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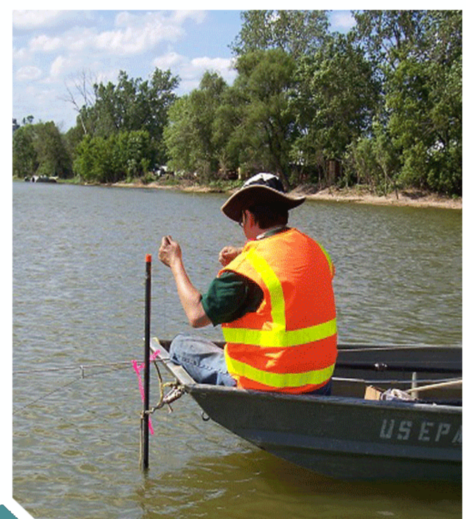
Linking Physical, Biological, and Social Sciences in Natural Resources: An Ecosystem Service Framework for Monetary Valuation of Environmental Impacts Related to Mining in Central Colorado



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Center for Environmental Solutions and Emergency Response
Land Remediation and Technology Division

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Foreword

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Abstract

Remediation of historical mine sites with a legacy of environmental impairment typically aims to restore water, soil, and sediment quality to levels that comply with relevant regulatory guidelines. The achievement of compliance goals improves surrounding ecosystems, which can then provide services to humans such as improved water quality for drinking, improved water quality for fish populations that are attractive to anglers, and desirable vistas, among others. Although the costs of varying degrees of remediation are clearly estimated, the value of ecosystem service improvements is not explicitly considered in remedial planning. The purpose of this study is to evaluate approaches for valuing ecosystem services affected by mine remediation and predicting the growth of those post-remediation benefits. The literature was surveyed to identify studies that value ecosystem service endpoints that can be linked to geo-environmental models of mine pollution remediation. Benefit valuation applicable to mine site pollution were identified for aquatic habitat, drinking water, groundwater, water supply reliability, lead contamination, air particulate matter, mercury emissions, residential views, natural land cover, and fish populations suitable for recreational angling (fish large enough to catch). To demonstrate the valuation of ecosystem services affected by mine remediation, this exercise is applied to catchable populations of brown trout in ten sample locations at two legacy mine sites on the National Priorities List—specifically the Leadville district (California Gulch Superfund site) in the Arkansas River watershed and the Gilman district (Eagle Mine Superfund site) in the Eagle River watershed. The impact of dissolved metals on water quality, aquatic macroinvertebrates, brown trout populations and their growth are modeled to allow the valuation of catchable brown trout populations. This application, combined with the identification of studies that value ecosystem services affected by mine pollution, outlines a framework for valuing changes in mine site pollution for future research where site data are more readily available. While this study evaluates the benefits of remediating mine site pollution, this framework may also be applied to an increase in mine site pollution if site data are sufficient to allow the linkage of geo-environmental modeling and ecosystem service endpoints.

Executive Summary

Historical mining commonly left a legacy of environmental impairment that affected surface water, groundwater, soil, sediment, and associated ecosystems. Remediation of legacy mine sites typically aims to restore water, soil, and sediment compositions to concentrations that comply with relevant regulatory guidelines. A consequence of meeting compliance goals around legacy mine sites is the restoration of a variety of ecosystem services, such as improved water quality for drinking, improved water quality for fish populations that are attractive to anglers, and desirable vistas. Despite the positive effects that remediation can have on ecosystem services, the value of those improvements is not an explicit goal of remedial planning. The purposes of this study are to evaluate approaches for the valuation of ecosystem services affected by mine remediation, and to explore approaches for predicting the growth of those post-remediation benefits.

Legacy mine sites on the National Priorities List were the target of this exercise because of the comprehensive and prescribed approach required to characterize environmental risks and ecological damages. Four different mineral deposit types, with two sites each, were initially selected for investigation, but inability to access and acquire data and reports from these Superfund sites resulted in limiting this exercise to the carbonate-hosted lead-zinc-silver deposits in central Colorado, specifically the Leadville district (California Gulch Superfund site) in the Arkansas River watershed, and the Gilman district (Eagle Mine Superfund site) in the Eagle River watershed. The primary data used for both areas are from reports by state and federal agencies. Within these two sites, data limitations constrained the ecosystem services scope of the study to recreational angling.

The deposits and mines in both districts experienced similar production histories, which resulted in similar environmental legacies. They were all mined by underground methods, the ore was crushed, sulfide concentrates were produced by flotation techniques, and mill tailings were disposed on the surface. The Leadville district has the additional feature of tunnels that were driven into the mountain side to facilitate easy removal of ore from the mine workings and to drain the mine workings. Drainage from the tunnels is currently being addressed by active water treatment plants. The remaining ecological and water-quality issues have largely been addressed through source control – removal and disposal of solid mine waste. The water-quality issues resulted from acid-mine drainage mixing with slightly alkaline river water. The mixing effectively removed iron by precipitation of hydrated ferric oxides. The low solubility of lead sulfate (anglesite) and lead carbonate (cerussite) and sorption on to the hydrated ferric oxides effectively limited dissolved concentrations of lead. Likewise, a significant amount of the copper was removed by sorption. Downstream exceedances of water quality criteria for the protection of aquatic life in both the Arkansas River and Eagle River watersheds have been dominated by zinc followed by cadmium and copper. The contribution of each element to the toxicity of the water is zinc: 68 to 80%, cadmium: 10 to 20%, and copper: 10%. Downstream reductions in the concentrations of these elements are due to dilution by groundwater and surface water influxes. The bulk of remedial activities was completed in the Arkansas River watershed in 2000 and in the Eagle River watershed in 2001, although minor environmental problems remain to be addressed.

The improvements in water quality in both watersheds due to remediation have resulted in steady improvement in the populations of aquatic macroinvertebrates and fish, specifically brown trout. The Upper Arkansas River fishery achieved "Gold Medal" status from Colorado Parks and Wildlife in 2014. Gold Medal status is reserved for waters with a combination of high population density and at least 12 trout 14-in. (or longer) per acre. This can be achieved either by natural populations or fisheries management. The growth in trout populations can be modeled using the logistic function that uses a starting population, a growth rate, and a carrying capacity as inputs. These input parameters were all estimated based on published trout population data for these watersheds. More sophisticated population dynamic models are available in the literature, but all require a larger set of input data that is not currently available for the Arkansas and Eagle River watersheds. Nevertheless, published studies that assessed the temporal variations in the coefficient of variation for abundance of brown trout in healthy streams in Colorado ranged between 15 and 82%, compared to an average for all trout species in the United States of 49%. This large range in the coefficient of variation for temporal variations suggests that the logistic function is adequate for predicting trout population growth in this study. In general, the logistic function adequately describes population growth in these two watersheds. Specific sites that fall significantly below the model typically have residual environmental issues, such as fluvial tailings, untreated tunnel drainage, or seepage of leachate from waste piles that represent ongoing sources of contamination.

For an ecosystem service valuation of recreational angling related to brown trout population at these two sites, the relevant ecosystem service endpoint (which links the natural and social science models) is the portion of the brown trout population that is large enough to be caught (hereafter referred to as "catchable"). The goal is to estimate the change in net economic value resulting from the increase in catchable trout population for its initial level to its carrying capacity. The ecosystem service valuation literature provides two approaches for estimating the change in net economic value to recreational anglers. The first approach directly values the catchable fish population by combining the model fish population estimates with the estimated value per catchable trout. The second approach multiplies a value estimate (for the number of fish caught by recreational anglers) by the ratio of fish caught and catchable fish – which also estimates the change in net economic value per catchable fish.

Linking biological and valuation models to determine the ecosystem service valuation of catchable trout population was only successful at 10 sampling sites at locations in the Upper Arkansas River and Eagle River. To provide a broader context, for the change in net economic value due to the California Gulch Superfund remediation, an ecosystem service valuation was conducted (without the benefit of geo-environmental modeling) along the impacted stretch of the Upper Arkansas River. Due to data limitations, this ecosystem service valuation focused on the use value of fish, i.e., those caught by recreational anglers.

Finally, outside of the specific study sites, the environmental and ecosystem service valuation literature was widely surveyed to assess the literature's ability to value impacts of mine site pollution through an ecosystem service framework. Catchable target fish populations are explicitly modeled while guidance for valuation of ecosystem services related to aquatic habitat, drinking water, groundwater, water supply reliability, lead contamination, air particulate matter, mercury emissions, residential views, and natural land cover are also provided to enable future research where site data prove to be more forthcoming.

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Introduction

Extractive industries, especially the mining of mineral resources, have historically played important roles in the economic development of the United States. Many parts of the United States owe their existence and growth to mining. The California Gold Rush in 1849 and the Colorado Silver Boom of 1879 advanced the development of those regions. This development led to the sustainable economies found today, even though metal mining has essentially ceased in the areas of initial focus. The development included not only the mines themselves, but also the physical, commercial, and social infrastructure to support the mines including, roads, railroads, energy supplies, stores, schools, hospitals, and other emergency services. However, the environmental legacy of the mines, particularly that of historical mines, can also affect the long-term economic benefits derived from ecosystems, otherwise known as ecosystem services, in the surrounding regions.

Defining ecosystem services and their endpoints

To evaluate the effect of the environmental legacy of mines on long-term economic benefits derived from surrounding ecosystems, the links between the natural and social sciences must be explored. The field of ecology breaks an ecosystem into *processes* and *services*. *Ecosystem processes* are the complex physical and biological interactions that underlie the natural world, such as nutrient cycling, regulation of water chemistry, and maintenance of biological diversity. By contrast, an *ecosystem service* is the result of ecosystem processes. Ecosystem services sustain or enhance human life.

To link natural and social science models, this analysis requires a definition of ecosystem services that satisfies the underlying ecological science, as well as the requirements of economic application. Ecologists historically define ecosystem services as the benefits that humans derive from ecosystems (Millennium Ecosystem Assessment, 2005, Wallace, 2007). This definition tends to "double count" the benefits that humans derive from the ecosystem because it encompasses ecosystem processes *and* ecosystem services. In other words, the value of the function and the value of the service are both counted.

In contrast, Boyd and Banzhaf (2007) defined ecosystem services as the final components of ecosystem processes. Instead of being benefits, ecosystem services are viewed as "*components of nature*, directly enjoyed, consumed, or used to *yield* human well-being." By this definition, ecosystem services are not the *benefits* humans obtain from ecosystems, but rather the *final ecological components* that flow from the ecosystem. As such, these endpoints are combined with other inputs to create human benefit (Boyd and Banzhaf, 2007, Boyd, 2007, Fisher et al., 2008). This definition allows economists to incorporate ecosystem services as inputs in utility functions, which they use to model human benefits. This advancement helps social scientists to measure how much people care about changes in ecosystem services.

To illustrate this definition, Boyd and Banzhaf (2007) provided the following example of how ecosystem services (surface waters, fish populations, and scenic surroundings) combine with human-made inputs (equipment, time, and access) to produce benefits for recreational anglers:

> *Consider, for example, the benefits of recreational angling. Angling requires ecosystem services, including surface waters and fish populations, and other goods and services including tackle, boats, time allocation, and access. For this reason, angling itself–or 'fish landed'–is not a valid measure of ecosystem services. More fish may be landed simply because better tackle are [sic] used... The fish population, surroundings, and water body are the 'ecosystem end products' directly used by anglers to produce recreational benefits. Thus, they are the ecosystem services that should be counted. The case of commercial fishing is similar, but here aesthetics are unimportant, so only the target fish populations need to be counted as ecosystem services.*

This example illustrates that ecosystem services are not human benefits. Rather, they are measurable, physical endpoints of ecosystem processes that are combined with other inputs by humans to create benefit. Ecosystem services provide an avenue for valuation of environmental quality because they are quantifiable. Changes in the quantity of ecosystem services impact human benefit and these impacts can be valued.

Ecosystem services impacted by mine site pollution

The legacy of historical mines can include water-quality degradation due to mine drainage, sediment-quality degradation from solid mine waste erosion, soil-quality erosion from wind-blown tailing dust, and degradation of vistas in mining areas. Water-quality issues result from the interaction of precipitation, surface water, and groundwater with rock within the mine workings, waste rock excavated to access ore, or waste from processed ore. Sediment-quality and soil-quality issues commonly result, respectively, from the water and wind erosion of mine waste, particularly mill tailings that have been inadequately disposed. Mill tailings (typically the size of sand or silt) are the waste materials produced by finely crushing ore and removing the ore minerals using a variety of physical and chemical processes. Mining also alters the visual appearance of the landscape around mines. Open pits are rarely filled once mining has ceased. Waste rock and tailings storage facilities are common features on mine landscapes that have historically been left with minimal restoration after mining has stopped. In contrast, best practices for modern mining seek to restore landscapes to useful purposes without impacting the environment. Mine buildings and structures such as head frames for mine shafts may be a significant part of the legacy of mining landscapes. In some cases, these historical structures may be an important part of the appeal of these areas for tourism.

Mine drainage can affect several ecosystem services. Acid or dissolved metals can affect aquatic organisms, such as aquatic invertebrates that form the base of lotic ecosystems, and fish. Fish support terrestrial ecosystems and can support recreational or subsistence fisheries depending upon location. Mine drainage can also contain one or more solutes that may exceed drinking water standards and affect the potability of groundwater or surface water in the vicinity of mines.

Sources of drinking water are treated prior to human consumption. In the case of a stream impacted by mining activity, there may be high concentrations of several metals present, such as copper, lead, or arsenic, leading to increased costs to treat the water to a level suitable for human consumption. Impacts of mine drainage on water quality in mining districts, therefore, have economic impacts on the ecosystem services supported by water quality.

Solid mine waste can also affect ecosystem services, leachate from waste piles can serve as a source of mine drainage, fine-grained mill tailings can be a source of wind-blown contamination for residential and agricultural soils, and ongoing erosion of poorly contained tailings (as well as catastrophic failure of tailings storage facilities) can serve as a source of fluvial tailings that are transported downstream and become deposited in quiescent settings. Such fluvial tailings can serve as a chronic source of contaminants to surface water.

Ecosystem service impacts related to mines can also extend beyond direct impacts to water, soil, or sediments. Many abandoned and existing mines are located in remote and often scenic terrains. Streams originating in these locations may also be used for recreational activities, such as fishing, rafting, hiking, and camping. Additionally, terrestrial and riparian vegetation may be lost from changes to the soil and sediment composition, resulting in undesirable aesthetics, loss of cover for camping, and/or change in stream temperature. While other types of activities may influence these same services, such as logging, clear cutting for electrical lines or wind turbines, or laying of pipelines, we chose locations that had mineral extraction as the dominant (or preferably sole) source of influence on the ecosystem.

Initial set of ecosystem services explored

In this research, we use existing data from mined locations to evaluate any losses of ecosystem services that may result from that mining activity. Specifically, we focus on sites from the U.S. Environmental Protection Agency (USEPA) National Priorities (Superfund) list from their Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) program. Sites from the Superfund list were chosen because of the detail and depth of data gathered as part of the prescribed remedial investigations and subsequent post-remediation monitoring.

Table 2.1 below presents the ecosystem services that were initially identified as potential foci for this research. In many cases, water quality is a metric. However, within water quality, there are a number of measurements that may serve as the basis for evaluation. Assessment based on a sole parameter, or analyte, may not be possible because all parameters contribute to the evaluation. Therefore, water quality is the metric, but parameters and analytes are how the metric will be evaluated. As described in the following sections, it was decided to focus on target fish populations as they related to sport fishing because this ecosystem service had the best data set available for this study.

Table 2.1. Benefits, Ecosystem Services, Metrics, and Valuations Considered

Benefits	Ecosystem Services	Metrics	Valuations
Human Health (Drinking water)	Quality of stream water, quality of groundwater	Concentrations of constituents in the water (for example, copper, cadmium, hardness), water quality parameters (for example, pH, dissolved oxygen, color, turbidity)	Avoided costs for treatment or replacement
Recreation (fishing, hiking, camping, rafting, swimming)	Target fish population	Numbers of fish	Travel cost surveys
Recreation (fishing, hiking, camping, rafting, swimming)	Surface water body existence and quality of water	Miles of streams Quality of stream water	Travel cost surveys, avoided costs for treatment or replacement
Recreation (fishing, hiking, camping, rafting, swimming)	Natural land cover	Acres of riparian regions, soil types	Travel cost surveys
Aesthetic values	Vistas	Acres of undisturbed land	Hedonic pricing method, contingent valuation
Aesthetic values	Water quality (clarity)	Turbidity and color	Hedonic pricing method, contingent valuation
Existence values	Species abundance	Variety/number of species	Contingent valuation, choice modeling
Existence values	Wilderness	Acres of undisturbed land	Contingent valuation, choice modeling

The endpoint problem: Difficulties in linking natural and social science research

Despite the advancement that the Boyd and Banzhaf (2007) definition brings to the linkage of natural and social science models, problems persist (Boyd, 2007, Kontogianni et al., 2010). Boyd

(2007) explained that "*if linked social and natural science is a relay race, endpoints are the baton. The problem is that the baton never gets handed off smoothly*." An example of the endpoint problem is provided by the quotation above from Boyd and Banzhaf (2007) (Section 1.1) on the ecosystem service of fish population and the benefits derived from recreational angling.

The endpoint problem is that anglers do not assign an economic value for the target fish population itself (Ng, 2011). Instead, they can assign a value for the trip that they take to an angling destination or the day that they spend fishing. As a result, economic valuation studies have focused on the value of an angling trip or day (Ng, 2011). Boyd and Banzhaf (2007) referred to the angling trip/day in their quote as “angling itself”, which is “not a valid measure of ecosystem services.”

Therefore, for an ecosystem service valuation, natural scientists can successfully model the ecosystem service (target fish population), while economists are only able to model the value of a related – but different – environmental good. In fact, the angling trip/day is *two* degrees removed from the ecosystem service (catchable fish). The number of catchable fish influences the number of fish caught (first degree of separation), and the number of fish caught influences the number, and value of angling days/trips (second degree of separation) (Johnston et al., 2006, Loomis and Ng, 2009, Mazzotta, 2015, Ng, 2011, USEPA, 2006a).

A body of economic research (USEPA, 2006a) and a journal publication (Johnston et al., 2006), resolved the second degree of separation by analyzing the valuation literature to determine what information it revealed about anglers’ value for catching another fish. In other words, USEPA (2006a) conducted an analysis of angling day valuation studies to estimate how changes in the number of fish caught affected the value, or number, of angling days. However, Boyd and Banzhaf (2007) referred to the number of fish caught in their quote as “fish landed", which is also not a valid measure of ecosystem services. Nonetheless, this research moves economists one step closer to be being able to estimate the value of catchable fish.

Out of the 405 recreational angling valuation studies surveyed by USEPA (2006a), only four were identified by this analysis as providing a possible remedy for resolving the first degree of separation. The first two studies (Johnson et al., 1995, Mazzota et al., 2015) addressed the gap by making static, proportional assumptions about the impact of changes in target fish population on changes in the number of fish caught by anglers. The second two studies (Loomis and Ng, 2009, and Ng, 2011) originated from the same body of research and addressed the gap by econometrically modeling the catch per day.

Johnson et al. (1995) assumed that 60% of stocked trout were caught by anglers. Mazzota et al. (2015) assumed that the percentage change in the catch rate is equal to the percentage change in the abundance of catchable fish. Loomis and Ng (2009) and Ng (2011), on the other hand, modeled the catch per day as a function of the number of catchable fish, angler skill, target species, and the number of hours fished each day. While these four studies help economists to value the ecosystem service in question, future research should be dedicated to the dynamic modeling of the relationship between fish population and fish catch, even in the presence of data constraints (Ng, 2011).

In conclusion, the “relay race” of linked natural and social science is still working to hand the baton off smoothly. For the purpose of this ecosystem service valuation, the endpoint problem is particularly prevalent for trout population. With the endpoint problem in mind, we describe which of the ecosystem services from Table 2.1 were selected for economic valuation modeling. It is important to state that these selections were made in light of serious data difficulties described in detail below.

Selection of ecosystem service endpoints capable of linking geo-environmental and valuation models

Button et al. (1999) provided guidance on the selection of ecosystem services for valuation of mine site pollution: “In terms of acid mine drainage (AMD) remediation, the importance of restoring water quality, the restoration of scenic beauty, and the reintroduction of fish stock emerge as key issues”. One ecosystem service that follows this guidance, captures the majority of impact value, and can be valued by the existing valuation literature is catchable target fish population.

Olander et al. (2015) suggested using ‘catchable’ target fish population as the endpoint for the ecosystem service of target fish population. This term focuses ecosystem service modeling on the segment of target fish population that is large enough to be caught by anglers. It also relegates the target fish population itself to the status of an ecosystem process. Catchable target fish population can increase in a stream segment as a result of mine-site pollution remediation. Alternatively, mine site development can decrease catchable target fish population by altering water flow, reducing habitat, or accidentally releasing toxic effluent. In conclusion, data issues limited the successful linkage of geo-environmental and valuation modeling to the single ecosystem service of catchable trout population.

Central Colorado study area

Central Colorado provides an excellent opportunity to investigate the economic impact of legacy mining on ecosystem services using a geologically based (geo-environmental) approach. Central Colorado has historically been mined for a number of commodities from a variety of mineral deposit types. The most significant deposit types include carbonate-hosted lead-zinc-silver deposits, porphyry molybdenum deposits, and epithermal gold deposits. This study will focus on the carbonate-hosted lead-zinc-silver deposits of Central Colorado, which provided the impetus for the early settlement of this region. The main historical mining districts include Gilman, Leadville, Kokomo, Aspen, and Sherman. The Leadville district is the most economically significant of these camps (Beaty et al., 1990). The Leadville (California Gulch) and Gilman (Eagle mine) districts were selected for this study because of the amount of data available from these areas. Both sites offer well documented case studies of environmental impacts caused by geologically similar abandoned mines, their remediation, and the subsequent and ongoing recovery of downstream surface water ecosystems.

