip-rapping the banks with cobble stones; although this approach appears to stabilize the banks, rip-rap is not a conducive fishery habitat (see Chapter 3, Box 3-1, and Chapter 7). During the design phase, the committee anticipates that EPA will give due consideration to fishery habitat restoration in any actions related to streambank stabilization.

Streambed remediation. The ROD (EPA 2002) calls for removing up to 2.6 million cubic yards of contaminated sediment from the streambed in natural deposition areas such as near Dudley. The rationale for this action is to reduce the movement of lead in surface water. The transport of lead particles by the river is the principal mechanism for transporting lead downstream. Bookstrom et al. (2004) estimate that 70-80% of the particulate lead entering Lake Coeur d'Alene is derived from the riverbed downstream of Cataldo and that 44-48 times more riverbed surface area is exposed to erosive water flows than riverbank surface area. Further, highly contaminated sediments are buried in the lower basin riverbed and they are susceptible to scouring and transport during flood events. The volume, lead concentration, and potential for transport make riverbed sediments a key component of any remedial strategy.

According to what is presented and costed in the ROD, EPA intends to dredge riverbed sediments, dewater the sediments, and treat the water in a settling pond before releasing the dewatering product back to the Coeur d'Alene River. In the ROD, EPA did not consider treating the aqueous dewatering product[11] which can contain high concentrations of zinc. To illustrate, in November 2000, USGS (Balistrieri et al. 2003) collected pore water from sediments at the river's edge at Cataldo. The sediments at this sampling location would be submerged when the Coeur d'Alene River rises during the summer, spring runoff, and flood events. Pore water samples, collected within the sediments at discrete depths ranging from 10 to 25 centimeters showed zinc concentrations ranging from about 13,000 to 36,000 μg/L. Further, oxidation of metal-bearing sediments during their removal and settling can lead to additional metals releases. The release of untreated water from the dredging operation would likely be unacceptable. Treatment of the dewatering product will produce sludge, which must be disposed of in a secure repository.

[11]The aqueous dewatering product is the river water that drains from the sediment after the sediments are removed from the riverbed.

In the riverbed, dredging is a temporary measure because the depositional areas of the river will fill back in with contaminated sediment transported from upstream primary and secondary sources. EPA has considered this and plans to redredge several times throughout the 30-year time frame of the interim ROD. Although dredging and redredging have merit, because sediment conveyed from upstream will be deposited in the same area, the volume of contaminated sediment that will be removed from the streambed is small compared with the total amount of affected sediment deposited in the entirety of the main stem of the Coeur d'Alene River. The committee questions whether removal of such a small amount of sediment will have any measurable effect on lead-enriched sediment transport and deposition downstream and also questions what effect dredging may have on fluvial behavior. Dredging was practiced near Cataldo for some 30 years starting in the 1930s; some lessons surely were learned from this dredging activity. It also needs to be considered that the sediments that refill an area (and are slated for redredging) will likely be lower in concentration than the highly contaminated historical depositions adjacent and elsewhere in the riverbed. As mentioned by Bookstrom et al. (2004), "the dredged river channel probably would re-fill with relatively dilute metal-bearing sediment, transported from the confluence of the North and South Forks, and containing about 2000 ppm of Pb." One thing is for certain—until contaminated sources that exist both upstream and in the lower basin riverbed are removed or otherwise stabilized, particulate lead transport down-river is inevitable.

The ROD states that "other sediment management techniques that may be viable alternatives to [riverbed] sediment removals for reducing particulate lead transport and providing long-term protection will … be evaluated in remedial design" (EPA 2002, p. 12-34).

According to EPA (Dailey 2004) the "ROD thus leaves open the possibility of (for example) capping, rather than dredging, riverbed sediment sources." Capping as an alternative to dredging was further explored by Bookstrom et al. (2004), as was a dredging approach that began at Cataldo and progressed down-river from there. The committee commends EPA for retaining the flexibility to consider alternatives based on new information. All alternatives should be considered on their likelihood of reducing downstream transport of metals and contamination of adjacent wetland areas. The committee also suggests that alternatives be examined to consider: effects on fishery habitat; the potential for release of metals during remedial work; and the effect on fluvial dynamics, particularly the potential for scouring of highly contaminated riverbed sediments. Further studies on the fluvial dynamics of the system will be needed to support these decisions.

Floodplain sediments. Cleanup plans for the wetlands and lateral lakes include removing the top foot of contaminated sediment, which is the sediment ingested by the waterfowl, disposing of this contaminated material in upland or subaqueous repositories, and capping deeper contaminated sediments with clean fill, possibly derived from clean wetlands, marshes, or lateral lakes in the vicinity. EPA also intends to further evaluate phosphate amendments to stabilize lead. To minimize possible recontamination from flood events, levees will be enhanced and floodgates installed.

The interim remedy proposes remediating about 25% (4,528 acres) of wetlands and lateral lakes in the lower basin that waterfowl use during their migration through the basin.[12] RI studies indicate that more than 18,000 acres of waterfowl habitat exceed the adverse-effects level of 530 mg/kg. Because the total contaminated floodplain area in the lower basin is so large, it was recognized that all areas needing long-term cleanup could not be addressed completely in the interim action. Thus, EPA prioritized specific areas. EPA states that these areas were selected based on the following criteria: (1) high use by waterfowl, (2) high levels of lead in sediments, (3) ease of site access, and (4) relatively low potential for recontamination during flood events. However, it is unclear to the committee how areas with low potential for recontamination were selected, as EPA provided to the committee that "adequate data were not available to rigorously delineate areas susceptible to recontamination based on projected average return intervals of flooding events. In particular, the maximum flood level elevations for potential design events and the detailed topography (1-foot contours) required to make such estimates were not available" (EPA 2004b, [June 23, 2004]).

EPA recognizes that available evidence is circumstantial as to whether cleaning up 25% of the contaminated feeding ground will result in a reduction of waterfowl mortality [EPA 2004b (April 6, 2004)]. The Fish and Wildlife Service, with whom the committee met, thought that even this partial cleanup would result in a significant decrease in risk to waterfowl (see discussion above in Ecologic Risks: Rationale for Determining Levels of Remediation). However, the committee is concerned about the potential of recontamination (see below) and the potential that remediated wetlands would be less desirable to waterfowl.[13] Overall, EPA recognizes that a partial effort is not enough to protect migratory birds under the Migratory Bird Treaty Act (EPA 2002).

[12]To be specific, the ROD proposes remediating 1,169 acres of wetland area and 1,859 acres of lake bottom (lake areas less than 6 feet deep) and converting an additional 1,500 acres of land currently used for agricultural purposes to safe waterfowl feeding areas. This 4,528 acres is approximately 25% of the estimated 18,000 acres of wetlands with lead concentrations greater than 530 mg/kg.

[13]Remediated wetlands could potentially be less desirable if vegetation is not reestablished or if that vegetation is not attractive waterfowl habitat. The ROD does not discuss reestablishing wetland habitats conducive to waterfowl following remediation.

Even with large monetary expenditures to remove contaminated sediments, store them in repositories, and construct levees and floodgates, the committee recognizes that severe flood events, which the valley has experienced in the past and will experience again in the future, can undo even the most well-designed and costly remedial actions. It is inevitable that recontamination will occur to some portion or all of what is remediated unless upstream and instream sources are removed and/or stabilized first. This issue is nicely summarized by Bookstrom et al. (2004):

> During episodes of high discharge, Pb-rich sediments will continue to be mobilized from large secondary sources on the bed, banks, and natural levees of the river, and will continue to be transported to the floodplain, and deposited during floods, which occur frequently. This probably will continue for centuries unless major secondary sources are removed or stabilized. It is therefore most important to design, sequence, implement, and maintain remediation in ways that will best limit recontamination.

The committee also cautions that flood control actions, such as enhanced levees, likely will affect river flow and could cause undesirable consequences. This also was considered by Bookstrom et al. (2004). The committee encourages EPA during the remedial design phase to carefully evaluate the consequences of flood control actions.

Also, although soil amendments with phosphate should be considered as a way to sequester lead, the committee cautions that nutrient-based amendments in particular could be problematic because of possible downstream eutrophication effects from excess nutrient runoff.

The committee encourages EPA's efforts to secure agricultural lands, converting them to high-quality feeding grounds. Although it has not been described which lands will be acquired, their level of contamination, or how effective such efforts may be in directing the waterfowl from contaminated areas, reestablishing wetlands in these areas is a laudable effort, particularly if these areas are less susceptible to contamination from flooding.

The other major efforts to protect waterfowl involve removing contaminated sediments from the bed and banks of the lower reach of the Coeur d'Alene River to reduce the likelihood that the cleaned-up areas will become recontaminated as well as to possibly reduce the transport of contaminated sediment through Lake Coeur d'Alene to the Spokane River. This appears to be a largely experimental effort, and EPA has not advanced criteria for evaluating whether it is successful.

According to the agency, the decision to remediate a portion of the wetlands was based on evaluation criteria for Superfund remedial alternatives, key issues associated with implementation of the alternatives, and the input of various stakeholders (states, tribes, federal trustees, and the public) (EPA 2004b [April 6, 2004]). It is unclear how Superfund remedial alternatives were considered, as many criteria (for example, protection of ecologic health, compliance with ARARs, long-term effectiveness, and permanence) likely will not be met. It appears likely that this decision was made primarily on input from various stakeholders. Regardless, decisions about remedial actions proposed in the floodplain of the lower basin need to seriously consider the impact and potential of recontamination as it can quickly undo costly, time-consuming, and resource-intensive remedies.

Lake Coeur d'Alene

Lake Coeur d'Alene is not included in the interim action, because its cleanup is to be addressed via a lake management plan (Coeur d'Alene Basin Restoration Project 1996, 2002; IDEQ 2004) under separate regulatory authorities. Lake Coeur d'Alene will be addressed in a future ROD (EPA 2004a).

There is currently uncertainty about the fate and transport of nutrients and metals after they are released from the lake sediments into the water column (benthic flux) and about the mass balance of metals in the lake on a seasonal basis (see discussion in Chapter 4). Lake Coeur d'Alene is currently the subject of a 3-year, integrated metal-nutrient flux study. Such studies to generate a greater understanding of metals dynamics are needed before a viable lake management plan can be developed and implemented for metals (also see discussion in Chapter 4).

Spokane River

For the Spokane River in the state of Washington, the ROD (EPA 2002) identifies cleanups for a limited number of sediment and soil sites in and adjacent to the Spokane River. These cleanups, estimated to cost between \$4.5 million and \$11 million, are specified for both human health and ecologic risks. Contamination with polychlorinated biphenyls, unrelated to past mining operations, appears to be a more serious issue than metal contamination.

EPA anticipates that implementation of the selected remedy will result in a reduction of dissolved metal loads in the Spokane River of approximately 16% (EPA 2002, p. 12-41). The 16% reduction is anticipated from the selected remedy based on analysis with the probabilistic model. As indicated in the earlier section "Assessing the Probabilistic Model," the committee questions the ability of this model to accurately estimate the effect of remedial actions.

The committee believes that, until upstream source areas are cleaned up, recontamination of remediated areas in and along the Spokane River will be highly probable.

Concluding Thoughts on Remediation of the Coeur d'Alene River Basin

It is apparent to the committee that EPA did not apply either a systems approach (see Chapter 4), which would consider all contaminant sources and all paths of contaminant transport, or a river continuum theory (Chapter 3, Box 3-1) that integrates the entire hydrologic system to the health of the fishery to the design of the selected remedy. Rather, it appears that EPA considered each region of the basin as a separate unit and attempted to develop a remedy for each unit or contaminant problem within that unit. As a result, the remedies are incongruent and do not address the contaminant problems of the basin in a prioritized, systematic manner. One consequence of not using a systems approach that is of particular concern is that recontamination of remediated areas is inevitable.

Particularly troubling is the fact that necessary repositories do not currently exist and potential locations are quite limited in the basin. The siting, design, and public comment stages will take years to complete if a suitable location can be established. Because the ecologic remedies are based primarily on removals of media that require secure storage, any proposed remedies will be delayed for a considerable time.

Another concern of the committee is that EPA primarily used average conditions in designing remedies. For example, average mass loadings were used, despite the fact that metal concentrations at low flows are higher, and, therefore, conditions at low flows are more toxic to aquatic life. At stream flows higher than average, particulate metal concentrations are higher and could result in recontamination of areas that were remediated based on average conditions. The committee believes that these variations may have a significant impact on the effectiveness of the proposed remedies.

Further, it is obvious that floods play a fundamental role in the resuspension and distribution of contaminants in the basin.[14] In particular, the scouring effect of these large floods mobilizes highly contaminated sediments that have been deeply buried. The timing, intensity, and duration of these floods markedly affect the potential for sediment transport. The

[14]"During low-flow periods, total lead loads as low as 30 pounds per day have been measured in the Coeur d'Alene River at Harrison. By contrast, during the 100-year flood event in February 1996, an estimated 1,400,000 pounds of lead were discharged to Coeur d'Alene Lake in a single day" (EPA 2002, p. 5-7).

negative impact of resuspended sediments on human and environmental health coupled with the expense associated with potential remediation and recontamination make it necessary to consider management of the entire watershed to reduce the intensity, frequency, and duration of floods. It is expected that watershed management practices (particularly road density) are linked to water yield and peak flood discharge in the basin (Isaacson 2004). Overall, the basin is experiencing "a more rapid response to runoff producing events [precipitation], with possibly greater peak flows (a flashier hydrograph) than historically occurred . . ." (Idaho Panhandle National Forests 1998, p. 48). To the extent that water yield and flooding can be managed through land-use practices, it is important to include them in the schemes designed to protect human and environmental health.

Given the unrelenting contribution of metal contaminants from sources in the upper and middle basins and the pervasive nature of the deposition of contaminants in the lower basin, it is entirely conceivable that the basin cannot be fully cleaned up by remedial efforts alone. There is even considerable uncertainty about whether remedial objectives set forth in the interim ROD are achievable. However, a number of remedial actions discussed in the ROD and considered in this section of the report are laudable efforts and should be pursed by EPA and others.

What is certain is that, until sources in the upper and middle basins are cleaned up, contaminants will continue to move downstream and mix with the relatively clean but large sediment load from the North Fork Coeur d'Alene River; these collective sediments will deposit in the streambed, stream banks, wetlands, marshes, and lateral lakes of the main stem of the river and eventually settle into Lake Coeur d'Alene.

Natural recovery is a central component of EPA's remedial action plan that predicts outcomes up to 1,000 years in the future. This process will be facilitated if source removal/stabilization in the South Fork and main stem of the Coeur d'Alene River occurs. Deposition rates throughout the lower basin are rapid enough that sediment loads would (if uncontaminated by sediments from the South Fork and resuspension of riverbed sediments in the main stem) expedite natural remediation of the basin.

Clearly, a great deal of new information has been collected by USGS, the Idaho Department of Environmental Quality, the Coeur d'Alene tribe, EPA, and others on sediment dynamics in the South Fork, the North Fork, and the lower basin. Much of this information has become available since the RI was released in 2001 and the ROD was issued in 2002. Many of the remediation plans proposed to mitigate damage to ecologic systems (particularly those involving lead in sediments) have been severely criticized, and recent studies tend to support some of the criticism. The committee believes it is appropriate that EPA develop a holistic methodology to remedial design using a systems approach for sediment dynamics, deposition, and biogeochemistry for the basin as a whole and a river continuum philosophy for habitat restoration that takes into consideration new scientific information.

CONSIDERATION OF NCP CRITERIA AND ADHERENCE OF ACTIONS TO SUPERFUND GUIDANCE

Adherence of Actions to Superfund Guidance

EPA's decision-making process regarding remedial actions in OU-3 of the Coeur d'Alene River basin followed the NCP (40 CFR 300), which is applicable to all Superfund sites. EPA expanded the Superfund site to include lands and waters outside the area surrounding Kellogg addressed in OU-1 and OU-2 after the agency determined the area met the criteria for listing a site on the national priorities list. The agency then proceeded through the RI/FS process of investigating the nature and extent of the contamination (see Chapter 4) and conducting risk assessments (see Chapters 5 and 7). EPA conducted a feasibility study and selected a remedy consistent with the NCP 40 CFR 300 and the CERCLA guidance for conducting an RI/FS (EPA 1988), cost estimating (EPA 2000a), and remedy decision making (EPA 1999). Under this process, EPA developed a range of remedial alternatives, presented in the FS (URS Greiner, Inc. and CH2M Hill 2001a) and described earlier in this chapter. EPA then worked with governmental stakeholders to develop a proposed plan (EPA 2001a) with a preferred alternative, and following a period for public and stakeholder review, developed a selected remedy (EPA 2002).

During this process, the agency has made a substantial effort to work with other federal, state, and local governmental (including tribal) organizations concerned about the human health and ecologic risks in the basin and to inform and receive comments from the concerned public about its findings and actions. A review in March 2004 by the EPA Office of Inspector General Ombudsman (EPA 2004d) found that Region 10 EPA had met and gone beyond requirements for soliciting and including community involvement during the process. Indeed, in the experience of the committee members, the number of cooperating organizations, processes established to provide avenues for citizen participation, and opportunities for the public to obtain information and provide written and verbal input have been substantially greater than what is normal at Superfund sites. Of course, the geographical extent of this site and the fact that it affects two states and two tribes as well as numerous localities necessitates more cooperation and public involvement than a more typical site. Nevertheless, the committee believes that the agency has been unusually open and inclusive in its process.

Although EPA adhered to the typical Superfund process, the Coeur d'Alene River basin is anything but a typical Superfund site, and the nature and extent of the site have created a number of difficulties.

Consideration of National Contingency Plan Criteria

One of the major problems has been the agency's difficulty in identifying remedies that satisfy the nine criteria for evaluating remedies described in Table 8-5 (40 CFR §300.430(e)(9)(iii)). The following sections discuss the extent to which remedial activities address these criteria.

Protecting Public Health and the Environment

The first of the two "threshold criteria" is "protection of human health and the environment." It is expected that cleanup of contaminated soils in yards, recreational facilities, and other sites is expected to be protective of human health, assuming that remediation leads to a decrease in lead intake in children (for further discussion see Chapter 5), and so long as these cleanups are maintained. Similarly, providing alternative sources of drinking water or point-of-use water filters to homes and businesses whose water supply does not meet ARARs is protective of public health.[15] As EPA points out, however, its proposed remedies do not allow for subsistence lifestyles or unlimited recreational use of contaminated areas, and they do not address future use of groundwater (EPA 2002, p. 12-2). Nor has the agency proposed a remedy to address contamination problems in Lake Coeur d'Alene. (EPA 2004b [June 14, 2004]).

The committee is less sanguine about the likelihood of success the proposed remedies will have in protecting the environment (see section Selected Remedy: Geographic Areas, Levels of Remediation, and Remediation Plans). The proposed remedies will not lower the amount of surface-water contamination (particularly from dissolved zinc) to levels specified in water-quality standards to protect native fisheries. Nor is it clear that cleaning up only 25% of the basin's wetlands will provide adequate protection to migratory waterfowl. Nineteen of the migratory bird species in the basin are considered to be at risk from the contamination in the basin (EPA 2002, p. 8-2). EPA recognizes that its proposed remedies may not fully protect human health and the environment and therefore has designated the selected remedies as interim measures, stating in explanation:

[15] One caveat on this conclusion is that the point-of-use water filters will have to be properly maintained if they are to continue to be effective. Indeed, improper maintenance can result in the quality of the output water being worse than the quality of the input water (Health Canada 2005).

> The Selected Remedy is designed to provide prioritized actions towards meeting the statutory requirement of protectiveness of human health and the environment. Accordingly, the Selected Remedy, by its nature, need not be as protective as the final remedy is required to be under the statute. Here, the Selected Remedy is sufficiently protective in the context of its scope, even though it does not, by itself, meet the statutory protectiveness standard that a final remedy would have to meet. (EPA 2002, Declaration, p. 6)

Compliance with ARARs

The second "threshold criterion" is that the remedies have to comply with all federal and state standards or other requirements that are relevant to the proposed cleanup. These standards and requirements are commonly called ARARs.

The ROD lists 35 ARARs and 10 additional guidance, policy, or other materials that EPA has to consider in selecting its final remedies (EPA 2002, pp. 13-7 to 13-16). The agency has sorted them by type as indicated in Table 8-11. The committee has not evaluated the relevance of the ARARs that EPA has identified, nor has it attempted to identify any that the agency has not. The committee does note, however, that (1) the agency did not identify any ARARs or other factors "to be considered" adopted by the tribes or local or regional governmental organizations;[16] (2) the proposed lake management plan may result in the adoption of policies or even regulations that will need to be included in the final list of ARARs; and (3) other environmental quality regulations have been or may be adopted by the state or federal governments before the final remedies are selected (presumably not for several decades at the least), and these too will become ARARs.

With respect to the ARARs that EPA identified, the remedies directed at protecting human health generally appear to satisfy the applicable rules. The only ARAR governing soil contamination was an EPA guidance document recommending a screening level for lead contamination in soil of 400 mg/kg. This recommendation was based on the results of applying the integrated exposure uptake biokinetic model with the "default parameters." In OU-3, a higher screening level was selected with site-specific parameters (see Chapter 6), which is consistent with EPA guidance.

Providing alternative water supplies or point-of-use water filters should be adequate to satisfy drinking water ARARs. Air pollution problems could be caused by soil blowing off construction areas and soil repositories, but wetting these areas, as called for in the remedies, is expected to

[16]However, as indicated below, EPA is evaluating the applicability of water-quality standards adopted by the Coeur d'Alene Tribe.

TABLE 8-11 Number of ARARs, by Category and Jurisdiction, Identified as Pertinent to Bunker Hill Mining and Metallurgical Complex Operable Unit 3

	Jurisdiction			
Category of ARARs	Federal	State	Tribe	Local
Waste management and repository design	2	5		
Air quality	1	3		
Surface-water quality	3	4		
Drinking water quality	1	1		
American Indian concerns and cultural resources protection	4			
Special status species	2	2		
Sensitive environments	3	2		
Other requirements	1	1		
Other policies and guidances to be considered[a]	9	1		
Total (ARARs, to be considered)	17, 9	18, 1	0, 0	0, 0

[a]These are not formal ARARs but rather guidance, policy, or other unpromulgated materials that are to be considered in selecting remedies (EPA 2002, pp. 13-7 to 13-16).

control these problems and satisfy the air pollution ARARs as well as the Idaho Rules for Control of Fugitive Dust.

With respect to achieving those ARARs pertaining to protecting fish and wildlife, however, the interim remedies are likely to be less successful. As the agency states in the ROD, “Although the Selected Remedy is not anticipated to be fully protective of the environment and achieve environmental ARARs, it represents what EPA believes is a significant step toward these goals” (EPA 2002, p. 10-8).

The biggest difficulty is in meeting water-quality standards for dissolved zinc, cadmium, and lead established to protect fish and other aquatic organisms. Currently, the agency argues only that its proposed actions will reduce the time required to achieve such standards, although it will still require hundreds of years to do so. Further, the ROD stated that at least a 50% reduction in lead loading may be needed to attain the AWQC in the Spokane River (EPA 2002, p. 12-110). Yet, it is not clear that actions in the selected remedy are intended to achieve that mark.

It is also unclear whether the interim remedies focused on cleaning up the wetlands and lateral lakes in the lower basin will provide adequate protection for the migratory bird species to satisfy the requirements of the Migratory Bird Treaty Act ARAR.

Several new rules, which probably will qualify as ARARs, have been adopted since the ROD was prepared. One of these is the total maximum daily load (TMDL) restrictions that are being imposed on surface waters not achieving water-quality standards. Proposed TMDLs for dissolved zinc, cadmium, and lead will create particularly serious challenges in the South Fork and main stem of the Coeur d'Alene River during low-flow periods. Because the amount of dissolved zinc entering the river from apparently uncontrollable groundwater flow (see Chapters 3 and 4 and discussion earlier in this chapter) is sufficient by itself to create violations of this standard, the agency will be forced to virtually prohibit any point source discharges of zinc during these periods. Such prohibitions presumably would severely limit the agency's ability to discharge dredging waters back to the river and also would affect the operation of its wastewater treatment facility in Kellogg.

A second new rule is the Idaho groundwater-quality rule, which includes numeric groundwater-quality standards (EPA 2000b, p. A-4). These standards are identical to the federal maximum contaminant levels (MCLs) for drinking water. The rule also lists secondary constituent levels equivalent to the federal secondary MCLs. EPA's initial determination is that the primary standards are "potentially relevant and appropriate" and that the secondary standards are "potentially to be considered" (EPA 2000b, p. A-4).

A third rule for water-quality standards was adopted by the Coeur d'Alene tribe in 2000. The applicable water-quality standards in this rule are virtually equivalent to those adopted by the state of Idaho except that the human health protection criteria are based on higher daily amounts of fish consumption than the EPA and Idaho standards. The agency apparently is still reviewing the tribe's rule. It is not clear what effect these standards would have on the proposed remedies, particularly in that they apply only to the southern portion of Lake Coeur d'Alene.

EPA does not claim to have satisfied all the ARARs with its interim measures, stating that

> The remedial actions selected in this ROD are not intended to fully address contamination within the Basin. Thus, achieving certain water quality standards, such as state and federal water quality standards and criteria and maximum contaminant levels for drinking water, are outside of the scope of the remedial action selected in this ROD and are not applicable or relevant and appropriate at this time. Similarly, special status species protection requirements under the MBTA [Migratory Birds Treaty Act of 1918] and ESA [Endangered Species Act] are only applicable or relevant and appropriate as they apply to the remedial actions included within the scope of the Selected Remedy. (EPA 2002, p. 13-2)

EPA can waive an ARAR for any of three primary reasons (EPA 1996, p. 6). The first is if the agency determines that achieving that ARAR is technically impractical. The second is if the agency determines that the proposed action will "provide a level of performance equivalent to the ARAR, but through an alternative design or method of operation." The third applies only to cleanups financed by EPA's dedicated cleanup fund and "may be invoked when compliance with an ARAR would not provide a balance between the need to provide protection at a site and the need to address other sites."[17] However, the agency has not yet undertaken an effort to waive any ARARs with respect to OU-3 and apparently does not intend to do so until all the interim remedies have been completed (EPA 2002, p. 12-2).

Long-Term Effectiveness and Permanence

The first of the balancing criteria (see Table 8-5) is the preference for permanent solutions. Although EPA states that it "has determined that the Selected Remedy represents the maximum extent to which permanent solutions and treatment technologies can be utilized in a practicable manner at the site" (EPA 2002, p. 13-19), few of the interim remedies selected by EPA strictly satisfy this criterion. Many have the potential to be undone by floods, which are common in the valley and most selected remedies will require continued monitoring and maintenance to retain their effectiveness. These issues were discussed earlier in this chapter (see "Feasibility and Potential Effectiveness of Remediation Plans").

Reduction of Toxicity, Mobility, and Volume

The interim remedies similarly do not rate well with respect to the second balancing criterion, reduction of toxicity, mobility, or volume through treatment (Table 8-5). EPA seems to recognize this weakness when it states "although the Selected Remedy is not intended to fully address the statutory mandate for permanence and treatment to the maximum extent practicable, the Selected Remedy does utilize treatment, and thus supports that statutory mandate. A comprehensive evaluation for preference for treatment will be conducted in subsequent decision documents" (EPA 2002, p. 13-20). The agency proposes three remedies (hydroxide precipitation with media filtration, permeable reactive barriers, passive treatment pond) or studies that would involve treatment (EPA 2002, Table 9.2-2). However, most of the proposed remedies do not involve treatment, although EPA is considering a proposal to use soil amendments to reduce the bioavailability

[17]These are the primary reasons for waiving ARARs, although the CERCLA legislation and the NCP list three others as well.

of lead contained in some of the sediments in the lower basin (EPA 2002, p. 12-111).

The remedies do include some provisions that will reduce the mobility of the contaminants. These include excavating contaminated sediments from the river channel and floodplain areas, placing the excavated materials in repositories with erosion-resistant caps, and stabilizing sources of contaminated sediments in situ (for instance, by the use of soil amendments). Some proposals such as installing grout curtains to contain and treat groundwater (for example, the efforts on Ninemile Creek at the Success Mine and Mill Site in Ninemile Creek) (Calabretta et al. 2004) would also serve to reduce the mobility of the contaminants, but the practicability and effectiveness of such approaches is highly uncertain. Placing erosion-resistant caps on repositories as well as removing contaminants from potential inundation by floodwaters may reduce the effective mobility of these materials.

Virtually nothing has been proposed to reduce the volume of contamination.

Short-Term Effectiveness

The remedies selected for protecting human health are expected to rate relatively high with respect to short-term effectiveness, assuming that yard remediations will limit lead absorption indoors (see Chapter 5 for discussion).

The short-term effectiveness of the remedies focused on protecting fish and wildlife is less certain (see section Selected Remedy: Geographic Areas, Levels of Remediation, and Remediation Plans in this chapter). The effectiveness of the upper basin remedies is uncertain. As mentioned, it has not been demonstrated that removing selected floodplain materials would decrease inputs of dissolved zinc. Implementing some of the lower basin remedies will substantially disrupt the wildlife habitat being "remedied," giving them a negative effectiveness in the very short term. The proposals to establish new wetland habitat on existing farm land will not suffer from these problems, but their short-term effectiveness will depend on whether and how quickly viable wetland communities can be established on these lands and on the success of these efforts in attracting waterfowl away from the more contaminated areas.

Implementability

Again, a distinction has to be made between those remedies focused on protecting human health and those focused on protecting environmental health. The former have already been demonstrated in the box and at other Superfund sites to be relatively easily implemented, although, as voiced at the public comment session at the committee's meeting in Wallace, Idaho, some land owners in the Coeur d'Alene River basin have exhibited a resistance to having their yards remediated.

As discussed earlier in this chapter, the implementability of some of the remedies proposed for environmental protection is less certain, and the agency has been frank in indicating that some of the proposals need to be tested through bench-scale and pilot-scale studies. One example is the effort to control the flow of zinc-rich groundwater by installing grout curtains. The effort to accomplish this in Ninemile Creek has had limited success because of the very low interception rate of groundwater (see Chapter 4).

The proposal to dredge the riverbed near Dudley is similarly uncertain, although in this case the question is not whether the dredging can be done—it has been done at this site in the past and presents no particular engineering problems. The question is how effective such an effort will be in reducing the flow of contaminated materials downstream, how long the effectiveness will last, and whether the dredging and disposal of dredge spoils can be done in such a manner as to avoid creating serious short-term environmental problems.

Another question about implementability is whether the agency will be able to find adequate repositories for all the contaminated soils it proposes to remove and sources for all the "clean" fill it proposes to use. The process of excavating contaminated soils and disposing of them in a secure landfill has been demonstrated at many Superfund sites. However, the Coeur d'Alene River basin presents special challenges because of the volume of materials proposed for excavation[18] and limited areas with geographic characteristics appropriate for siting a repository. The FS was undertaken with the assumption that such sites could be found, but none has been identified except the repository being used for the relatively limited removals involved in the yard cleanups. Similarly, the geology of the basin provides limited sources of clean fill without seriously disrupting human and natural environments.

Cost

The law establishing Superfund (CERCLA) requires that the selected remedy be cost-effective (40 CFR 300.430(f)(1)(ii)(D)). In its strictest sense, the term cost-effective means that, if alternative remedies will provide the

[18]For example, the proposal to dredge the riverbed near Dudley is expected to produce 1.3 million cubic yards of excavated material (2.6 million cubic yards if the project is "demonstrated to be compliant with ARARs and cost-effective") (EPA 2002, p. 14-1). The removal of the Coeur d'Alene River banks is expected to produce approximately 400,000 cubic yards. In comparison, the approximately 256 acre CIA contains 24.2 million cubic yards of material (URS Greiner, Inc. and CH2M Hill 2001h, Appendix J, Table A-8).

same protection to human health and the environment, EPA must select the least expensive of these alternatives. However, the alternatives identified in the FS provide different degrees of protection. Thus, the cost-effectiveness criterion, as strictly defined, is not relevant.

EPA, however, uses a somewhat looser definition of cost-effectiveness, stating that "a remedial alternative is cost effective if its 'costs are proportional to its overall effectiveness'" (40 CFR 300.430(f)(1)(ii)(D)). The agency explains that the cost criterion enters into the remedy selection process in two ways:

1. A remedial alternative is cost effective if its 'costs are proportional to its overall effectiveness' (40 CFR 300.430(f)(1)(ii)(D)). Overall effectiveness of a remedial alternative is determined by evaluating the following three of the five balancing criteria: long-term effectiveness and permanence; reduction in toxicity, mobility and volume (TMV) through treatment; and short-term effectiveness. Overall effectiveness is then compared to cost to determine whether the remedy is cost-effective (*id.*) (EPA 1996, p. 5).

2. Cost is evaluated along with the other balancing criteria in determining which option represents the practicable extent to which permanent solutions and treatment or resource recovery technologies can be used at the site. This balancing emphasizes two of the five criteria (long-term effectiveness and permanence, and reduction of TMV through treatment) (40 CFR 300.430(f)(1)(ii)(E)). However, in practice, decisions typically will turn on the criteria that distinguish the different cleanup options most. The expectations anticipate some of the likely tradeoffs in several common situations, although site-specific factors will always play a role (EPA 1996, p. 5).

In essence, the agency looks at the tradeoff between the amount of protection provided by the alternative remedies and the costs of these remedies, and then makes a judgment about which of the alternatives appears to provide adequate protection at a reasonable cost.

In the Coeur d'Alene River basin, however, some of these judgments are very difficult, for—at least in the case of environmental protection—none of the alternatives considered is expected to provide the amount of protection required by law. The agency is not particularly clear about how it made these judgments but asserts that "the Selected Remedy achieves a significant reduction in residual risk relative to its cost. It would be cost effective as its costs are proportional to its overall effectiveness" (EPA 2002, p. 10-9). High costs were a consideration in EPA's decision not to select the large-scale cleanup that would provide the amount of protection required by law (EPA 2002, p. 10-3). Instead, EPA crafted the less-ambitious selected remedy to achieve a significant reduction in residual risk.

In addition to these issues, questions can be raised about the cost estimates themselves. Although the cost estimates for yard remediation appear to be accurate,[19] cost estimates for excavating and disposing of large amounts of material in the lower basin, for instance, are very uncertain because EPA has not identified any repositories for these materials, and, therefore, transport distances, methods, and operating costs are not known. The uncertainty about the costs associated with some of the more experimental remedies is even greater.[20]

Another question is whether all the costs of the proposed remedies have been considered. For example, EPA informed the committee that its dredging cost estimates included the cost of settling ponds (located either on a barge or on the land), but no additional treatment for the discharges from such ponds (EPA 2004b, [July 27, 2004]). As discussed earlier, this discharge may well require expensive treatment to remove dissolved metals before being discharged back to the river. In addition, it is highly likely that some of the areas that the agency proposes to clean up will be recontaminated by flood deposited sediments, and it is not clear that the agency has adequately taken account of the cost of redoing the remedies in these areas. The cost estimates should reflect the likelihood of a cleanup action being vulnerable to recontamination by flooding.

As a result, EPA's statement that this "order-of-magnitude engineering cost estimate" is expected "to be within +50 to –30% of the actual project cost" (EPA 2002, p. 12-37) may, for a number of reasons, be overly optimistic. However, it is not clear that improved cost estimates would affect the relative attractiveness of the different alternatives identified in the FS, although substantially higher costs might cause EPA to reduce its expectations of what it can afford to do in the valley.

Perhaps more problematic are the externalities or indirect costs associated with many of the proposed remedial actions. For instance, the proposed remedies involve excavating and transporting millions of cubic yards of materials. One commenter estimated that 1,170,000 truck trips would be required to implement Alternative 3 identified in the FS and that, assuming an average distance of 20 miles per trip, the total distance driven by these trucks would exceed 23 million miles (ASARCO 2001; URS Greiner and CH2M Hill 2001a, Appendix I; Temkin 2004).

Although the remedy selected in the ROD would involve less excavation and material movement than Alternative 3 (and therefore fewer truck

[19]Costs for the actual cleanup work conducted under contract in the box are very close to the original estimate and could actually end up lower than estimated (GAO 2001).

[20]The committee also found that there were a number of errors and inconsistencies in the cost estimates for at least one remedial action (removal of riverbed sediments in the lower basin around Dudley) it examined (EPA 2004b [September 10, 2004]).

miles traveled), the impact of such traffic could impose significant costs, which are not included in the cost estimates, on the valley communities.[21] Examples of such costs include wear and tear on roads and bridges, increased maintenance costs and inconvenience for other vehicles using these roads, vehicle accidents,[22] air pollution, and noise.

Other components of the remedies also could create such external costs. Such externalities, of course, are likely to be associated with any Superfund cleanup or other large construction project. What makes them particularly significant in the case of the Coeur d'Alene project is their magnitude and duration, as well as the topography of the valley.

Another external cost, of a different nature, that concerned several people making presentations to the committee, was the possible impact that designating the valley as a Superfund site would have on its economic prospects. As indicated in Chapters 2 and 3, the economy of the valley has suffered since most of its mines and the Bunker Hill smelter closed. Some residents and potential developers hope that the natural beauty and historical significance of the valley will make it attractive for recreational and second-home developments and fear that the Superfund designation may severely limit this potential.

It is impossible to assess the significance of this potential effect without substantial uncertainty, and there is little that the agency can do to avoid it even if it is significant. It is perhaps unfortunate in this regard that some statements describing this site refer to the entire 1,500-square-mile project area, whereas the contaminated area designated as OU-3 is very much more limited.

There is also some anecdotal evidence that the impact may not be as serious as some valley residents fear. Indeed, recreational developments are being built in Kellogg inside the box, which was initially the most-contaminated area in the basin (Kramer 2004). Perhaps the developer rationalized that the cleanup conducted under OU-1 and OU-2 has addressed

[21]EPA also indicates in the ROD that it thinks that dredged material may be transferred by pipeline.

[22]The average accident rate for heavy trucks is approximately 50 per 100 million miles of travel. The comments referenced above estimated that, using national average rates, the amount of travel required to implement Alternative 3 would result in more than fifteen injuries and, more likely than not, at least one fatality. Most of these would occur to other drivers and pedestrians, not the truck operators. Although the selected remedy would involve less transportation than Alternative 3, the accident rate (in terms of the number of accidents per million miles driven) could well be higher given the narrow, twisting roads that are typical in the valley. This issue is addressed briefly in the FS in the evaluation of the short-term effectiveness of the ecologic alternatives (URS Greiner, Inc. and CH2M Hill. 2001a, Part 3, p. 6-49). However, the agency appears to consider it to be something that can be controlled with adequate safety measures.

the health risks and has limited the liability he might face compared with building in a part of the valley where cleanup has not occurred.

In such ways, the cleanup might generate some external benefits as well as external costs. Other obvious examples are the long-term employment opportunities for valley residents that such a massive project will create and the economic stimulus that valley merchants will likely experience as a result of all this activity. The valley may even end up with better roads as a result of the improvements that will likely be needed to handle the projected truck traffic.

Such costs and benefits, of course, are very difficult to quantify in monetary terms. However, this does not make them any less significant. In projects as large as this, they are sufficiently significant that the committee concludes they should be explicitly considered when comparing alternative approaches and remedial actions even if they are not included in the quantitative cost estimates.

State Acceptance

As indicated earlier in this chapter, EPA has apparently made substantial efforts to coordinate its plans and proposals with other governmental organizations. As a result, it has received the required concurrence of the states involved.

Community Acceptance

From the extensive comments made to the committee during its public sessions, the agency clearly has been less successful in obtaining community acceptance. Although the positions taken were not unanimous, many residents of the upper basin generally opposed the project, wanted the site delisted, and hoped never to see an EPA employee or EPA contractor again, whereas residents living downstream tended to argue that the agency was not doing enough and that the project would leave many potential human health and environmental problems. Indeed, even those committee members who have had substantial experience with Superfund projects found an exceptionally high level of contentiousness in the Coeur d'Alene River basin in spite of the efforts the agency has made to communicate with residents. Some of the contentiousness could be due to the high degree of uncertainty in EPA's ability to develop quantitative estimates of time, costs, and reduction in risk. The committee finds this situation very unfortunate but was not asked to and did not attempt to recommend how it can be substantially improved.

CONCLUSIONS AND RECOMMENDATIONS

This section provides the committee's conclusions and recommendations regarding EPA's scientific and technical practices in establishing Superfund site remedial objectives and approaches in the Coeur d'Alene River basin.

Conclusion 1

EPA has followed the procedures and requirements as understood by the committee set forth in the legislation establishing the Superfund program and in the NCP for determining the nature and extent of contamination at National Priorities List sites and for selecting remedies to reduce the risks to human health and the environment resulting from this contamination.

The agency has gone to great lengths to provide the public with information about its activities and to provide opportunities for the public to comment on its plans, findings, and decisions.

Conclusion 2

EPA has adequately characterized the feasibility of alternative actions it could take to protect human health in the basin, and the selected remedies should provide adequate protection to the most significant risks. The effectiveness of the remedial actions for human health protection, where they have occurred, needs to be further evaluated.

The agency has implemented similar measures in OU-1 and OU-2 and at other sites. However, EPA has not, as it points out, addressed human health risks that might be associated with subsistence living, unlimited recreational use of contaminated areas, or future use of groundwater. It also has not proposed a remedy to address contamination problems in Lake Coeur d'Alene, although no significant human health risks resulting from this contamination had been identified at the time the ROD was released.

Conclusion 3

EPA has not adequately characterized the feasibility and effectiveness of actions to protect fish and wildlife resources in the basin.

In several cases, substantially more investigation and experimentation are needed to determine whether the selected remedies are effective and feasible. Even if they prove to be so, it is highly unlikely that they will sufficiently reduce the risks resulting from the basin's contamination to meet Superfund requirements to protect the environment and satisfy ARARs. The agency recognizes this weakness and therefore has designated its proposals "interim remedies." The agency has begun some of the investigation and experimentation needed, and the committee supports these efforts.

Recommendation

EPA should support the substantial additional characterization that will be required to determine whether the interim remedies proposed are feasible and to what extent they will effectively reduce environmental risks. EPA and the state of Idaho also should investigate the feasibility of developing biologically based water-quality criteria that could provide alternatives to concentration-based ARARs. In addition, a strategy is needed for evaluating the performance and efficiency of the selected remedies.

Conclusion 4

The lack of repositories for contaminated soils and sediments is particularly problematic and is a primary concern to the committee regarding the feasibility and implementability of the proposed remedial actions in the basin.

The selected remedy proposes removing large quantities of materials that, at present, have no location for disposal. The siting, design, and construction of repositories will take a long time, if these actions are even possible, especially considering the geography of the basin and the contentious political climate.

Conclusion 5

None of the remedies proposed for cleanup and risk management in the Coeur d'Alene River basin is permanent.

Remediated sites are likely to suffer from recontamination from sediment carried by the frequent floods in the basin. These floods can also erode protective caps covering contaminated areas, thereby eliminating the protection that the caps provide. The need for lifetime maintenance of remedies selected for management of risks to human health has already been demonstrated in the box where, in 1997, floods recontaminated remediated areas. The state of Idaho and the Panhandle Health District have established a process for monitoring the integrity of the human health protection measures and apparently were successful in re-establishing the human health protection measures after the flooding. However, the process will have to remain in place essentially in perpetuity to respond to problems created by future floods and other events that compromise the integrity of remedies.

Recommendation

A plan should be developed to create appropriate institutions and funding to maintain selected remedies through time. Such maintenance will be required for hundreds of years.

Conclusion 6

The Coeur d'Alene River basin is a system where floods play a fundamental role in the resuspension and distribution of contaminants. The timing, intensity, and duration of these floods markedly affect the potential for sediment transport.

The negative impact of resuspended sediments on human and environmental health coupled with the expense associated with potential remediation and recontamination make it necessary to consider management of the entire watershed to reduce the intensity, frequency, and duration of floods, as it is expected that watershed management practices (particularly canopy removal in forests and road building) are linked to water yield in the basin.

Recommendation

To the extent that water yield and flooding can be managed through land-use practices, it is important to include these in the schemes designed to protect human and environmental health.

Conclusion 7

Ultimately the contamination problems in the Coeur d'Alene River basin, Lake Coeur d'Alene, and the Spokane River will be solved only when the contaminated sediments in the river basin have been removed or stabilized.

Efforts to remove contaminated sediments in the lower basin are likely to be of limited value until the problems of sediment transport from the upper and middle basins have been adequately addressed. Even when sediments have been physically stabilized, as they have in the embankment of Interstate 90 and the former Union Pacific Railroad bed, groundwater seepage through these materials still may contain high levels of dissolved metals and may need to be collected and treated.

Recommendation

The committee recognizes that it is not feasible to remove all the sediments but strongly supports the proposed remedies that call for the removal or stabilization of potentially mobile sediments in the upper and middle basin and urges EPA to explore additional opportunities for such actions.

Conclusion 8

Recontamination is a major issue relating to the protection of waterfowl and their habitat, and the committee has significant concerns about the likely effectiveness and long-term viability of many of the remedies proposed to reduce waterfowl mortality. The committee supports measures such as restoring wetlands on agricultural lands in the lower basin and upgrading the quality of the habitat in existing wetland areas that have the least likelihood of being recontaminated.

Many of the wetland and lacustrine areas in the lower basin are likely to be recontaminated by the first major flood that occurs after their remediation, and the likely effectiveness of some of the measures proposed to reduce such recontamination is very uncertain. Recontamination is less problematic in areas such as the lower basin agricultural lands that formerly were wetlands and some wetlands and lacustrine areas historically protected from extensive flooding. Increasing the available area of high-quality waterfowl habitat may reduce waterfowl mortality; however, these reductions can occur only if the availability of the restored or enhanced habitat substantially reduces the use of more heavily contaminated areas by waterfowl.

Recommendation

The committee recommends that EPA proceed in implementing those remedies that are most likely to be successful and durable, particularly regarding recontamination of remediated areas. It will be essential to monitor the success of these efforts both in attracting waterfowl to the wetlands that have been remediated and in reducing waterfowl mortality.

Conclusion 9

The riverbed downstream of Cataldo represents the largest repository of lead-contaminated sediments susceptible to transport during severe flood events. The mobilization of these deposits results in further contamination of adjacent riverbanks and wetlands as well as downstream transport into Lake Coeur d'Alene and eastern Washington.

The riverbeds hold most of the lead in the lower basin. These sediments contain high concentrations of lead and present a large surface area susceptible to the erosive and scouring effects of floods. Monitoring has demonstrated that, during flood events, lead concentrations increase in the river downstream of Cataldo and that riverbed sediments in the lower basin are redeposited on the banks and adjacent wetlands. It is estimated that the riverbed of the lower basin is the source of 70-80% of the particulate lead entering Lake Coeur d'Alene. Without corrective measures, it is expected that these sediments will continue to move downstream.