Statement of Problem

Abandoned mines are complex in terms of the source, transport, and fate of contaminants and the effect of these contaminants on the surrounding ecosystems and the services that these ecosystems provide. The source of contaminants is influenced by the geology of the deposit, the mining methods used, the ore-processing used, the waste management practices used, the hydrologic setting, and climate – among other factors. The transport and fate of contaminants are influenced by the local climate and hydrologic setting, the geology of the watershed as it relates to the chemistry of receiving water bodies, and the specific contaminants themselves. The impact of mine drainage and solid mine waste on aquatic organisms is equally complex and depends on climate, hydrologic setting, the physical characteristics of the in-stream habitat, the number of contaminants, and a variety of processes that serve to either increase or decrease the concentrations and/or bioavailability of contaminants in surface water.

The recovery of the habitat in these streams impacted by abandoned mines is complicated by the success in addressing water-quality issues, the distribution of residual sources of contamination, the state of the habitat available once remediation is complete, and the availability of food sources for aquatic organisms. The linkage of remediation to the value of an ecosystem service requires knowledge of specific pathways of contaminants to the ecosystem services that they impact, how remediation will influence those pathways in both the short and long term, and how those changes will affect the value of those ecosystem services in the present and future.

For the present study, the exercise is restricted to the ecosystem services related to recreational fishing because reasonably clear links can be established among water quality, the abundance of catchable fish – specifically brown trout, and the value of those fish to anglers. The Leadville (California Gulch) and Gilman (Eagle Mine) districts have geologically similar carbonate-hosted lead-zinc-silver deposits that were mined and processed using similar approaches. Both sites were addressed through the USEPA CERCLA program for remediation, and both sites have site-specific environmental data spanning the period from prior to remediation, through remediation, to after most remedial activities have been completed. The data encompass the physical and chemical hydrology of downstream habitat, including information about fish and macroinvertebrate populations extending at least 10 km downstream of the abandoned mine sites.

The overall goal of this project is the valuation of ecosystem services to estimate the benefits for what will be gained from remediation of abandoned mine sites having similar characteristics to these existing sites. Specifically, the main objectives of this project are:

1. To develop a geologically based (geo-environmental) mineral-deposit model for carbonate-hosted lead-zinc-silver deposits in Central Colorado that identifies specific geochemical risks in a context that is relevant for the hydrogeologic setting of these deposits,

2. To incorporate damages to ecosystem services, specifically for recreational fishing, into a watershed scale context spanning the period from before remediation to after remediation,
3. To develop a model linking changes in water chemistry and aquatic habitat to changes in fish populations, and
4. To link changes in fish populations to changes in their value as an ecosystem service, specific sport fisheries.

Scope Of Study

The original plan for this project was to consider a number of mineral deposit types in a number of hydrologic and climatic settings. The original list included porphyry copper deposits (for example, Bingham Canyon, Utah, Morenci, Arizona), epithermal gold deposits (for example, Cripple Creek, Colorado, Golden Sunlight, Montana), volcanic-associated massive sulfide deposits (for example, Holden, Washington, Greens Creek, Alaska), carbonate-hosted lead-zinc-silver (Pb-Zn-Ag) deposits (for example, Leadville and Colorado Eagle, Colorado), and Mississippi Valley-type lead-zinc deposits (for example, southeastern Missouri). The list of potential deposit types to include in the study was narrowed based on the availability of relevant secondary environmental and ecosystem services data meeting the data-quality criteria of having been acquired by documented and approved methods with defined data quality objectives, i.e., accuracy and precision, etc., that were either consistent with methods approved by the federal government or are universally accepted. The data availability exercise required the study to be focused on the carbonate-hosted Pb-Zn-Ag deposits of Central Colorado (Leadville and Gilman mining districts) because those areas are the only sites for which adequate environmental data are available. Furthermore, the ultimate adequacy of data varied between those two sites with the Leadville district having the most suitable data set.

A similar exercise was conducted to select ecosystem services for this study. Initially, plans were to include the value of drinking water, vistas, aquatic habitat, and recreational fishing, but insufficient data were identified to allow quantitative evaluation of the first three ecosystem services. This project focuses exclusively on the value of sport-fishing recreation.

Sources Of Data

Data fell into three broad categories: geologic, environmental, and economic. Data were sought and evaluated based on the data-quality hierarchy described below, in order of descending quality:

1. U.S. EPA or U.S. Geological Service (USGS) data collected under the respective agency's quality assurance/quality control (QA/QC) program,
2. State or tribal data collected under respective approved QA/QC programs,
3. Data collected by other federal agencies (and their contractors) having approved QA/QC programs (for example, Fish and Wildlife Service, Bureau of Land Management),

4. Data collected, peer-reviewed and published by academic organizations having described QA/QC protocols, and
5. Published data with limited QA/QC oversight (for example, independent contractors that may or may not have established protocols) or publications that may or may not be peer-reviewed (for example, conference proceedings from sources other than items 1-3 above).

Data in categories 1-4 above were used quantitatively, and data from category 5 was used only qualitatively.

Environmental data

A significant portion of the hydrologic, hydrochemical, and biologic data used in this study was derived from USGS data sources or reports published by the USGS and state agencies - for example, Clements et al. (2010) and Woodling et al. (2005). The USGS National Water Information System (NWIS, http://nwis.waterdata.usgs.gov/usa/nwis/qwdata/) was used for stream flow and some water-quality data (pH, specific conductance) in both the Arkansas River and Eagle River watersheds. Sources of geologic, hydrologic, geochemical, and biologic data are summarized in Table 5.1.

Table 5.1. Sources of Geologic, Hydrologic, Geochemical, and Biologic data

Source	Data Types	Comments
General		
Beaty et al. (1990)	Geologic setting	Descriptive paper for geologic background that includes grade and tonnage details
Beaty (1990)	Geologic setting	Descriptive paper for geologic background
Thompson et al. (1990)	Geologic setting	Descriptive paper for geologic background
Wallace (1993)	Geologic setting	Descriptive paper for geologic background
California Gulch		
Clements et al. (2010)	Water quality, sediment quality, macroinvertebrate populations, fish populations	Study includes two sites upstream and two sites downstream of California Gulch. Study spans 17 years.
Eagle River		
Woodling et al. (2005)	Water quality, sediment quality, macroinvertebrate populations, fish populations	

Economic data

Economic data were primarily obtained from peer-reviewed studies as detailed in Table 5.2 below. These studies were all meta-analyses of environmental valuation literature that proved capable of valuing ecosystem services relevant to this project. Creel survey data – including information about angling hours/days, the number of fish caught, their location – were obtained from the Colorado Parks and Wildlife annual fisheries inventory reports concerning the Upper Arkansas River from the years 2012 and 2013. Sources of economic data are summarized in Table 5.2.

Table 5.2. Sources of Economic and Ecosystem Services Data

Source	Data Types	Comments
General		
USEPA (2006a)	Value estimates for an angler's willingness to pay to catch an additional fish	A meta-analysis of the non-market valuation literature on recreational angling, with the goal of estimating the angler's willingness to pay to catch another fish of the angler's target species
Johnston et al. (2006)	Value estimates for an angler's willingness to pay to catch an additional fish	This data source is the journal article publication of the research conducted in USEPA (2006a) report described above.
California Gulch		
Policky (2012)	Creel census data (angling hours/days, catch rate, proportion of out-of-state anglers) for the years 1995, 2008, 2012	Colorado Parks and Wildlife Fisheries Inventory report for the year 2012
Policky (2013)	Extrapolation of creel site data to the broader river reach that the creel site is within - for the years 1995, 2008, 2012	Colorado Parks and Wildlife Fisheries Inventory report for the year 2013 – which provided angling estimates for each river reach from its respective creel study site

Economic Valuation Background

This section defines and describes various economic values that can inform benefits derived from ecosystem services to illustrate the effect that mine site pollution can have on the economic benefits of surrounding ecosystem services. Methods of estimating these economic values, in the absence of formal markets, are discussed with a particular focus on the use of existing economic valuation literature – a practice known as benefit transfer. Finally, a summary discussion of the existing mine site valuation literature is provided to lend context to the benefit transfer model described in the following chapter.

Measures of Economic Value

An excerpted discussion from Boyle et al. (1998) is provided in Figures 6.1 and 6.2 to illustrate several measures of economic value. This discussion is clear and is directly applicable to the primary ecosystem service values estimated in the benefit transfer model for catchable trout population and in the Upper Arkansas River ecosystem service valuation application.

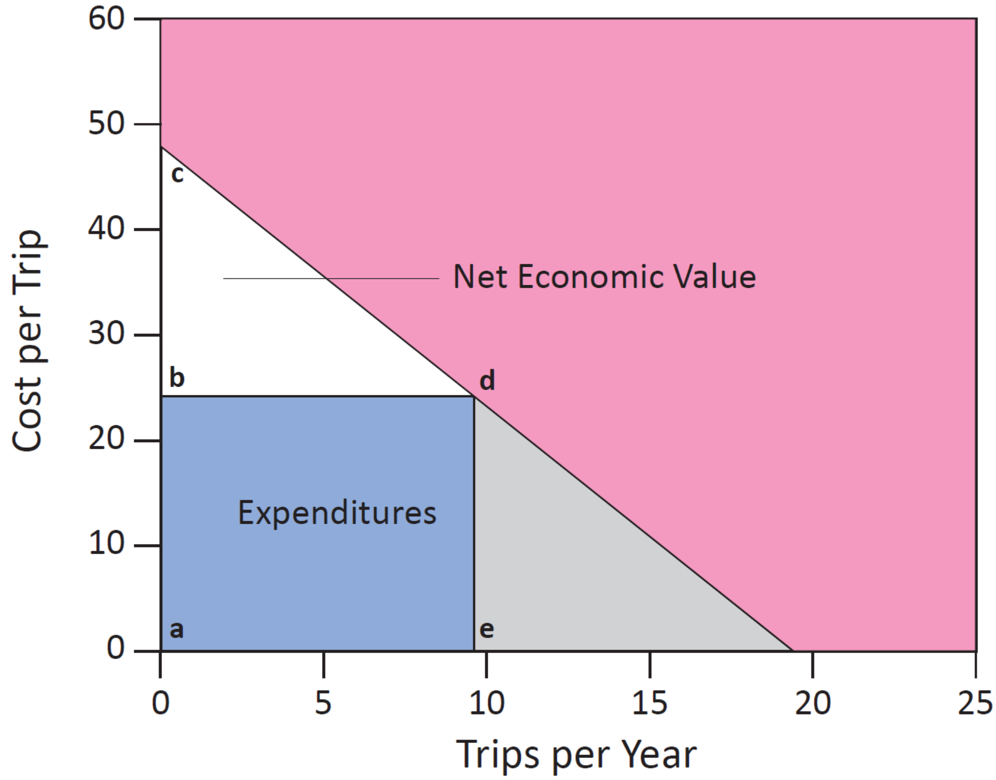

Figure 6.1: Illustrative demand curve for fishing trips by an individual angler. Modified from Boyle et al. (1998).

> Figure 6.1 shows a demand curve for a representative angler. "The downward sloping demand curve represents marginal willingness to pay per trip and indicates that each additional trip is valued less by the angler than the preceding trip. All other factors being equal, the lower the cost per trip (vertical axis) the more trips the angler will take (horizontal axis). The cost of a fishing trip serves as an implicit price for fishing since a market price generally does not exist for this activity.
>
> At $60 per trip, the angler would choose not to fish, but if fishing were free, the angler would take 20 fishing trips. At a cost per trip of $25 the angler takes 10 trips, with a total willingness to pay of $375 (area acde in Figure [6.1]). Total willingness to pay is the total value the angler places on participation. The angler will not take more than 10 trips because the cost per trip ($25) exceeds what

he would pay for an additional trip. For each trip between zero and 10, however, the angler would actually have been willing to pay more than $25 (the demand curve, showing marginal willingness to pay, lies above $25).

The difference between what the angler is willing to pay and what is actually paid is net economic value. In this simple example, therefore, net economic value is $125 (($50 – $25) 10 ÷ 2) (triangle bcd in Figure [6.1]) and angler expenditures are $250 ($25 × 10) (rectangle abde in Figure [6.1]). Thus, the angler's total willingness to pay is composed of net economic value and total expenditures. Net economic value is simply total willingness to pay minus expenditures. The relationship between net economic value and expenditures is the basis for asserting that net economic value is an appropriate measure of the benefit an individual derives from participation in an activity and that expenditures are not the appropriate benefit measure.

Expenditures are out-of-pocket expenses on items an angler purchases in order to fish. The remaining value, net willingness to pay (net economic value), is the economic measure of an individual's satisfaction after all costs of participation have been paid.

Summing the net economic values of all individuals who participate in an activity derives the value to society. For our example let us assume that there are 100 anglers who fish and all have demand curves identical to that of our typical angler presented in Figure [6.1]. The total value of this sport fishery to society is $12,500 ($125 × 100)" [emphasis added]

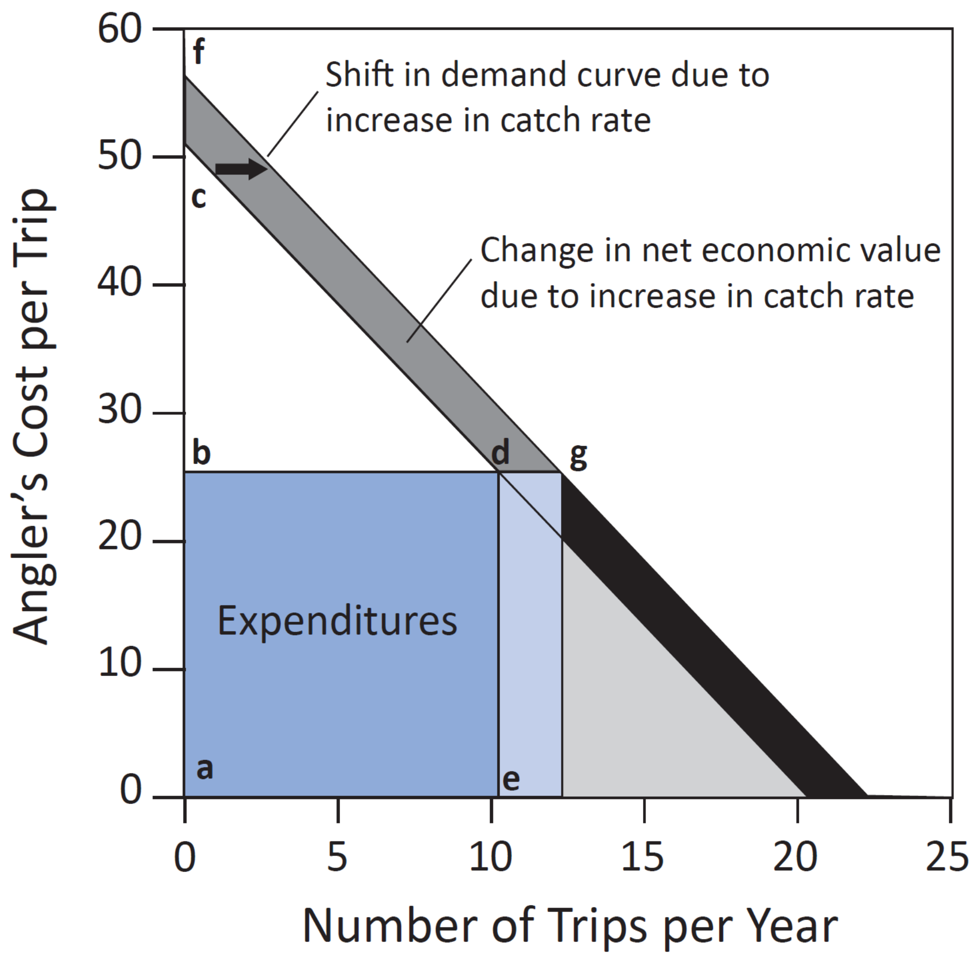

Figure 6.2: Impact of an increase in catch rate on the angler demand curve for fishing trips. Modified from Boyle et al. (1998).

> "In many instances, all or nothing values, as shown in Figure [6.1], are not appropriate. Rather, a change in quality shifts the demand curve, thereby resulting in a change in net economic value (Figure [6.2]). In these instances, the change in net economic value is the appropriate benefit measure.
>
> For example, assume a management activity will increase catch rates for anglers by 10 percent. This change in the resource results in a shift of the demand curve upward and to the right, as presented in Figure [6.2]. The benefit to the angler of this increase in catch rate is the area cfgd. Estimation of this area is possible by including harvest rates as explanatory variables in the estimated [value] equations" [emphasis added]

In other words, when the following analysis mentions benefits, it refers to the net economic value (triangle bcd) – which is also known as the net willingness to pay (WTP) and the consumer surplus. Similarly, the term 'benefits of remediation' refers to the monetary value of the area cfgd in Figure 6.2 above. The final important point is that the change in net economic value is the appropriate benefit measure that we will focus on.

Non-market valuation techniques

Economic research informs environmental decisions through non-market valuation, which allows a more direct comparison of costs and benefits relating to environmental change, such as environmental degradation related to abandoned mines or the subsequent remediation of those mines (Carson et al., 1992, Costanza et al., 1998). The goal of this section is to evaluate the capacity of the environmental valuation literature to quantify environmental damage from mine site pollution (from all types of mine sites) in monetary terms.

Economists estimate the benefit from directly using an environmental service through environmental valuation. Value derived from use is known as *use value*. Economists can also estimate the intrinsic value that humans place on environmental services – known as *non-use value*. Non-use values are received from the existence of a service that the individual would never use. In some cases, non-use values can be quite large.

Economists use four different techniques to value environmental services: replacement cost, revealed preference, stated preference, and benefit transfer. In the first technique, a replacement cost for a lost environmental service is calculated. Replacement cost techniques measure the cost of employing human capital or labor, in lieu of an environmental service. This is an intuitively appealing method for examining the value of an environmental services, however, it is often the least desirable method because cost does not necessarily equate with value (Loomis, 2000).

For example, a valuable environmental service (such as potable water) may be relatively inexpensive to replace. In this case, the replacement cost represents a minimum value of benefits that the environmental service provides. In contrast, the replacement of a fishery in an area not inhabited by humans would be costly and provide no value. Therefore, using replacement costs as a measure of value can be misleading for economic decision-making.

Revealed preference methods estimate environmental service values from consumer behavior that is observed in real markets. Examples of revealed preference methods include travel cost valuation, hedonic valuation, averting behavior valuation, and production function valuation. Travel cost valuation employs travel time and additional expenses incurred by individuals to value recreational sites. Hedonic valuation uses market data on property to isolate the value of a particular environmental service - such as a view. Averting behavior valuation sums up expenses imposed due to poor environmental services. Finally, the production function valuation technique estimates the value of environmental services as inputs to the production process. The observation of actual human behavior in real markets is an advantage of using the revealed preference technique. However, a shortcoming of this approach is that only *use values* are measured.

Stated preference methods gather environmental service values through surveys that detail hypothetical changes in non-market services, for example air quality, and ask respondents what they would be willing to pay for those hypothetical changes. Stated preference methods use carefully crafted surveys to allow respondents to directly state their willingness to pay to avoid a loss of environmental service quality. Examples of stated preference methods are contingent valuation and conjoint choice modeling. The stated preference technique is the only technique that can capture non-use value, which is a major advantage of using stated preferences methods (Haab et al., 2013). However, some economists see the fact that the methods rely on hypothetical markets, instead of real transactions, as a drawback (Hausman, 2012). Over the years, there has been a lively debate regarding the validity of stated preference methods (Carson et al., 2001, Diamond and Hausman, 1994, Hanemann, 1994, Portney, 1994).

A detailed literature review of all of these methods would be voluminous. Instead, the purpose of describing these methods is to demonstrate that a plethora of approaches are available for valuing environmental services.

Benefit transfer

The valuation techniques described in the previous section all produce *primary* valuation studies. Primary valuation studies are conducted for a specific site, time, and context. By their nature, primary valuations are expensive. The large number of mine sites that could be studied precludes primary valuation of many of them, let alone all.

Constrained financial resources and the high cost of primary valuation studies provided incentive to find cheaper ways to determine the value of non-market services at new sites (Bingham et al., 1992). The result came to be known as *benefit transfer*. This technique transfers a benefit estimate across time and space from the primary study site (known as the *study site*) to another site where a policy is being evaluated (known as the *policy site*). Wilson and Hoehn (2006) explained that:

> *[B]enefit transfer uses economic information captured at one place and time to make inferences about the economic value of environmental goods and services at another place and time.*

Hypothetically, if there are two identical populations, environmental services, and contexts, then the valuation of the environmental service should be the same for both sites. The need for environmental service valuation, coupled with the expense of conducting primary valuation studies, has propelled benefit transfer forward as a widely employed method to approximate the value of environmental services at different locations (Wilson and Hoehn, 2006). Since the early 1990's, benefit transfer has been used in federal regulatory impact analysis for non-market, environmental goods (Boyle et al., 2010). The literature on mine site pollution valuation (Appendix A) supports the use of benefit transfer:

> *Conducting very detailed case studies on any prospective location to ameliorate [acid mine drainage] is costly. A hybrid approach is, therefore, advocated, using existing information coupled with new, case-specific analysis. Although there is the need to gather some information on individual sites and those directly affected by remediation schemes, the use of literature reviews, meta-analysis, and other techniques (i.e., benefit transfer) facilitates value transfers* (Button et al., 1999).

Initially, benefit transfer comprised: 1. an evaluation of the policy site, 2. selection of a corresponding primary valuation study from the existing literature, and 3. direct transfer of the primary study's results to the study site. This is known as unit value benefit transfer (Loomis, 1992). However, benefit transfer is not limited to a single primary study, but can be employed using many primary studies. Using additional primary valuation studies can facilitate more accurate benefit transfer estimates. To advance this improvement, valuation experts began to isolate the effect of explanatory variables such as income, study site characteristics, and region on the valuation result (willingness to pay) that primary studies had generated. Using this information, a benefit transfer function is constructed with the intent of further improving the accuracy of benefit transfer (Loomis, 1992).

The use of meta-analysis for benefit transfer

As more benefit value estimates emerged for the same environmental service, it became clear that the resulting estimates were seldom of the same magnitude. Some even had conflicting signs. Whereas literature reviews were useful in qualitatively evaluating valuation result disparities, a more quantitative approach was required (Boyle et al., 1994). Economists began to evaluate primary valuation studies statistically via meta-analysis (Boyle et al., 1994, Carson et al., 1996, Smith and Huang, 1995, Smith and Kaoru, 1990, Woodward and Wui, 2001).

Meta-analysis is a quantitative analysis of valuation analyses that uses regression to determine the factors that cause variation between primary study results. First, the meta-analyst identifies determinants of variation between primary study estimates of willingness to pay (WTP) to avoid a reduction in environmental quality (Nelson and Kennedy, 2009). Usually these determinants include population income, population demographics, primary study site characteristics, study method, pollutant type, and publication method (Bergstrom and Taylor, 2006, Navrud and Ready, 2007).

Once a meta-analysis function is estimated, the explanatory variables are set to reflect the policy site as closely as possible. The result is a *meta-regression model benefit transfer* (Kirchhoff, 1998, Rosenberger and Loomis, 2000, Shrestha and Loomis, 2001). Meta-regression model benefit transfer reduces the error between benefit transfer estimates and site-specific estimates when study and policy sites were not particularly similar (Kaul et al., 2013), which is the majority of the time in benefit transfer.

Meta-regression models are currently the state-of-the-art instruments for synthesizing and transferring benefit estimates from the environmental literature to unstudied (‘policy’) sites. Therefore, in subsequent chapters, meta-regression models are relied upon heavily to construct a mine site pollution benefit transfer model and to apply this model to mine site pollution.

Summary of environmental valuation literature regarding mine sites

To employ the benefit transfer valuation technique, primary valuation studies must be located that correspond to the ‘policy’ site in question. Given that the goal of this analysis is to value changes in ecosystem services related to mine sites, the following section summarizes a more detailed review of the environmental valuation literature that relates to mine site pollution. See Appendix B for the full literature review. The number of environmental valuation studies of abandoned, proposed, or operating mine sites is rather small. The purpose of this summary is to illustrate why studies from this literature were not helpful in the construction of the mine site pollution benefit transfer model discussed in the following sections.

The first set of studies in this literature comprises attempts at social cost-benefit analysis of mine remediation schemes. Valuable information can be gleaned from these social cost-benefit analyses of mine site remediation. However, a benefit transfer model for the purpose of this analysis cannot be built upon these studies because of one or more of the following flaws. First, some of these studies confuse the economic values of expenditures and net WTP (Randall et al., 1978). Second, most of these studies conducted the valuation at the level of the whole site, rather than focusing on specific ecosystem services (Damigos and Kaliampakos, 2003, Farber and Griner, 2000, Michael and Pearce, 1989, Neelawala et al., 2013, Williamson et al., 2008). Third, some studies were unable to produce a benefit estimate (Button et al., 1999, Mendes et al., 2007). Finally, some study sites do not correspond to sites in the United States (Ahlheim et al., 2004, Burton et al., 2012, Lienhoop and Messner, 2009).

The second set of studies in this literature comprises attempts at social cost-benefit analysis of mine development schemes. Whereas these studies are also informative, they cannot be used for benefit transfer because they are conducted at the site-level (Trigg and Dubourg, 1993) or they are benefit transfer exercises themselves (Damigos and Kaliampakos, 2006, Unaldi et al., 2011).

The final source of valuation studies is natural resource damage assessments (NRDA). Estimates of economic benefit values are sparse in the NRDA literature. However, the few economic benefit estimates that are provided prove to be difficult to use. The contending NRDA valuations of the Eagle Mine are an example (Rowe and Schulze, 1985, Ward et al., 1992). Instead of valuing each environmental component separately, the plaintiff's valuation (Rowe and Schulze, 1985) used a contingent valuation method conducted at the site level. On the other hand, the defendant's valuation (Ward et al., 1992) estimated the cost to limit exposure to pollution from the Eagle Mine. Cost estimation misses the point of valuation by confusing the concepts of expenditure and net WTP.

In contrast to the Eagle Mine NRDAs stands the valuation of the BP Oil Spill (Board et al., 2013). Board et al. (2013) broke the impact down into the value of each ecosystem service that enters human utility functions to create benefit. By assigning a value to the magnitude of each ecosystem service's impact, that ecosystem service's value can be transferred to dissimilar sites that share a common ecosystem service.

In other words, economists need to work out exactly which environmental services are being valued and how the value of each service impacts the total value of damage or benefit. Focusing on specific services is important to economists because of substitution, scale effects, adding up, internal consistency, and external consistency (Carson et al., 1992, 2001, Diamond, 1996, Diamond and Hausman, 1994).

Ecologists agree with this sentiment and argue that economists do not understand what they are valuing (Limburg et al., 2002). Ecologists argue that this misunderstanding leads to double counting and confusion of the human subjects surveyed during non-market valuation (Limburg et al., 2002). A remedy is provided by the ecosystem service framework, which the following benefit transfer model follows closely.

Benefit Transfer Model for Valuation of Selected Ecosystem Services

The goal of this section is to apply environmental valuation methods to ecosystem services that are affected by mine site pollution. The purpose of this goal is to construct a model that measures how much people care about changes in environmental quality as a result of changes in mine site pollution. This model relies on three basic concepts. The first concept is that economists can estimate how important a change in environmental quality is to a sample of people by ascertaining the amount they are willing to pay to avoid the change[1]. The second concept is that ecosystems are made up of interactive processes among minerals, water, and biota, which create ecosystem services that people combine with time, effort, and equipment to derive benefit. The third concept is that if economists estimate an ecosystem service's value for a population sample, then it is feasible to apply the results to similar ecosystem services for similar population samples.

Combining these three concepts allows environmental valuation to communicate with ecosystem modeling and produces a scientifically rigorous valuation model by linking natural and social sciences. Such a model can transfer benefit estimates from relevant environmental valuation papers to unstudied mine sites and elucidate environmental costs and benefits of changing mine site pollution. Ecosystem service valuation helps to incorporate the value of ecosystem services into decisions regarding abandoned mine lands, legacy sites, operating sites, proposed mines, and closure of operating sites. For example, application of the ecosystem service valuation model quantifies the benefits of the legacy site remediation at California Gulch.

The ecosystem service chosen for this project is catchable target fish populations. This ecosystem service was chosen for two main reasons. First, clear links can be established between the ecosystem service and the underlying geology, geochemistry, hydrology, and mining method. Second, this ecosystem service encapsulates the majority of costs and benefits related to mining projects (Button et al., 1999). This section reviews the literature and methods used to construct the benefits transfer tool for catchable target fish population.

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[1] Or, conversely, the amount they would need to be compensated to accept the change.

Catchable target-fish population for recreational angling

The example provided by Boyd and Banzhaf (2007) regarding ecosystem services related to the benefits of recreational angling helps to clearly delineate what is being valued by this segment of the benefit transfer model. For the purpose of ecosystem service valuation, the Boyd and Banzhaf (2007) definition implies that the total value from angling is the sum of value from all of the inputs to angling. For example, the angler's value per day represents the value of *angling itself.* The angler's value for *fish landed* during the day is some portion of the angler's total value per day. Further, some portion of the value for *fish landed* represents the angler's value for the ecosystem service of *fish population*.

Finally, other portions of the angler's total value per day represent the value of benefits from inputs not related to fish, some may be ecosystem services (scenic surroundings, fresh air, clean water, wildlife, sunshine) and some may not (time with a friend, a cigar, a new fishing rod). In other words, if the angler's total value per day is a pie, the value of fish landed is one slice of that pie, and the value of the catchable target fish population is a portion of that slice.

WTP for an Angling Day

Traditionally, recreational angling has been valued by estimating an angler's willingness to pay (WTP) for a day of fishing (Boyle et al., 1998, Loomis and Ng, 2009, Loomis and Richardson, 2007, Ng, 2011, Vaughan and Russell, 1982). As discussed above, WTP estimates for an angling day include the value of additional ecosystem services – such as scenic view sheds, surface water for boating, bird watching, wildlife viewing, and fresh air. Therefore, WTP estimates for an angling day are included here only to provide context for the ecosystem service valuation of catchable target fish population.

Loomis and Richardson (2007) conducted a meta-analysis of this extensive valuation literature and provided WTP estimates for four species groups in six regions of the United States. The estimates that were applicable to this analysis were for cold water species in the Intermountain range – median net WTP per angling day of $51.27 and average net WTP per angling day of $67.91. The majority of trips included in the meta-analysis that generated these estimates were only for a single day.

WTP for Fish Caught

To value changes in the number of target fish caught, this project uses a benefit transfer tool created by USEPA (2006a)[2] in a "Regional Benefits Analysis for the Final Section 316(b) Phase III Existing Facilities Rule June 2006." The overarching goal of USEPA (2006a) was to value reductions in fish-kill from new regulations on power plant cooling water intake systems for streams, rivers, and lakes in the United States. Foregone fishery yield was modeled via a model that required estimates of species-specific size-at-age, stage-specific schedules of natural

[2] The results of this research were also disseminated in the journal article Johnston et al. (2006). However, for the sake of brevity, this research will be referred to by USEPA (2006a) only.

mortality, and fishing mortality (USEPA, 2006a). Estimates of foregone fishery yield were matched with species-specific estimates of anglers' willingness to pay to catch a fish. The meta-analysis of recreational angling valuation literature that generated these value estimates is described below.

USEPA (2006a) conducted an extensive literature review. All relevant studies in the published economic literature, academic dissertations, and conference presentations were evaluated. Forty-eight studies that provided marginal values of catching an additional fish were selected as the study sample. Each study contained multiple values calculated based on various sample characteristics and various model specifications – 391 WTP estimates in all. The 48 studies varied in aspects such as study methodology (for example, stated vs. revealed preference), elicitation method (for example, phone interview, survey, or in person interview), fish species, study location, study date, baseline catch rate, and human sample characteristics. How species in the primary valuation studies were aggregated for the meta-analysis is depicted in Table 7.1.

USEPA (2006a) econometrically estimated a regression on the 391 WTP estimates to estimate the marginal value of catching an additional fish. This regression estimated the influence of primary study variables such as baseline catch rate, species, angler income, and study methodology. Once the influences of these variables are estimated, USEPA (2006a) used the resulting meta-regression function for benefits transfer. This estimation involved setting the function variables to correspond to the new site in question and predicting anglers' WTP to catch an additional fish. For more detail, refer to USEPA (2006a) and Johnston et al. (2006). The constant marginal value per fish results of USEPA (2006a), updated to 2013 dollars, is reported in Table 7.2. These estimates represent mean value estimates. Confidence bounds to the estimates shown in Table 7.2 are provided in Table 7.3.

Table 7.1. Aggregate Species Groups

Aggregate Group	Number of Observations	Species Included[a]
Big game	30	Billfish family, dogfish, rays, shark, skate, sturgeon, swordfish, tarpon family, tuna, other big game
Small game	74	Barracuda, bluefish, bonito, cobia, dolly varden, dolphinfish, jacks, mackerel, red drum, sea trout, striped bass, weakfish, other small game
Flatfish	46	Halibut, sand dab, summer flounder, other flatfish
Other saltwater	89	Banded drum, black drum, chubby, cod family, croaker, grouper, grunion, grunt, high-hat, kingfish, lingcod, other drum, perch, porgy, rockfish, sablefish, sand drum, sculpin, sea bass, smelt, snapper, spot, spotted drum, star drum, white sea bass, wreck fish, other bottom species, other coastal pelagics, "no target" saltwater species
Salmon	44	Atlantic salmon, chinook salmon, coho salmon, other salmon
Steelhead	16	Steelhead trout, rainbow trout (in Great Lakes only)[b]
Muskellunge	1	Muskellunge
Walleye/pike	12	Walleye, northern pike
Bass	14	Largemouth bass, smallmouth bass
Panfish	11	Catfish, carp, yellow perch, other panfish, "general and no target" freshwater species
Trout	54	Brown trout, lake trout, rainbow trout, other trout

[a] Studies evaluated WTP for groups of species that did not fit cleanly into one of the aggregate species groups established by EPA. In those cases, the group of species from the study was assigned to the aggregate speces group with which they shared the most species.

[b] Rainbow trout in the Great Lakes were clasified as steelhead trout because they share similar physical characteristics and life cycles with true anadromous steelhead. Although they have different common names, rainbow trout and steelhead both belong to the species Oncorhyrychus mykiss. Source: USEPA, 2006a.

Table 7.2. Marginal Recreational Value per Fish (by Region and Species[a)])

Species	California	North Atlantic	Mid-Atlantic	South Atlantic	Gulf of Mexico	Great Lakes	Inland
Small game	$7.54	$6.17	$6.13	$5.94	$5.85		$5.56
Flatfish	$10.14	$6.19	$5.83				
Other saltwater	$3.07	$3.10	$3.03	$2.96	$2.89		
Salmon						$13.78	
Walleye/pike						$4.27	$4.25
Bass						$8.89	$9.36
Panfish			$1.10			$1.38	$1.10
Trout						$9.79	$2.94
Unidentified	$3.22	$3.12	$3.37	$2.97	$3.80	$6.46	$2.32

[a] All values are in 2013 Dollars (2013$). Source: USEPA, 2006a.

Table 7.3. Confidence Bounds on Marginal Recreational Value per Fish[a]

Species	California	North Atlantic	Mid-Atlantic	South Atlantic	Gulf of Mexico	Great Lakes	Inland
5% Lower Confidence Bounds[b]							
Small game	$4.12	$1.95	$2.06	$2.42	$2.53		$1.47
Flatfish	$4.85	$3.59	$3.45	$3.59			
Other saltwater	$1.62	$1.62	$1.65	$1.83	$1.80		
Salmon						$10.38	
Walleye/pike						$2.61	$2.28
Bass						$6.04	$5.49
Panfish			$0.59			$0.91	$0.59
Trout						$7.24	$1.50

Species	California	North Atlantic	Mid-Atlantic	South Atlantic	Gulf of Mexico	Great Lakes	Inland
Unidentified	$1.69	$1.63	$1.71	$1.84	$2.02	$4.43	$1.29
95% Lower Confidence Bounds[b]							
Small game	$13.76	$19.14	$17.94	$14.31	$13.31		$20.74
Flatfish	$20.89	$10.73	$9.95	$9.47			
Other saltwater	$5.86	$5.94	$5.60	$4.82	$4.65		
Salmon						$18.29	
Walleye/pike						$7.02	$8.03
Bass						$13.12	$15.98
Panfish			$2.01			$2.12	$2.01
Trout						$13.31	$4.46
Unidentified	$6.17	$5.99	$6.88	$4.87	$7.33	$9.47	$4.14

[a] All values are in 2013$.
[b] Upper and lower confidences bounds based results of the Krinsky and Robb (1986) approach. Source: USEPA, 2006a.

This project uses the Table 7.2 inland value of trout at $2.94 for central Colorado, updated to 2013 dollars. To provide context for this figure, Table 7.4 shows the sub-sample of valuation studies from USEPA (2006a) that apply directly to trout in Colorado.

Table 7.4. Trout Specific Values Corresponding to Eagle and Leadville 'Policy' Sites

Author and Year	State/Region	Study Methodology	Type of Trout	Marginal Value per Fish (June 2013$)
Boyle et al (1998)	U.S. Fish and Wildlife Service Mountain Trout Region	Contingent Valuation	General	$4.16
Johnson (1989)	Colorado	Contingent Valuation	Rainbow	$3.27
Johnson (1989)	Colorado	Contingent Valuation	General	$1.10
Johnson (1989)	Colorado	Contingent Valuation	General	$1.44
Johnson (1989)	Colorado	Contingent Valuation	General	$2.04
Johnson (1989)	Colorado	Contingent Valuation	General	$2.19
Johnson et al (1995)	Colorado	Contingent Valuation	General	$3.72
Johnson et al (1995)	Colorado	Contingent Valuation	General	$2.03

Author and Year	State/Region	Study Methodology	Type of Trout	Marginal Value per Fish (June 2013$)
Johnson et al (1995)	Colorado	Contingent Valuation	General	$1.85
Johnson et al (1995)	Colorado	Contingent Valuation	General	$1.71
Johnson et al (1995)	Colorado	Contingent Valuation	General	$1.55
Johnson et al (1995)	Colorado	Contingent Valuation	General	$1.42
Johnson et al (1995)	Colorado	Contingent Valuation	General	$1.28
Johnson et al (1995)	Colorado	Contingent Valuation	General	$1.14
Johnson et al (1995)	Colorado	Contingent Valuation	General	$2.18
Johnson et al (1995)	Colorado	Contingent Valuation	General	$0.90
Johnson et al (1995)	Colorado	Contingent Valuation	General	$0.69
Johnson et al (1995)	Colorado	Contingent Valuation	General	$2.30
Johnson et al (1995)	Colorado	Contingent Valuation	General	$1.71
Johnson et al (1995)	Colorado	Contingent Valuation	General	$1.38
Johnson et al (1995)	Colorado	Contingent Valuation	General	$1.14
Johnson et al (1995)	Colorado	Contingent Valuation	General	$0.98
Johnson et al (1995)	Colorado	Contingent Valuation	General	$0.87
Johnson et al (1995)	Colorado	Contingent Valuation	General	$0.77
Johnson et al (1995)	Colorado	Contingent Valuation	General	$1.02
Vaughan and Russell (1982)	USA	Travel Cost	General	$1.44
Overall Average				$1.70
Weighted Average by Study				$1.08

The criteria used to judge the correspondence between the primary valuation study and the central Colorado sites were species (trout) and geographical region (Rocky Mountain Region). Table 7.4 indicates that the value of an additional trout is within the range of $0.77 – $4.16 with a study weighted average of $2.28. This range indicates that the value for trout of $2.94, from USEPA (2006a) shown in Table 7.2, is well supported.

Johnson et al. (1995) provided a simplistic method to derive the value of the catchable target fish population from the value of fish caught, which has been provided by USEPA (2006a). The logic of this method is that the estimate of the percentage of target fish caught out of the catchable target fish population can be used to determine how many catchable fish must be added for an angler to catch another fish. Therefore, this method first estimates the ratio of the fish caught to the catchable fish population and then multiplies the value of fish caught by this ratio.

For example, if the value of catching an inland trout is $2.94 (from Table 7.2) and one trout is caught out of a catchable population of three trout, then the value of each trout in the target population is $2.94 multiplied by one-third. It must be pointed out that many problems arise from this simple solution. First, the percentage of fish caught from the catchable population now has a large influence on the value of the catchable population. Second, there are obvious dynamic effects between the number of fish caught and the number of catchable fish in the population (Ng, 2011). Whereas the modeling of such dynamic fisheries is outside the scope of this analysis, this solution provides a mechanism for transferring the results of USEPA (2006a) to the ecosystem service of catchable target fish populations.

WTP for Catchable Fish

Few studies have been dedicated to estimating the value of a catchable fish, even though such a value would prove exceptionally useful for management decisions regarding fish stocking (Ng, 2011). Loomis and Ng (2009), as well as Ng (2011), directly addressed this problem through their valuation survey of anglers at trout-stocked reservoirs in Colorado (Loomis and Ng, 2009). The goal of Loomis and Ng (2009)[3] was to inform fishery management decisions at the reservoirs by estimating net WTP for angling trips, angling days, fish caught, and fish stocked for trout and non-trout species. Loomis and Ng (2009) used survey data from 265 anglers to estimate the value of angling trips via contingent valuation and travel cost methods.

The travel cost model was selected to estimate net WTP per angler trip. The dependent variable was the number of annual trips to the site. The independent variables were the total cost of gas per person, the hourly catch rate, a dummy for motorboat use, the number of people in the party, the number of working household members, and the highest level of formal education. Mean net WTP was the inverse of the estimated coefficient for total cost of gas per person. Finally, and most importantly, the catch rate was modeled based on the number of catchable fish per acre, skill, and a dummy for whether trout was the target species.

The endpoint problem surfaced for Loomis and Ng (2009) when the survey results indicated that surveyed anglers had not traveled to their angling destinations just for fishing. They also participated in other outdoor recreation activities such as camping, whitewater rafting, hiking, horseback riding, photography, and scenery viewing (Loomis and Ng, 2009, Ng, 2011). The result was that survey respondents' stated value for an 'angling trip' included the value of recreational benefits additional to angling.

Once net WTP per angler trip was estimated, the travel cost model and the catch rate model were used to estimate the net WTP per angler day, net WTP per trout caught, and net WTP per trout stocked (Loomis and Ng, 2009). Net WTP per angler day was calculated by dividing the net WTP per angler trip by the average number of days per trip. Net WTP per trout caught and per trout stocked were calculated by first doubling the average number of catchable (stocked) trout per acre in the catch rate model, then estimating the impact on average catch rate, and finally estimating the resulting increase in number of fishing trips[4]. Finally, the resulting increase in net WTP was divided by the increased number of fish caught and by the increased number of fish

[3] Ng (2011) was the PhD dissertation that resulted, in part, from the research of Loomis and Ng (2009).

[4] Notice that this approach directly mirrors the discussion above from Boyle et al. (1998) regarding proper measurement of changes in net economic benefit (aka: net WTP).

stocked. The resulting WTP per trout caught was $25.91, and the resulting WTP per catchable trout was $0.60 in 2013 dollars.

Net WTP per angler day, trout caught, and trout stocked were all calculated from the base of the inflated trip value. This result is clear when the Loomis and Ng (2009) net WTP per angler day ($173.66) is compared to a meta-analysis of net WTP per angler day values from the recreational angling valuation literature ($67.91) (Loomis and Richardson, 2007). The same result emerges when the net WTP per trout caught ($25.91) is compared to a meta-analysis of net WTP per trout caught values from the recreational angling valuation literature ($2.94) (USEPA, 2006a).

The estimation of net WTP per stocked (catchable) trout is a novel accomplishment because it values the ecosystem service itself. It also raises an important ecosystem service valuation question. If the base value of the angling trip is inflated by the value of additional recreational benefits, then does the increase in net economic value, which results from the increased number of trips from doubling the catchable trout population, represent an ecosystem service value for catchable trout? To explore this question, the net WTP per catchable trout from Loomis and Ng (2009) will be compared with the net WTP per catchable trout derived from scaling the estimates from USEPA (2006a) by the percentage of target fish caught.