Recommendation

Priority should be given to remedial measures that address the largest potentially mobile sources of lead-contaminated sediments. High priority should be given to understanding the process of flood scouring of the channel below Cataldo. Remedial designs to stabilize or remove this source will need to consider the impacts to fluvial behavior from dredging or riverbed-armoring operations, potential downstream migration of suspended sediments from potential dredging operations, and elevated zinc in settling pond effluents in potential dredging operations. If dredging is selected, riverbed recontamination will be another important consideration, especially until upstream areas are removed or stabilized, as continuing deposition of contaminated sediments (albeit at a much lower concentration) is ongoing (see Conclusion 7).

Conclusion 10

Riverbanks possess a relatively small proportion of the lead that is available for transport in the system; they have a high likelihood for recontamination; and there is insufficient information available to assess the risks that existing riverbank materials present to environmental receptors.

Riverbank remediation is intended to reduce particulate lead loading in the river and soil toxicity to songbirds, small mammals, and riparian plants. The rationale for excavating the riverbanks is questionable because only a small percent of the lead in the depositional environment of the lower basin resides in the riverbanks, and, compared with the riverbed, a small surface area is exposed to surface-water flows. Further, limited evidence exists linking the presence of lead-contaminated riverbanks to exposure and impacts to songbirds and small mammals. In addition, remediated riverbanks will be highly susceptible to recontamination by the deposition of contaminated sediments derived from the riverbed or upstream sources during flood events.

Recommendation

EPA should not give priority to the less-certain proposed remedies until it can better demonstrate the likely effectiveness of these efforts.

Conclusion 11

The likely effectiveness of the interim remedies EPA has proposed to reduce risks to aquatic life is uncertain.

The threat to aquatic life results primarily from the influx of groundwater containing high levels of dissolved metals, particularly zinc during the late summer low-flow season. A substantial portion (modeled at 41%) of the dissolved zinc in the lower basin results from groundwater seepage through the box area, but EPA has excluded this area from consideration in OU-3. It appears unlikely that the agency will be able to achieve water-quality standards downstream from the box without reducing the amount of zinc coming from this source. Based on removals that have been conducted up to this point, the committee has not seen evidence suggesting that removals in the basin have decreased surface-water concentrations of zinc, although that would be anticipated if the materials were contributing zinc to the surface water. The agency has proposed some innovative approaches to reduce zinc loadings from the upper basin streams, such as Canyon Creek and Ninemile Creek. Although the committee endorses continued experimentation with such techniques, it notes that they have had limited success, and these approaches are not likely to be effective where large volumes of water require treatment. Because passive systems are probably inappropriate for treatment of large volumes where very large areas are not available to provide for long detention times (for example, in Canyon Creek), the agency will have to explore alternative approaches if it is to reduce zinc loadings from these larger volume sources. The committee also questions the wisdom of using phosphate as a sequestering agent, because this may result in eutrophication problems in Lake Coeur d'Alene.

Recommendation

Characterization needs to be conducted to locate the specific sources contributing zinc to groundwater (which subsequently discharges to surface water) and set priorities for their remediation. Groundwater should be addressed directly if loading to the groundwater is determined to stem from subsurface materials too deep or impractical to be removed. Further, EPA should continue to support research on and demonstration of low-cost innovative groundwater-treatment systems. In particular, the agency should place a high priority on identifying possible methods of reducing metal loading in groundwater from the box and highly affected tributaries.

Conclusion 12

EPA proposes using adaptive management in implementing interim ecologic-protection remedies; however, EPA's approach to remediation does not include all the elements needed for an effective adaptive management approach.

Adaptive management is not synonymous with trial and error. Rather, adaptive management is a multistep, interactive process for defining and implementing management policies for environmental resources under conditions of high uncertainty concerning the outcome of management actions. Development of explicit remediation objectives and performance benchmarks, together with a monitoring program to measure progress toward the objectives, is critical to achieving maximum benefits from the adaptive approach. Many of the performance benchmarks and monitoring indicators described in the ROD and the BEMP, especially those that relate to terrestrial biota and habitats, are insufficiently specific to support a truly adaptive approach.

Recommendation

EPA should improve its use of the adaptive management approach by establishing unambiguous links between management objectives, management options, performance benchmarks, and quantitative monitoring indicators for all the habitats and biological communities addressed in the ROD.

Conclusion 13

The reliability of the model for predicting postremediation concentrations of dissolved zinc (probabilistic model) is highly questionable because it appears to be based on an untested hypothesis that is not supported by theoretical or experimental evidence. Furthermore, the time variation contained within the model is incorrect.

The probabilistic model is used to estimate relative loading potentials based on estimated total volume of contaminated material, estimated concentration of available zinc, and estimated effectiveness of various remediation methodologies in reducing metal loading. There are no leach test data from sediments or tailings that would provide rates and quantities of metal release over time, allowing extrapolation of relative loading potential. There are no measurements of groundwater-quality upgradient or downgradient of the various source types used in developing the model, and there is no evidence of the effectiveness of proposed remediation methodologies in reducing relative loading potential. The probabilistic model has not been calibrated in a rigorous sense other than the calibration that is inherent in the model's use of statistical results from historic monitoring data as the preremediation condition.

Recommendation

EPA should support the development of a predictive tool based on sound scientific principles and supported by site-specific information on leaching potential, groundwater movement, and other such factors to allow them to accurately assess the likely effectiveness of remedial actions on dissolved metal loadings from various sources along the river.

Conclusion 14

The transport of contaminated sediment through the basin and the rest of the project area is a key factor in determining the likely effectiveness and durability of proposed remedies.

EPA has not developed a sediment-transport model for the basin that would allow these factors to be evaluated. USGS has collected and is collecting some very useful information about flood flows and sediment transport in the basin that would support the development of such a model. Such a tool would be very useful in assessing the likely long-term effectiveness of proposed remedies focusing on reducing the risks resulting from lead-contaminated sediments.

Recommendation

EPA should develop a quantitative model using a systems approach for sediment dynamics, deposition, and geochemistry for the basin as a whole and should use the results of this model in designing and establishing priorities for proposed remedies.

Conclusion 15

Implementing remedies at a Superfund project as large and complicated as the Coeur d'Alene River basin can generate significant indirect costs and environmental impacts that the agency has not adequately considered in evaluating the alternative remedies.

The indirect costs include, among other items, likely accidents, wear and tear on basin roads, traffic congestion, and other costs associated with the large volume of traffic that could be required to implement some of the remedies. Potential environmental impacts include, for example, silt mobilized by dredging and excavation in aquatic environments, reduction in the quality of habitat for aquatic organisms, and air emissions from the truck traffic and construction machinery. The committee also cautions that flood-control action, such as enhanced levees, can affect river flow and cause undesirable consequences. The committee encourages EPA during the remedial design phase to carefully evaluate the consequences of flood-control actions.

Recommendation

In establishing priorities for designing and implementing remedial actions, EPA should consider the potential indirect costs and environmental impacts of the remedies being considered.

Conclusion 16

The large uncertainties in the present understanding of the mechanisms of release of metals and nutrients from Lake Coeur d'Alene sediments and their transport and fate after release will limit development of an effective lake management plan.

Lake Coeur d'Alene is currently the subject of a 3-year, integrated metal-nutrient flux study. Such studies to generate a greater understanding of metals dynamics are unquestionably needed before a viable lake management plan can be developed and implemented to limit the effects of metals loading to the lake on environmental and human health risks—including those associated with the Spokane River.

Recommendation

Comprehensive studies of Lake Coeur d'Alene should be given a high priority to support development of an effective lake management plan.

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9

Mining Megasites: Lessons Learned

The final charge to the committee was to assess "lessons from the Coeur d'Alene case that may be applicable to other similar Superfund sites." The committee believes that there are some lessons to be learned. Certainly, it has observed a number of problems in the expansion of the Superfund process to operable unit 3 (OU-3) in the Coeur d'Alene River basin. Some of these problems resulted from the way the expansion was undertaken, and others appear to be inherent in the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) process. However, the committee does not question the overall goals of protection and restoration of human and ecologic health that are embodied in the CERCLA legislation. In touring the Coeur d'Alene River basin and reviewing studies extensively detailing the ubiquity of contamination, it was obvious that there were human and ecologic health risks in the basin that require remediation. The potential adverse economic implications, such as reduced real estate values, created opposition to a Superfund site designation both locally and within the Idaho State Government, although such tension is not unique to the Coeur d'Alene site. This chapter addresses issues and opportunities associated with large complex mega-mining sites under Superfund. The discussion is informed both by perspectives gained through experience with the Coeur d'Alene River basin site, as well as committee members' broader insights and observations. It is not a comprehensive review, but a digest of the issues and an outline of conclusions and approaches for facilitating the effective management of these large and complex sites.

APPLICABILITY OF THE SUPERFUND PROCESSES TO MEGASITES

The Coeur d'Alene River basin is one of the largest mining-related Superfund sites in the United States. It is not, however, unique. For example, just east of the Idaho-Montana border, the Clark Fork Operable Unit of the Milltown Reservoir-Clark Fork River Superfund site includes 120 river miles of the Clark Fork River contaminated with metals stemming from mining activities in upstream reaches (EPA 2004a). A 2004 report by the U.S. Environmental Protection Agency (EPA) Office of Inspector General (EPA 2004b) identified 63 hard rock mining sites (which do not include coal mining) listed on the National Priorities List (NPL), another 82 that were on Comprehensive Environmental Response, Compensation and Liability Information System (CERCLIS) but had not yet been listed on the NPL, and 11 potential CERCLIS/NPL sites.[1] These represent only a small portion of all the abandoned hard rock mining sites in the United States. A Western Governors' Association survey estimated that there were at least 400,000 abandoned or inactive hard rock mining sites in the West (WGA 1998a,b) and the Mineral Policy Center estimated in 1995 that, nationwide, there were 557,000 abandoned mines (Custer 2003). Although many of these are small sites presenting little or no human health risks, the EPA Inspector General found that the total cost of cleaning up the sites on the EPA inventory could be as much as $24 billion and that at least 19 of the sites already listed on the NPL are likely to have cleanup costs of $50 million or more (EPA 2004b).

By one formulation, these would be considered "megasites."[2] A Resources for the Future study has assessed the impact of such megasites on the budgetary state of Superfund (Probst et al. 2001), and a recent EPA advisory committee report (NACEPT 2004) discussed the issue of megasites and possible management options but provided no recommendations.

Mining megasites such as Coeur d'Alene typically involve multiple contaminants and contaminant sources and large volumes of waste material that have accumulated over many years of mining activity and are dispersed over wide areas. Large quantities of mining-related contaminants may have been deposited many miles from the original sources. Soils, sediments, surface water, and groundwater may be contaminated, and the hydrological relationships between these media may be complex and difficult to characterize.

The Superfund process has some serious difficulties in addressing this type of site. The following discussion focuses specifically on large mining

[1]CERCLIS contains a list of all hazardous waste sites that are on the NPL or are being considered for the NPL. Many sites included in CERCLIS are unlikely ever to be listed on the NPL.

[2]An EPA advisory committee characterized a Superfund site as a "megasite" if any combination of remedial action costs excluding long-term remedial actions exceeds $50 million (NACEPT 2004).

sites like the Coeur d'Alene River basin. Although other Superfund sites may show similar characteristics and, therefore, experience some of these problems as well, the extrapolation to all megasites as conventionally defined in monetary terms is limited, as many of the issues stemming from large mining areas relate to the large areal extent and complex nature of the site, and not simply projected costs.

PROBLEMS OBSERVED IN APPLYING SUPERFUND TO MEGA-MINING SITES

As it reviewed the work that was done in attempting to identify remedies for OU-3 and problems at other mining areas being cleaned up under CERCLA, the committee observed a number of problems in applying CERCLA and the National Oil and Hazardous Substances Pollution Contingency Plan (NCP) to mining megasites. As indicated later in this chapter, EPA attempted to overcome some of these problems within Superfund regulations, but some appear to be inherent in the program. In recognizing these problems, the committee is not suggesting that CERCLA be amended to allow it to deal with them. The law was intended to address specific problems associated with environmental contamination that poses risks to human health and the environment and should remain focused on eliminating that contamination. Other federal, state, and local programs that can address the limitations observed in Superfund often already exist.

No Final Remedy

The focus of the NCP governing implementation of CERCLA is on identifying and implementing a final remedy (40CFR 300.430 (f)(4)), but the concept of a final remedy may not be appropriate for some megasites because this term implies that there is a final solution that can be clearly defined in advance of remediation (Moore and Luoma 1990). In the case of large mining sites, where remediation may involve many decades of sequential remedial actions, and institutional controls may be required in perpetuity, there may never be a final remedy. Indeed, EPA believes that more than half of the mining sites currently listed on the NPL will require operation and maintenance in perpetuity (EPA 2004b).

The most obvious problem with "cleaning up" megasites such as the Coeur d'Alene River basin is the massive quantities of contaminated waste materials (including waste rock, tailings, and tailings-contaminated sediments) that cover a large geographic area in a variety of upland, wetland, and aquatic environments. This complexity and volume of contaminated material practically eliminate the potential to completely remove, cap, and treat the contaminated materials, and make practical and effective remedies very difficult to design and implement. Indeed, the volume of mining wastes present in the Coeur d'Alene River basin is so large that it is doubtful that complete removal can ever be attained. As indicated in Chapter 3, there are more than 100 million cubic yards of contaminated materials in the basin, much of which underlies buildings, roads, and railroads. Even if there were sufficient money and consensus to remove all these materials, it would be very difficult to find a place to put them where they would not create a threat of recontamination.

Even the limited removals proposed for OU-3 will be costly, difficult, and disruptive. In some cases (particularly the removals proposed to protect fish and wildlife), they may not even be feasible. The extent to which proposed remedial measures would reduce dissolved metals concentrations in the river is unclear. And the proposed removals can generate significant external costs in the form of large numbers of truck trips and associated road maintenance, noise, traffic, and accidents and will affect local populations and infrastructure over many decades. Other solutions (for example, chemical fixation and capping) may be feasible at some locations but could not be applied throughout the basin. In short, there are no obvious engineering solutions to a contaminated region as large and geographically complex as the Coeur d'Alene River basin. Remediation must be viewed as a long-term process involving numerous individual remediation projects, only some of which can be specified at the beginning.[3] Given the inevitably high uncertainty about the design and ultimate success of the proposed remedies, any estimates of the duration and cost of the remediation are necessarily crude approximations.

A Long-Term Process

Because of the difficulty of implementing a final remedy, the cleanup of a site like Coeur d'Alene will require a long-term commitment to implement and maintain the cleanup actions that are undertaken. Although the committee has concluded that the remedies proposed for the protection of human health will likely be effective in achieving their goals, they will require continued efforts to control land use, protect the integrity of the remedies, and deal with flood-related recontamination, which is inevitable in a watershed like the Coeur d'Alene River basin.

The need for long-term commitment is even greater in the case of the remedies to protect the environment. Here, EPA admits that the expenditure of hundreds of millions of dollars over three decades will be only a first

[3]Recognizing this problem, the EPA National Remedy Review Board recommended that the environmental protection remedies proposed for OU-3 be designated "interim" remedies (NRRB 2001).

step in achieving its environmental protection goal and, if nothing more is done, it will still take hundreds of years to achieve the water-quality standards established to protect aquatic resources.

For all the above reasons, cleanup of mining megasites necessarily must be viewed as a long-term process with an uncertain outcome. Management of these sites over the many decades needed to complete the remediation process requires the development of institutions with the capability to oversee engineering operations, minimize the impact of remediation on local communities, and maintain the institutional controls needed to maintain human exposures at acceptable levels. It also requires the implementation of a long-term monitoring strategy that will (1) provide more specific information on the causes of the human health and environmental risks and the sources of contamination causing these risks, (2) evaluate the effectiveness of remediation efforts, and (3) monitor the overall changes in human and environmental health being experienced.

Limited Scope

The Superfund process was established to address a particular, limited problem—risks to human health and the environment posed by contaminated wastes. But, particularly in megasites like the Coeur d'Alene River basin, the contamination is likely to be only one of the problems creating these risks. Lack of access to adequate health care, unemployment, poverty, and a number of other factors can have as much of an impact on community health as the contamination from mining wastes.

On the environmental side, even if the concentrations of all metals in water and soil could be reduced to nontoxic levels, the degree of habitat modification that has occurred within the basin is probably sufficient to prevent fish and wildlife resources from returning to the conditions that existed before mining. The success of these efforts could be substantially influenced by factors such as how the forests in the basin are managed.

It probably would be much more effective and efficient to address the human health and environmental problems in these areas with a program that could address all these different factors in an integrated fashion. However, most of these contributor problems lie outside the purview of Superfund, and, therefore, its funds cannot be used to address them, even if by so doing the agency could reduce the total cleanup costs.

These other factors are also likely to limit the effectiveness of the cleanup efforts in achieving the goals of protecting human health and the environment. For instance, aquatic communities are limited by impaired habitats as well as chemical exposures. Reducing chemical concentrations to safe levels will not lead to ecologic recovery if physical aspects of the habitat remain impaired. Healthy aquatic ecosystems can exist in the presence of modestly elevated levels of contaminants, but, even in the absence of chemical stressors, healthy aquatic systems will not exist in degraded habitats.

The Liability Problem

The Superfund legislation incorporates what many consider to be the most stringent liability provisions in federal law—retroactive, perpetual, joint and several,[4] and absolute. As appropriate as these standards may be for holding "responsible parties" liable for paying for the cleanup of these sites, they are said to discourage contractors from becoming involved in the cleanup activities and particularly discourage the use of innovative and other nontraditional cleanup approaches. This may be an issue at hard rock mining sites where the wastes contain valuable minerals.

One possible approach to such sites is to re-mine the wastes with modern technologies that remove these minerals (NRC 1999, p. 72).[5] Such an approach would have several advantages: (1) contaminants would be removed from the basin environment and the potential for recontamination eliminated (Moore and Luoma 1990); (2) the net cleanup costs would be reduced by the value of the recovered minerals; and (3) such an approach would be one of the few options that would satisfy the preference in CERCLA for remedies that reduce the toxicity of the wastes. As indicated in Chapter 2, tailings have been re-mined and reprocessed in the past in parts of the Coeur d'Alene River basin.

The strict liability provisions of CERCLA, however, discourage the re-mining approach. This option likely could be undertaken only by an established mining company with adequate technical expertise and financial resources. But such a company, if it were to become involved, could be putting itself at risk of being designated a PRP (potentially responsible party) responsible for the entire cleanup cost. Any established mining company with the resources necessary to undertake such an effort likely would be reluctant to put itself at such risk, particularly when the financial rewards probably would be limited.

The 1986 SARA (Superfund Amendments and Reauthorization Act) amendments to CERCLA established a special liability category for firms involved in cleaning up Superfund sites (42 USC § 9607(b)). Their liability changed from an absolute liability to one based on a standard of negligence. However, the joint and several provisions still apply so that a company

[4]Joint and several liability means that all responsible parties are jointly responsible for the entire cleanup cost, and each of them individually can be held responsible for paying these costs.

[5]The committee did not assess whether such re-mining might be a viable option in addressing the contamination in the Coeur d'Alene River basin.

involved in cleanup theoretically could be held responsible for cleaning up the entire site.

In some cases, government agencies have indemnified organizations involved in cleanup operations. EPA did so before passage of the 1986 amendments, and other government entities have done so in special circumstances—for instance, when they are the owners of the contaminated site (and, therefore, are liable in any case). The effect of these liability provisions on remediating mining sites is described in recent reports by Trout Unlimited, a conservation group that has partnered with the U.S. government in an effort to remediate abandoned mine sites (Trout Unlimited 2004a,b).[6]

Funding Limitations

The need for long-term management of these sites and the desirability of addressing issues beyond contamination resulting from the disposal of wastes highlights the limitations on funding available under Superfund. Initially, CERCLA established a special dedicated tax on oil and chemical companies to fund cleanup activities where there was no financially viable responsible party. This taxing authority, however, has expired, and Congress now funds the program from general revenues through annual appropriations.[7] Particularly under current budget conditions, the availability of adequate funding in the future is uncertain. The lack of a secured funding stream raises serious concerns about how a remediation program expected to last for decades if not centuries can be successfully implemented.[8] Funding interruptions would not only disrupt the remediation efforts but could even make the situation worse (for instance, if a wetlands restoration project were disrupted after the excavation stage but before the appropriate vegetation could be reestablished).

A second limitation associated with Superfund funding is that use of the funds is restricted to furthering the purpose of the legislation; they cannot be used, for instance, for general community improvement, wildlife management, or economic development projects.[9] These restrictions inhibit adoption of the comprehensive management approach discussed above.

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[6]"Existing laws may actually create a disincentive for private entities such as TU to cleanup abandoned mines, and funding is woefully scarce for restoration efforts." Chris Wood, Trout Unlimited Vice President for Conservation Programs.

[7]Even if the special Superfund tax were still in effect, the companies paying this tax could reasonably object to substantial amounts of these funds being used to clean up hard rock mining sites for which they had no responsibility.

[8]Funding options for long-term stewardship approaches have been discussed in a recent NRC report (NRC 2003) and Resources for the Future has analyzed different approaches for addressing this problem through establishing trust funds (Bauer and Probst 2000).

[9]Funds recovered from a Natural Resources Damage Assessment can be used for wildlife improvement projects.

A third funding issue relates to payment of costs associated with cleanup versus operation and maintenance (O&M) costs. At sites like the Coeur d'Alene River basin, when the government is paying for most of the cleanup work because there is no financially viable responsible party, the federal government pays for 90% of the construction costs, with the state paying the other 10%. However, the state is solely responsible for paying all the O&M costs starting a year after construction is declared to be complete.[10] Thus, even if a long-term management option was determined to be substantially less expensive than a construction option achieving the same result, the state would have a strong financial incentive to favor the construction alternative if the long-term management option was deemed to fall in the category of O&M. Such incentives have the potential to bias the remedy-selection process because the state must concur with the selected remedy.

NCP Threshold Criteria

Although not unique to megasites, some of the criteria for remedy selection under Superfund make the process more difficult, at least as they are usually interpreted. The threshold criteria, according to the NCP, are to "protect public health and the environment" and "satisfy ARARs [applicable or relevant and appropriate requirements]." Any proposed remedy must meet these threshold criteria. In the case of the Coeur d'Alene River basin, EPA's modeling studies indicate that hundreds of years will be required to meet these goals, regardless of how much remediation is performed. Unless one envisions a remediation program lasting for several centuries, one must question whether these types of ARARs are appropriate criteria for remedy selection. Villa (2003) refers to this as "Perhaps the most intractable problem for ecologic protection":

> Now, here's the rub: if CERCLA requires remedies to attain ARARs, and ARARs for the Coeur d'Alene River Basin remedy include water-quality criteria, yet such criteria could not be met for less than 200 years at best, how can CERCLA be satisfied? The answer lies in the inherent flexibility of the Superfund statute and its implementing regulations. The statute itself authorizes ARARs "waivers" in specified circumstances. However, these waivers only apply to satisfaction of ARARs. There is no statutory

[10]For some types of cleanup, particularly those related to groundwater and surface-water cleanup, the operation of treatment systems or other measures for a period of up to 10 years is considered part of the remedial action, and the state's obligation to fund O&M begins after this period has ended (GAO 2003; 42 USC § 9604(c)(6) [2003]; 40 CFR § 300.435(f)(3) [2005]).

> waiver for the other threshold criterion of protecting human health and the environment. In the Coeur d'Alene River Basin, not only are water-quality criteria exceeded, but the aquatic life intended for protection by such criteria are also at risk. Therefore, waiving the ARARs in this case would offer no relief from the independent statutory obligation to protect the environment.

Particularly at sites as extensive and complex as the Coeur d'Alene River basin, it appears more reasonable to define protection of the environment in terms of restoration of normal ecologic functions rather than reduction in chemical concentrations below theoretically protective thresholds. These statements should not be construed to indicate that a decreased level of environmental protection is acceptable. Rather, measured end points and goals should be based on achieving characteristics of healthy aquatic ecosystems (for example, macroinvertebrate diversity, numbers, and composition; habitat indices; and fishery markers) and not on achieving a specified concentration of contaminant.

This approach is, in fact, consistent with recent trends in water-quality management throughout the United States. With active encouragement and technical support from the EPA Office of Water, many states are using "biocriteria" (indices of aquatic community composition) to supplement or replace numerical concentration standards as a means for determining whether water bodies can support their designated uses (Barbour et al. 1999). At the Lower North Potato Creek site in Polk County, Tennessee (discussed further below), Tennessee's biocriteria are being used to define the performance goals for site remediation.

A Bureaucratic Process

To many observers, cleaning up a site under Superfund appears to be a very bureaucratic, cumbersome, and inefficient process. Millions of dollars and many years can be spent undertaking studies, producing massive reports, and attempting to come to agreement on a "remedy" that will adequately protect human health and the environment while complying with the other requirements of CERCLA. This is done according to the extensive procedures established under the NCP. However, this process was established initially to address more limited industrial waste sites, and it is not clear that the process is appropriate for cleanup at a large geographically complex mining megasite like the Coeur d'Alene River basin.

Complexities inherent in an ecosystem as multifaceted as the Coeur d'Alene River basin do not mesh well with the rigidity of the Superfund process. The Superfund process calls for EPA first to gather all the necessary information (the remedial investigation [RI] phase), then evaluate alternatives for addressing all the human health and environmental risks identified in the information-gathering stage (the feasibility study [FS] stage), and then decide on the best remedies for reducing these risks to acceptable levels (the record of decision [ROD]). Conceptually, each stage is completed before the next one begins (although, in practice, the RI and FS are often combined).

At most sites, the OU being assessed addresses only one or two closely related problems, and this process works reasonably well. In the Coeur d'Alene OU-3, however, there are a large number of different problems. Some, like the contamination of yards, are fairly easy to assess. Others, like the reduction of dissolved metals in the main stem of the river are much more difficult. By combining these different problems into one OU and subjecting them to the process established in the NCP, EPA must attempt to answer all the questions for all the problems before it can attempt to remedy any of them.

As a result, the agency must delay action on addressing the more tractable problems until it has all the information it needs to decide what to do about those that are less easily addressed, or, alternatively, it must propose remedies for some of the problems with inadequate information.[11] In OU-3, the first option would have resulted in substantial delays—perhaps decades—in efforts to reduce human health risks while the agency collected information and conducted experiments on possible ways of solving the basin's very complicated environmental problems. The agency adopted the second option, which allows it to begin work on reducing the human health risks but leaves substantial confusion about how it will address many of the environmental problems. It has proposed remedial actions for addressing these environmental risks, but this may have been largely a paper exercise because there is so much uncertainty about the effectiveness of the proposed remedial actions, or even whether they can be implemented. Although these considerations also exist for smaller, less complex Superfund sites, the complexity of these large geographically diverse sites like the Coeur d'Alene River basin dramatically increases the difficulty in developing workable remedies for every problem before beginning action on any of them.

This dilemma was very apparent during the committee's information-gathering and deliberation process. Questions to EPA about specific operations or technologies noted in the selected remedy were often answered with uncertainty, as the actual process was not yet known or formally selected, and decisions were deferred to the remedial design stage. As stated

[11]EPA can conduct emergency removal actions under the NCP without preparing the series of reports required to decide on an appropriate remedy.

by EPA, "While the ROD establishes the general concept, intent, and goals of the remedy, RD [remedial design] and RA [remedial action] are where design and construction details are developed and implemented" (EPA 2004c). Thus, much of the effort that has gone into evaluating and costing alternatives may not be used for the final solution.[12]

The development of decision documents that subsequently went unused was particularly apparent in the review of environmental protection remedies. For example, little use is made of the extensive detailed analyses and development of preliminary remedial goals presented in the ecologic risk assessment in developing the selected remedy. The FS presents voluminous documentation and goes to great lengths to select, document, cost, and compare five alternative strategies. However, none of these remedies was selected. The ROD selects a remedial strategy that may or may not be conducted owing to on-the-ground considerations. This is not a fault of EPA but rather an artifact of the Superfund process that requires development of decision documents in this fashion, in an environment not conducive to encompassing descriptions and predictions.

As an area increases in complexity, the certainty of cost, volume, and remedial efficacy estimates decreases as does the certainty that selected decisions will be conducted. In reality, these large geographically complex sites like the Coeur d'Alene River basin cannot be remediated in a short time frame, and efforts to describe the entirety of the problem and chart a path to completion (as attempted in the Superfund process) become less realistic with increasing complexity of the site. These decision documents—even when based on best understanding and engineering practices and considering the uncertainty involved—open the agency to criticism that the decisions are not being followed and/or are incorrect. Under the current system, this may be unavoidable.

[12]One example is the extensive effort made to describe, cost, and compare remedial activities within Canyon Creek. Five alternatives were considered. Approaches outlined in these alternatives included excavation and removal of floodplain deposits and waste rock, adit water treatment, pipeline construction, active and passive treatment systems, groundwater treatment, bioengineering controls, in-stream deflectors, and repositories. However, none of these alternatives was selected because they all "would be very difficult, costly, and time consuming" and the agency wanted to "focus on identifying cost-effective technologies for improving downstream water-quality" (EPA 2002, p. 12-25). The selected remedy described in the ROD states that "one potentially cost-effective approach that will be evaluated is to intercept the creek water in lower Canyon Creek and remove metals using passive treatment." For this "potential approach," the ROD includes a detailed cost estimate ($15 million), provides an engineering drawing, and estimates a reduction of 322 pounds of zinc per day. The committee later learned from EPA during a tour of the basin that there were no longer plans for the passive treatment system described in the ROD.

OPPORTUNITIES UNDER SUPERFUND

Can these problems be fixed within the existing Superfund framework? Villa (2003) argues that the Superfund program is the only program comprehensive enough to deal with sites as complex as the Coeur d'Alene River basin and that the program is flexible enough to satisfy all contingencies. He points out that EPA attempted to use its other authorities to address the contamination problems outside of the "box" and these authorities were inadequate.

EPA does, in fact, take advantage of much of the flexibility that the Superfund program can provide. For instance, many Department of Energy (DOE) and Department of Defense sites are very large and complex, often experiencing extensive contamination in a variety of ways and from a variety of sources. Cleanup of these sites is performed under Federal Facilities Agreements between the agency responsible for the site, EPA, and responsible state regulatory agencies. Whicker et al. (2004) describe the remediation approach adopted for DOE's weapons complex that involves a combination of institutional controls, land-use planning, and active remediation. Substantial acreages at several of these sites have been set aside as natural areas. Because these areas have been protected from human intrusion for more than 50 years, they provide habitat quality that generally is substantially higher than is present in the surrounding landscapes. DOE, EPA, and state agencies have agreed that in many of these cases the adverse effects associated with remediation would be greater than the harm caused by current chemical and radiological exposures. Cleanup standards for these areas may be relaxed compared with standards for areas slated for industrial or residential development, because human exposures are expected to be limited by institutional controls. These sites, of course, have the advantage over the Coeur d'Alene River basin that the government owns the entire site and, therefore, has full control over how the site will be managed and what access will be provided to the site in the future.

The East Tennessee Copper Basin is a nongovernment site where EPA has demonstrated substantial flexibility under Superfund (EPA 2004d). This former mining and ore-processing district in Polk County, Tennessee (the Copper Basin), is one of the largest contaminated sites in the eastern United States. Soil, sediment, and water throughout the basin have been severely degraded by metals contamination and acid rock drainage. Severe soil erosion has occurred, resulting in deposition of several feet or more of sediment in the two creeks that drain the basin. Remediation of one of these areas, the North Potato Creek Watershed, is being managed by the responsible party (Glenn Springs Holdings) under the Superfund Alternatives Program. In this program, EPA has secured settlement agreements for PRP-led cleanups without listing the site on the NPL. Settlements and cleanups at

Superfund alternative sites are intended to be equivalent to settlements and cleanups at sites listed on the NPL and should provide for timely action that meets the same cleanup standards as if the site were officially designated (EPA 2004e).

At the East Tennessee Copper Basin site, requirements for remediation of the North Potato Creek watershed are defined in a consent order between Glenn Springs Holdings (GSH) and the Tennessee Department of Environmental Conservation (TDEC). The consent order requires GSH to restore the "biological integrity" of North Potato Creek, as defined in state water-quality regulations. However, the order does not prescribe a specific remedy, and there is no explicit timetable for completion. GSH must continue the remediation until the biological performance goal is met. Because TDEC defines biological integrity in terms of the characteristics of benthic invertebrate communities present in unimpaired streams, waste removal, acid drainage control, revegetation, and in-stream habitat restoration will all be required to meet the site performance goal. TDEC and GSH have implemented a site-wide biological monitoring program intended to measure progress toward the goal and to identify the specific chemical and physical stressors contributing to the impairment of different on-site stream reaches. GSH intends to apply an adaptive management approach to the site, in which metrics for both engineering performance and biological performance are used to measure the success of each remediation project and determine the need for further actions.

From an institutional perspective, the Copper Basin site had the advantage that there were viable private responsible parties capable of and agreeable to performing the cleanup work under a consent decree. Availability of a willing PRP permitted the site to be managed under the Superfund Alternatives Program and facilitated the implementation of an unusually flexible and innovative approach to remediation.

The Clear Creek Watershed in Colorado provides another example of conducting a cleanup under an "informal" basin-wide approach (Pring 2001; EPA 2004f). EPA listed the entire upper watershed of this basin on the NPL in 1983 but has attempted to promote the cleanup of much of the basin through a Clean Creek Watershed Forum that includes more than 50 governmental and nongovernmental organizations. Some of the work is being conducted under Superfund, some by private companies, and some by state or local governments and environmental organizations. Part of this cleanup involves re-mining of mining wastes.

EPA has also demonstrated substantial flexibility in cleaning up the Coeur d'Alene River basin. For instance, as frustrating as it may be for basin citizens and others attempting to review the agency's plans, the agency's approach to deferring the final decision about how proposed remedial actions will be implemented is practical and reasonable at sites involving such inherent complexities and uncertainties as Coeur d'Alene.

The agency has demonstrated its flexibility in the Coeur d'Alene River basin in a number of other ways as well. Its agreement to establish a Basin Environmental Improvement Project Commission (BEIPC) made up of representatives from Idaho, Washington, the Coeur d'Alene tribe, and county officials as well the EPA Region 10 Administrator is an innovative management approach.[13] The BEIPC is responsible for setting priorities, directing and coordinating an annual work plan, and generally overseeing environmental remediation and natural resource restoration projects in the Coeur d'Alene River basin (BEIPC 2004). To support its efforts, it has established a technical leadership group (TLG), composed of 23 government entities, and a citizens' coordinating council. This is apparently the first time that EPA has assigned such responsibilities to such an organization (EPA 2004g).

Another example of EPA flexibility is the agency's inclusion of other agencies such as the U.S. Geological Survey, the Fish and Wildlife Service, the Forest Service, and the Bureau of Land Management in helping characterize the contamination problems and implementing the cleanup program. The efforts of all these agencies are coordinated under the auspices of the BEIPC, and they are all represented on the TLG responsible for evaluating proposed technical studies and remedial activities. Few Superfund sites have as broad participation from federal agencies as the Coeur d'Alene River basin.

The Basin Environmental Monitoring Plan the agency has developed is much more extensive and comprehensive than normal for a Superfund site. This plan appears to recognize the complexities and uncertainties of the system and should provide much of the information needed to make informed decisions about the most important and effective cleanup approaches.

Finally, EPA deferred action on cleaning up Lake Coeur d'Alene to allow the state, tribal, and local authorities to develop and implement a lake management plan addressing the human and environmental health risks that the lake may present.

Thus, in many ways, the current cleanup strategy appears to recognize the complexities of the system while working within the constraints of CERCLA and the NCP. At this and other sites, the agency has demonstrated an ability and willingness to take advantage of the flexibility that Superfund provides, particularly if there are viable parties willing and able to accept responsibility for the cleanup activities.

The flexibility that Superfund presents, however, does not appear sufficient to address all the issues identified by the committee. The fund cannot

[13]The committee was not charged with considering the structure, development, or effectiveness of the BEIPC and has not done so in this report.

be used to support the full range of activities that may be desirable to establish healthy communities and ecosystems, and there is no guarantee of long-term funding that is necessary for projects that will take as long to implement and maintain as those proposed in the Coeur d'Alene River basin.[14] Current rules cannot resolve the competing incentives resulting from the distinction between payment of construction costs and O&M costs under fund-financed cleanups. Finally, the liability problems that may be interfering with the adoption of some potentially effective approaches to cleanup remain a problem.

CONCLUSIONS AND RECOMMENDATIONS

Given these problems, the committee believes that an effective program for mining megasites should emphasize long-term management of sites, recognizing that the remediation process inevitably will take decades to complete. The objectives of the program would be to protect human health and the environment, using a combination of institutional controls, active remediation, and habitat restoration. The desirable characteristics of such a program would include the following:

- A stable management structure, which includes federal, state, and local representation
- State and local involvement in defining remediation/restoration goals, considering present and future desired land use
- The ability to address socioeconomic as well as health and environmental aspects of remediation, including the need for economic assistance for low-income communities and provision of health support services for communities living with human health risks
- Long-term commitment to funding, from a mix of state, federal, and private sources

The recommendations below are intended to address problems the committee has observed in the process currently used to remediate large, geographically complex mining sites under Superfund. Most of these rec-

[14]This limitation results more from the federal budget process than from any restrictions in the Superfund program. Under the federal budget process, an agency cannot obligate any funds that have not been appropriated. The agency conceivably could work within this restriction by obligating all the funds needed for future work out of current appropriations. However, such an approach is not feasible for two reasons. One is that, given the uncertainty inherent in such a complex site as Coeur d'Alene, there is no way to accurately predict how much money will be required in the future. A second is that any such obligation, even if there were sufficient funds currently available to fulfill it, would divert funds from other sites and substantially disrupt their cleanup.

ommendations can be accomplished within the existing Superfund framework, and some reflect actions that EPA has already undertaken in the Coeur d'Alene River basin. Some recommendations may not be possible under the current Superfund framework. However, even these problems may be addressed in part by Superfund, particularly if there are private sources of funding available. The committee recommends the following:

1. From the beginning, design the data collection, evaluation, and decision-making process so that it is focused on establishing a durable process for long-term management of mining megasites, rather than selecting "final" remedies that cannot truly be final. Because of the long-term commitment required, active involvement by the affected states and local communities is essential. Long-term management requires long-term management structures.

2. Focus on the basic purposes of CERCLA, protecting human health and the environment, and be ready to waive specific ARAR requirements if an effective monitoring program demonstrates that it is not necessary to achieve these numeric standards to achieve these basic purposes. In taking this approach, it is important that the agency specifically define what will be necessary to achieve these goals and what monitoring information will be needed in order to determine when they have been achieved. The goals of protecting human health and protecting the environment are open to multiple interpretations. Experience both within the Coeur d'Alene River basin and with other large sites such as the DOE weapons complex shows that protecting human health can involve a combination of cleanup and institutional controls, depending on the long-term land use projected for a site. The best approach to protecting the environment is to define biological performance goals that are also a function of future land use, and a remedy or suite of remedies should be designed to meet those performance goals.

3. Where it is unlikely that final remedies can be identified and implemented, establish a rigorous adaptive-management process as discussed in Chapter 8, with well-defined performance milestones, monitoring strategies, and evaluation criteria and focus the data collection and analysis activities on supporting this process. An adaptive approach to remediation should be applied consistently. The adaptive approach recognizes that the information needed to design a remedy that will meet all performance goals may not be available when remediation begins. The adaptive-management approach involves establishing goals and developing a monitoring program that measures progress toward the goals and provides data needed to adjust the remedy to meet the goals. This approach also emphasizes continuous real-time evaluation of remediation success and replacement of ineffective or inefficient approaches by more cost-effective approaches. Use of an independent technical advisor panel (see below) to provide oversight could

substantially improve the results obtained from the adaptive management approach.

4. Establish an independent external multidisciplinary scientific review panel to evaluate and advise the agency on critical needs for characterization and remediation decisions at mining megasites as a quality control mechanism. Although establishing an expert review panel may appear to add to the bureaucratic process, at particularly complex sites it may well speed up the cleanup, help avoid unnecessary costs and costly mistakes, as well as provide an acceptable mechanism for resolving technical disagreements. EPA does not have sufficient technical resources to devote to a particular site to conduct the types of technical reviews that are necessary.

5. Broaden the goals of the cleanup to include economic assistance to impacted communities as well as provision of comprehensive medical support services which acknowledge that the effects of toxic waste sites have broad impacts on health. Services would include increased medical support to prevent, diagnose, and counsel community members on the increased risk of cancer, learning/behavioral disabilities, hypertension, pulmonary/cardiovascular disease, and psychiatric illness associated with exposure to environmental toxic waste. Restoration of habitat for ecologic resources should also be provided to the extent required to meet biological performance goals. If these activities cannot be financed under Superfund, explore the possibility of obtaining the necessary support from other federal, state, and nongovernmental entities. If there are viable PRPs associated with the site, the funds they contribute could be allocated to these types of activities.

6. Encourage alternative and innovative technologies including responsible re-mining to clean up at least some of the contamination. If this appears to be a viable option but liability concerns interfere with its implementation, consider offering indemnification to participants, agreeing that any liability will be limited to problems resulting from the remediation activity.[15] It would also be very helpful for EPA to maintain a publicly available source of information on examples of mine-site remediation alternatives that have both succeeded and failed along with general information on their costs and examples of their implementation.

7. Look for opportunities to provide long-term support for implementing and maintaining the cleanup activities and stewardship of the land. Possible sources of such support might include trust funds established from special appropriations by Congress or made available by public and private organizations interested in the site.

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[15]Such relief obviously should not be afforded to any responsible party at the site that has not entered into a binding settlement agreement with EPA regarding their cleanup liability.

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Appendix A

Statement of Task and Committee Biosketches

STATEMENT OF TASK

A multidisciplinary committee will independently evaluate the Coeur d'Alene River basin Superfund site in northern Idaho as a case study to examine the U.S. Environmental Protection Agency's (EPA's) scientific and technical practices in Superfund site area characterization, human and ecological risk assessment, remedial planning, and decision making. The committee will assess the adequacy and application of EPA's Superfund guidance—in this case, in terms of currently available scientific and technical knowledge and best practices. Recognizing that substantial actions have already been taken to assess and remedy some of the risks attributable to the Coeur d'Alene site, the committee will strive to provide guidance to facilitate scientifically based and timely decision making for this site in the future. The committee will discuss remedial options but will not recommend a specific remedial strategy for this site.

The committee will assess the scientific and technical aspects of the following:

- Determining the geographical extent of areas contaminated by waste site sources. What types of data and analysis are necessary to assess the extent of contamination? In this case, did the approaches used to collect and analyze the data provide results that adequately support EPA's conclusions? Were the sources, transport, and fate of identified contaminants properly considered?

- Assessing and apportioning risks to humans from multiple contaminant exposures related to waste site sources as well as other sources (for example, lead exposure via soil and house-paint dust). What techniques should be used to identify contaminants of concern and estimate the human health risks attributable to waste site sources? In this case, were risks attributable to sources other than mining and smelting activities adequately analyzed?
- Estimating blood lead levels in children with the integrated exposure uptake biokinetic model. Are the design, input data, and assumptions of this model consistent with current scientific understanding? In this case, was the model appropriately applied given the local and regional characteristics? Were alternative tools appropriately used to assess and interpret the model results?
- Assessing the ecological risk from waste site contaminants in the context of multiple stressors. What are the necessary data and appropriate analyses to estimate the ecological risks attributable to waste site contaminants? In this case, how well were these analyses applied to estimate the risks, including the effects of lead on migratory fowl? Were risks attributable to sources other than mining and smelting activities adequately analyzed?
- Defining the remediation objectives. What factors should be considered in selecting the remediation objectives? In this case, did EPA use an appropriate scientific rationale in selecting the remediation objectives, including the spatial extent and levels of remediation? Was this scientific rationale adequately explained? Were the limitations of the analyses appropriately described?
- Evaluating the remediation approaches. In this case, were the feasibility and potential effectiveness of the remediation plans adequately characterized, given best engineering and risk practices and the site-specific characteristics? Was an adequate set of alternatives considered?
- Lessons from the Coeur d'Alene case that may be applicable to similar Superfund sites. Do new approaches need to be developed in the Superfund program to assess the extent of contamination, the resulting health and ecological risk, and possible remediation strategies where water and/or air have distributed contamination over extensive geographical areas?

COMMITTEE BIOSKETCHES

David J. Tollerud *(Chair)* is professor of public health, medicine, and pharmacology/toxicology at the School of Public Health and Information Sciences, University of Louisville, and chair of the Department of Environmental and Occupational Health Sciences. He holds specialty board certifications in internal medicine, pulmonary and critical care medicine, and occupational medicine. He has extensive experience in epidemiology and population studies, particularly those involving the use of immunological biomarkers, and in environmental and occupational health research focusing on prevention of injury and illness. In addition to his work in public health, he supervises clinical trials data management and data analysis activities for the multidisciplinary Institute for Cellular Therapeutics at the University of Louisville. Dr. Tollerud has a 10-year history of service to the Institute of Medicine and has been a National Academies Fellow. He currently serves as a member of the Board on Health Promotion and Disease Prevention, and he is the Board Liaison to the Committee on Poison Prevention and Control. He served as chair for the Institute of Medicine Committee to Review the Health Effects in Vietnam Veterans of Exposure to Herbicides and the National Research Council Committee to Assess the Distribution and Administration of Potassium Iodide in the Event of a Nuclear Incident. Dr. Tollerud received his MD from Mayo Medical School, his MPH from the Harvard School of Public Health, and his BS in mechanical engineering from Stanford University.

Herbert E. Allen is a professor of environmental engineering at the University of Delaware and director of the Center for the Study of Metals in the Environment. Previously, he was the director of the Environmental Studies Institute and professor of chemistry at Drexel University. Preceding that, he was on the faculty of the Department of Environmental Engineering at the Illinois Institute of Technology. Dr. Allen's research is on the fate and effects of trace metals in aquatic, sediment, and soil environments; bioavailability of trace metals; environmental chemistry; ecological risk assessment; and the development of waste-site-specific criteria. Dr. Allen has served on the National Research Council Committee on Technologies for Cleanup of Subsurface Contaminants in the U.S. Department of Energy Weapons Complex. He received his PhD in environmental chemistry from the University of Michigan.

Lawrence W. Barnthouse is the president and principal scientist of LWB Environmental Services, Inc. His consulting activities include evaluations for nuclear and non-nuclear power plants, Superfund ecological risk assessments, natural resource damage assessments, and risk-based environmental restoration planning. He was formerly at Oak Ridge National Laboratory where he organized an ecological risk assessment group that was responsible for all ecological risk assessments performed on the U.S. Department of Energy sites at Oak Ridge, Tennessee; Portsmouth, Ohio; and Paducah, Kentucky. After leaving Oak Ridge National Laboratory, he was a consultant with McLaren-Hart, Inc., prior to establishing LWB Environmental Services. He is a member of the Atlantic States Marine Fisheries Service Cumulative Impacts Assessment Panel and chair of the Society of Environmental Toxicology and Chemistry's Population-Level Ecological Risk Assessment Work Group. He has served on the National Research Council Board of Environmental Studies and Toxicology and on several National Research Council committees, and was a member of the peer review panel for the U.S. Environmental Protection Agency's Guidelines for Ecological Risk Assessment. Dr. Barnthouse holds a PhD in biology from the University of Chicago.