Additional ecosystem services

The economic valuation literature has the capacity to model the value of many more ecosystem services than were selected for this analysis. These ecosystem services were not modeled because USEPA data concerning biological, geological, hydrological, and spatial characteristics of intended study sites proved to be elusive. This deficiency prevented the modeling of changes in additional ecosystem services and, instead, encouraged the focus on the one ecosystem service with the most impact on total value (Button et al., 1999). Others have noted similar data difficulties when attempting to value ecosystem services related to mine site pollution (Burton et al., 1999, Button et al., 2012, Williamson et al., 2008). Nonetheless, Appendix B provides an extended discussion of the potential to model the value of additional ecosystem services impacted by mine site pollution.

Summary of Valuation Model

The benefit transfer model values catchable target fish population. Relevant value estimates are summarized n Table 7.9. This benefit transfer model can be combined with a geo-environmental model because it corresponds directly to the ecosystem services in question. By translating changes in ecosystem services to monetary values of cost and benefit, decision makers can compare services on a consistent basis when making decisions relating to ecosystem services. To complete this process for the central Colorado sites, this project applies the benefit transfer model to the outputs provided by the environmental model in Section 10.

Table 7.9. Summary Table of Values Used by the Project ($2013)

Net Economic Benefit	Value	Units
WTP per angling day	$67.91	$ per angler per day
WTP to catch another trout	$2.91	$ per trout caught
WTP per catchable trout	$0.60	$ per catchable trout

Study Site Description

Physical setting

The study areas are located in the Rocky Mountains of central Colorado. The Leadville and Gilman districts are on opposite sides of the continental divide. California Gulch drains the Leadville area. California Gulch is a tributary of the Arkansas River near its headwaters. The Arkansas River joins the Mississippi River approximately 2,400 km downstream from Leadville. The Eagle River is on the western side of the continental divide from Leadville. It is a tributary of the Colorado River.

The study area is in the ecoregion described by Bailey et al. (1994) as the Southern Rocky Mountains Steppe – Open Woodland – Coniferous Forest – Alpine Meadow Province. Average low temperatures for the area range from -16 °C in December and January to 3 °C in July and August. Average high temperatures range from -1 °C in December and January to 22 °C in July. The area receives an average of 29.4 cm of precipitation per year with 9 cm (31%) arriving in July and August.

Regional geologic setting

The geologic history of central Colorado is summarized from Wallace (1993). The history began more than 1.8 billion years ago in the Early Proterozoic with the accretion of volcanic arc and back-arc complexes to the southern margin of the Archean Wyoming craton. These rocks were deformed and then intruded by large Early and Middle Proterozoic batholiths. During Paleozoic and Mesozoic time, the Proterozoic basement complex was buried beneath several kilometers of marine and continental sediments, and it was partially exhumed during Pennsylvanian uplift. Subduction-related calc-alkalic magmatism and uplift affected the region during the Late Cretaceous-Early Tertiary Laramide orogeny. Post-subduction Oligocene and younger extension generated the north-trending Rio Grande rift zone, which was accompanied by magmatic activity. Most of the mineral deposits in the central Colorado mineral belt are associated with Oligocene subduction-related magmatism or later rift-related activity. Laramide-aged deposits are relatively small, and a few carbonate-hosted deposits may be of Mississippian age.

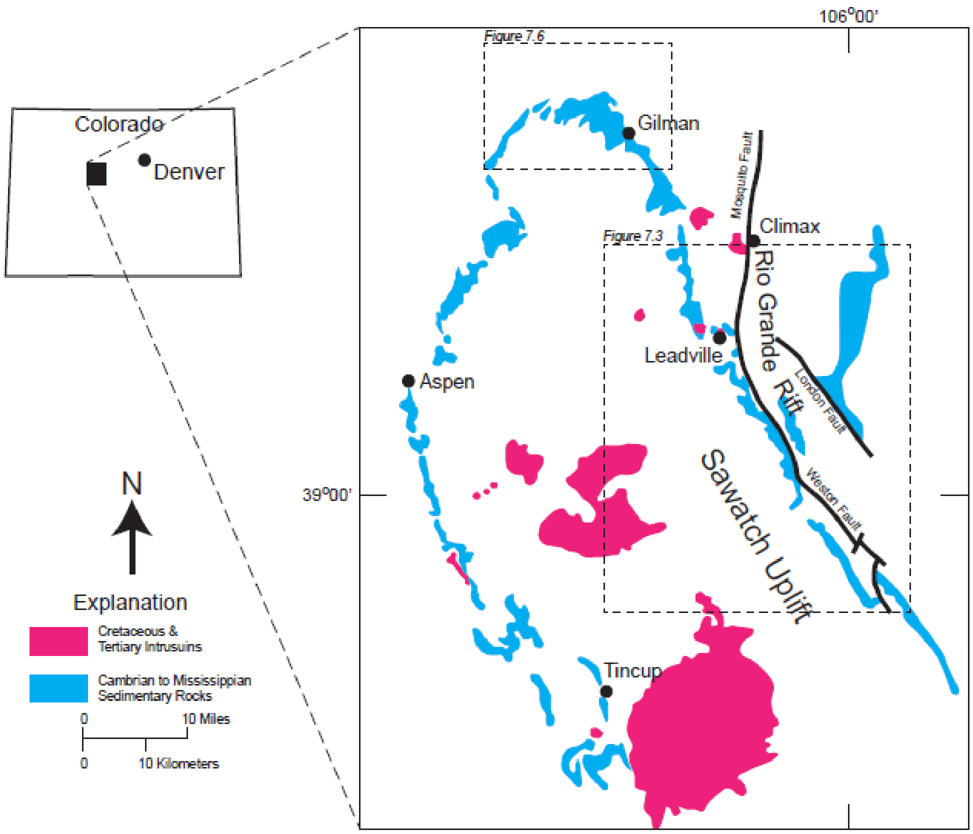

Figure 8.1. Generalized Geologic Map of Central Colorado (modified from Wallace, 1993). Blue areas are Cambrian through Mississippian sedimentary rocks. Tertiary intrusive rocks are shown in pink. Unshaded areas are undifferentiated, but span that age range from Precambrian to Quaternary.

The Leadville and Gilman districts lie on the eastern flank of the Sawatch Range. Rifting exposed Paleozoic sedimentary rocks that overlie Proterozoic granites and were intruded by Late Cretaceous and younger igneous rocks. Orogenic sediments were deposited in the graben during uplift and erosion of the adjacent Sawatch Range and Mosquito Range to the east. Quaternary glaciation further modified the landscape and locally redistributed the sediments in the district.

The oldest, volumetrically most important rocks exposed in the area are granites of the Middle Proterozoic (approximately1.4 Ga [billion years old]) Saint Kevin Granite (Tweto et al., 1978). These rocks are overlain by shallow marine Paleozoic limestones, dolomites, sandstones, and quartzites. The lower part of the stratigraphic section is composed predominantly of quartzite with subordinate amounts of carbonate rocks and shale, ranging in age from Late Cambrian to Late Devonian (Figures 8.1 and 8.2). Overlying these units are roughly 150 m of principally carbonate rocks, including the Late Devonian Dyer Dolomite, Early Mississippian or Late Devonian Gilman Sandstone, and the Early Mississippian Leadville Dolomite.

Central Colorado was subjected to major intrusive events in Late Cretaceous-Early Tertiary time (approximately 72-64 Ma [million years ago]) and again in the Middle Tertiary (43 to 39 Ma). The intrusive activity produced sills, dikes, and small stocks of granodioritic to monzogranitic composition (Bookstrom, 1990). Magmas invaded many faults, including shallow-dipping Laramide thrust faults and high-angle younger faults, forming structurally controlled dikes. The Pando Porphyry was emplaced at about 72 Ma. The Gray Porphyry includes igneous rocks formed during several early to middle Tertiary intrusive events, units include the Lincoln Gulch (66 Ma) and Evans Gulch (range in age from approximately 72 Ma to 30 Ma).

The more recent geology of the area was dominated by uplift and glaciation. After Middle Tertiary uplift of the Sawatch and Mosquito Ranges and formation of the Arkansas River valley, erosion of the ranges deposited sediments into the graben. At Leadville, erosion exposed many orebodies, which consequently became oxidized during prolonged surface exposure. As sedimentation in the graben continued during the late Tertiary, the orebodies, and probably much of the area of the modern Leadville district, were progressively covered by sediments. These poorly consolidated sediments are composed of sandy silt and interbedded sand and gravel layers.

Stratified rocks of the Leadville district dip moderately to the east, forming a homocline that, prior to Neogene rifting, once formed the eastern flank of the Sawatch uplift. This homocline is cut by a complex network of faults, most of which dip steeply, but a few of which, as noted by Thompson and Arehart (1990), are low-angle thrusts that presumably formed during the Laramide orogeny. Trends of the principal faults in the district are approximately N15°E and N20°W, consistent with the trends of major regional faults in the district (Tweto, 1960, 1968). The mineralized rocks of the district owe their exposure in large part to formation of the Rio Grande rift, a major intracontinental rift that extends northward from west Texas into at least central Colorado.

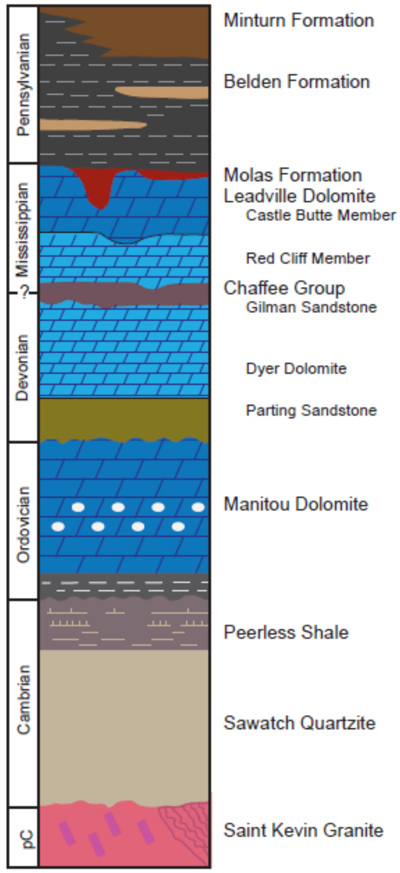

Figure 8.2: Pre-Tertiary Stratigraphic Section of the Leadville District. The replacement ore deposits of the district occur principally in the Leadville Dolomite and the Dyer Dolomite. Modified from Thompson and Arehart (1990).

Economic geology and mining history

The discovery of the carbonate-hosted sulfide deposits of the central Colorado Mineral Belt prompted much of the economic development of this part of the United States. Placer gold deposits were discovered in the Leadville area in 1860, but these deposits were essentially depleted by 1868 (Beaty et al., 1990). The carbonate replacement ores in the area were not recognized until 1874. This discovery led to a prospecting rush focused on silver from 1877 to 1879 that resulted in the identification of similar deposits at Aspen, Gilman, Red Cliff, Tincup, Kokomo, and Alma (Figure 8.1). The Black Cloud Mine in the Leadville district, the last operating carbonate-hosted mine in the region, closed in 1999. Collectively, the Leadville and Gilman districts produced over 1.5 million metric tons of zinc (Zn), 1.1 million metric tons of lead (Pb), 100 thousand metric tons of copper (Cu), 10 million kg of silver (Ag), and 100 thousand kg of gold (Au), which was recovered from over 35 million metric tons of ore (Tables 8.1 and 8.2). This study will focus on the Leadville and Gilman districts.

The deposits of the Leadville and Gilman districts are predominantly hosted by the Leadville Dolomite and the Dyer Dolomite. The Manitou Dolomite also locally hosts ores at Leadville, and the Sawatch Quartzite locally hosts ores at Gilman. The ores formed during a major mineralizing event at about 39 Ma by wholesale replacement of the Paleozoic carbonate rocks by silver-, lead-, zinc-, and gold-rich sulfide minerals. The carbonate rocks are also silicified adjacent to the orebodies. Pyrite (iron sulfide), galena (lead sulfide), and sphalerite (zinc sulfide) are the most common sulfide minerals in the replacement deposits of the Leadville district, with relatively minor amounts of chalcopyrite (copper sulfide), tennantite-tetrahedrite (copper-arsenic-antimony-sulfosalt), and magnetite (iron oxide). Silver principally occurred as argentite (silver sulfide) with some argentiferous tetrahedrite, and gold is in its native form (Tweto, 1968). Manganosiderite (manganese-iron carbonate) and quartz (mostly as fine-grained jasperoid) are the principal nonsulfide gangue minerals (Beaty, 1990, Beaty et al., 1990, Thompson and Arehart, 1990, Wallace, 1993). At both Leadville and Gilman, mineralization formed as mantos, veins, and chimneys. Mantos are tabular replacement deposits typically confined by the surrounding sedimentary stratigraphy. Chimneys are funnel-shaped bodies that may represent feeder zones for the manto deposits. Igneous rocks are inferred to have been the source of both the metals and sulfur for the Leadville-type deposits, and acid neutralization through reaction of saline hydrothermal fluids with the carbonate hosts rocks is thought to have been the primary depositional process for these ores (Beaty et al., 1990).

In the Leadville district, the orebodies are developed around the Breece Hill stock and cover an area of approximately 6 km by 5 km (Thompson and Arehart, 1990). Typical sulfide ore grades range from 3 to 8% lead, 6 to 30% zinc, 68 to 204 g/metric ton silver, and 1.7 to 7 g/metric ton gold. The Zn:Pb ratios ranged from 1:1 to 4:1. Copper was present, but not in sufficient quantities to warrant recovery. The main orebody at Gilman (Eagle Mine) supported most of the mine production and covered an area of approximately 1 km by 2 km east of the Eagle River. The ore processed at the mill from 1929 to 1977 averaged 2.0% lead, 11.6% zinc, 0.2% copper, and 37.5 g/metric ton Ag (Beaty, 1990) with a Zn:Pb ratio of 5.8:1.

Table 8.1. Metal Produced from Selected Carbonate-Hosted Districts in Central Colorado through 1987 (from Beaty et al., 1990).

District	Zinc metric tons	Lead metric tons	Copper metric tons	Silver kilograms	Gold kilograms
Gilman (Eagle)	866,000	147,000	96,000	2,116,000	12,400
Leadville	714,000	1,000,000	48,000	8,087,000	97,600
Kokomo	34,000	14,000	0	56,000	800
Aspen	10,000	267,000	minor	3,140,000	0
Sherman	0	3,000	0	228,000	0
Tincup	4,000	75,000	500	1,302,000	0

Table 8.2. Cumulative Grade and Tonnage of Carbonate-Hosted Ore Produced in the Central Colorado Mineral Belt from Selected Districts (from Beaty et al., 1990).

District	Data Interval	Ore metric tons	Zinc %	Lead %	Copper %	Silver mg/kg	Gold mg/kg
Gilman (Total)	1880 - 1987	11,400,000	8.8	1.5	0.9	220	1.4
Gilman manto ores	1880 - 1987	8,600,000	11.6	2.0	0.2	38	0.7
Gilman chimney ores	1914 - 1987	2,800,000	0.0	0.0	3.0	777	3.4
Kokomo	1905 - 1965	466,029	7.3	3.0	0.05	120	1.8
Leadville	1873 - 1987	23,800,000	3.0	4.2	0.2	320	3.7
Aspen	1880 - 1987	4,000,000	2.0	8.0	0.0	1,000	0.0
Sherman	1973 - 1984	645,005	4.0	0.8	0.1	485	0.0

Hydrologic setting

California Gulch

California Gulch is a 17-km long tributary of the Arkansas River that joins the Arkansas River less than 32 km from its headwaters (Figure 8.3). The discharge from upper Arkansas River is dominated by snowmelt typically peaking between early to mid-June (Figures 8.4 and 8.5). The peak typically wanes throughout the summer, reaching base-flow conditions in early fall (Figure 8.5). The annual variation in discharge in the Arkansas River spans nearly two orders of magnitude. The streambed is predominantly medium to large cobbles underlain by pebbles and coarse sand (Clements et al., 2010).

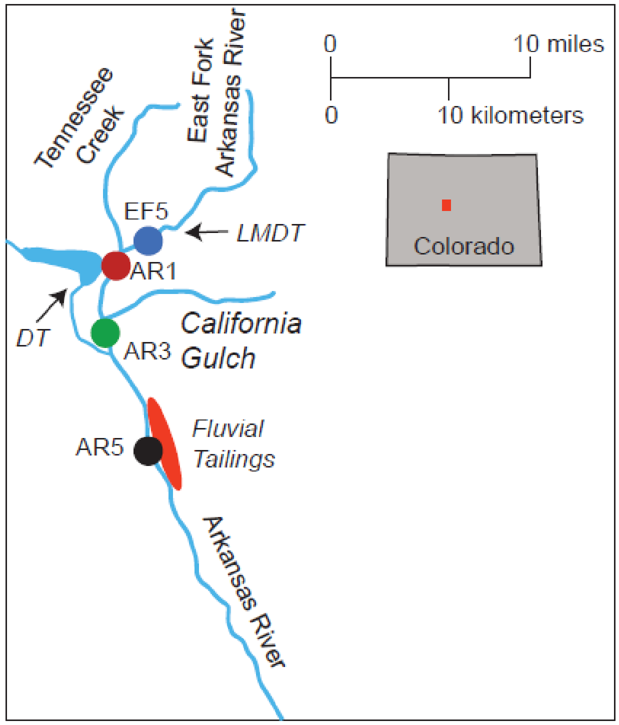

Figure 8.3. Map of Upper Arkansas River Showing Sample Sites. California Gulch, LMDT: Leadville Mine Drainage Tunnel, and DT: Dinero Tunnel (modified from Clements et al., 2010).

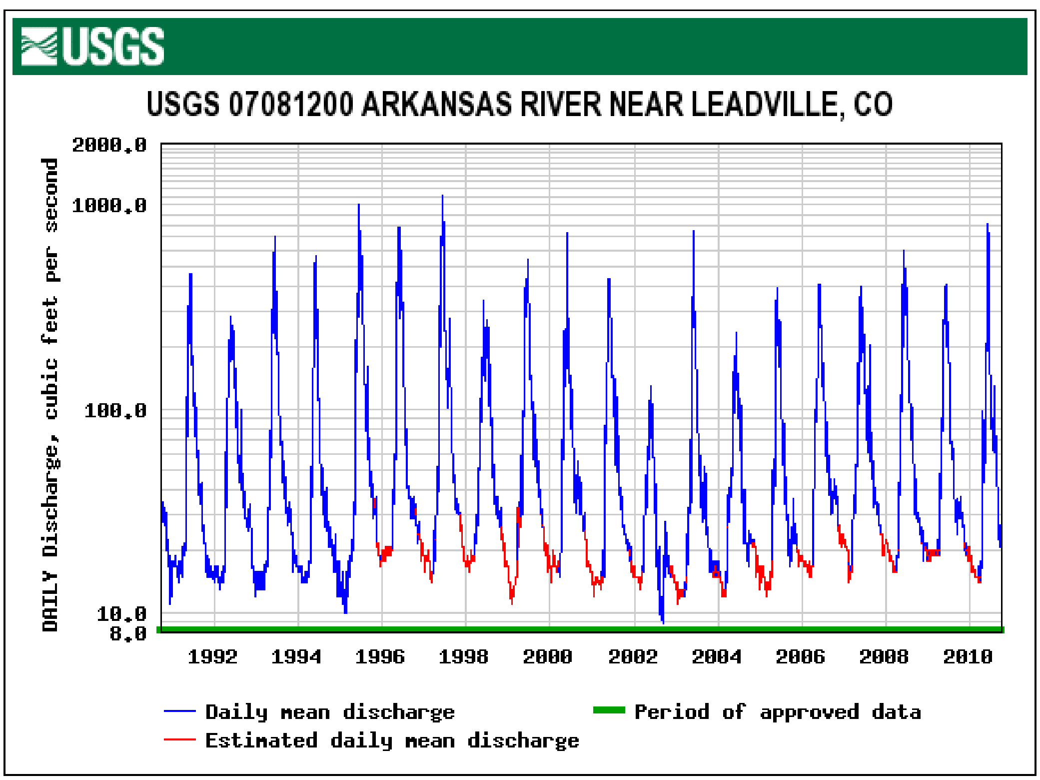

Figure 8.4. Variation of Daily Discharge in the Arkansas River Upstream from California Gulch from 1990 – 2010 (https://waterdata.usgs.gov/nwis).

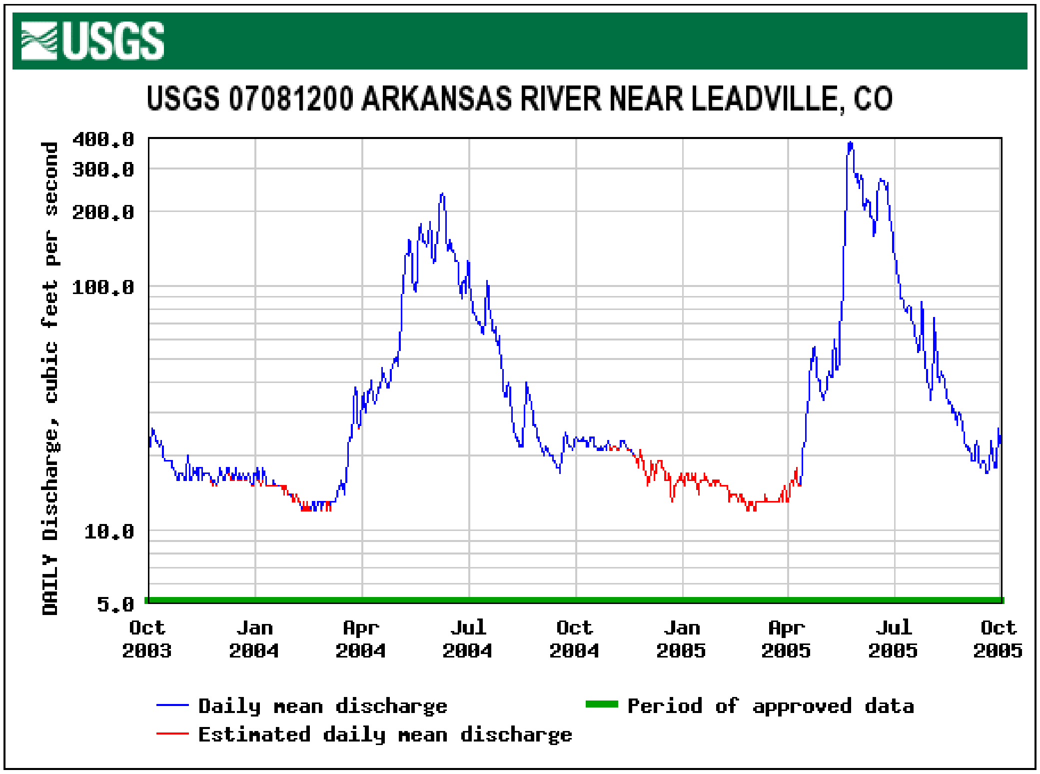

Figure 8.5. Detail of the Variation of Daily Discharge in the Arkansas River Upstream from California Gulch from October 1, 2003 – October 1, 2005 (https://waterdata.usgs.gov/nwis).