Corale L. Brierley (NAE) provides technical and business consultation to the mining and chemical industries and government agencies through Brierley Consultancy LLC. Previously, Dr. Brierley worked as chemical microbiologist at New Mexico Institute of Mining and Technology as the chief of environmental process development for Newmont Mining Corporation, as a general partner at Vista Tech Partnership, Ltd., and as the president of Advanced Mineral Technologies. Her research interests include the application of chemical, physical, biological treatment and management of metal-bearing aqueous, solid, and radioactive wastes and biotechnology applied to mine production. She is a member of the Division Review Committee for the Risk Reduction and Environmental Stewardship Division at Los Alamos National Laboratory and is a member of the International Advisory Committee for the Biohydrometallurgy Symposia and the Editorial Board for Hydrometallurgy Journal. Dr. Brierley is a member of the National Academy of Engineering (NAE), serving on the NAE Program Committee and Committee on Membership, and has served on several National Research Council committees, including the Committee on Technology for the Mining Industries, the Committee on Earth Resources, the Committee on Novel Approaches to the Management of Greenhouse Gases, and chaired the Committee to Review the USGS Mineral Resources Program. Dr. Brierley holds a PhD in environmental sciences from the University of Texas at Dallas.

Edwin H. Clark II is president of Clean Sites Inc. in Alexandria, VA. He is the former secretary of natural resources and environmental control for the state of Delaware, vice president of the Conservation Foundation, and associate assistant administrator for pesticides and toxic substances in the U.S. Environmental Protection Agency. He has served as a member of the National Research Council Board on Environmental Studies and Toxicology and on several committees, including the Committee on Risk-Based Criteria for Non-RCRA Hazardous Waste. He holds a PhD in applied economics from Princeton University.

Thomas W. Clarkson (IOM) is the J. Lowell Orbison Distinguished Alumni Professor of Environmental Medicine and Professor of Biochemistry & Biophysics, and Pharmacology & Physiology in the University of Rochester School of Medicine and Dentistry. His research is on the pathways, mechanisms, and disposition of toxic metals in the body to seek a cellular-level understanding of how metals cross diffusion barriers in the body. Much of his recent research has focused on the effects of human exposure to methylmercury. Dr. Clarkson was elected to the Institute of Medicine in 1981. He received his PhD from the University of Manchester and an MD (Honoris causa) from the Umea University School of Medicine, Sweden.

Edmund A.C. Crouch is a senior scientist with Cambridge Environmental, Inc. He has published widely in the areas of environmental quality, risk assessment, and presentation and analysis of uncertainties. He has coauthored a major text in risk assessment, *Risk/Benefit Analysis*. Dr. Crouch serves as an expert advisor to various local and national agencies concerned with public health and the environment and has served on three National Research Council committees. He has written computer programs for the sophisticated analysis of results from carcinogenesis bioassays; has developed algorithms (on the levels of both theory and computer implementation) for the objective quantification of waste site contamination; and has designed Monte Carlo simulations for purposes of fully characterizing uncertainties and variabilities inherent in health risk assessment. He received his PhD from the University of Cambridge, England, in high-energy physics.

Alison C. Cullen is an Associate Professor at University of Washington's Daniel J. Evans School of Public Affairs. Her specialization areas include environmental risk analysis, environmental science and policy, quantitative uncertainty analysis, and statistical decision theory. Previously, she held positions in the Water Quality Branch of EPA and was on the faculty of Harvard University's School of Public Health. Her research involves the analysis of environmental health risk, decision making in the face of risks that are uncertain or varied across populations, and the application of value of information and distributional techniques. She is active in environmental exposure assessment projects in the United States and internationally. Also, she has served as a technical consultant to the Natural Resources Defense Council, the Environmental Defense Fund, and on the Risk Assessment Advisory Committee for the state of California. She holds a ScD from Harvard University School of Public Health.

Joseph H. Graziano is a professor of environmental health sciences and pharmacology and associate dean for research at Columbia University. Previously, he served on the faculties of the Rockefeller University and

Cornell University Medical College. He was the founding director of Columbia University's National Institute of Environmental Health Sciences (NIEHS) Center for Environmental Health in Northern Manhattan. In addition, he was the principal investigator of a 15-year NIEHS-funded prospective study of childhood lead poisoning carried out in the mining town of Kosovska Mitrovica, Yugoslavia. Dr. Graziano is the founding director of the Columbia University Superfund Basic Research Program on health effects and geochemistry of arsenic and lead. Dr. Graziano received his PhD in physiology from Rutgers University.

David L. Johnson is a professor of environmental chemistry at the State University of New York College of Environmental Science and Forestry. His research interests are in the development of analytical techniques for the determination of the chemical and physical forms of heavy metals in soils and atmospheric and aquatic samples, as well as quantitative relationships between soil lead and blood lead. In the past, he worked with the application of automated scanning electron microscopy/image analysis techniques for individual particle analysis. His current activities seek to combine geography with urban geochemistry to study the spatial and temporal resolution needed for addressing pollution abatement and remediation of metals in urban soils and for the creation of geography-based exposure assessments in environmental health studies. Dr. Johnson received his PhD in oceanography from the University of Rhode Island.

Ronald J. Kendall is the founder and director of The Institute of Environmental and Human Health at Texas Tech University and Texas Tech University Health Sciences Center and founding chair and professor of the Department of Environmental Toxicology, Texas Tech University. Dr. Kendall founded and directed The Institute of Wildlife and Environmental Toxicology at Clemson University and was the founding department head of the Department of Environmental Toxicology at Clemson University. He also previously directed the Institute of Wildlife Toxicology and was professor of environmental toxicology at the Huxley College of Environmental Studies at Western Washington University. Dr. Kendall is the past-president of the Society of Environmental Toxicology and Chemistry (SETAC) and has served on its board of directors and executive committee as well as served on the SETAC Foundation for Environmental Education Board of Directors. He was chairman of EPA's Joint Science Advisory Board/Science Advisory Panel Review on Data from Testing of Human Subjects and a member of EPA's Science Advisory Board Mercury Review Subcommittee. Dr. Kendall served as chair of EPA's Scientific Advisory Panel on the Federal Insecticide, Rodenticide, and Fungicide Act (FIFRA). He served on the National Research Council Environmental Status and Trends Program and the Committee on Risk Assessment Methodologies. Dr. Kendall received his PhD in fisheries and wildlife sciences from Virginia Polytechnic Institute and State University, Blacksburg.

John Kissel is an associate professor in the Department of Environmental and Occupational Health Sciences at the University of Washington, where he is the program director of the Environmental Health Department in the School of Public Health and Community Medicine. His research focuses on human exposure to environmental contaminants, including soil-borne metals such as lead and arsenic. Dr. Kissel's work on human dermal contact has been used in the development of exposure factors used at Superfund and other contaminated sites. Dr. Kissel also conducts research on the predictive capability of regulatory exposure models, including determinations of the relative contributions of the ingestion, inhalation, and dermal absorption routes. Among other honors, he is the past president of the International Society of Exposure Analysis and previously the chair of the Exposure Assessment Specialty Group of the Society for Risk Analysis. His externally funded research history includes projects supported by EPA, the U.S. Department of Energy, and the National Institute of Environmental Health Sciences. Dr. Kissel is an environmental engineer and received his PhD in civil engineering from Stanford University.

Thomas W. LaPoint is a professor in the Department of Biological Sciences and director of the Institute of Applied Sciences at the University of North Texas. Previously, he was a professor in the Department of Biological Sciences and leader of the aquatic toxicology section within the Institute of Environmental and Human Health at Texas Tech University. Prior to that, he was a professor of environmental toxicology and leader of the aquatic toxicology section at the Institute of Wildlife and Environmental Toxicology at Clemson University. In addition, he was the assistant chief biologist at the National Fisheries Contaminant Research Center in Columbia, MO. Dr. LaPoint's primary research and teaching interests are in contaminant effects on freshwater aquatic communities and in understanding linkages among fisheries and benthic population dynamics and how these are influenced by anthropogenic perturbations. He also conducts research on the distribution of chemical pollutants and how they affect community structure and function. Dr. LaPoint holds a PhD in aquatic biology from Idaho State University.

David W. Layton is a senior environmental scientist at Lawrence Livermore National Laboratory (LLNL), University of California. His research at LLNL has focused mainly on assessing health risks of contaminants in environmental media and foods and on the environmental impacts of energy technologies. In addition, he has broad expertise in applying models to simulate human exposures and uptake of environmental contaminants. His research has included assessments of soil-based exposures of plutonium and uranium and evaluation of cleanup criteria for contaminated soils; assessments of models for predicting chemical exposure and transport; and studies of the penetration and transport of particles to residences. Dr. Layton has also conducted major studies on the environmental chemistry and toxicology of conventional ordnance, field-water quality standards for military personnel, and geothermal energy. At LLNL, he has conducted risk assessments of hazardous gas releases, contamination at a Superfund site and Department of Energy facilities, heterocyclic amines in cooked meats, and nuclear wastes dumped in the Arctic Ocean. To improve exposure assessments for airborne contaminants, he developed a metabolically based model for determining breathing rates. He has also conducted studies on modeling the environmental transport and fate of transportation fuels and associated additives such as ethanol and MTBE. Dr. Layton holds a PhD in water resources administration from the University of Arizona.

C. Herb Ward is the Foyt Family Chair of Engineering at Rice University, where he is also professor and chair of Civil and Environmental Engineering and professor of Ecology and Evolutionary Biology. Dr. Ward has directed the EPA-sponsored National Center for Ground Water Research and the Department of Defense Advanced Applied Environmental Technology Demonstration Facility. He is currently chair of the Scientific Advisory Board of the Strategic Environmental Research and Development Program and chair of the Division Review Committee of the Risk Reduction and Environmental Stewardship Division of the Los Alamos National Laboratory. His research interests include the microbial ecology and bioremediation of hazardous waste sites, aquifer restoration, and environmental remediation technology development. He has chaired National Research Council committees including the Committee on Technologies for Cleanup of Subsurface Contaminants in the U.S. Department of Energy Weapons Complex and the Committee on the Department of Energy-Office of Science and Technology's Peer Review Program and has served on several other NRC committees. He received his PhD degree from Cornell University and an MPH from the University of Texas School of Public Health. He is the founding and current editor-in-chief of the international journal *Environmental Toxicology and Chemistry*, a professional engineer in Texas, and a certified environmental engineer by the American Academy of Environmental Engineers.

Dr. Spencer Wood is a professor in the Department of Geosciences at Boise State University. He has wide-ranging expertise in geology, geomorphology (including modern erosional and sedimentary processes), seismology, hydrogeology, and tectonics. His current research involves studying the geophysical log expression of lacustrine sedimentary facies of aquifers, researching the geomorphology of floodplains, geologic mapping, and studies of the Quaternary faulting and geomorphic evolution. His recent research support has been provided through the U.S. Geological Survey, the Idaho Department of Water Resources, and Boise State University. Recent field research has involved evaluations of erosional events through analysis of alluvial and lacustrine stratigraphy, analysis of the geologic controls of recharge in river valley groundwater systems, channel morphology and bed materials in riverine systems, and the use of high-resolution geophysics to study the geometry of aquifer systems. Dr. Wood has been participating in geologic investigations for the past 40 years, most recently in Idaho, Thailand, and Nepal. He has multiple peer-reviewed publications and is currently completing a book on the geology of Idaho. Dr. Wood received his MS in geophysics and his PhD in geology from the California Institute of Technology.

Robert Wright is an attending pediatrician at the Children's Hospital in Boston, MA and assistant professor of environmental health at the Harvard School of Public Health. His research focus is in childhood neurodevelopment examining both genetic and environmental predictors of developmental performance, especially relating to using nutritional and toxic metals as predictors of neurodevelopment. Dr. Wright is the principal investigator (PI) or co-PI on several studies funded by the National Institute of Environmental Health Sciences (NIEHS), the National Heart, Lung, and Blood Institute (NHLBI), and the Kresge Center for Environmental Health. He is the project leader in National Institutes of Health-funded study examining a birth cohort in Tar Creek, OK—a Superfund site contaminated with lead and manganese. This study will evaluate health disparities of infant development with an emphasis on genetic susceptibility to lead and manganese exposure. Dr. Wright has published extensively in peer-reviewed journals and books and is board certified in general pediatrics and medical toxicology by the American Board of Pediatrics. He received his MD from the University of Michigan and his MPH from the Harvard School of Public Health.

Appendix B

Evaluation of the Methodology to Determine Background Concentrations in the Lower Basin

The determination of background concentrations for compounds of potential concern (COPCs) in the lower basin is described in the *Final Technical Memorandum (Rev. 3) Estimation of Background Concentrations in Soil, Sediment, and Surface Water in the Coeur d'Alene and Spokane River Basins* (URS Greiner, Inc. and CH2M Hill 2001) (Background Technical Memo). Although the upper basin, lower basin, and Spokane River are addressed in the memo, only the lower basin is considered in this Appendix.

The data to determine these background concentrations were derived from an ambitious coring study conducted in the lower basin to determine the vertical extent of metal contamination and estimate the volume of contaminated sediments within the basin (URS Greiner, Inc. and CH2M Hill 1998). In this study, a multitude of cores were taken in the lateral lakes, lower basin floodplain, and the river.

The metals concentration data from these cores were assembled into a database, which was processed by the ten-step method described in the Background Technical Memo (Section 3.2, pp. 3-4 to 3-6) and is evaluated below.

It appears that the proposed basis of the ten-step method is this statement made in Step 1:

- For each COPC, the distribution of the pooled data was identified as lognormal and a lognormal CFD (cumulative frequency distribution) of the pooled data set (283 samples for each COPC) was plotted with log concentration in milligrams per kilograms (mg/kg) as the independent variable and the normal standard variate of the population as the dependent variable using the methods described in Section 3.1 (see URS Greiner, Inc. and CH2M Hill 2001, Fig. A-11).

- On a lognormal CFD plot, a pooled data set containing both background and contaminant concentrations will ideally show two distinct populations identifiable by their distinct slopes, separated by a transition zone of rapidly escalating concentrations. The population with lower concentrations represents background, while the population to the upper right of the distribution is taken to represent contaminated sediments.

No clearer definition of what is considered background is provided; it appears from the procedures adopted that the "distinct population" with lowest concentration is assumed to be the distribution of background concentrations, and this is how we interpret the data below. It is not described how "the pooled data was identified as lognormal," but they clearly are not for any COPC. Single lognormal distributions would plot as approximately a single straight line on the plots constructed,[1] and the pooled data clearly do not fall along such single straight lines.

It appears to be implied that the observed data are necessarily a probabilistic sum of two lognormal distributions that would plot as two distinct straight lines. However, this implication is false. A probabilistic sum of two lognormal distributions does not plot as two straight lines, and there is no guarantee that there are only two component distributions, nor is there a guarantee that any component distributions are lognormal. In practice, the data on individual COPCs often show plots that approximate the description given in Step 2, and the distributions for individual COPCs often can be approximated as a sum of lognormals, but it is not necessarily possible to discern by eye on such plots how many component lognormals are necessary to fit the data adequately.

Practically, there is reason to suggest that the assumption of two populations—background and contaminated sediments—is too simplistic, especially considering the environment being modeled. These proposed sediment populations would exist in a continuum with each other and vary greatly through time as background sediments and tailings interacted in varying proportions based on the dynamic interaction of flooding events,

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[1]The "normal standard variate" described in the first paragraph of Step 1 is an approximation to the expected value of the order statistic for a normal distribution. One of the best available omnibus tests for normality makes use of the correlation coefficient calculated between (better approximations for) the expected values of these order statistics and measured data, using empirically derived curves to associate correlation coefficients with probabilities (Royston 1993, 1995).

tailings production, changing mining technologies (for example, stamp and jig techniques versus flotation), tailings disposal practices, secondary releases of tailings, and input of sediments from unaffected watersheds and floodplains. Also, as mentioned in the text of Chapter 4, the large sample intervals used in the coring studies have the potential for sampling both pre- and postmining sediments in a single analysis.

Steps 2 and 3 of the procedure are subjective because they call for visually selecting a straight line "through the lower bound population" and selecting a location where the data plot "diverges from" this straight line.

Steps 4 and 5 call for plotting on a similar "lognormal CFD plot" the data lying below the point of divergence identified in Step 3 and the least-squares fitting of a line to those data. Although least-squares fitting is an objective procedure, there is no objective basis for selecting an unweighted least-squares fitting procedure, and there is good reason not to, because even for a true lognormal distribution the variation of plotting points away from their expected values is heteroskedastic.

Step 6 calls for constructing a line bisecting the two lines constructed so far (the "visually fit tangent line and the lower bound data population regression line"). No basis is supplied for selecting a bisecting line rather than any other. Step 7 selects the 95th percentile on this line (the value of the abscissa at ordinate 1.645). Again, no basis is supplied for the selecting the 95th percentile.

Steps 8-10 then select the data points below the selected 95th percentile as being representative of the background lognormal distribution and use least-squares fitting to estimate the parameters of it.

The overall effect of this ten-step process is to obtain estimates that artificially truncate the background distribution of concentrations, assuming that it is lognormal.

The Background Tech Memo states (p. 3-6) the following:

> This approach is believed to provide a reliable means of estimating background concentrations for each COPC in the Lower Basin. This approach is supported by both empirical testing and statistical evaluation of the best-estimate background data set. In all cases, the identity of the best estimate background data set as a distinct population representative of background is supported by high r^2 values.

No indication is given of what empirical testing or what statistical evaluation has been performed. Overall, the evaluation indicates that the procedure is subjective and contains several assumptions unsupported by any documented statistical theory. However, as mentioned in the text, the background concentration for lead in lower basin sediments appears reasonable, considering evaluation of the metals analysis data from the cores and other studies assessing background concentrations in the lower basin.

If this type of mathematical analysis is to be used, the following suggestions are provided:

- Explicitly define the assumptions behind the analysis applied to obtain estimates of background distribution.
- Adopt objective techniques to obtain the parameters of interest with known uncertainty bounds (for example, the ten-step process relies on subjective approaches).
- Use appropriate statistical techniques, either explicitly proving any required statistical properties or citing literature for such support (for example, there is no evidence that the ten-step process is reasonably unbiased, and no estimator of its uncertainties is available).
- Implement adequate quality control to ensure that all the data used are included in the report—for example, the data for zinc concentrations in sediments of the lower basin are not provided in the report as they are for the other metals (URS Greiner, Inc. and CH2M Hill 2001, Table C-2).

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Appendix C

Detailed Comments on the Integrated Exposure Uptake Biokinetic Implementation Code

The uncertainties and inaccuracies listed below are referenced to sections of the Technical Support Document (TSD) for the integrated exposure uptake biokinetic (IEUBK) model (EPA 1994).

1. The bone weight (WTBONE, as given by equation B-5g (p. A-10 of the TSD) is not continuous, because the two equations do not match at 12 months with the given definition for WTBODY. At 12 months, $0.111 \times \text{WTBODY} = 1.1192265$, whereas $0.838 + (0.02 \times 12) = 1.078$, about 4% lower.

2. Equation B-2b (p. A-7 of the TSD) defines TRBCPL as

$$\text{TPLRBC} \times (\text{RATBLPL} - 0.55/[0.55 + 0.73]).$$

The text (p. 40 of the TSD) simply states that TRBCPL is the product of TPLRBC and RATBLPL minus a constant, without any explanation why. If TRBCPL is being estimated by the usual assumption that the ratio of TRBCPL and TPLRBC is equal to the steady-state mass ratio (p. 29, paragraph 2 of the TSD), then the "constant" here is not in fact quite a constant, because then:

$$\text{TRBCPL/TPLRBC} = \text{RATBLPL} - (\text{VOLPLASM/VOLBLOOD})/([\text{VOLPLASM/VOLBLOOD}] + [\text{VOLECF/VOLBLOOD}])$$

3. VOLECF/VOLBLOOD = 0.73 (equation B-5d of the TSD), but VOLPLASM/VOLBLOOD is not the constant 0.55 implied in equation

B-2b. Although this ratio is fairly constant, it is only as low as 0.55 for ages less than 0.4 month and exceeds 0.6 for all ages between 5 and 84 months (with the parameter values given in equations B-5a and B-5c). None of this makes any substantial difference, but the discussion on page 29 needs to be amplified to indicate where this "constant" comes from.

4. On the same matter, to agree with the statement that the ratio of times is equal to the ratio of steady-state masses (p. 29 of the TSD), it should not be the ratio of TRBCPL and TPLRBC that is set to this mass ratio but the ratio of TRBCPL to TPLRBC2, because TPLRBC2 is the actual-time constant.

5. The definition of TPLRBC2 given in equation B-2.5 of the TSD is not physical, since it relates to VOLRBC (t – 1), which presumably is supposed to be the volume of red blood cells at the previous time step, and, of course, the time step of a computer program has nothing to do with the mathematical definition of the problem. It might be a viable approximation in a computer program to use the value in the previous time step, but in the actual computer code, the value in the previous month is used not the value in the previous time step.[1]

6. On p. A-10 of the TSD, equations B-5a, B-5b, and B-5c define the blood, plasma, and red blood cell volumes, but the required relationship VOLBLOOD = VOLPLASM + VOLRBC does not hold at all times. It is not clear what the difference is supposed to represent. With the values given, this difference turns out to have different signs at different ages, suggesting that the equation just given is supposed to hold (as one would expect, unless there is supposed to be another compartment to hold the other cellular components of blood). This is an example of an unnecessarily introduced approximation that would be trivial to correct.

7. On p. B-7 of the TSD, a definition of HCT0 is given in such a way that numerically it differs from 1 – VOLRBC(0)/VOLBLOOD(0). This is again an unnecessary approximation.

8. On page A-18 of the TSD, the initial conditions are defined. However, the source of these initial conditions is not clear. The statement after equations B-7a through B-7d that equations B-7a through B-7d are "numerically equivalent to the following equations" is incorrect. For example, equation B-7d could be numerically equivalent to the corresponding equation below only for HCT0 = 1.284, which is physically impossible. B-7b could be numerically equivalent to its corresponding equation below only accidentally. Indeed, neither set of equations corresponds to the assump-

[1]The system requirements and design document for the IEUBK model (EPA 2002) indicate that *t* refers to the month (which corresponds to the code). As mentioned, use of the value from the previous time step would be a viable approximation, but instead the previous month is used.

tions described earlier in the TSD. If some other set of assumptions is being used, then it should be documented how those assumptions lead to the equations of p. A-18. In the computer code, both sets of equations are present, and indeed both are executed; but only the second has any effect.

9. On page A-19 of the TSD, equations B-7e and B-7l contradict the statements made under MCORT*(t)* on page B-9, and MTRAB*(t)* on page B-11. In both cases, it is stated that there is an assumption that the bone (cortical or trabecular) lead concentration/blood lead concentration ratio is equal to the bone (composite) lead concentration/blood lead concentration ratio (so cortical and trabecular bone lead concentration/blood lead concentration ratios should be equal). Equations B-73 and B-7l give different concentration ratios (78.9 for cortical, 51.2 for trabecular).

10. Equations B-4a through B-4d (p. A-9) are stated (p. B-4 and B-5) to come from an analysis of the data of Barry (1981). However, at age 0 they are contradicted by the initialization conditions given in equations B-7e through B-7l (p. A-17), which are said to be based on the same data (p. B-9, B-10, B-11). For kidney, liver, and other tissues, the tissue/blood concentration ratios implied by equations B-4a, B-4b, and B-4d at time 0 are 0.777, 1.1, and 0.931 L/kg, whereas equations B-7f, B-7g, and B-7h give 1.06, 1.30, and 1.60 L/kg, respectively. Here is another internal inconsistency, because equation B-4c gives a bone/blood concentration ratio of 6.0 L/kg at t = 0, whereas equations B-7e and B-7f give separate ratios at t = 0 of 7.89 and 5.12 L/kg for cortical and trabecular bone, respectively.

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Appendix D

Procedures Used in Model Comparisons

Similar to the Agency for Toxic Substances and Disease Registry (ATSDR) Health Consultation (ATSDR 2000), data from the Field Sampling Plan Addendum (FSPA06) conducted in support of the remedial investigation (RI) (URS Greiner and CH2M Hill 2001) were used in this analysis. For the present study, however, the number of homes was slightly different for two reasons: (1) data for two houses originally tabulated in the RI were not used in the ATSDR comparison—these were added for the committee comparisons. (2) The ATSDR analysis used geometric mean house-dust values for seven houses where those data were not originally collected. In the present comparison, those houses were dropped from consideration, and the results are based solely on residences where both soil and house dust measurements were available. The data set used in these calculations (referred to below as the 75 homes' data) is presented in Table D-1 of this appendix.

THE ONTARIO MINISTRY OF ENVIRONMENT AND ENERGY BIOKINETIC SLOPE FACTOR MODEL

The Ontario Ministry of Environment and Energy (OMOEE) has established an intake of 3.7 micrograms (µg) lead per kilograms (kg) of body weight/day as the level of intake for which more than 95% of children will have blood lead values less than 10 µg per deciliter (dL). This intake of concern (IOC) is divided by 2 to provide a safety factor; the resulting IOC is 1.85 µg of lead/kg of body weight/day. For the model comparisons, lead

TABLE D-1 FSPA06 Data Used in Calculations

House	Arithmetic Mean of Yard Soil, 0-1 in. (mg/kg)	Geometric Mean of Community Soil (mg/kg)	Vacuum Bag Dust	House	Arithmetic Mean of Yard Soil, 0-1 in. (mg/kg)	Geometric Mean of Community Soil (mg/kg)	Vacuum Bag Dust
1	663	419	606	38	278	419	427
2	804	419	480	39	1,423	568	1,020
3	174	419	764	40	364	352	341
4	448	419	173	41	766	628	682
5	4,796	110	3,140	42	769	419	23
6	1,189	419	1,000	43	688	368	1,820
7	1,610	628	1,620	44	16,026	771	6,150
8	1,080	419	978	45	718	568	2,430
9	870	419	528	46	503	419	769
10	259	419	390	47	500	568	387
11	623	257	525	48	3,054	568	2,730
12	239	257	422	49	843	568	619
13	979	419	154	50	852	771	3,300
14	290	257	389	51	56	368	626
15	665	257	765	52	319	419	504
16	342	419	332	53	256	419	492
17	760	419	1,260	54	3,026	419	621

18	3,491	352	604	55	787	419	1,550
19	5,566	628	1,960	56	735	257	315
20	794	419	1,200	57	544	368	504
21	1,014	568	1,660	58	642	568	384
22	276	352	680	59	353	368	833
23	796	419	818	60	2,711	568	353
24	871	419	512	61	1,165	771	778
25	451	771	639	62	188	257	232
26	1,337	771	1,350	63	284	568	1,680
27	1,687	771	798	64	563	419	655
28	977	419	808	65	2,701	628	1,540
29	813	568	703	66	1,194	352	937
30	438	568	84	67	1,094	771	780
31	682	419	762	68	2,788	568	1,380
32	622	568	349	69	479	568	727
33	1,322	628	767	70	1,381	568	405
34	437	568	383	71	321	771	942
35	1,576	568	1,020	72	3,837	419	362
36	827	628	710	73	2,861	628	2,840
37	3,603	568	1,020	74	694	368	2,400
				75	807	419	1,000

SOURCE: Data provided by Idaho Department of Health and Welfare, unpublished material, 2004.

intake from soils, dusts, water, air, and food is calculated from measured media concentrations and added to background default levels in non-measured media. The factor by which the estimated intake exceeds the IOC is obtained by dividing the result by 1.85 μg lead/kg body weight/day. The percentage of locations for which exposure estimates are less than a factor of 2 above the IOC is taken as the percentage of children whose blood lead values are less than 10 μg/dL.

BATCH OPERATION OF THE INTEGRATED EXPOSURE UPTAKE BIOKINETIC MODEL

The 75 homes' data were used for blood lead estimates using the batch mode capability of the integrated exposure uptake biokinetic (IEUBK) model. For these comparisons, the estimated blood lead level at an age of 20 months was obtained. This age matches closely the age corresponding to maximum blood lead concentration and also corresponds approximately to the 16 kg body weight for which the OMOEE IOC computation is made.

IMPLEMENTATION OF THE O'FLAHERTY MODEL

The physiologically based, transport limited biokinetic model of O'Flaherty (O'Flaherty 1998) was applied to the 75 homes' data for comparison with the other models. Such comparisons are not exact because of differences in how the models specify input of exposure regimes and the way bioavailability is incorporated in the computations. Another impediment is the sensitivity of the O'Flaherty model to year of birth for the individual being simulated. As noted in the TRW adult lead model review (EPA 2001, Appendix K), a variety of model parameters may be adjusted in the exposure specifications to establish baseline conditions against which variations in soil and dust lead concentrations may be examined. For the O'Flaherty model implementation here (Advanced Continuous Simulation Language [ACSL] platform) the following variable values were used for model runs: year of birth, yob = 1980; frlung = 0.32 (bioavailability of inhaled lead—same as IEUBK); cair2 = 0.1 μg/m^3 (same as IEUBK); concentration of lead in water, cwater = 4 μg/L (same as IEUBK); rfood2 = 20 μg of lead/day ingested by adult; rfood3 = 15 μg lead/day ingested by child; and the concentration of lead in infant formula, cfmla = 0.01 μg/L. For tabulation in Table 6-3, the midpoint between blood lead at ages 12 and 24 months was used.

ADAPTATION OF MODELS FOR PREDICTIONS UNDER THE BUNKER HILL SUPERFUND SITE "BOX MODEL" CONDITIONS

The study of von Lindern et al. (2003) established a set of IEUBK model conditions that best fit the observed blood lead distribution for children living within the Bunker Hill Superfund site (BHSS). Discussion of this model and an evaluation of its application to predictions of blood lead levels for children living in the Coeur d'Alene River basin outside the BHSS box is detailed in the body of the report. Important points for the present comparison of model results are as follows: (1) the soil and dust exposure regime was weighted as 40% from household dusts, 30% from the residential soil, and 30% derived from the community-wide soils; and (2) bioavailability for soil and dust ingestion was set at 18%.

Soil lead values for the 75 homes' data (BHSS box conditions) were tabulated on a geographical location basis as the average between the individual residential lot surface-soil value and the geometric mean soil value for the community where the residence was located. The latter values were derived from the human health risk assessment for operable unit 3 (TerraGraphics et al. 2001, Table 6-48). To account for the lower bioavailability of lead in soils and dusts used in the box model, concentration values for these inputs were reduced to 60% of their original values before each model's invocation. This corresponds approximately to the change in bioavailability used in the box model version of the IEUBK model, since the default bioavailability from soil in the IEUBK is 30%. This approach was adopted because bioavailability, the fraction of lead intake that is taken up in the blood, could not be adjusted in the ATSDR model. The modification of the soil concentration achieves the same effect, because the model exhibits a linear response over the concentration ranges of interest. In the O'Flaherty model, the user cannot specify bioavailability, but the ACSL program constants were adjusted to reflect 40% dust and 60% soil inputs to the exposure module of the program. The O'Flaherty model uses age-specific soil/dust-ingestion rate functions that are not accessible in the executable program structure but whose average value is about 60% of the average IEUBK default ingestion regime.

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Appendix E

Crustal Element Analyses for Following Soil Lead Transport

Measurements of the crustal elements iron and manganese in yard soils, entry mats, and dust were made as part of remedial investigation (RI) studies supporting the human health risk assessment (HHRA) of nonlead contaminants in the basin (URS Greiner Inc. and CH2M Hill 2001, TerraGraphics et al. 2001). These data can be used to assess the sources of lead in indoor dust. Table 7-2 in the human health risk assessment (HHRA) presents a summary of the results of the residential sampling that shows that the concentrations of and in yard soils and entryway floor mats were essentially the same, with ratios of mat dust to surface soil of 0.94 and 0.97, respectively. In contrast, the ratios of vacuum dust to surface soil for the two elements were 0.57 and 0.53, respectively. The similarity in the concentrations of iron and manganese in the outdoor soils and indoor mats is consistent with tracking of soil into the houses sampled. In contrast, the elevated level of lead in mat dust compared with yard soils (described below) could be due to either indoor lead sources (lead-based paint) or the preferential tracking indoors of soil particles that have higher lead concentrations than the bulk soil samples processed using a 175 micrograms (μm) sieve. For example, lead concentrations on fine particles might be enhanced, whereas iron and manganese are crustal elements, so their concentrations would be expected to be independent of particle size—including those under 175 μm. Unfortunately, little is known about the particle sizes that are most effectively transported on footwear, and there is no clear physicochemical explanation for a particle-size-dependent concentration profile of lead in surficial soils—unless perhaps the ore processing methods and subsequent weathering processes of lead tailings preferentially produce lead in fine particles, or perhaps the majority of tracked particles are very fine lead particles deposited from air.

The dilution effect of indoor-derived organic-rich particles on the concentrations of crustal elements associated with tracked-in soils has been analyzed by Trowbridge and Burmaster (1997). For a series of crustal elements with no significant indoor sources (aluminum, cerium, iron, hafnium, lanthanum, manganese, sodium, phosphorus, scandium, samarium, thorium, and vanadium), the geometric mean (GM) dilution ratio (defined as the ratio of the concentration of the crustal element in house dust to its concentrations in yard soil) was 0.42, with a geometric standard deviation (GSD) of 1.44. A ratio of 1 would indicate that indoor dust is entirely of outdoor origin, whereas a ratio of 0 implies that outdoor soil does not contribute to indoor dust. The U.S. Environmental Protection Agency (EPA) default value for this ratio in the IEUBK model (defined as the M_{SD} parameter, or the mass fraction of soil in dust, grams [g] of soil/g of dust) is 0.70 (EPA 1998), which is higher than the apparent dilution values noted above. However, IEUBK model runs conducted in support of the HHRA (Tables 6-11a-h) used measured concentrations of lead in household dust and yard soil.

As a means of further exploring the relationships between the crustal elements and lead in soil and dust, we evaluated the analytical results of the sampling campaigns that included measurements of iron, manganese, and lead in yard soils, entryway mats, and vacuum bag dust (data provided by the Idaho Department of Health and Welfare from FSPA06). The data included residences in the towns of Kingston, Osburn, Mullan, Silverton, and Wallace, along with residences in the Side Gulches, Nine-Mile, and Burke. To minimize the potential impacts of different sampling techniques and geographical regions on the exploratory analysis, we restricted the evaluation to the basin towns and used only the top surface-soil samples (0 to 0.08 feet) from the borehole samples of the yards (thereby excluding surface grab samples and hand auger samples). The soil concentration, assumed to be representative for the multiple yard samples obtained at each residence, was calculated as the GM of the samples. A total of 37 residences had paired measurements of the crustal elements and lead in the soil, mat, and vacuum bag media. Three residences included data outliers for one or more of the soil constituents and therefore were removed from the analysis, leaving 34 residences for the analysis. The resulting concentration data for the soils, mats, and vacuum bags were then used to calculate ratios for mat/soil, vacuum bag/mat, and vacuum bag/soil. These ratios are presented in Table E-1.

The concentrations of iron and manganese in yard soils exhibit less variability than that of lead, which is reasonable given that the crustal

TABLE E-1 Summary Statistics for the Concentrations of Iron (Fe), Lead (Pb), and Manganese (Mn) in Yard Soils, Entryway Mats, and Vacuum Bags as Well as Computed Concentration Ratios for a Sample of 34 Residences in the Coeur d'Alene River Basin

		Elements					
		Fe		Pb		Mn	
Parameter	Units	GM	GSD	GM	GSD	GM	GSD
Yard soil	μg/g	19924	1.24	542	1.95	892	1.46
Entry mat	μg/g	19627	1.41	1029	2.09	904	1.43
Vacuum bag	μg/g	11841	1.96	626	2.48	516	2.17
Mat/soil ratio	Unit less	0.98	1.39	1.90	1.52	1.01	1.37
Bag/mat ratio	Unit less	0.60	1.92	0.61	2.03	0.57	1.95
Bag/soil ratio	Unit less	0.59	1.95	1.16	2.08	0.58	2.05

Abbreviations: GM, geometric mean; GSD, geometric standard deviation.

elements are from weathered soils, whereas soil lead in these communities is the result of complex transport processes from the many sources (for example, redistribution of flood sediments and mine tailings). Levels of iron and manganese are essentially the same in yard soils and mat dust samples, but lead is about a factor of two higher in the mat dust than yard soils, as seen also in the results of the Agency of Toxic Substances and Disease Registry (ATSDR)-funded study (ATSDR 2000). Nevertheless, the linear correlation coefficient, *r*, between the concentrations of lead in yard soils and mat dusts was 0.87, compared with 0.30 and 0.74 for iron and manganese, respectively. The *r* value for iron concentrations in mat dusts and soils, though, increases to 0.65 after removing three data sets in which iron levels in mats might have been sampling/analysis artifacts. The strong correlation between lead in soils and mats indicates that the apparent particle fractionation-enrichment process between yard soil and mats occurs in a systematic fashion among the sampled residences. We also calculated correlation coefficients of 0.71 between soil manganese and soil lead and 0.52 between soil iron and soil lead. Although more analyses are warranted, the congruence between the concentrations of crustal elements in soil and residual lead indicates that waste ore/tailings mixed with host soils have also changed the elemental composition of soils.

The vacuum bag/entry mat dilution ratios for iron and manganese have geometric mean values of 0.60 and 0.57, respectively, with geometric standard deviations (GSDs) of nearly 2, which are greater than the GSD of 1.44 reported by Trowbridge and Burmaster (1997) in their review of other studies. The Ln-transformed iron and manganese concentrations are highly correlated, as shown in Figure E-1, with $r = 0.93$. The dilution ratios are also substantially higher than the value of 0.42 reported by Trowbridge and

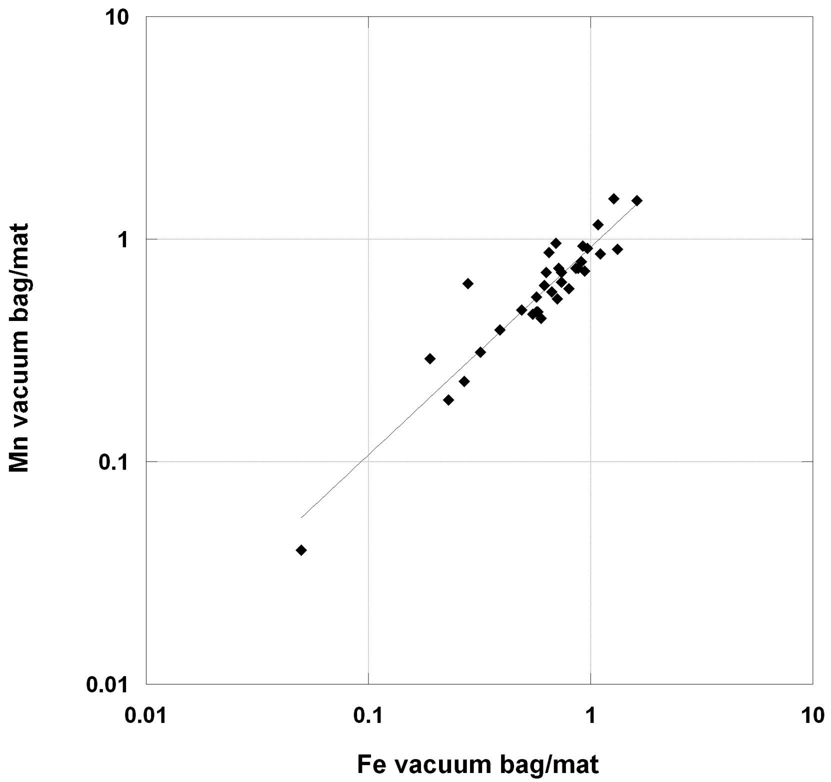

FIGURE E-1 Correlation between 1n-transformed concentrations of iron (Fe) and manganese (Mn) in vacuum bags and entryway mats for 34 basin residences. The correlation coefficient between the log-transformed concentrations is 0.93.

Burmaster (1997)—an indication that outdoor soils may be a more significant component of the indoor dusts in these communities. However, four of the vacuum bag/mat dilution ratios for iron were above 1, whereas three of the manganese ratios exceeded 1, suggesting that there were indoor sources of these elements (or possibly analytical artifacts). Other crustal elements may in fact be better tracers for characterizing the migration of soil lead to the indoor environment and in-house dilution processes in this mining region (Fe and Mn were targeted for sampling in the RI for human health considerations—not to study contaminant transport processes). Interestingly enough, the concentration reduction of lead between mat samples and vacuum bag samples is about the same as for iron and manganese. The correlations, though, between the log-transformed concentrations for lead and iron and lead and manganese (r values of 0.66 and 0.75, respectively) are lower than the correlation between the iron and manganese ratios (r = 0.93). Possible explanations are the presence of indoor lead sources such as lead-based paint particles and the differential transport of indoor lead due to particle-size-dependent processes of resuspension, deposition, and tracking.

We compared the vacuum bag/mat concentration ratios for manganese and lead by dividing the lead ratio by the manganese ratio to determine the potential extent of indoor lead sources. Figure E-2 presents a log probability plot of the resulting ratios. The GM of the ratios is 1.07, with a GSD of 1.61. More than half the ratios are greater than 1, which indicates that lead in vacuum dust may have nonoutdoor sources, such as lead-paint particles. This is particularly so for the many houses in the basin that were built before the phase out of lead-based paints.

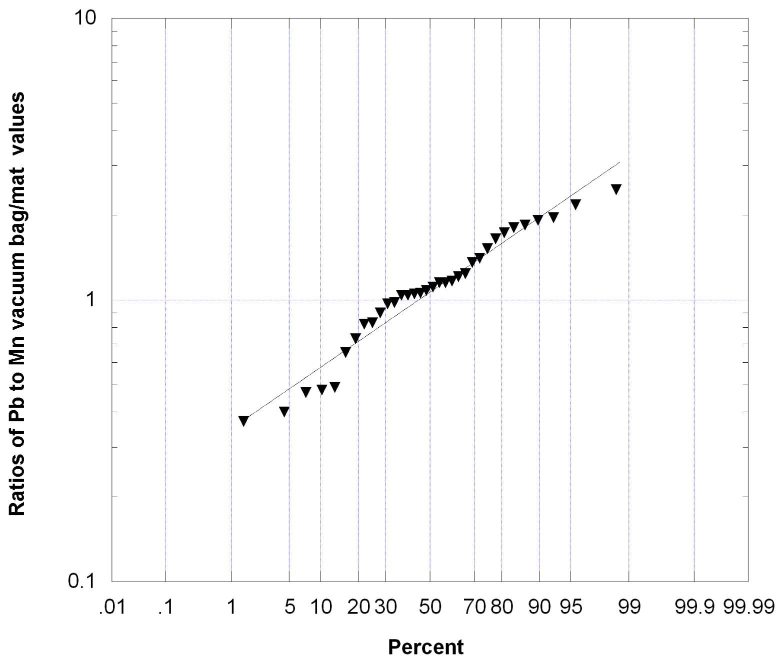

FIGURE E-2 Log probability plot of the ratios lead (Pb) vacuum bag/mat to manganese (Mn) vacuum bag/mat for 34 basin residences. The GM of the ratios is 1.07 with a GSD of 1.61.

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Appendix F

Assessment of the Probabilistic Model for Estimating Metal Loading and Effectiveness of Remedial Action

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The Probabilistic Analysis of Post-Remediation Metal Loading Technical Memorandum (Revision 1) (URS Greiner, Inc. and CH2M Hill 2001a) (PTM) describes its purpose very well:

> The probabilistic analysis is a risk management tool that can help quantify the certainty, conditional on available information and its interpretation, that a proposed remedy would meet cleanup goals. (PTM p. 1-1)
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> The purpose of the probabilistic analysis is to help support informed risk management decision-making. It does so by helping to quantify the certainty that a remedial alternative or a proposed remedy could actually meet cleanup goals. . . . (PTM, p. 1-3)

It formulates an approach intended to meet these objectives. The formulation can be readily summarized,[1] and this summary is presented first without any comment on its correctness or applicability. It is assumed that dissolved metal loading to the Coeur d'Alene River (e.g., in pounds/day, the

[1]Understanding this appendix will require access to and some familiarity with the PTM.

unit used throughout the PTM) at some specific location on the river, can be calculated as follows:

$$L = \sum_{j=1}^{K} RLP_j Z_{ref} \left\{ \sum_{i=1}^{N_j} V_{ij} \right\}, \tag{1.1}$$

where[2]

L = (preremedial) metal loading in the Coeur d'Alene River at the specific location examined (pounds/day);

Z_{ref} = "loading potential" per unit volume (pounds/day/cubic yard) for the reference source type for the location on the river under examination (averaged over all sources of that type affecting that location);[3]

RLP_j = "relative loading potential" for the location on the river under examination for a contamination source of type j, averaged over the sources of that type that affect the river location under examination (for the reference source type, the RLP is unity);

V_{ij} = volume (cubic yards) of a source of type j with index i that affects the river at the location examined, all such sources being indexed;

K = number of different types of sources that affect the river at the location examined; and

N_j = number of sources of type j that affect the river at the location examined.

The contamination sources are generally volumes of contaminated soil, sediment, and rock, categorized by type. The source types used in the PTM (pp. 2-18 to 2-19), for the upper basin, conceptual site model (CSM) Units 1 and 2, are adits (these are treated specially, by using measured flows and concentrations and deriving an effective volume for them), tailings-impacted floodplain sediments, unimpounded tailings piles, impounded tailings piles at inactive facilities, impounded tailings piles at active facilities, waste rock piles in floodplains, waste rock piles in upland areas, and deeper impacted floodplain sediments (unremediated sources).

The reference source type (with RLP = 1) is taken to be tailings-affected floodplain sediments.

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[2]The notation of the PTM is adopted, except that all symbols are italicized to agree to the degree possible with standard notational conventions (which are not observed in the PTM). The only possible confusion is between the symbols L and L which have distinct meanings in the PTM; however I avoid this confusion by using a different symbol, W, for what the PTM calls L.

[3]Only ratios of quantities each of which multiplies what I here call Z_{ref} are required in the PTM, so no such term is defined anywhere in the PTM. The exposition is made more concise and direct by introducing Z_{ref} explicitly.

For the lower basin, CSM Unit 3, the source types used are riverbed sediments, banks and levees, wetland sediments, lake sediments, other floodplain sediments, Cataldo/Mission Flats dredge spoils, and a composite of all the source types (unremediated sources).

At some time t after remediation (the time of which defines $t = 0$), the dissolved metal loading $F(t)$ at the same specific location is written as

$$F(t) = R(t)L, \tag{1.2}$$

where $R(t)$ is a remediation factor at time t.[4] $R(t)$ is a moving 1-year time average (it is defined in the PTM, p. B-4, as representing "one-year averages over each water-year"). The immediate effect of remedial action on each contamination source is supposed to be a reduction in the relative loading potential of that source by a "remedial action effectiveness" factor R_{ij} for the source type j and with index i, so that immediately after remediation the remediation factor is given by

$$R(0) \equiv R_0 = \frac{\sum_{j=1}^{K} RLP_j \left\{ \sum_{i=1}^{N_j} V_{ij} R_{ij} \right\}}{\sum_{j=1}^{K} RLP_j \left\{ \sum_{i=1}^{N_j} V_{ij} \right\}}. \tag{1.3}$$

For future times, $R(t)$ is written as

$$R(t) = R_0 \exp(-\beta t), \tag{1.4}$$

where the decay rate β is estimated as the ratio of the preremedial "total effective mass" of metal (TEM') available for leaching and the average preremedial rate W at which metal is removed via the river, or as the ratio of the same quantities immediately postremediation[5]

[4]Transient effects due to the remediation efforts themselves (e.g., stirring up sediments during remedial actions) are explicitly ignored and implicitly assumed to have no lasting effect.