Eagle River

The Eagle Mine is located on the northeastern bank of the Eagle River approximately 33 km downstream from its headwaters (Figure 8.6). Similar to the Arkansas River, the discharge from Eagle River is dominated by snowmelt typically peaking between early to mid-June (Figures 8.7 and 8.8). The annual variation in discharge in the Eagle River spans nearly two orders of magnitude.

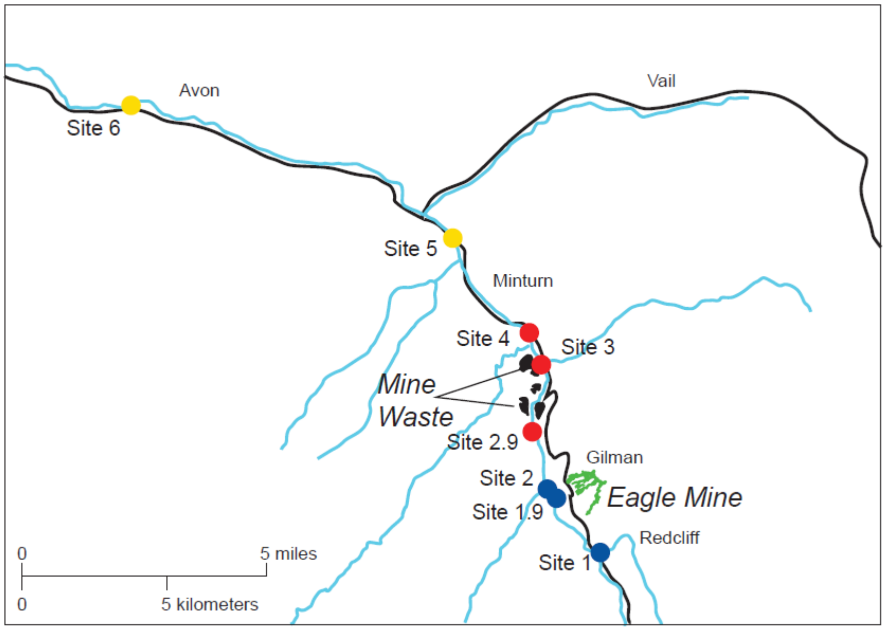

Figure 8.6. Map Showing Sampling Sites along the Eagle River used in this Study from Woodling et al. (2005). Potential sources of contamination exist from just north of Site 1 to just south of Site 4. The Redcliff Mine workings lie along Turkey Creek (not shown), which flows southwest to the Eagle River, entering near Site 1. The Eagle Mine workings are located on the northeast side of the Eagle River between Sites 1 and 2. Mine waste piles were located on both sides of the river between Sites 2.9 and 4. Modified from Woodling *et al.* (2005).

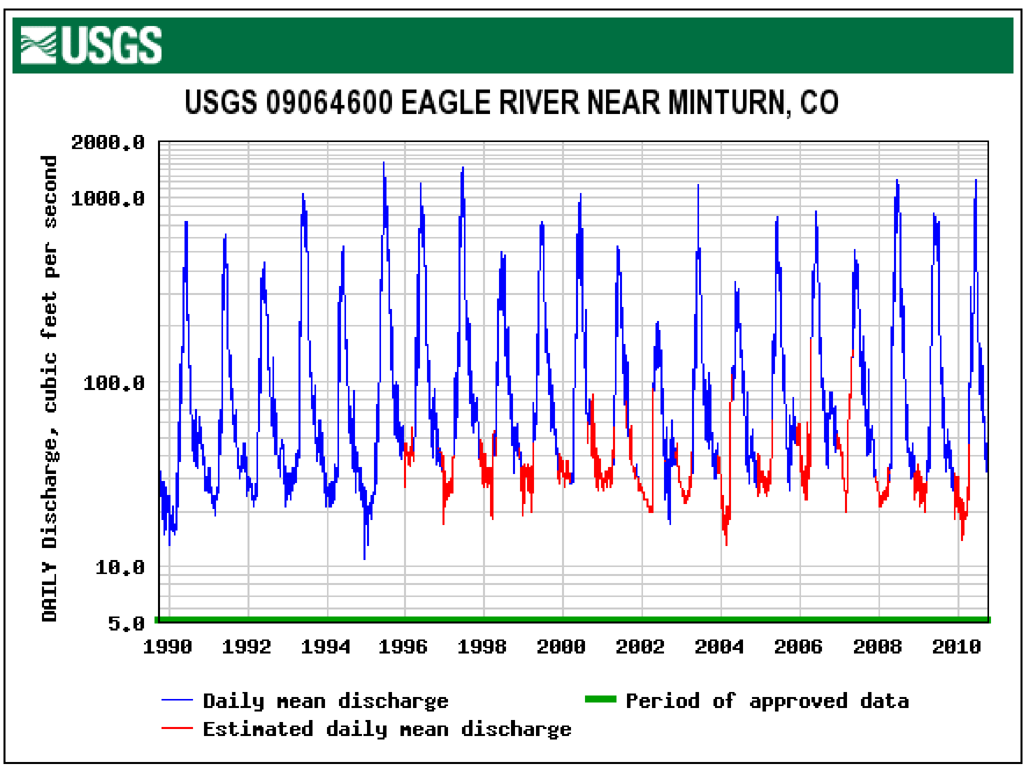

Figure 8.7. Variation of Daily Discharge in the Eagle River Downstream from the Eagle Mine from 1990 – 2010 (https://waterdata.usgs.gov/nwis).

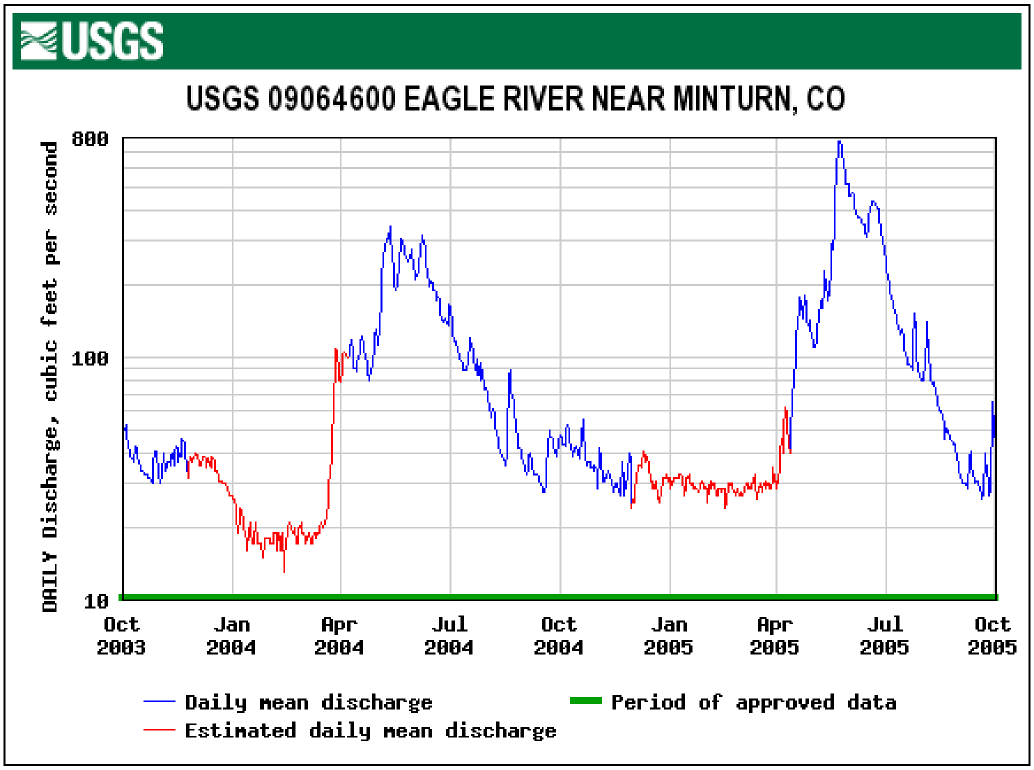

Figure 8.8. Detail of the Variation of Daily Discharge in the Eagle River Downstream from the Eagle Mine from October 1, 2003 – October 1, 2005 (https://waterdata.usgs.gov/nwis).

Water-quality variations

California Gulch

The long-term water quality of the Arkansas River is characterized by near neutral pH and moderate specific conductance. Daily measurements of pH and specific conductance were taken from 1990 to 1997 (Figures 8.9 and 8.10). The pH value generally ranged between 7.5 and 8.5 with limited excursions above and below this range and little or no seasonal variation. In contrast, a distinct seasonal variation in specific conductance is evident with the lowest values (greatest dilution) occurring in the late spring during snow melt and the highest values occurring during base-flow conditions in the late summer through winter (Figure 8.10). The specific conductance generally ranged between 50 and 300 µS/cm.

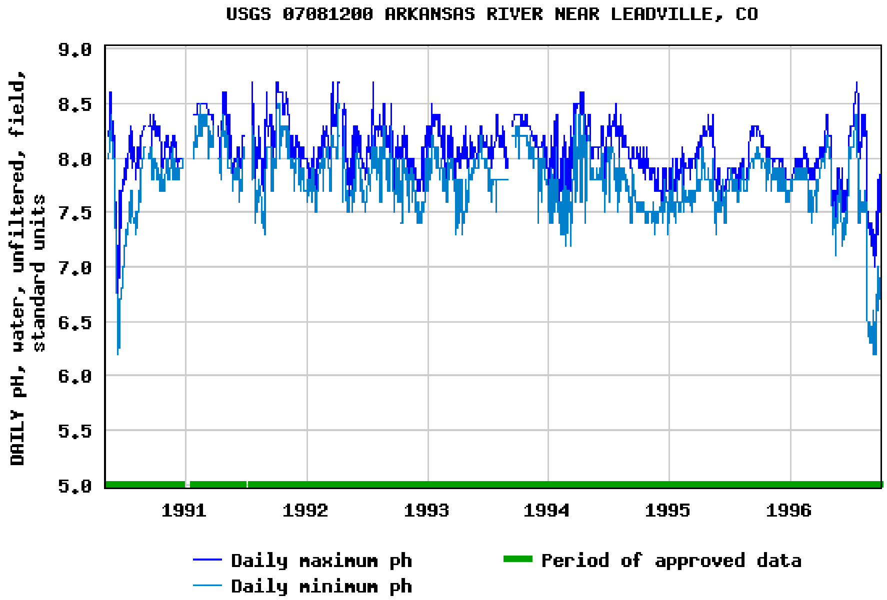

Figure 8.9. The Variation of pH from 1990 – 1997 in the Arkansas River Upstream from California Gulch. From the USGS National Water Information System (http://nwis.waterdata.usgs.gov/usa/nwis/qwdata/).

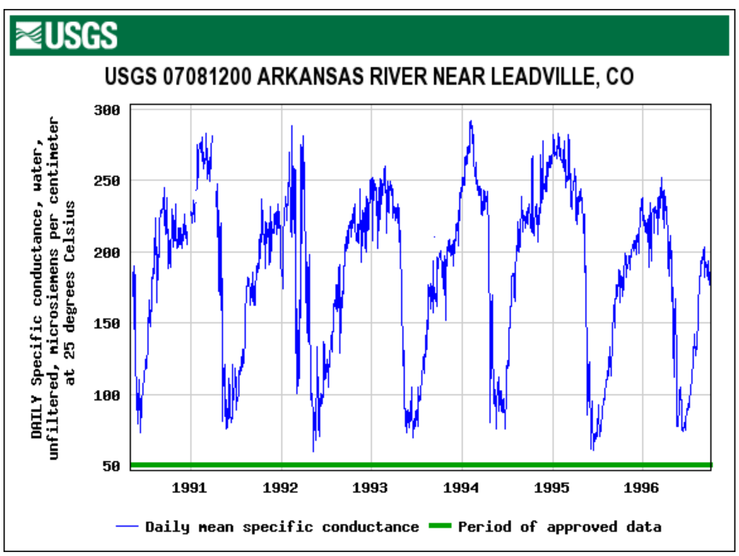

Figure 8.10. The Variation of Specific Conductance from 1990 – 1997 in the Arkansas River Upstream from California Gulch. From the USGS National Water Information System (http://nwis.waterdata.usgs.gov/usa/nwis/qwdata/).

Water-quality data, including pH, specific conductance, alkalinity, hardness, and trace element (Zn, Cd, Cu) concentrations combined with biological data for the Arkansas River in the vicinity of the Leadville district are provided by Clements et al. (2010) for the time period from 1989 to 2006. The longer-term data from Clements et al. (2010) are similar to the USGS data for pH and specific conductance shown in Figures 8.9 and 8.10. The alkalinity, hardness, and trace element concentrations do not show any significant correlations with pH (Figures 8.11, 8.12, and 8.13). Neither alkalinity nor hardness are shown to have any systematic variation upstream (sites EF5 and AR1) and downstream (sites AR3 and AR5) from California Gulch (Figures 8.11 and 8.12).

Significant differences in trace metals are evident upstream vs. downstream from California Gulch. The downstream sites (AR3 and AR5) have higher dissolved zinc concentrations than the upstream sites (EF5 and AR1, Figure 8.13). Dissolved cadmium and zinc have a general correlation at higher concentrations (Figure 8.14), presumably reflecting a common source – the mineral sphalerite in mine waste (Seal and Hammarstrom, 2003). Zinc is typically approximately 10 to 1,000 times more abundant than cadmium, on a mass basis, in the watershed. Compared to copper, dissolved zinc is generally 1 to 1,000 times more abundant (Figure 8.15).

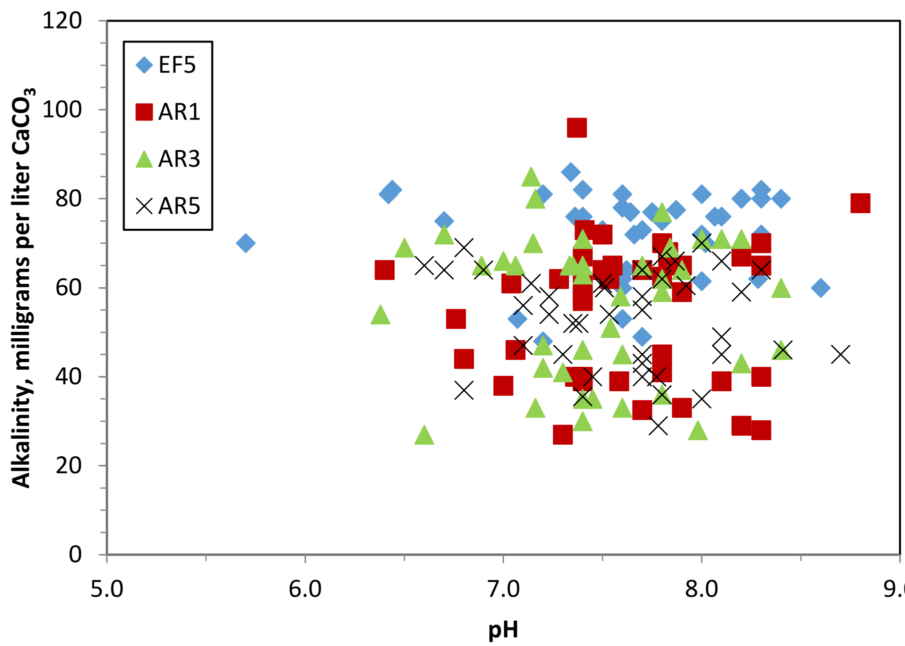

Figure 8.11. Graph Showing pH and Alkalinity at Four Sites in the Arkansas River. Site locations are shown in Figure 8.3. Data from Clements et al. (2010).

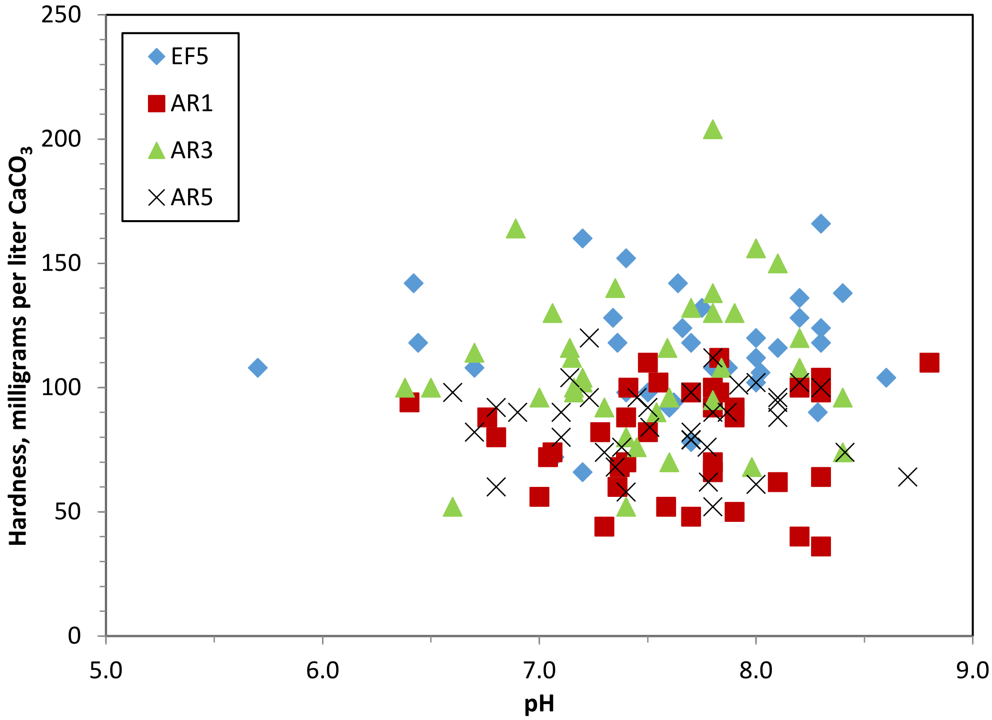

Figure 8.12. Graph Showing pH and Hardness at Four Sites in the Arkansas River. Site locations are shown in Figure 8.3. Data from Clements et al. (2010).

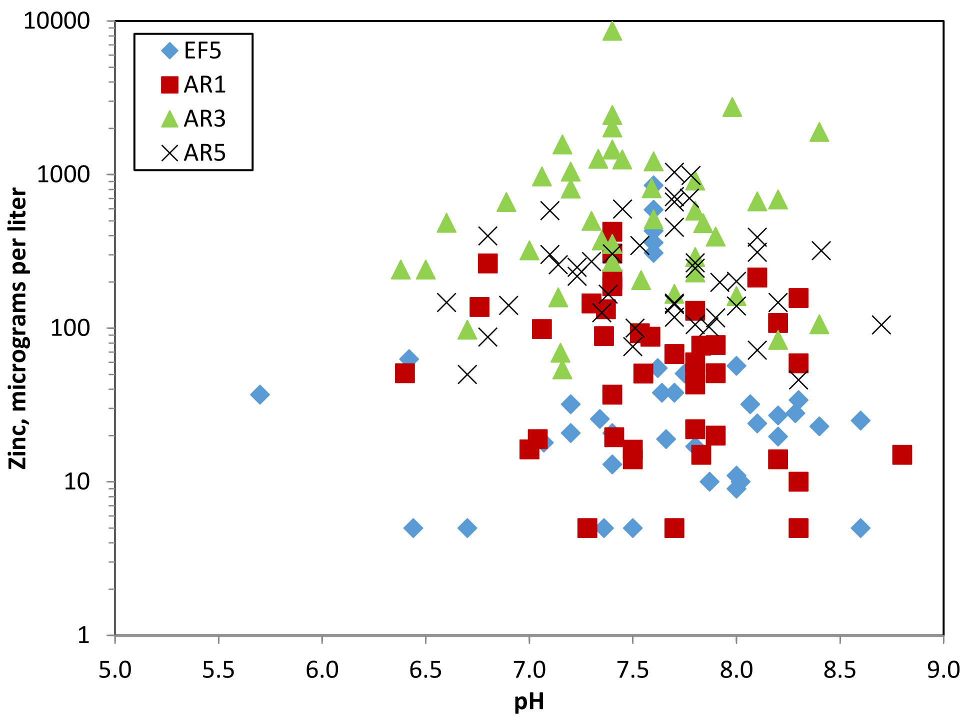

Figure 8.13. Graph Showing pH and Dissolved Zinc Concentration at Four Sites in the Arkansas River. Site locations are shown in Figure 8.3. Data from Clements et al. (2010).