[5]The PTM uses the symbol L to represent what I here call W. The notation in the PTM becomes confused, particularly in section B.2.2.2 starting on p. B-16, in not distinguishing 1-year time averages from instantaneous values. In Equation 1.5 above, $F(0)$ is used as in the PTM (p. B-18, equation 6), but what is meant is a time-averaged version of F, because F is defined as proportional to L, which is not time-averaged (PTM, p. B-4, equation 1, and Equation 1.2 above).

$$\beta = W / TEM' = F(0) / TEM'', \tag{1.5}$$

where

$$W = \frac{1}{\tau}\int_0^{\tau} L dt, \tag{1.6}$$

and τ is 1 year, while the total effective mass of metal (*TEM′* preremediation and *TEM″* postremediation) available for leaching are assumed to be computable as

$$\begin{aligned} TEM' &= \gamma C_s \sum_{j=1}^{K} RLP_j \sum_{i=1}^{N_j} V_{ij} \\ TEM'' &= \gamma C_s \sum_{j=1}^{K} RLP_j \sum_{i=1}^{N_j} R_{ij} V_{ij} \end{aligned} \tag{1.7}$$

where γ = volumetric unit weight of the reference source type, and C_s = volumetric average metal concentration of the reference source type.

Finally, the "load ratio", $Lr(t)$, is defined by

$$Lr(t) = F(t) / C_L, \tag{1.8}$$

where C_L, the "loading capacity," is the product of ambient water-quality criterion (*AWQC*) and river flow rate *Q*:

$$C_L = AWQC * Q. \tag{1.9}$$

The *AWQC* is a concentration of the dissolved metal in water and is defined by regulation at a value that is supposed to be protective of freshwater life. For many metals (and zinc in particular), the *AWQC* increases with the hardness of the water, and the hardness of the water in the Coeur d'Alene River varies inversely with the flow rate. The *AWQC* represents the target for most ecological cleanup efforts, in particular for the cleanup of the Coeur d'Alene River, so that a load ratio of unity represents the ultimate cleanup target.

The PTM evaluates estimates only for dissolved zinc, claiming that results for other dissolved metals *except lead* could be obtained approximately from those of zinc by using suitable scaling factors (PTM, p. 1-8, section 1.4).

The above summary makes no mention of uncertainties in measurement of the various quantities discussed (for many of the quantities), of their variability due to their unpredictable fluctuations with time, or of the correlations between such uncertainties or variabilities. The PTM attempts to account probabilistically for the uncertainties and variabilities. It does this analytically by assuming wherever necessary that uncertainty and variability distributions are lognormal and matching means and coefficients of variation (equivalently, standard deviations). That is, uncertain or variable quantities included in equations are assumed to have that uncertainty or variability represented by lognormal distributions, and the mean and coefficient of variation for the quantity on the left-hand side of the equation are obtained as the mean and coefficient of variation of the expression on the right-hand side of the equation (even if, strictly speaking, the combination of uncertainty distributions on the right-hand side of the equation does not result in a lognormal distribution).

DEFICIENCIES OF THE PTM

The PTM suffers from multiple invalidating deficiencies in its formulation and application. The formulation in the PTM goes into considerably more detail (PTM, appendix B) than indicated by the summary given above (which itself contains invalidating deficiencies); however, most of that detail is trivial, in the sense that it is just application to specific cases of the general methodology given in PTM appendix A for combining lognormal distributions. Addition of that detail is unnecessary and substantially reduces the comprehensibility of the PTM. Moreover, there are several instances (described below) where that detail is incorrect either conceptually (through confusion of uncertainty and time variation) or because the equations are incorrect (apparently because of typographical errors in most cases). I did not examine the implementation of the methodology described in the PTM (in the form of the PAT1 and PAT2 spreadsheets;[6] PTM, p. 3-1) in sufficient detail to comment on that implementation, because of the deficiencies identified here.

It is claimed that: “The analysis results are estimates: engineering approximations based on interpretation and synthesis of information available at this time” (PTM, p. 1-5).

It is further claimed that: “The estimates are objective within common standards of engineering practice and applied science. They are scientifically sound and technically defensible within the limits of available information and adequately support informed risk management decisions” (PTM, p. 1-5) (the same claims are made in PTM, p. C-1).

[6]The committee was provided with copies of these spreadsheets. It is unclear if they were part of any public record until that time. I believe they should have been.

Unfortunately, however, simply stating such claims does not make them true; in this case, they are not true. The analysis presented in the PTM lacks any scientific basis. Four reasons for this conclusion are summarized here: the dependence of the entire analysis on an untested hypothesis; the incorrect treatment of time variation; the use of undocumented, un-validated, and nonreproducible values for parameter values; and incorrect handling of certain probabilistic aspects of the analysis.

THE BASIS OF THE ANALYSIS IS AN UNTESTED HYPOTHESIS

The analysis in the PTM is based entirely on an untested hypothesis for which no theoretical or experimental evidence is presented. The PTM is explicit in admitting that its entire basis is a hypothesis; for example:

> The relative load reduction is hypothesized proportional *on average* to the volume remediated *for a given source type and alternative-specific remedial action*. This hypothesis generalizes the practical *approximation* that the load reduction from a given source and remedial action is proportional to the volume remediated. (PTM, p. 1-14, italics in original)

> It was hypothesized that post-remediation loading reductions *for a given source type and remedial action* (e.g., removal and placement of impacted sediments into a repository) were proportional, on average, to the volume remediated. (PTM, p. 2-29, italics in original)

But there is no attempt to justify the use of this hypothesis *in the context of remedial actions* either by reference to any experimental data or by presentation of plausible theoretical ideas. The statement that the hypothesis "generalizes the practical *approximation*" begs the question, because there is no demonstration of any such practical approximation in the PTM. An attempt is made (PTM section B.2.2, pp. B-20 to B-25) to justify the hypothesis as the "most credible," but that attempt is irrelevant to the hypothesis stated; it addresses a different problem entirely—namely, the time rate of change of loading (which is addressed separately below). The failure to present any evidence for the hypothesis would not necessarily render the claims of the PTM incorrect if the hypothesis were in fact correct or a reasonable approximation. Some theoretical ideas suggest that it is not correct;[7] but the lack of any leaching experiments on any of the materials in

[7]For example, the PTM at (p. B-11) points out that loading from each source will occur as the result of at least four mechanisms: erosion, infiltration of surface water, leaching caused by groundwater fluctuations, and leaching by groundwater flow. Under certain physical conditions these mechanisms could produce loads proportional to source area for the first three examples (erosion, infiltration, groundwater fluctuations) or source linear dimensions for the

the basin, the lack of concentration measurements in groundwater, and the very limited information on groundwater flow deny the information needed to evaluate the hypothesis or propose any more correct one on which to build a plausible analysis.

THE EVALUATION OF TIME IS INCORRECT

Even if the principal hypothesis used in the PTM was correct and the calculation of the immediate postremediation situation was adequately approximated, the treatment of time variation following remediation is incorrect. This treatment is essentially captured in the summary above by Equations (1.4), (1.5), and (1.7). The PTM claims (PTM, section B.2.2.2, starting at p. B-16) that the decay rate β is the same for all times and all remedial scenarios.

Unfortunately, the analysis leading the PTM to such conclusions is incorrect in two ways. (1) The "relative loading potential" (*RLP*) introduced by the PTM is defined to account for the rate of leaching of metal from source material; it does not in any way represent the total mass of metal ultimately available for leaching or erosion. Even the original definition of β (Equation 1.5) thus does not define a decay rate for the available leachable metal. (2) The PTM analysis that purports to show that there exists a constant decay rate, β, is based on (at least) two incorrect assumptions and is itself incorrect.

The Time Scale for Loading or Concentrations Varies with Remedial Option

The first of these problems is easy to detect. The PTM analysis purports to show that the exponential decay rate for annual average loading or concentration is the same for all remedial actions (including no action). Assuming for arguments sake that the loading and concentration do decrease exponentially, it is obvious that the decay rate cannot be the same for different remedial scenarios. Only one remedial option (chemical fixation) has the potential to substantially change the total amount of metal that ultimately could leach or erode down the Coeur d'Alene River (all other options simply reduce the rate of leaching or erosion). Because the expo-

last (groundwater flow). If all sources were the same depth, the first three might be considered proportional to source volume, but the fourth would not. However, under different physical conditions these mechanisms would produce loads that differed in their relationship to source volume. Even if the physical conditions were just right to produce loading proportional to source volume, it does not follow that loading *reduction* is proportional to the reduction in source volume due to remediation, because remedial action may alter the relevant physical conditions.

nential decay rate is just the ratio of the rate of transport down the river to the total mass ultimately available for leaching, reducing the rate of transport (the aim of the remedial actions) necessarily will decrease the decay rate (unless the only remedy applied is chemical fixation). All the available metal ultimately will leach or erode into the river and be carried downstream; if the rate of leaching and erosion is reduced, the time scale over which leaching or erosion occurs is correspondingly increased.

To explain where the fallacy arises in the PTM analysis, recall that the "relative loading potential" (*RLP*) is introduced (PTM pp. 2-17 to 2-18) in an attempt to take account of the differences between various source materials in the combination of metal concentration and mass, its relative mobility, and its exposure to leaching or erosion. Conceptually, therefore, the *RLP* is not proportional to metal mass available for leaching or erosion (that is, conceptually at least, two source types with substantially different average metal masses per unit volume available for leaching or erosion may have identical *RLP* values, and two source types with substantially different *RLP* values may have identical average metal masses per unit volume available for leaching or erosion; in practice, as discussed below, it is unclear how the *RLP* values were derived). For example, the *RLP* for waste rock piles in upland areas may be very low compared with tailings-affected sediments (the PTM, p. C-6, gives an estimate value of 0.001 to 0.005 for upland waste rock, compared with 1 for the reference source, tailings-affected sediments), but that tells us nothing about the relative mass per unit volume ultimately available for leaching in these two source types.

The preremedial total effective mass (TEM') introduced in the PTM (Equation 1.7 and PTM p. B-15, equation 8) is thus conceptually related to loading, and the same goes for the postremedial total effective mass (TEM''), so that in concept it may be adequate to write the loading as proportional to this total effective mass; that is (PTM, p. B-16, equation 1, but see footnote 5)

$$\begin{aligned} W &= \beta TEM' \quad \text{pre-remediation} \\ F(t) &= \beta TEM'' \quad \text{post-remediation} \end{aligned} \tag{1.10}$$

However, even if the preremediation total effective mass (TEM') were somehow to represent the total mass available for leaching or erosion (as could happen in principle if the metal in all sources were present at the same concentration, equally mobile, and equally exposed to leaching or erosion), the same would not be true of the postremedial total effective mass (TEM''), because this is conceptually obtained by incorporating the remedial action effectiveness factors R_{ij}. These factors measure the extent to which remedial actions reduce the loading potential—that is, the rate of leaching or erosion; in principle, they have nothing to do with changing the mass that is available for leaching or erosion. Only one of the potential remedial actions (chemical fixation) is likely to have any substantial effect on the total mass ultimately available for leaching or erosion.

It is therefore incorrect to write (PTM, p. B-16, equation 2)

$$\frac{dTEM''}{dt} = -\beta TEM'' \tag{1.11}$$

The right-hand side represents the loss of metal mass down the river, but the left-hand side bears no relation to the rate of change in total metal mass ultimately available for transport down the river. Equation 1.11 therefore is not a mass balance equation, and all the arguments about mass balance in the PTM (e.g., p. B-18) fail for the same reason—that *TEM′* and *TEM″*, despite the name given to them, have nothing to do with the total metal mass available for leaching or erosion. As a consequence of this failure, the PTM fails to appreciate that the time scale for leaching and erosion will change under different remedial options, and the entire evaluation of the future course of concentrations and loadings is completely incorrect (and even for the unremediated case, the "decay rate" obtained is incorrect).

The Timecourse of Loading or Concentration Is Not Exponential

Another error in the PTM analysis occurs in the assumption that the time course of loading or concentration will be exponential either before or after remediation. There is a long argument given (PTM, pp. B-20 to B-25) that purports to demonstrate that relationships of the form

$$\begin{gathered} F(t) = \beta_n TEM''(t)^n \\ \frac{dTEM''}{dt} = -F(t) = -\beta_n TEM''(t)^n \end{gathered} \tag{1.12}$$

(PTM, p. B-20, equations 12 and 13), with (implicitly) constant n, are sufficiently general to be all that must be examined, and that the value $n = 1$ is the "most credible" (PTM, p. B-24).[8]

However, this argument is based on multiple fallacies, among which are the following:

[8] I have combined equations 12 and 13 (PMT, p. B-20) because this is the *only* way in which $TEM''(t)$ is anywhere defined for arbitrary time t; the definition of $TEM''(0)$ (immediately postremediation) is given in Equation 1.7.

A Belief in the Generality of Equations 1.12

It is stated that "By varying exponent n, the relationship $F(t) = \beta_n TEM''(t)^n$ could allow loading to be any hypothetical yet plausible continuous function of total effective mass" (PTM, p. B-20), and then, after allowing the coefficient β_n to be essentially an arbitrary function of time,[9] "it would tentatively appear that $F(t) = \beta_n TEM''(t)^n$ could approximate the net effect of any plausible theory of geochemical dependence between metal mass and loading." These statements are either trivial (and useless) or meaningless. At time $t = 0$, the first equation with $n \neq 1$ is incorrect by definition of $TEM''(0)$ (see Equation 1.7), because $TEM''(0)$ was explicitly constructed (all its terms were defined) so that the loading (F) was proportional to it. To make any meaningful statements requires definitions that can support some meaningful interpretation, and no such definitions are provided in the PTM; in this sense, alternatively, the PTM argument is trivial (but useless) in that it can mean whatever anybody wishes. Even if the statements were not meaningless or trivial, they would not be correct as used in the arguments, where β_n and n are treated as constants. There are many potential leaching behaviors that cannot be represented by such functional forms (e.g., a constant leaching rate for some period followed by a decline that can be modeled as an error function, as might occur for infiltration of groundwater into a waste pile).

The Fallacy of Equating TEM″ to the Mass of Metal Available for Leaching or Erosion

This was already pointed out. The second of Equations 1.12 has no physical meaning; it is not the mass balance equation that the PTM assumes. Although the right-hand side could represent the rate of loss of mass (if the definition of TEM'' were to be suitably modified at arbitrary times to account for $n \neq 1$), the left-hand side is not the rate of change of mass available for leaching or erosion.

The Fallacy That "n = 1 Is the Only Non-zero Value of n That Yields Physical Reasonable Results That Are Independent of Arbitrary Changes in Loading History" (PTM, p. B-24)

This obtuse phrase is used to represent the (false) conclusion of the PTM (obtained on p. B-24) that the solution of the second of Equations (which is PTM equation 13 on p. B-20) is multivalued ("for any . . . arbitrary time periods such that $t_{p1} < t_{p2} < t_{p3} < \ldots < t_{pX}$ . . . load $F(t_{pX})$

[9]But subsequently (in the argument) it is treated as a constant.

depends on the arbitrary time periods t_{p1} through t_{pX}", PTM, p. B-24).[10] The error in the PTM probably arises from the careless use of notation—the substitution $\beta_n = \beta/TEM''$ on p. B-20 followed by $\beta_n = F_0/TEM''^n$ on p. B-21 apparently without the realization that this makes β a function of F_0. As a result, equation [18] (PTM, p. B-21) would more clearly be written as follows:

$$F(t) = F_0\left(1 + \beta_n^{1/n} F_0^{(n-1)/n} t(n-1)\right)^{n/(1-n)}, \tag{1.13}$$

in which form it is immediately apparent that no such problem arises as imagined in the PTM (p. B-24), and the solution $F(t)$ exists and is single-valued for all finite positive real t and all n (including $n = 1$ and $n = 0$ as limiting cases).

In reality, the time course of loading even from a single uniform homogeneous source need not be exponential. For example, consider the average[11] load due to infiltration of rainwater through an initially uniform waste pile, in which there is sufficient time for the infiltrating water to reach equilibrium with the waste before exiting at the bottom of the pile. In this situation, there may be a long period when the average load is constant as the infiltrating water removes contaminant from the upper part of the waste pile, exiting the waste pile with a constant concentration equal to the equilibrium concentration. The location of the dividing line between leached and unleached waste will travel downward through the waste pile until it reaches the bottom, when there may be a relatively rapid drop in loading from that waste pile (that in some circumstances can be modeled by an error function). Many other situations can easily be envisioned, and the physical situation for erosion, infiltration, groundwater leaching, and other mechanisms may all be different.

It is then obvious that the time course of loading (in particular to the Coeur d'Alene River) can be extremely complex, as it will be the sum of many components from different sources each (potentially) with a different time behavior. For example, in the unlikely event that all sources do exhibit exponential behavior (but with different decay constants), the overall loading to the river will be a weighted sum of many exponentials with those different decay constants. It is plausible that this weighted sum might behave approximately as a power law with time (e.g., consider the case of

[10] Carrying the "argument" of the PTM to its logical conclusion, $F(t)$ is indeterminate for any $t > 0$ unless $n = 1$, contrary to a general theorem on the existence and uniqueness of solutions of differential equations!

[11] The argument given here applies to a time average over periods of over 1 year. Actual loads of course fluctuate on a shorter time scale due to variation in rainfall, pressure, temperature, variation in covering vegetation, and other conditions.

decay afterheat in a nuclear reactor; this is the weighted sum of many exponentials with different decay constants and behaves roughly as a power law, at least over suitably defined intervals), but prediction of such behavior requires evaluation of all the sources separately, and no a priori guess about the behavior is likely to be adequate.

Evaluation of the time dependence of leaching behavior in each source requires some information on leaching behavior of the materials involved and an evaluation of the mechanisms acting on each waste source. None of this information is presented in the PTM, and there is no evidence presented that any such information was considered in the necessary detail.

THE PTM USES UNVALIDATED AND NONREPRODUCIBLE VALUES FOR PARAMETER VALUES

The PTM analysis makes use of quantitative estimates for many input values. The great majority of these estimates do not appear to be based on any empirical evidence (none is presented or cited in the PTM)[12] or on extrapolations from empirical evidence (no such extrapolations are presented or cited) or on theoretical analyses[13] (again, no such analyses are presented or cited). In particular, this is true for

- All the relative loading potential (RLP_j) estimates (PTM, pp. C-5 to C-9).
- All the remedial action effectiveness (R_{ij}) estimates (PTM, pp. C-11 to C-14).

There is no semblance of objectivity for these estimates; indeed, in most cases it is impossible to discern their origin. The descriptions of how the estimates were obtained are entirely qualitative; indeed, it is even claimed that the analyses performed were almost entirely qualitative, yet at the end a number somehow appears.

To illustrate, it is claimed in the main text (PTM, p. 2-21) that

> RLPs were estimated from interpretation of available information, including consideration of metal concentrations, mobility, and exposure to leaching and erosion, analysis of simple loading models, and professional judgment. The uncertainty in the estimates was handled probabilistically by characterizing the RLP estimates using an expected value and coefficient

[12]In a single case there is one reference to one experiment, but using simulated groundwater.

[13]The treatment of adits is based on a theoretical analysis that shoehorns them into the structure of the model, but adit loading is subsequently assumed to behave exactly as any other.

> of variation and assuming that the uncertainty in estimates followed a lognormal distribution.

However, no data on mobility, exposure to leaching, or erosion are presented or summarized. The reader is given no information on what "simple loading models" were considered or how they were considered. It is not stated whose professional judgment was sought, what was the connection of those professionals with this site, what their professional judgment was based on if not on the preceding information, or what extrapolations from other situations were used by those professionals in obtaining the values presented.

Similarly, in discussing the remedial action effectiveness estimates, the main text (PTM, pp. 2-27 to 2-28) states:

> For each alternative, effectiveness estimates were based on an assessment of each remedial action and an engineering interpretation of the range of potential effectiveness, as documented in Appendix C. Estimates for Alternatives 2, 3, and 4 were based on engineering interpretation of the range of potential effectiveness for the typical conceptual designs (TCDs) used in the alternatives, as documented in the FS. These interpretations used qualitative engineering analysis, limited quantitative performance modeling, experience with similar remedial actions, and professional judgment. Professional judgment was used to set context and frame the interpretations, determine what questions to ask, and synthesize information to make the estimates. Experience with similar remedial actions generally considered how well actions have performed in the past, and included considerations inherent in the technology screening documented in FS Section 3. Qualitative engineering analysis was based on knowledge of scientific and engineering principles and construction limitations and used to consider how effective the TCDs are likely to be for the potential range of site-specific conditions. The analyses were qualitative except for HELP analyses used to evaluate potential cover performance in terms of infiltration and percolation. (Ridolfi 2000 as cited in in PTM, p. 4-2)

However, Appendix C, to which the reader is directed, contains no "engineering interpretations," "performance modeling," or documentation of any of the other approaches mentioned. There is no documentation of the "contexts" and "evaluations." There are no references to measurements that document "experience with similar remedial actions," or even any mention of which such actions are considered similar. There is no information on the HELP analyses that were performed.

Again, the information required to make objective estimates for most of these input values does not exist, primarily because of the lack of any leaching experiments for any materials in the basin and the very limited information on groundwater flow and metal concentrations in groundwater.

The PTM gives no indication of how the RLP_j and R_{ij} values were obtained in any way that would allow reproduction or challenge of their values; indeed, it is unclear how any reader could determine a preference for the sets of values given in the PTM over almost any other set of plausible but arbitrary values. It is claimed that (PTM, p. A-31): "Professional judgments and interpretations are documented and quantified, as scientifically and practically appropriate."

However, the complete lack of documentation on such judgments and interpretations prohibits their evaluation. If expert judgment is to be used in a situation like this, there are documented procedures for debriefing those experts in such a manner that the basis of the final estimates can be tracked and reproduced (e.g., Kaplan 1992). The procedures require the experts to state a basis for extrapolation to the situation in hand and to justify the models and heuristics that should be applied to that basis to make the extrapolation. The justifications for the basis and for the extrapolation methods, and the extrapolation itself, are then documented and the extrapolation is performed by others (e.g., risk assessors who are not the experts). There may be one or more rounds of feedback in which the experts examine the results and modify (for stated and documented reasons) the proposed bases and extrapolations (and, of course, correct any errors in documentation). With such documentation, one could be reasonably confident in knowing where values come from and have a basis to challenge their reliability; without it, the values might as well have come from a (biased) random number generator.

THE PTM HANDLES VARIOUS PROBABILISTIC ASPECTS OF THE ANALYSIS INCORRECTLY

At various points, the PTM confuses the time variation (fluctuations) of some physical quantity (such as loading in the Coeur d'Alene River) with the uncertainty in some physical quantity (such as estimates of the remediation factor). This confusion appears to extend to the most basic level. Specifically (PTM, p. A-2), it is claimed:

> Natural variability is the combination of two effects: (1) the practically irreducible uncertainty due to our limited quantitative and predictive knowledge of the fundamental physical mechanisms and interactions underlying the phenomenon of interest, and (2) the fundamentally probabilistic nature of the phenomenon itself. In principle, advancements in fundamental knowledge could reduce the first effect, at a cost, but not the second. From a practical standpoint, natural variability can be considered "intrinsic, fundamental, irreducible" uncertainty, reflecting the inexactitude of available knowledge.