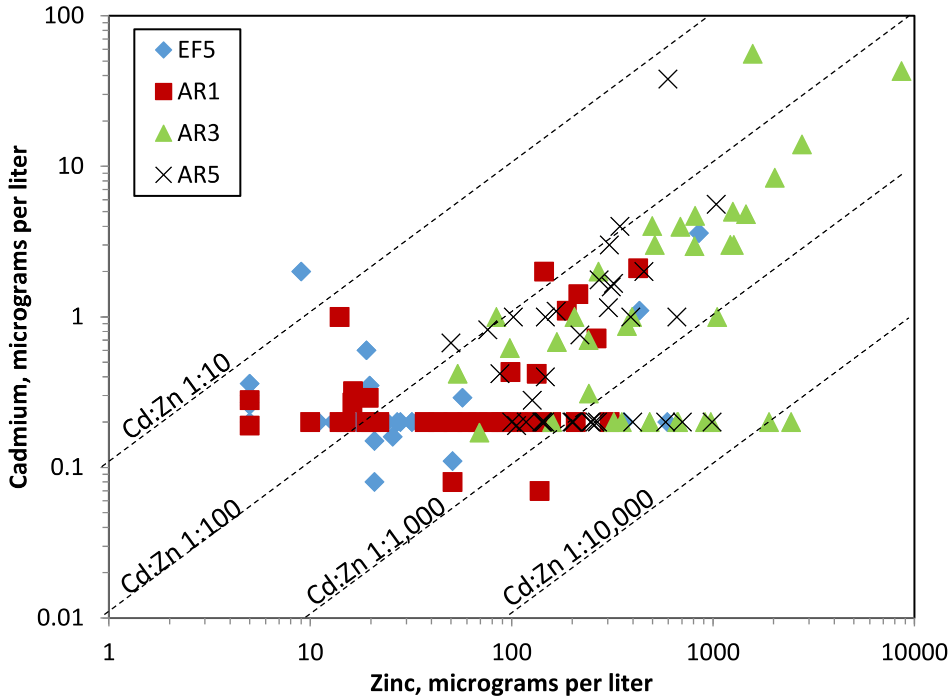

Figure 8.14. Graph Showing Dissolved Zinc and Dissolved Cadmium Concentrations at Four Sites in the Arkansas River. Values that are below the analytical detection limit were plotted as the detection limit, as is evident in many of the cadmium values falling at 0.2 µg/L. Site locations are shown in Figure 8.3. Data from Clements et al. (2010).

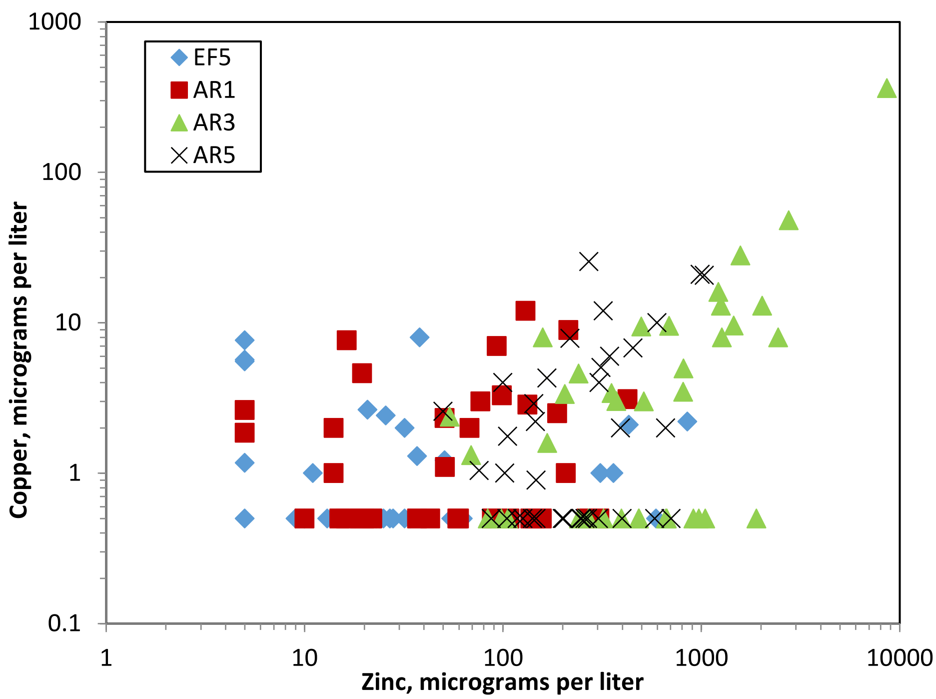

Figure 8.15. Graph Showing Dissolved Zinc and Dissolved Copper Concentrations at Four Sites in the Arkansas River. Values that are below the analytical detection limit were plotted as the detection limit, as is evident in many of the copper values falling at 0.5 µg/L. Site locations are shown in Figure 8.3. Data from Clements et al. (2010).

Eagle River

This river's long-term water quality is characterized by near neutral pH and moderate specific conductance. Daily measurements of pH and specific conductance were taken periodically from 1989 to 2015 (Figures 8.16 and 8.17). The pH values generally ranged between 7.5 and 8.5, which is similar to the range observed in the Arkansas River near Leadville. The specific conductance range in the Eagle River was similar to that observed in the Arkansas River near Leadville.

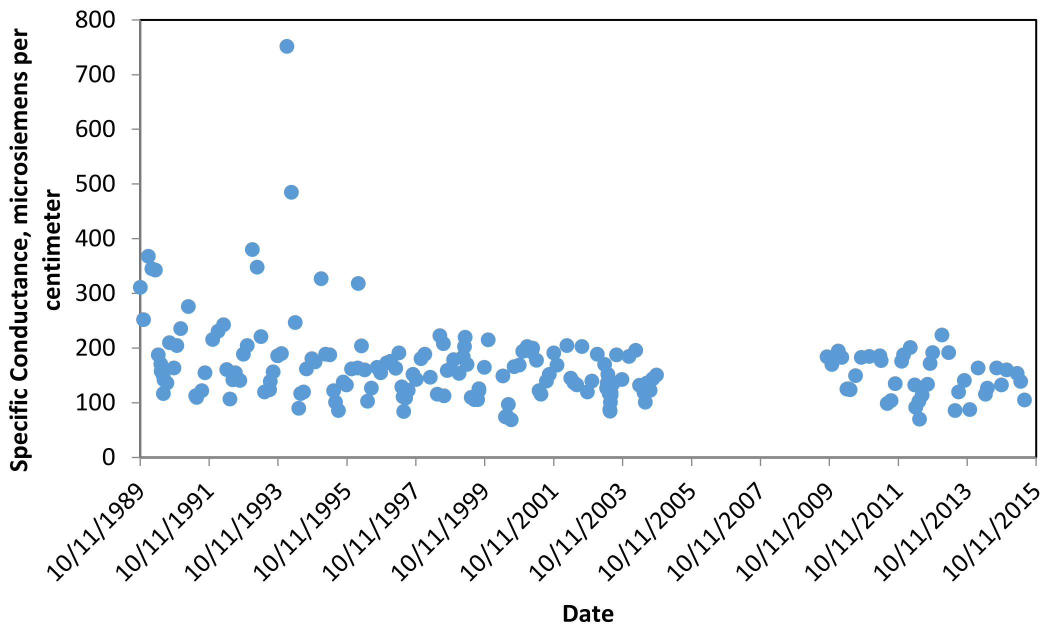

Figure 8.16. The Variation of Specific Conductance from 1989 – 2015 in the Eagle River Downstream from the Eagle Mine (USGS Site ID 09064600). From the USGS National Water Information System (http://nwis.waterdata.usgs.gov/usa/nwis/qwdata/).

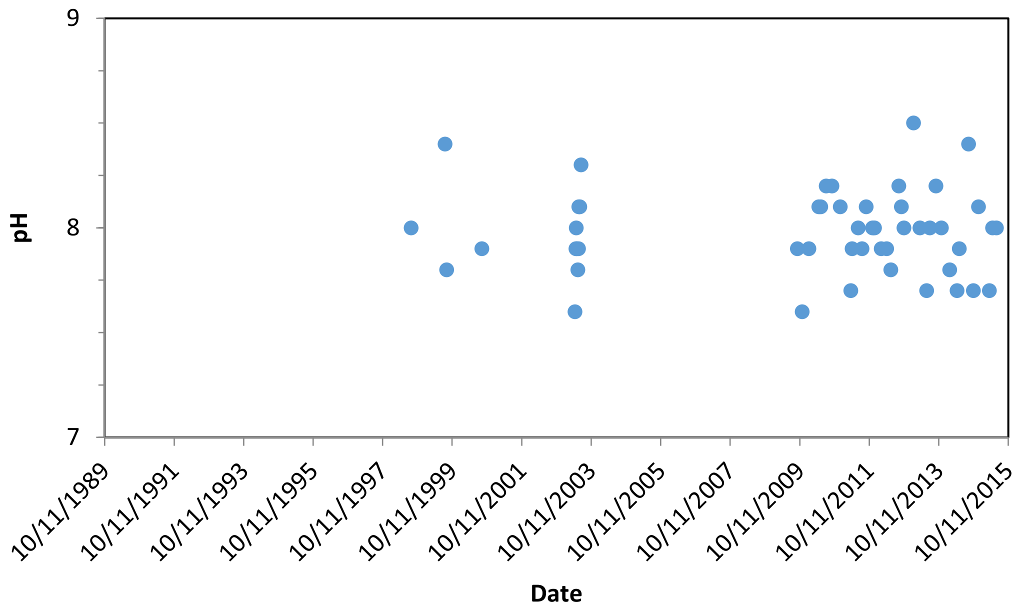

Figure 8.17. The Variation of pH from 1989 – 2015 in the Eagle River Downstream from the Eagle Mine (USGS Site ID 09064600). From the USGS National Water Information System (http://nwis.waterdata.usgs.gov/usa/nwis/qwdata/).

Water-quality data, including pH, alkalinity, hardness, and trace element (Zn, Cd, Cu) concentrations combined with biological data for the Eagle River in the vicinity of the Eagle Mine are provided by Woodling et al. (2005) for the time period from 1990 to 2005. The relationships among major and minor water-quality parameters are somewhat different from those observed in the Arkansas River near Leadville. The alkalinity, hardness, and trace elements concentrations show moderate correlations with pH (Figures 8.18, 8.19, and 8.20). Neither alkalinity nor hardness indicates any systematic variation upstream (Sites 1 – 2) and downstream (Sites 2.9 – 6) from the Eagle Mine, although the upstream and farthest downstream sites tend to show the highest pH, hardness, and alkalinity values and the sites from within the mine zone tend to have the lowest pH and highest dissolved zinc values (Figures 8.18 and 8.19). In contrast, the downstream sites (except for Site 6 where zinc concentrations are lower than Site 3- because of the diluting effect and higher pH of Gore Creek) have higher dissolved zinc concentrations than the upstream sites (Figure 8.20). Dissolved cadmium and zinc also have a general correlation (Figure 8.21). Zinc is typically approximately 100 to 1,000 times more abundant than cadmium on a mass basis in the watershed. A significant correlation is not apparent for zinc and copper (Figure 8.22).

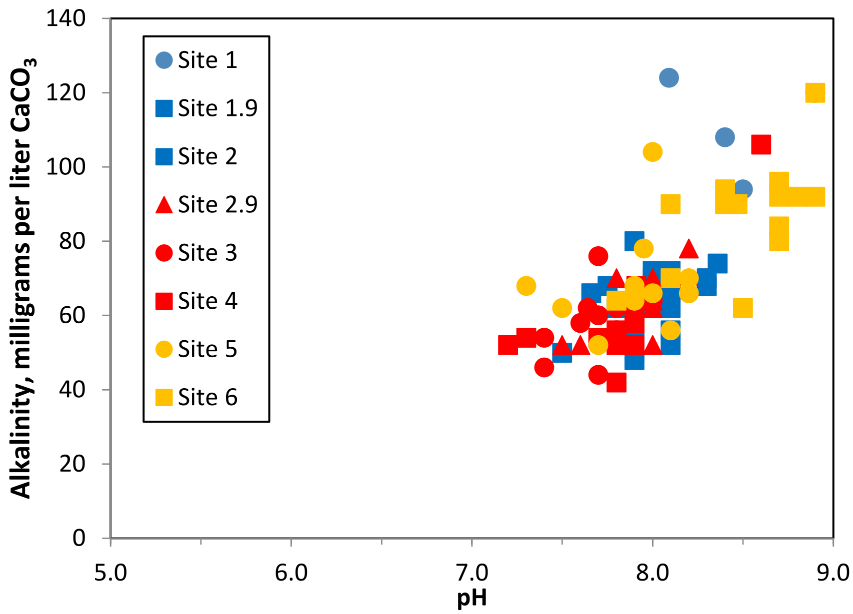

Figure 8.18. Graph Showing pH and Alkalinity at Eight Sites in the Eagle River. Site locations are shown in Figure 8.6. Data from Woodling et al. (2005). Blue symbols are sites upstream from significant mining activity, red symbols represent sites from the reach of mining impacts on the land surface, and yellow symbols represent sites downstream from surface disturbance related to mining.

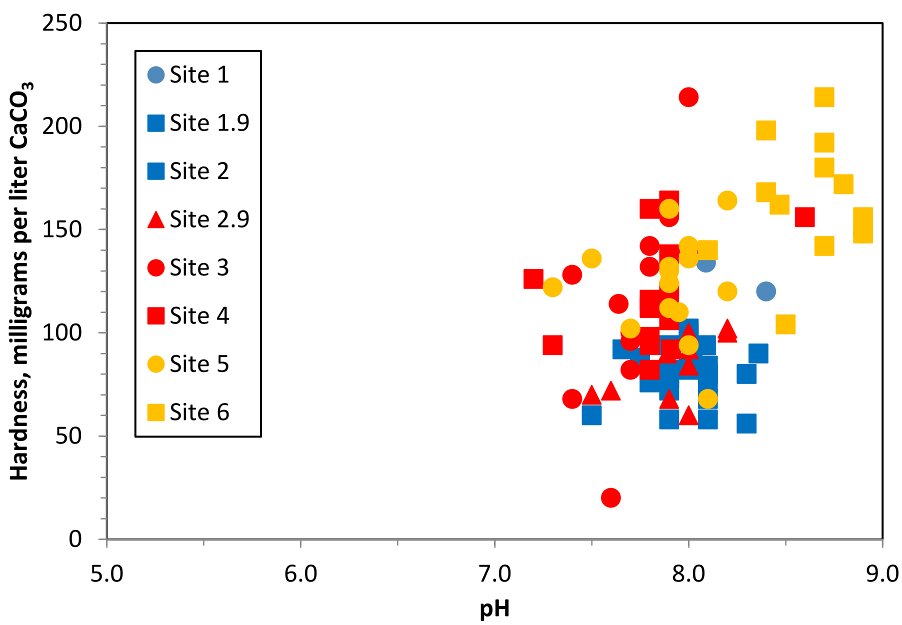

Figure 8.19. Graph Showing pH and Hardness at Eight Sites in the Eagle River. Site locations are shown in Figure 8.6. Data are from Woodling et al. (2005). Blue symbols are sites upstream from significant mining activity, red symbols represent sites from the reach of mining impacts on the land surface, and yellow symbols represent sites downstream from surface disturbance related to mining.

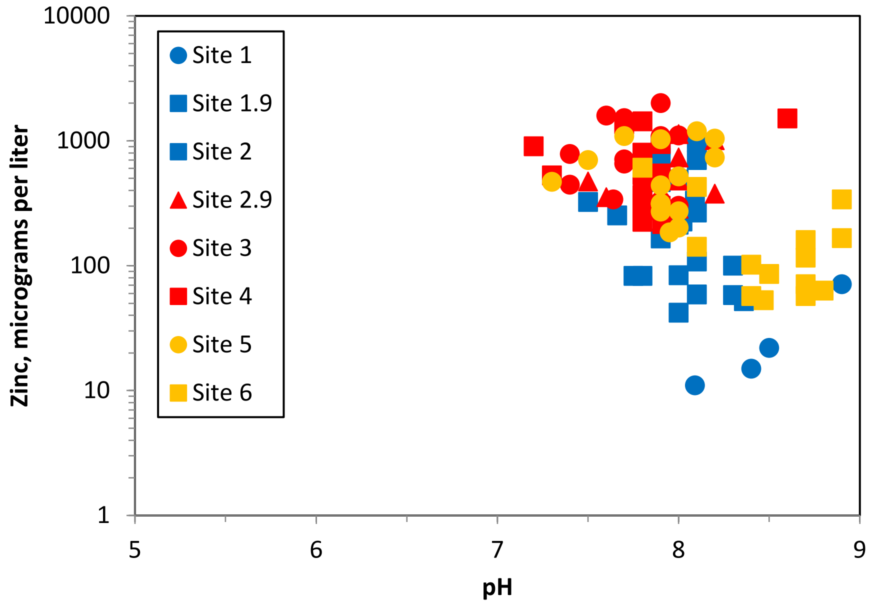

Figure 8.20. Graph Showing pH and Dissolved Zinc Concentrations at Eight Sites in the Eagle River. Site locations are shown in Figure 8.6. Data are from Woodling et al. (2005). Blue symbols are sites upstream from significant mining activity, red symbols represent sites from the reach of mining impacts on the land surface, and yellow symbols represent sites downstream from surface disturbance related to mining.

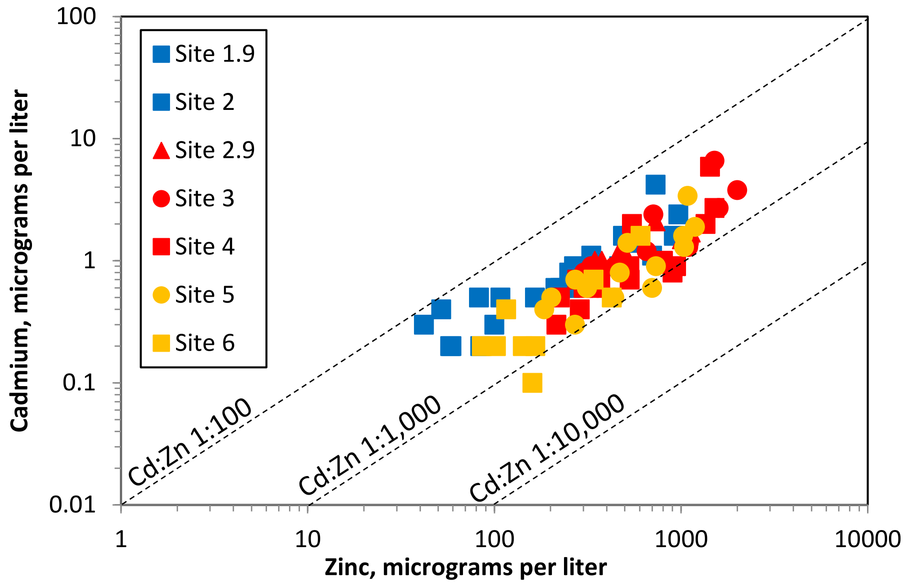

Figure 8.21. Graph Showing Dissolved Zinc and Dissolved Cadmium Concentrations at Eight Sites in the Eagle River. Site locations are shown in Figure 8.6. Data are from Woodling et al. (2005). Blue symbols are sites upstream from significant mining activity, red symbols represent sites from the reach of mining impacts on the land surface, and yellow symbols represent sites downstream from surface disturbance related to mining.

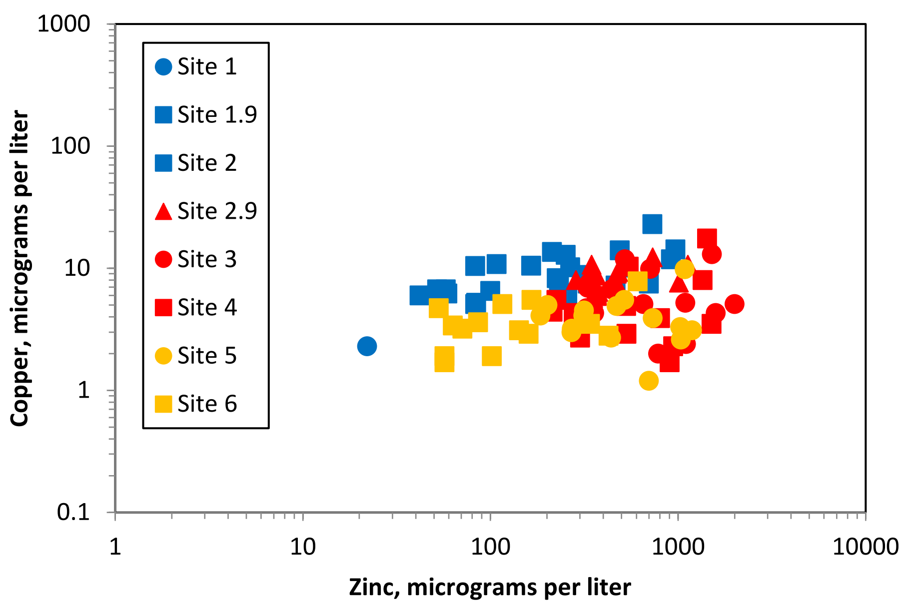

Figure 8.22. Graph Showing Dissolved Zinc and Dissolved Copper Concentrations at Eight Sites in the Eagle River. Site locations are shown in Figure 8.6. Data are from Woodling et al. (2005). Blue symbols are sites upstream from significant mining activity, red symbols represent sites from the reach of mining impacts on the land surface, and yellow symbols represent sites downstream from surface disturbance related to mining.

Mining-environmental landscape

Leadville District (California Gulch)

The mining-environmental landscape of the Leadville district is adequately described by the Operable Units defined by EPA for the site. The California Gulch (Leadville) Superfund Site includes 47 km^2 (18 mi^2) of affected land. EPA has divided the site into 12 operable units that include mine drainage, surface water, groundwater quality, sediment, waste rock, mill tailings (in situ and fluvially transported), smelter sites including pyrometallurgical slag, and residential soils (Table 8.3). California Gulch includes many operable units where remediation activities have been completed and the operable units have been "deleted" from the National Priorities List.

Table 8.3. Summary of operable units (OUs) at the California Gulch (Leadville) Superfund site.

OU	Name	Description	Status
1	Yak Tunnel	Mine drainage	Ongoing treatment
2	Malta Gulch	Mill tailings	Deleted 7/23/2001
3	Denver and Rio Grande Railroad Slag Piles, Railroad Easement, Railroad Yard, and Mineral Belt Trail	Slag	Operation and Maintenance
4	Upper California Gulch	Surface water	Deleted 10/24/2014
5	ASARCO Smelter/Colorado Zinc-Lead Site	Smelter and mill site, smelter waste, waste rock, mill tailings	Deleted 10/24/2014
6	Stray Horse Gulch	Waste rock, mill tailings, surface water, groundwater	Remedial design
7	Apache Tailings	Mill tailings	Deleted 10/24/2014
8	Lower California Gulch	Mill tailings, soils, waste rock, stream sediments	Deleted 1/12/2010
9	Populated residential areas	Soils	Deleted 9/21/2011
10	Oregon Gulch	Mill tailings	Deleted 4/16/2001
11	Arkansas River floodplain	Fluvial tailings	Field work completed
12	Site-wide surface water and groundwater quality	Surface water, groundwater	Remedial design

Remediation of California Gulch and adjacent areas has been implemented in stages. The Leadville Mine Drainage Tunnel water treatment plant, operated by the U.S. Bureau of Reclamation, was completed in 1992. The Yak Tunnel water treatment plant, operated by ASARCO, was completed in 1992. Tailings remediation in California Gulch and restoration of riparian areas was completed in 1999 (Clements et al., 2010).

Gilman District (Eagle Mine)

The mining-environmental landscape of the Gilman district is adequately described by the Operable Units defined by EPA for the site. The Eagle Mine Superfund site is much smaller than the California Gulch Superfund site. The Eagle Mine Superfund site only covers 95 hectares (235 acres). EPA has divided the site into three operable units that include mine drainage, surface water, groundwater quality, soils, waste rock, mill tailings, and roaster piles (Table 8.4).

Table 8.4. Summary of Operable Units (OUs) at the Eagle Mine (Gilman) Superfund Site

OU	Name	Description	Status
1	Eagle Mine, Roaster Pile, Waste Rock Piles, Rex Flats, Old Tailings Pile, Consolidate Tailings Pile	Mine drainage, surface water, groundwater, waste rock, mill tailings	Ongoing treatment, site remediation completed by 2001
2	Gilman	Soils	
3	North Property	Mill tailings, roaster piles	Remedial investigation

Remediation of the Eagle Mine and adjacent areas has been implemented in stages. A permanent water treatment plant was constructed in 1990. The water treatment plant treats water collected from the mine, groundwater beneath the tailings pile, and contaminated surface and groundwater collected from multiple locations across the site. Most solid mine waste remediation was completed by 2001, but additional relocation of roaster waste near Gilman occurred in 2006.

Environmental Setting

Water

Influences on water quality

Mining influenced water (MIW) is water that has had its chemical composition affected by mining or mineral processing activities (Wildeman and Schmiermund, 2004). The downstream effects of MIW on aquatic organisms throughout the remediation histories of the sites are best considered in terms of the exceedance of water quality guidelines for metals of concern (Zn, Cd, and Cu) in these watersheds. The toxicity of Zn, Cd, and Cu to aquatic organisms can be modeled as a function of water hardness (Figures 7.12 and 7.19, USEPA, 2006b). Thus, the inferred toxicity of dissolved metals (Zn, Cd, and Cu) in both rivers can be modeled using the hardness-dependent aquatic life criteria in conjunction with site-specific water quality data from the Arkansas River and the Eagle River. The Arkansas River water-quality data are from Clements et al. (2010) and the Eagle River data are from Woodling et al. (2005). The concentration of a metal for acute toxicity is expressed in terms of a criteria maximum concentration (CMC), which can be compared to metal concentrations at sample sites. Dissolved metal concentrations can be compared, or normalized, to their respective CMCs through a hazard quotient (HQ), as described in Equation 9.1 below:

$$HQ = m/c \qquad (9.1)$$

where m is the measured concentration of the metal and c is the hardness-adjusted CMC for that metal in that specific sample. The simultaneous toxicity of multiple metals, such as Zn, Cd, and Cu, can be evaluating by summing hazard quotients for individual metals in a sample as a hazard index (HI), also known as a cumulative criterion unit (CCU), expressed as shown in Equation 9.2 below:

$$HI = CCU = \Sigma m_i/c_i \qquad (9.2)$$

where m_i is the measured concentration of the ith metal and c_i is the hardness-adjusted CMC for the ith metal. An HQ or HI less than 1 implies metal concentrations that should not be toxic to aquatic organisms. An HQ or HI above 1 implies toxic conditions.

In the Arkansas River, the cumulative (Zn, Cd, and Cu) hazard indices for the upstream reference sites (EF5, AR1) averaged 0.95 and 0.78, respectively, over the course of the study (1991 – 2006), and those for the downstream sites (AR3, AR5) averaged 4.76 and 1.68, respectively. In the Eagle River, the hazard indices averaged below 1 for the upstream reference site (Site 1), and the sites from the reach with mine disturbance and downstream from mine disturbance ranged from 0.99 (in the most downstream site) to 10.35 over the course of the study (1990 – 2005). In other words, dissolved concentrations of zinc, cadmium, and copper locally exceed the aquatic life guidelines.

The predicted relative contributions of zinc, cadmium, and copper to the aquatic toxicity of MIW in the Arkansas River are Zn 68%, Cd 21%, and Cu 11% (Figure 9.1), in the Eagle River, they are Zn 80%, Cd 9%, and Cu 11% (Figure 9.2). Thus, in both watersheds, zinc is the dominant aquatic stressor and will be the focus of the following discussion.

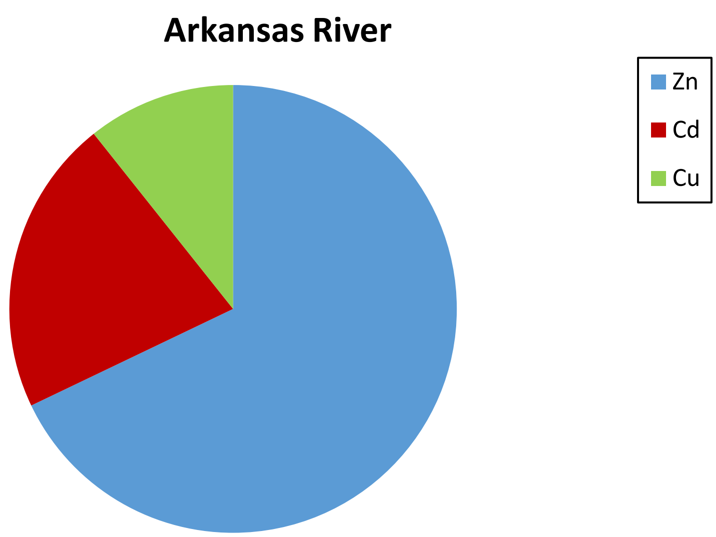

Figure 9.1. Pie Diagram Showing the Relative Contributions of Zinc, Cadmium, and Copper to the Predicted Toxicity (Expressed as Hazard Index) of Water in the Arkansas River Downstream of California Gulch.

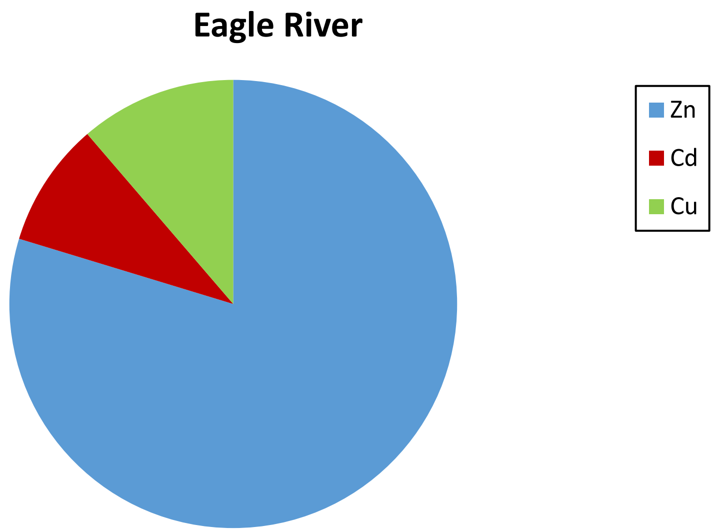

Figure 9.2. Pie Diagram Showing the Relative Contributions of Zinc, Cadmium, and Copper to the Predicted Toxicity (Expressed as Hazard Index) of Water in the Eagle River Downstream of the Eagle Mine.

The variations among sites with time in terms of predicted toxicity for the combination of Zn, Cd, and Cu compared to Zn alone, the predominant contaminant of concern, are very similar (Figures 9.3 and 9.4). The similarities include upstream and downstream sites, influxes of metals, and responses to remediation. Thus, it can be concluded that zinc is a reasonable proxy for predicted metal toxicity in the Eagle River watershed. Similar conclusions can be reached for the Arkansas River near Leadville (Figure 9.5).

For the Eagle River, there has been a general, but erratic, decrease in predicted toxicity of Zn, Cd, and Cu over the course of the remediation project, although none of the sites near the sources are below predicted toxicity limits as of 2005, the last year of data in the report (Figures 8.3 and 8.4, data from Woodling et al. (2005)). In contrast for the Arkansas River, all of the sites have achieved, with time, zinc concentrations below that predicted acute toxicity limit (Figure 9.5).

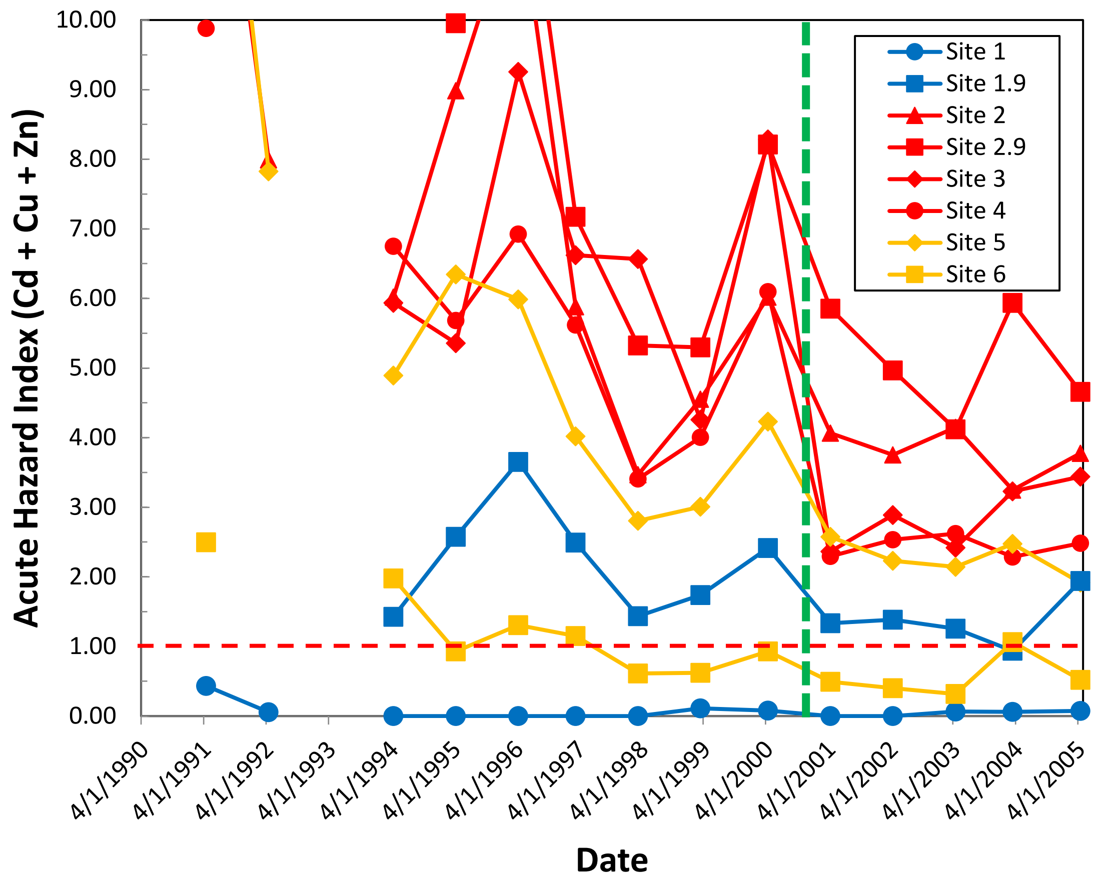

Figure 9.3. Variations of the Cumulative Acute Hazard Index (Zn + Cd + Cu) with Time for all Study Sites. Note: A hazard index or quotient below 1 is considered "not toxic" (shown as the red horizontal dashed line). The Eagle Mine Superfund site transitioned from active remediation to operation and maintenance (O&M) in 2001 (shown as the vertical dashed green line). The site was placed on the National Priorities List in 1986. Upstream sites are shown in blue, potential source region sites are shown in red, and sites downstream from all known sources are shown in yellow. Data from Woodling et al. (2005).

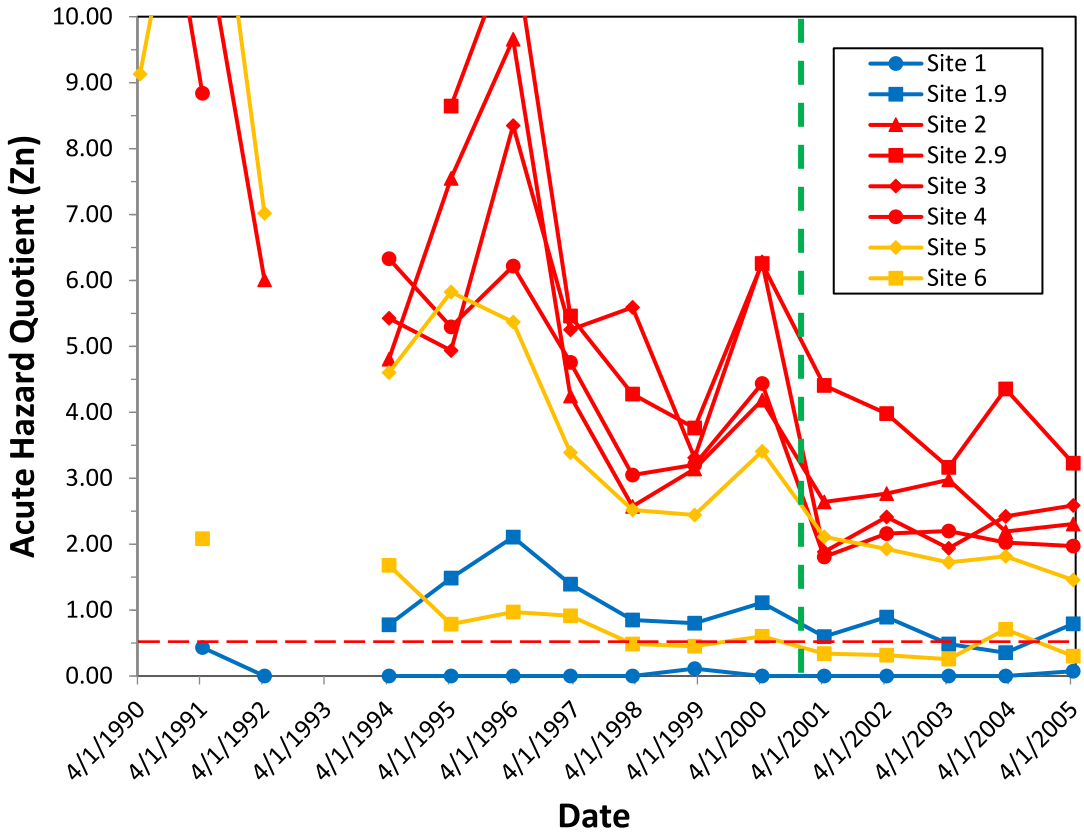

Figure 9.4. Variations of the Acute Hazard Quotient for Zn with Time for all Study Sites. Note: A hazard quotient below 1 is considered "not toxic" (shown as the red horizontal dashed line). The Eagle Mine Superfund Site transitioned from active remediation to operation and maintenance (O&M) in 2001 (shown as the vertical dashed green line). The site was placed on the National Priorities List in 1986. Upstream sites are shown in blue, potential source region sites are shown in red, and sites downstream from all known sources are shown in yellow. Data from Woodling et al. (2005).

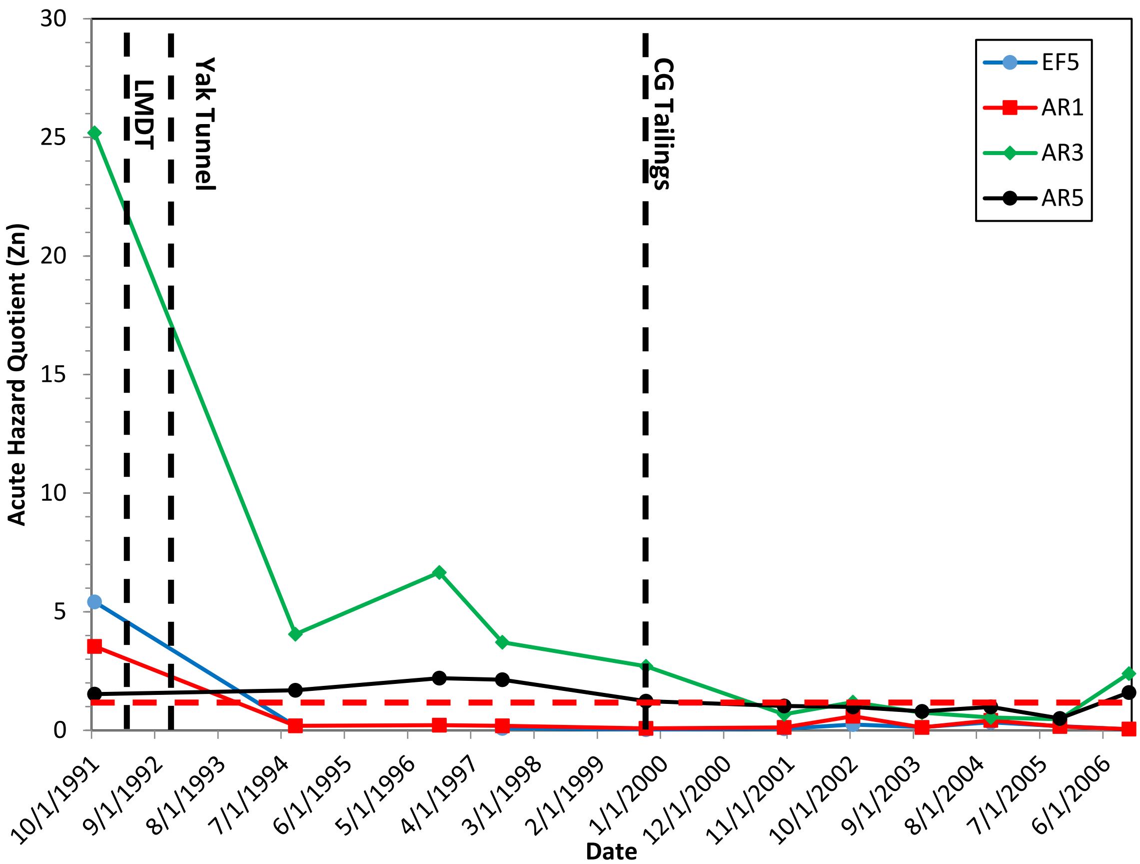

Figure 9.5. Variations of the Acute Hazard Quotient for Zn with Time for all Study Sites in the Arkansas River (California Gulch Superfund Site). Note: A hazard quotient below 1 is considered "not toxic" (shown as the red horizontal dashed line). Significant remediation events are shown as vertical dashed lines. These events include the start of the Leadville Mine Drainage Tunnel treatment plant (LMDT), the start of the Yak Tunnel treatment plant, and remediation of the California Gulch (CG) tailings.

The downstream reductions in Zn concentrations, as reflected by the lower hazard quotients, may be due to several processes, the most important of which is dilution due to the influx of groundwater or surface-water tributaries to the Eagle and Arkansas rivers. This process is particularly evident at Site 6 in the Eagle River that is downstream from the confluence with Gore Creek. Removal by sorption on to hydrated ferric oxides or clay minerals is a less likely possibility because the sorption edge for zinc is at a fairly high pH (Section 10) and neutralization of MIW by receiving water bodies should have effectively precipitated most of the hydrated ferric oxides at the site of mixing, and thus would have removed downstream sources of trace metal sorbents (Section 10).

To verify that zinc is behaving conservatively in the watershed below all sources, the dissolved concentrations that would be expected simply from dilution using the discharge data found in Woodling et al. (2005) were calculated. Because potential sources of contaminants exist from Site 1 to Site 4 (Figure 8.6), these calculations were made starting with Site 3 and again starting with Site 4. The comparison of the estimated concentrations with the measured concentrations is shown in Figure 9.6.

Both sets of calculations yield strong correlations between measured and estimated concentrations. Linear regression of both simulations yields r^2 values greater than 0.86 (Figure 9.6). This strong correlation confirms that zinc behaves conservatively downstream from sources, that is sorption and precipitation are not important in-stream processes affecting zinc concentrations. This type of approach is not possible upstream from Sites 4 or 3 because of the potential for multiple sources of contamination from mine waste piles and mine workings, although this approach may have value in identifying reaches that still have sources of zinc. Given that zinc behaves conservatively in the Arkansas and Eagle rivers, its toxicity should be linked to the amount of dilution that has occurred at any point along the river downstream from the primary sources of zinc.