Including item 1 in this list as natural variablity is incorrect; item 1 describes uncertainty, not natural variability. Only item 2 corresponds to natural variability, and nothing involving our knowledge of it will change it.[14] In the case of loading or stream flow, for example, the natural variability is represented in the PTM by a probability distribution representing the fluctuations that occur from time to time in load or flow.

The implication is that this distribution would be obtained by accurate measurements made at random times. Improvements in knowledge would certainly allow changes in our ability to predict stream flow or loading at particular times, but no improvement in our knowledge will change this probability distribution. Improvements in measurement also might allow us to estimate the parameters of the stream flow or loading distribution more accurately, but that has no effect on the distribution that is being measured.

The failure to distinguish time variation and uncertainty extends to the metric that the PTM is attempting to evaluate. This metric is never explicitly or precisely defined. It appears to be some measure of the uncertainty distribution for AWQCs to be exceeded. The conflation of time variation and uncertainty in the PTM implies that the PTM attempts to obtain the uncertainty distribution for the ratio of water concentration to AWQC *at a random time*. Other metrics may be of greater interest to the regulator, however, although there is no discussion of any other metrics. For example, for fishery conservation it may be of more interest to know the uncertainty distribution for the average ratio of water concentration to AWQC during different seasons, or the uncertainty distribution for the expected period during a given season that the ratio of water concentration to AWQC exceeds a given value or for the expected time intervals between such exceedances.

I list below a few instances in which time variability and uncertainty have been confused in such a way as to affect the analysis of the PTM. In this discussion, I interpret the PTM as attempting to evaluate the uncertainty distribution for the ratio of water concentration to AWQC at a random time, because no other interpretation of the PTM appears to be possible.

An Attempt to Draw Conclusions About Distribution Shapes

It is stated that

> Because the underlying phenomena leading to lognormality will not be changed by remedial action, it is expected that post-remediation loading

[14]The only exception to this statement occurs in quantum systems where an observer is strictly part of the system, and observer knowledge about the system represents part of the state of the whole system. However, at that level of detail the "natural variability" of the system is actually a quantum uncertainty that is fundamental to all physical systems.

> will also be lognormally distributed. An important implication of both pre- and post-remediation loading being lognormal is that the effects of remedial action should also be lognormal (because products and quotients of lognormal distributions are also lognormal. . . . (PTM, p. 2-10)

However, the claim is a complete nonsequitur and simply incorrect. The lognormality of the distribution representing *variability in time* of concentrations, stream flows, loadings, or other physical quantities has nothing to do with the shape of the *uncertainty* distribution for the effects of remedial actions. All the probability distributions for remedial actions presented in the PTM are uncertainty distributions (the PTM is not explicit, but no other interpretation is plausible). If, by some chance, the *variability in time* is what is contemplated in the PTM for one or more of the distributions given for the remedial actions, then there is no implication. In that case, the remedial actions are presumably consistent with the "underlying phenomena" (whatever those are supposed to be). Moreover, as stated elsewhere (PTM, p. A-14),

> In addition, although theoretically, the sum of independent lognormal distributions is not lognormal, it can be demonstrated by simulation that the sum closely approximates a lognormal PDF. Therefore, the sum of independent lognormal distributions can also be *approximated* as lognormal.

Thus, the analysis is based on approximations anyway; so one might as well admit from the start that it is approximate, the same approximations would apply to the remedial actions, and no such conclusion can be drawn about any distribution for remedial actions.

Erroneously Implying a Correlation

It is concluded that there is some correlation between L and $R(t)$ (PTM pp. B-37 to B-38):

> Estimates of the correlation between L and R(t) (as measured by $p_{\ln L,\ln R}$) were based on professional judgment and interpretation of potential remedial action behavior. Although there is no practical way to quantitatively predict the correlation, it is expected that remedial action will generally be relatively more effective at reducing high loadings (which correlate with high flow conditions) than reducing low loadings (which correlate with low flow conditions) such that L and R(t) will be negatively correlated. The midrange value of $p_{\ln L,\ln R} = -0.5$ was considered reasonable.

Apart from the total lack of basis for any particular numerical value, as explicitly admitted, the whole concept of this correlation is erroneous. L is the loading, with a distribution arising from its *variability in time*, particularly its variability during the year. $R(t)$ is explicitly defined to be a time average over a year (PTM, p. B-4); there can be no correlation on this basis alone.[15] More to the point, the distribution associated with $R(t)$ is an uncertainty distribution, with nothing whatever to do with variability in time, so the concept of correlation does not even apply. What has been done in the PTM is to cancel out (by applying a negative correlation) some of the *uncertainty* in $R(t)$ with the *time variability* of L! The "correlation" that is described in the cited paragraph is more accurately a claim that there is a functional relationship between the parameters of the distribution representing the time variability of L and the actual value of $R(t)$—specifically, that the upper end of the distribution of L is modified by the value of $R(t)$. A potential way of modeling such an effect would be to treat the standard deviation of the distribution of L as a function of $R(t)$. In this case, however, there is no basis provided that the claim is accurate and that "remedial action will generally be relatively more effective at reducing high loadings (which correlate with high flow conditions) than reducing low loadings (which correlate with low flow conditions)." Whether this claim is true depends on details about leaching and erosion from each source, details that are not documented or (apparently) even examined in the PTM in reaching its conclusion.

An Attempt to Estimate the Wrong Correlation

The load ratio is defined by Equation 1.8 above (PTM, p. B-53, equation 1); that is,

$$Lr(t) = F(t) / C_L, \tag{1.14}$$

and the metal loading $F(t)$ is given by Equation 1.2 above (PTM, p. B-4, equation 1); that is,

$$F(t) = R(t)L, \tag{1.15}$$

so that

$$Lr(t) = R(t)L / C_L. \tag{1.16}$$

Both L, the preremedial loading, and C_L, the loading capacity, vary with time throughout the year, whereas $R(t)$ is defined to be a yearly average.

[15] I discount as too unlikely the possibility that the PTM was implying a correlation between the uncertainty distributions for the parameters of the (current) loading and the (future) remedial effectiveness.

Associated with $R(t)$ is an uncertainty distribution but no unpredictable time variability ($R(t)$ varies with time, but smoothly and in a predictable fashion), whereas the distributions associated with both L and C_L are due to their (unpredictable) time variation (strictly, there are also uncertainty distributions associated with the parameters of the distributions describing their time variation, because of finite numbers of measurements, but these are ignored here, just as they are ignored in the PTM). There is a very high correlation between measured values for L and C_L (the correlation coefficient between their logarithms is approximately 0.95)[16] but none between $R(t)$ and L or C_L (as discussed for L; the same arguments apply to C_L as to L).

The PTM (p. B-53, equations 1, 2, and 3), however, obtains the uncertainty distribution for $F(t)$ at a random time within about a year of t by combining the time-variability distribution for L with the uncertainty distribution for $R(t)$. It then attempts to argue about the correlation between the resulting uncertainty distribution and the time-variability distribution for C_L based on the correlation between L and C_L. It states (pp. B-53 to B-54):

> The future correlation between lnF(t) and $\ln C_L$, measured by $p_{\ln F, \ln CL}$, is expected to be very high. This expectation is based on an almost perfect correlation (p = 1.0) between $\ln C_L$ and lnQ and a virtually certain high future correlation between lnF(t) and lnQ, just as there has been historically between discharge and loading (which, being a function of discharge, induces correlation). In addition, as further discussed in Section B.3.4.1, and independent statistical analysis of the zinc concentrations, water hardnesses, and discharge data corresponding to that used in developing the TMDL loading capacities for SF271 (EPA 2000) showed a correlation coefficient of 0.95 between the natural logs of zinc loadings (computed as the product of concentration and discharge) and the equivalent loading capacities (computed as the product of the zinc AWQC(H) and discharge). Consistent with this information, a value of $p_{\ln F, \ln CL} = 0.9$ was used in the analysis.

There is no basis for the selection of the particular value 0.9. It is not possible to state whether it is "consistent with this information" without further examination, but in general it is not consistent with that information. The effect of assuming a high correlation between $\ln(F(t))$ and $\ln(C_L)$ is to substantially cancel the *uncertainty* in $R(t)$ with the *time variability* in C_L; but this cancellation is purely fictitious. This error compounds the

[16] C_L is measured by measuring the hardness of the water and the flow rate simultaneously, computing the *AWQC* from the hardness, and forming the product of *AWQC* and flow rate. L is measured by measuring the metal concentration and flow rate and forming the product.

previous erroneous cancellation of the uncertainty of $R(t)$ by the time variability of L discussed above.

The effect of these two incorrect cancellations can be large. This may be illustrated by supposing that what is required is the uncertainty distribution for $Lr(t)$ at a random time, so that it is legitimate to (correctly) combine the uncertainty of $R(t)$ with the time variability of the ratio L/C_L in Equation 1.16. For dissolved zinc at location SF271 on the Coeur d'Alene, the measured standard deviation of (the time variability of) $\ln(L)$ is 0.525, that of $\ln(C_L)$ is 0.643, and that of the logarithm of their ratio, $\ln(L/C_L)$, is 0.225[17] (obtained from the joint measurements of concentration, hardness, and flow rate; [EPA 2000, for hardness and flow measurements; URS Greiner Inc. and CH2M Hill Inc. 2001b, for dissolved zinc and flow measurements]).[18]

With these measured standard deviations for $\ln(L)$ and $\ln(C_L)$, Table F-1 shows the correct calculation of the random-time uncertainty for $\ln(Lr(t))$ compared with that obtained by including the two erroneous correlations introduced in the PTM for various values of the standard deviation of $\ln(R(t))$. The error introduced is clearly substantial for any plausible estimates for uncertainty in $\ln(R(t))$.

INCORRECT OR MISLEADING STATEMENTS IN THE PTM

The following is an incomplete sampling of various incorrect or misleading statements and equations in the PTM. Attempting to list all such erroneous statements and equations would be too time-consuming, so the failure to list any statement or equation in this list cannot be considered an endorsement of the correctness of any statement or equation not listed here.

- The term "power series" is used incorrectly throughout Appendix A. Where "power series" is used, the correct term would be something like "power product." The discussion is not of power series in one or more random variable, but the product of powers of random variables.
- "Minimum statistical assumptions are required" (PTM, p. A-13). No basis is provided for this statement. One can assume anything, but that does not make it correct, or even consistent, or useful.

[17]The distributions for $\ln(L)$ and $\ln(C_L)$ are not distinguishable from normal ($p = 0.42$, 0.44, respectively, Shapiro-Wilk test). The distribution for L/C_L is closer to normal than lognormal ($p = 0.39$, 0.09, respectively; Shapiro-Wilk test) using the available data.

[18]Measurements taken on the same day were assumed to be simultaneous, and multiple measurements on the same day were averaged for the analysis. Only the subset of data with simultaneous hardness, flow rate, and concentration data are included in the statistics given here.

TABLE F-1 Effect of the Two Erroneous Correlation Calculations Introduced in the PTM

	Standard Deviation of $\ln(Lr(t))$	
Standard deviation of $\ln(R(t))$	Correct	PTM
0.0	0.225	0.285
0.3	0.375	0.306
0.6	0.641	0.280
0.9	0.928	0.347
1.2	1.221	0.541
1.5	1.517	0.791

- "A lognormal PDF is believed to be a *maximum entropy* PDF for the log of variables where only the expected value and coefficient of variation of the distribution is known or estimated. Maximum entropy estimates give the 'least prejudiced, or least biased, assignment of probabilities'" (Harr 1987) (PTM, p. A-13, footnote 9, italics in original). No connection is proposed between "minimum statistical assumptions" and "maximum entropy." Nor is any application to the problem at hand proposed; on what basis, for example, is it supposed that only the expected value and coefficient of variation are known for the log of variables, and how does this connect, for example, with the evaluation of probability to exceed the AWQC?
- "a correlation coefficient of –1.0 implies perfect inverse linear correlation (i.e., X_1 and X_2 are inversely proportional)" (PTM, p. A-8). This is incorrect; perhaps what was intended is that if the correlation coefficient between logarithms $\ln(X_1)$ and $\ln(X_2)$ is –1.0 then X_1 and X_2 are inversely related (but not necessarily in direct inverse proportions).
- "Unbounded positive values are allowed (which is generally conservative because it tends to overestimate true values)" (PTM, p. A-13). It does not follow that lognormal distributions lead to "generally" conservative estimates, without specifying the universe of discourse. For example, if some variable is (erroneously) assigned a lognormal distribution, and that variable occurs in the denominator of an expression, the result may be an underestimate rather than an overestimate. On the other hand, the inverse of a lognormal distribution is also lognormal, so the preceding example also shows that (erroneously) assigning a lognormal distribution to an expression in the numerator can lead to underestimates—because a lognormal distribution also allows unboundedly small values.
- "Any PDF can be conservatively approximated using a lognormal PDF that envelopes the PDF over the range of interest" (PTM, p. A-13). Again, this statement is meaningless without a definition of "conservatively," "envelopes," and "range of interest" at the least. Even with such

definitions, it is likely to be untrue in general. Indeed, it is quite likely that a converse theorem holds—for any lognormal approximation to a given PDF, there exist statistics of that PDF that are not conservatively estimated by the lognormal approximation.

- "Variables C_{DS} and C_S are, respectively, the metal (zinc) concentration of the deeper sediments and floodplain sediments having RLP = 1. These sediment concentrations will be positively correlated" (PTM, p. B-47). "Also, because of the way C_{DS} and C_S were estimated, they would be positively correlated" (PTM, p. C-9). It is quite plausible that the concentrations of deeper sediments and floodplain sediments are correlated spatially—that is, the concentration would tend to be higher in the deeper sediments beneath floodplain sediments with higher concentrations. Such a spatial correlation is entirely irrelevant, however, for variables C_{DS} and C_S, which are defined to be "volumetric average concentrations in the deeper impacted sediments" and "volumetric average concentration in the impacted sediments having an *RLP* = 1" (PTM, p. B-14). Any spatial correlation is entirely removed by the averaging. What is required is any correlation between the *uncertainty* distributions for these volumetric averages. No such correlation is induced "because of the way C_{DS} and C_S were estimated." The only documented "estimation methods" are given in section C.2.4, where uncertainty confidence intervals for the values of C_{DS} and C_S are supposedly (very loosely) based on observed data in the BHSS and a background estimate based on measurements outside the BHSS. Nothing in the measurements supposedly used or in the described derivation correlates these uncertainty distributions; the fact that the same value is used as the lower uncertainty confidence bound for one and the upper uncertainty confidence bound for the other is the only connection between them, *and that has no such effect*. The subsequent estimate of a value of 0.5 for the correlation coefficient of this hypothetical, nonexistent correlation is simply incorrect.
- "An estimate of CV[M] = 0.5 was used in the analysis" (PTM, p. B-48). There is no basis given for this estimate. Nor is it clear why it was introduced, except to arbitrarily increase the uncertainty estimate.
- "Since the estimates for *L* and TEM′ are independent of each other, $\rho_{\ln L, \ln TEM'}$ was set to zero in the analysis. This lack of correlation in the *estimates* should not be confused with the positive correlation that must exist in the *true* values of *L* and TEM′, and is otherwise inherent in the data used to make the estimates. To the extent there was (positive) correlation between the estimates of L and TEM′, it would decrease both E[β] and CV[β]" (PTM, p. B-48). This statement demonstrates complete confusion, apparently stemming from a misunderstanding of what "correlation" means or perhaps the confusion in this document between measurement uncertainties, variability in time, and functional relationships. There is obviously

no correlation possible between *true* values of L and TEM′,[19] which are single values.

- "The BHSS data do not represent the true values of C_{DS} and C_S, which are uncertain" (PTM, p. C-10). True values cannot be uncertain, although they may be unknown, so that we are uncertain about what they are.
- "For example, the correlation coefficient between the natural logs of Q and H is 0.96 for the SFCDR at SF271 . . . the correlation coefficient between the natural logs of AWQC and Q at SF271 is also 0.96 for the TMDL data set" (PTM, p. B-26). Both these correlation coefficients are –0.96, not +0.96.
- Page 1-1, footnote 1, the conversion factor is actually 0.005394 to 4 significant figures, or 0.00539 to 3 significant figures. The value used should at least be the correct rounding of the exact value.
- Page A-24, equation [2] is incorrect. The correct expression is

$$E(X) = \left\{ \prod_{i=1}^{n} E[X_i]^{a_i} \left(1 + CV[X_i]^2\right)^{a_i(a_i-1)/2} \right\} \Omega, \tag{1.17}$$

and there is no need to introduce the variables X_i'. Indeed, the entire exposition would be greatly clarified by working with statistics of the logarithms of the variables. For example, define $T_i = \ln(X_i)$, $T = \ln(X)$, and let X_i have mean m_i and coefficient of variation c_i, T_i have mean μ_i and standard deviation σ_i, and similarly for X and T (with no subscripts). Then we have

$$\begin{aligned} m &= \exp(\mu + \sigma^2 / 2) \\ c^2 &= \exp(\sigma^2) - 1, \end{aligned} \tag{1.18}$$

and similarly for all subscripted variables. Then equations 1 through 3 of PTM (p. A-24) become the considerably simpler equations:

$$\begin{aligned} \mu &= \sum_{i=1}^{n} a_i \mu_i \\ \sigma^2 &= \sum_{i=1}^{n} a_i^2 \sigma_i^2 + 2 \sum_{i<j} \rho_{ij} a_i a_j \sigma_i \sigma_j , \end{aligned} \tag{1.19}$$

where ρ_{ij} is the correlation coefficient between T_i and T_j, and it is trivial to move between statistics for variables and their logarithms using Equations 1.18.

- Page B-10, the second equation for L_j/V_j in the middle of the page is incorrect; that is,

[19]The term L here corresponds to our W.

$$L_j / V_j = \sum_{s=1}^{S} L_{js} \bigg/ \sum_{s=1}^{S} V_{js} \neq S \bigg/ \sum_{s=1}^{S} (L_{js} / V_{js})^{-1}, \tag{1.20}$$

and the inequality applies except in certain special cases (which do not apply in general in this application).

- Page B-12 and C-5, the last two entries in equation 4 of p. B-12, and the same equations repeated in section C.2.3 for RLP_j are incorrect if any attempt is made to interpret them according to standard conventions. The first and second entries of equation 4 of p. B-12 correspond to the definitions given. It is just possible to interpret the last two entries in equation 4 of p. B-12 and the same equations in section C.2.3 in the correct sense if the phrases "per unit volume of source type j and FP" and "per unit load of source type j and FP" are interpreted as applying separately to the numerators and denominators of the respective equations, contrary to any standard convention; coming on these equations by themselves (without the correct definition) in section C.2.3 is disconcerting.
- Page B-26, "The analysis used the same discharge and H(Q) and AWQC(Q) relationship used in EPA 2000 for the TMDL." This statement is incorrect, and the approach taken in the PTM is inconsistent with the intent of performing an uncertainty analysis. First, the statement is incorrect because the relationship assumed in EPA (2000) was linear between hardness itself and the logarithm of flow rate, whereas the relationship assumed in the PTM is linear between the logarithm of hardness and the logarithm of flow rate. Second, the approach taken in the PTM is inconsistent, because the PTM analyzed the loading capacities derived for regulatory purposes in EPA (2000). However, those loading capacities already have built into them the results of an uncertainty analysis; the loading capacities are derived as 90th percentiles of an uncertainty distribution.[20] That uncertainty analysis should be incorporated in the PTM as part of the overall uncertainty analysis—the PTM should evaluate the original data, not the summary statistics produced by EPA (2000).
- Pages A-18 to A-19 and B-33 to B-34, the technique used to estimate parameters (mean and standard deviations of the logarithm) of lognormal distributions by regressing order statistics of the logarithms of measurements against the "plotting points" (p. A-18, equation 8, and p. B-34, equation 1 has nothing to recommend it. The "plotting points" used are

[20]EPA (2000) states (p. 22) that the values were lower bounds of a 90th percentile confidence interval, thus at the 95th percentile. This is incorrect, however. The values obtained by EPA (2000) are the 90th percentiles (lower bounds of an 80th percentile confidence interval).

only approximations of the expected values of the normal order statistics, so the technique is approximate at best (better approximations of normal order statistics are available (Royston 1993, 1995). The values obtained for mean and standard deviation are almost certainly biased and have unknown statistical properties. On the other hand, simply computing the mean and (sample) standard deviation of the logarithms of measured values gives unbiased estimates with known (and optimal for certain purposes) statistical properties for these parameters. Unless the PTM justifies the methodology used (by demonstrating, for example, superiority in some sense of the estimates obtained), standard (and simpler) approaches should be used.

- Page B-35, equation 6, the right-hand side erroneously uses CV[L] where what is required is CV[C]. The expression for Ω erroneously omits p.
- Page B-35, equation 8, the right-hand side erroneously uses CV[L] where what is required is CV[C].
- Page B-35, footnote 17, the expressions could be somewhat simplified if the trivial identity

$$\{\exp(A)\}^{1/2} = \exp(A/2). \tag{1.21}$$

were applied. Better yet would be adoption of the suggestion discussed in the comment on p. A-24.

- Page B-36, first equation on page (carried over from equation 10 of p. B-35), the expression for Ω erroneously omits p.
- Page B-37, equation 3, the expression for Ω erroneously omits p.
- Page B-43, equations 6 and 7, in both these equations the denominators have been written incorrectly, because

$$\left(E\left(\sum_{j=1}^{K}\sum_{i=1}^{N_j} RLPj * Vij\right)\right)^2 \neq \sum_{j=1}^{K}\sum_{i=1}^{N_j} E(RLPj * Vij)^2. \tag{1.22}$$

The left side of Equation 1.22 is what is required inside the square root in the denominator of equations 6 and 7, but the right side is what is written.

- Page B-57, "The analysis showed the following principal results: Both the AWQC(H) and the equivalent loading capacities were lognormally distributed with respective r^2's of 0.94 and 0.97." The list continues with similar statements about ratios of zinc loadings to loading capacities, zinc concentrations, loadings, and hardness. However, the given information is not sufficient to support the conclusion of lognormality for these quantities—some values of r^2 would be obtained whether or not any particular distribution was lognormal. It is quite feasible to test whether a set of samples is consistent with lognormality—for example, by using the Shapiro-Wilk test (Royston 1982, 1993, 1995). Applying this test suggests that it is somewhat unlikely that the measured zinc concentrations ($p = 0.002$), AWQC(H) ($p = 0.014$), or hardness ($p = 0.018$) are lognormal, although zinc loading measurements ($p = 0.3$) and loading capacity ($p = 0.5$) are consistent with lognormality.[21] It is already pointed out in footnote 17 that the ratio of zinc load to the load capacity is more consistent with normality than lognormality.

- Page B-52 (section B.3.3.3), "For these reasons and because of its general theoretical and practical basis, Eq 1 was considered a valid and reasonable approximation for estimating CV[R(t)] for the lower basin, with further savings of effort." But there is no theoretical basis whatever for equation 1, because it is purely an empirical approximation found for the upper basin using the specific values for the upper basin.[22] Therefore, there is no basis whatever for extending this empirical approximation to the lower basin (with different source types, different mixes of sources, and so forth)—the results obtained there could be substantially different.

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[21]These statistics are obtained using all the available data, in the same manner as apparently used in the PTM. Footnote 17 gives similar statistics for the subset of these data for which all of hardness, zinc concentration, and flow were simultaneously measured.

[22]It is likely to be substantially incorrect, as indicated by the other comments made here, but that is irrelevant to the current argument.

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