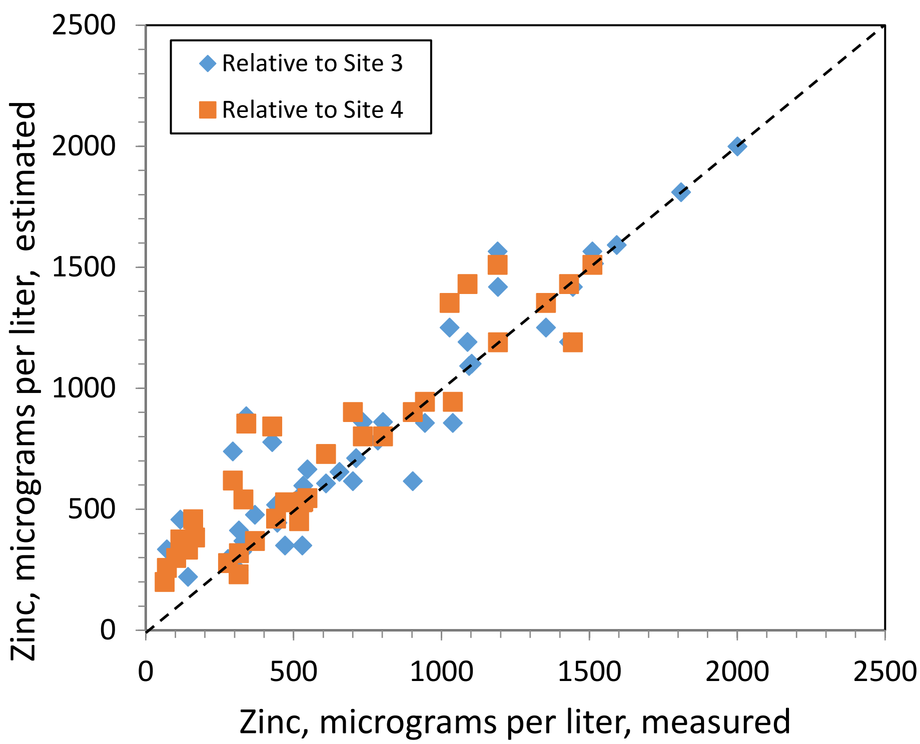

Figure 9.6. Scatter Plot Comparing Measured Concentrations of Zinc in the Eagle River with Concentrations Estimated Assuming that Dilution is the Only Process Causing Decreases in Concentration. The dashed line represents perfect agreement with measured and estimated values.

The variation of fish populations in the Eagle River by site with time is shown in Figure 9.7. Most sites, including the most upstream site, Site 1, exhibited depressed fish populations for the early part of the Woodling et al. (2005) study. After remediation was completed in 2001, most sites experienced a steady increase in fish populations. The relationship between the fish population and the hazard quotient for zinc for the Eagle River is shown in Figure 9.8. In general, the lower hazard quotient corresponds to the higher fish populations. However, the greatest increase in fish population is at a hazard quotient below 3 or 4 rather than the theoretical value of 1.

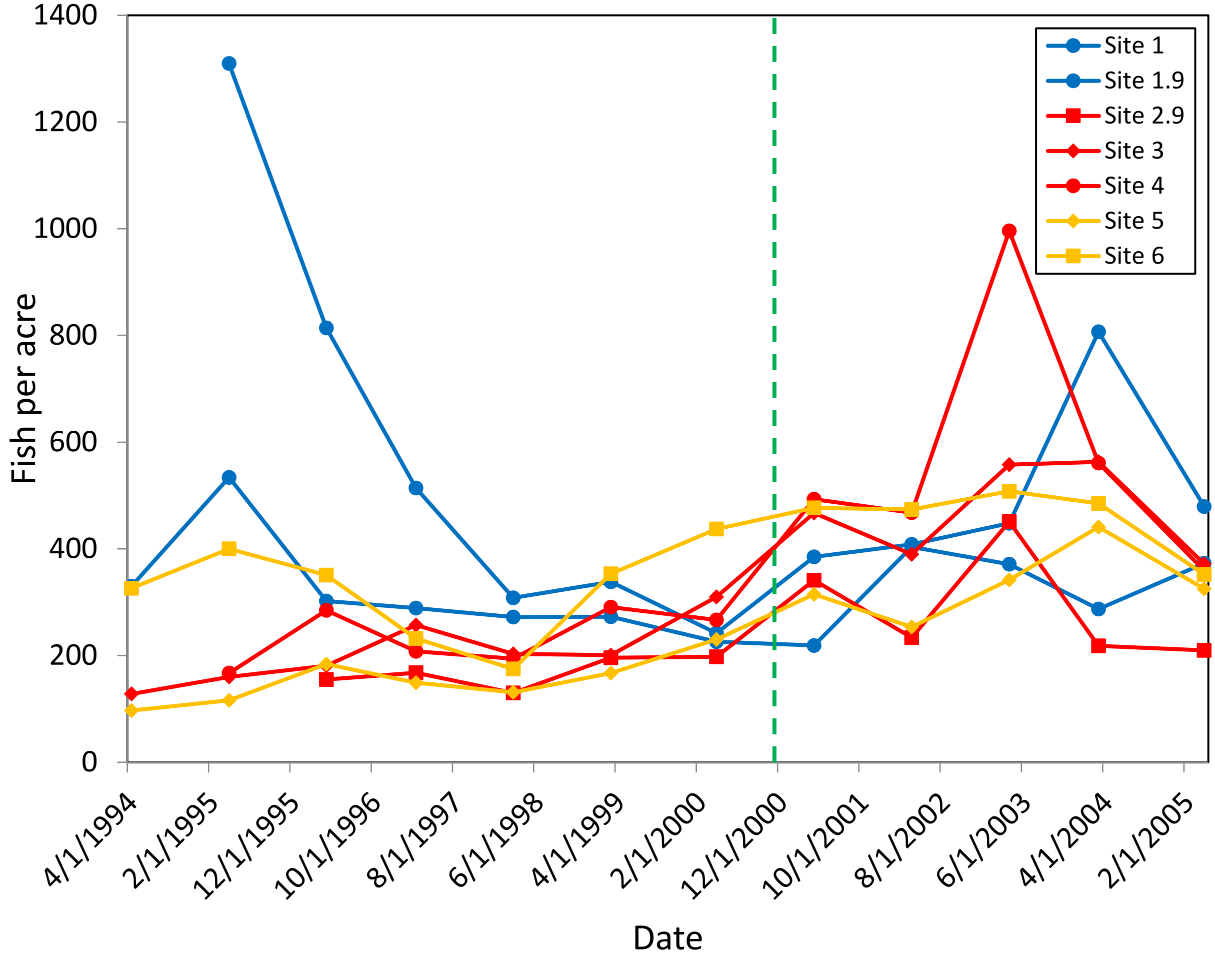

Figure 9.7. Variations of the Abundance of Fish (Predominantly Brown Trout) with Time for all Study Sites in the Eagle River. The Eagle Mine Superfund Site transitioned from active remediation to O&M in 2001 (shown as the vertical dashed green line). The site was placed on the National Priorities List in 1986. Upstream sites are shown in blue, potential source region sites are shown in red, and sites downstream from all known sources are shown in yellow. Data are from Woodling et al. (2005).

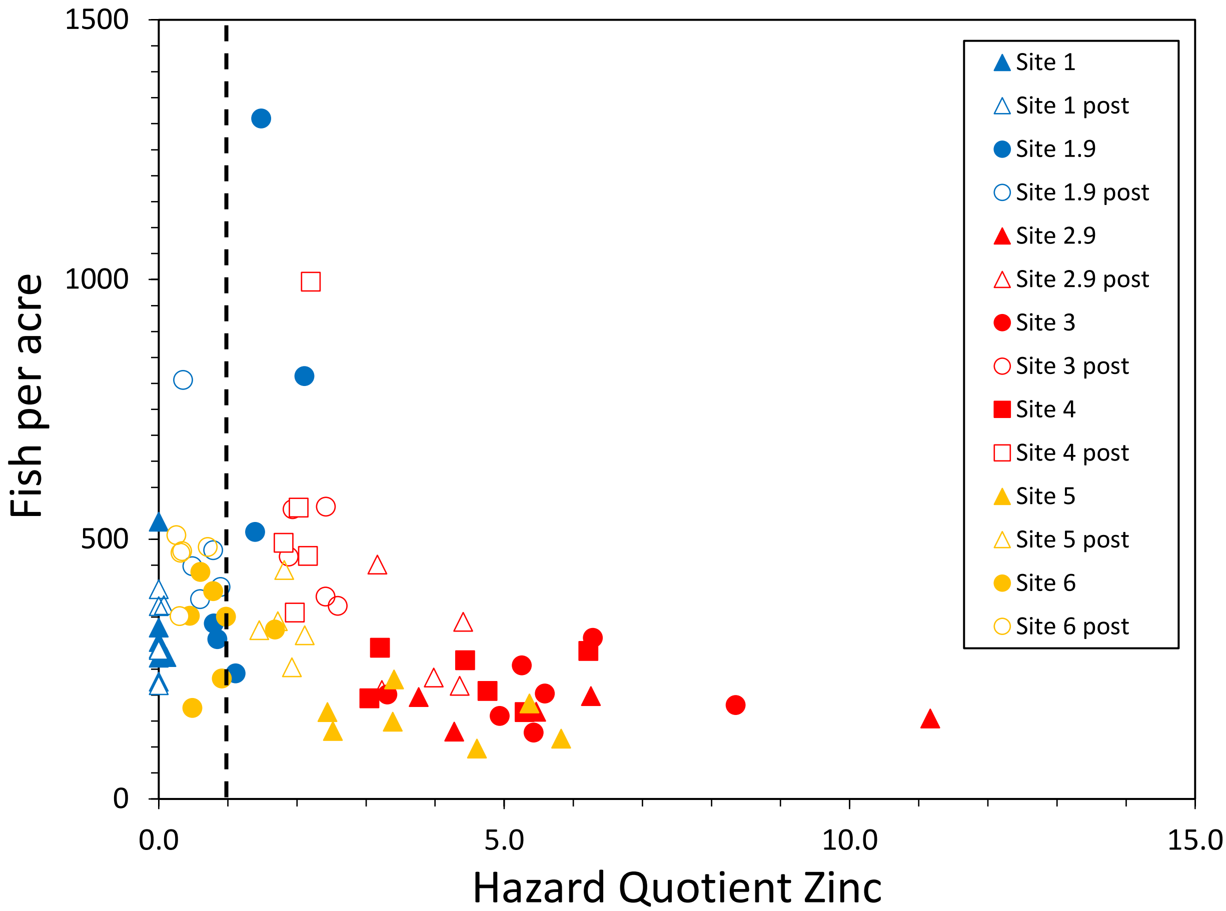

Figure 9.8. Scatter Plot Showing the Relationship Between Fish Populations (Predominantly Brown Trout) and the Acute Hazard Quotient for Dissolved Zn in the Eagle River. Upstream sites are shown in blue, potential source region sites are shown in red, and sites downstream from all known sources are shown in yellow. Sample taken before O&M started (2001) are shown as solid symbols. Those taken after the start of O&M are shown as open symbols. The vertical black dashed line marks a hazard quotient of 1. Data are from Woodling et al. (2005).

The variation of fish populations in the Arkansas River by site with time is shown in Figure 8.9. Most sites, including the most upstream site, EF 1, experienced depressed fish populations for the early part of the Clements et al. (2010) study. After remediation was completed in early 2000, a steady increase in fish populations was noted for all sites. The relationship between the fish population and the hazard quotient for zinc for the Arkansas River is shown in Figure 9.10. In general, the lower hazard quotient corresponds to the higher fish populations. However, the greatest increase in fish population is at a hazard quotient below 10 rather than the theoretical value of 1, or the value of 3 found in the Eagle River. Marr et al. (1995) described how brown trout acclimate to sub-lethal doses of metals and become more resistant to their adverse effects than rainbow trout. The fish liver is known to rapidly eliminate zinc, whereas copper in known to accumulate in the liver (Marr et al, 1995).

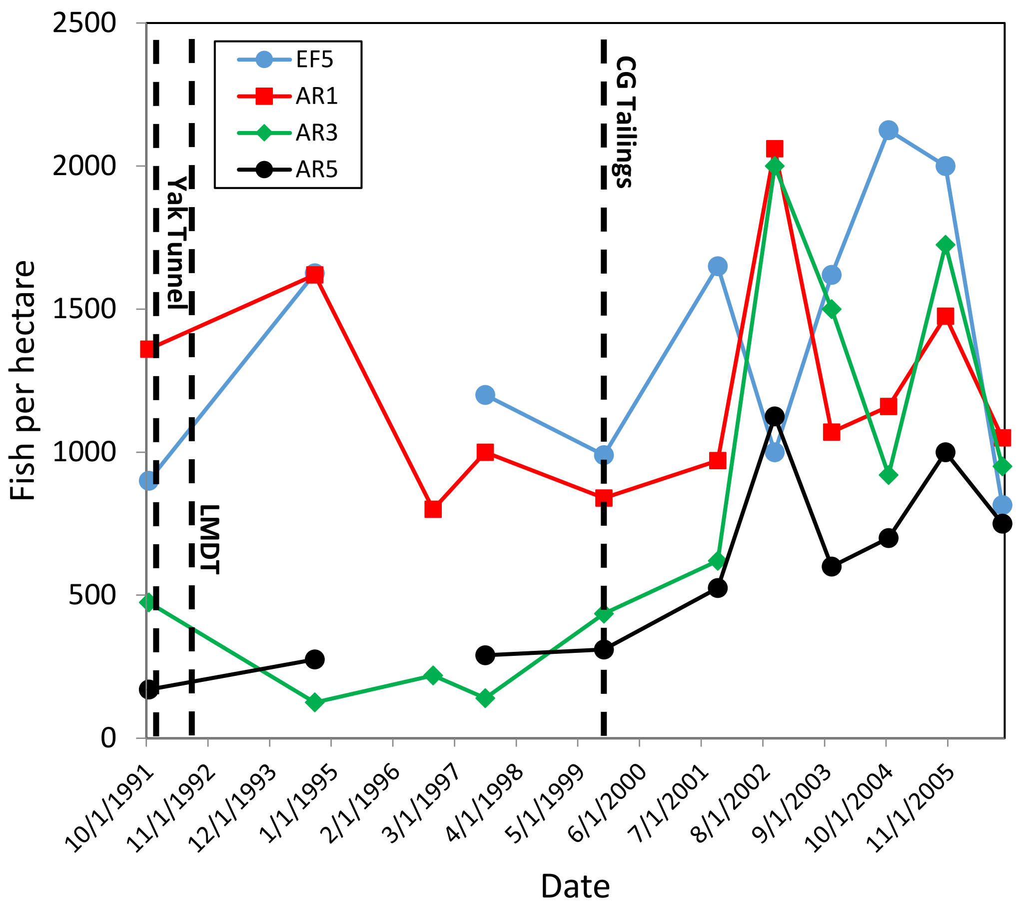

Figure 9.9. Variations of the Abundance of Brown Trout with Time for all Study sites in the Arkansas River. Significant remediation events are shown as vertical dashed lines. These events include the start of the Leadville Mine Drainage Tunnel treatment plant (LMDT), the start of the Yak Tunnel treatment plant, and remediation of the California Gulch (CG) tailings. Data are from Clements et al. (2010).

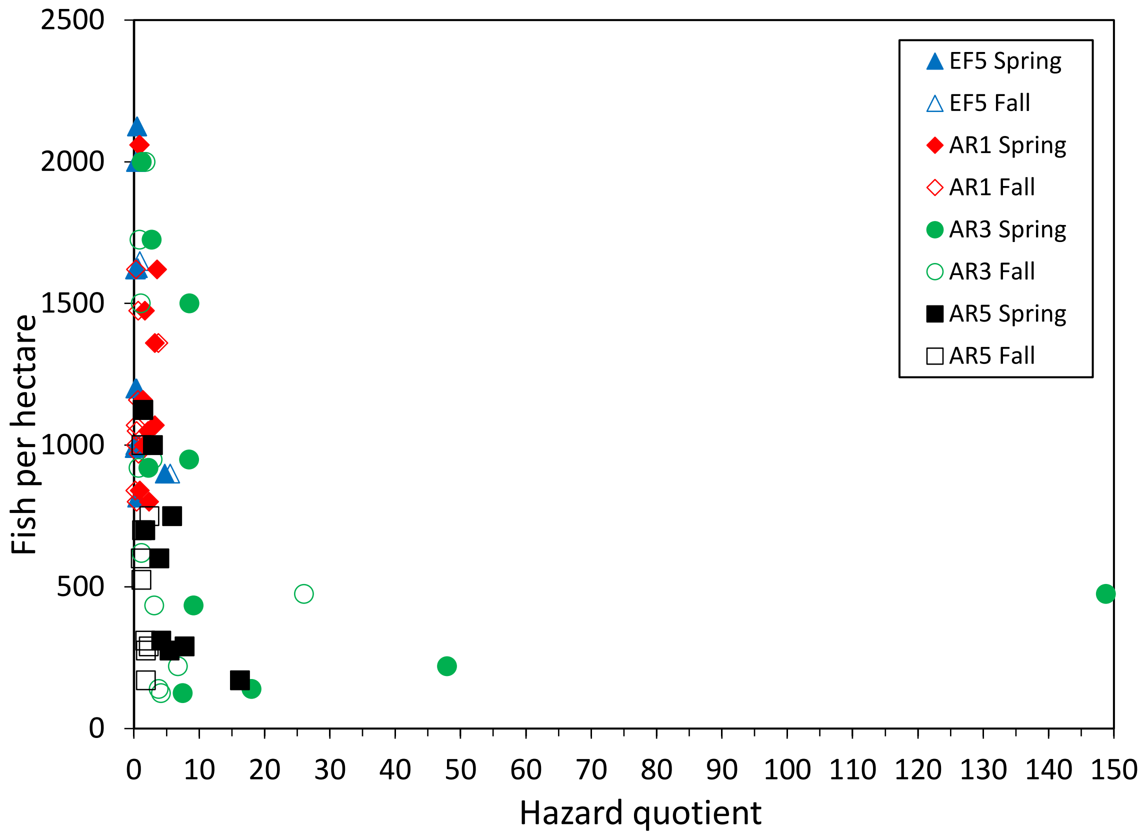

Figure 9.10. Scatter Plot Showing the Relationship Between Brown Trout Populations and the Acute Hazard Quotient for Dissolved Zn in the Arkansas River. Both Spring and Fall fish surveys are shown. Data are from Clements et al. (2010).

Aquatic organisms

Arkansas River/California Gulch

Macroinvertebrate and brown trout (*Salmo trutta*) data from 1989 to 2006 for four sites in the Arkansas River in the vicinity of Leadville are described and presented by Clements et al. (2010). Macroinvertebrate data include total abundance per area and number of mayfly species per area and were sampled in both spring and fall each year. Brown trout data include abundance and biomass per area and were sampled in August. The macroinvertebrates demonstrated steady recovery after restoration events reaching levels above reference streams (Figure 9.11). The brown trout data also indicate steady recovery downstream after restoration events (Figure 9.12). The reader is referred to Clements et al. (2010) for more detailed discussion of their biologic data.

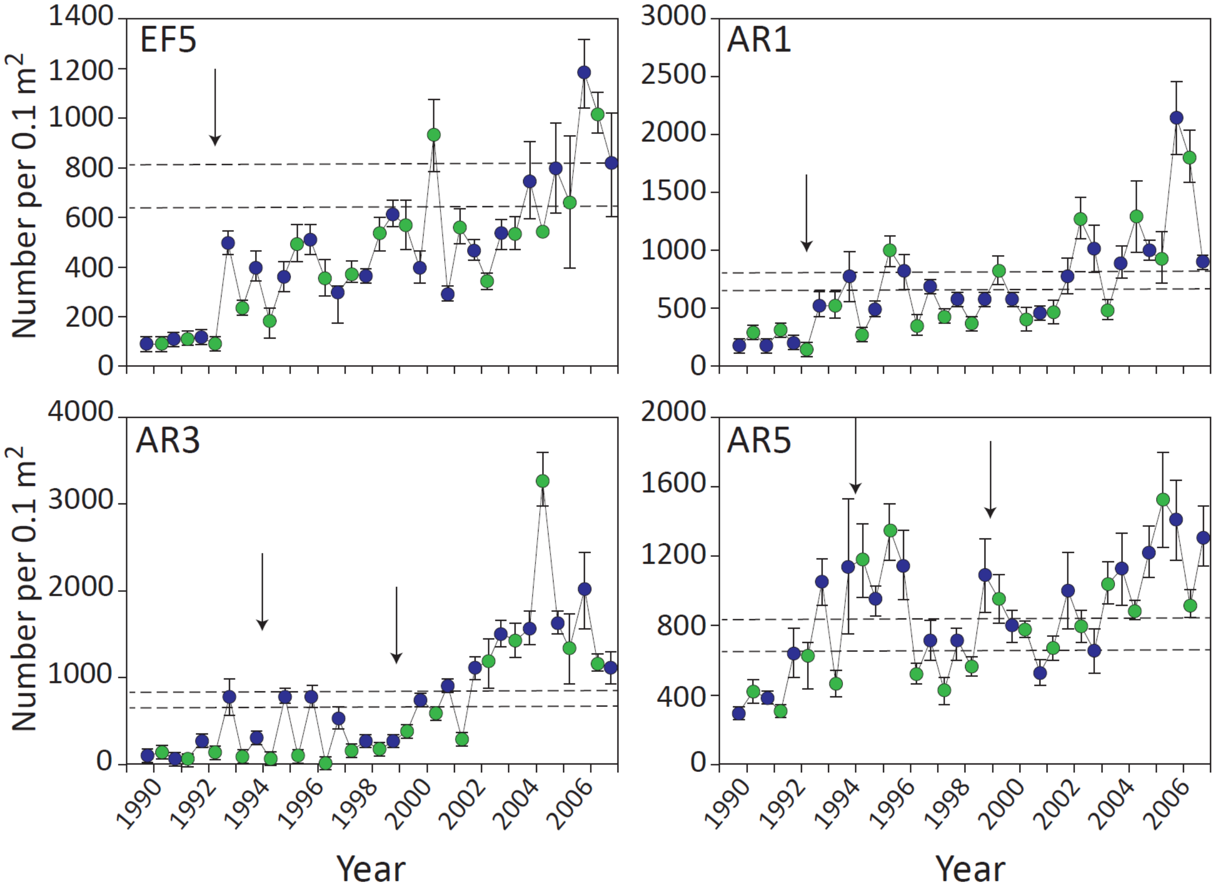

Figure 9.11. Variations in Total Macroinvertebrate Abundance in the Arkansas River from Sites Upstream (EF5, AR1) and Downstream (AR3, AR5) from California Gulch. Arrows indicate the completion of major restoration projects, and the horizontal lines bracket the mean value for reference streams with good water quality in Colorado. Restoration events upstream are shown by solid arrows, and downstream events are dashed. Upstream events include the start of operation of the Leadville Mine Tunnel and Yak Tunnel treatment plans, and downstream events are tailings removals. Modified from Clements et al. (2010).

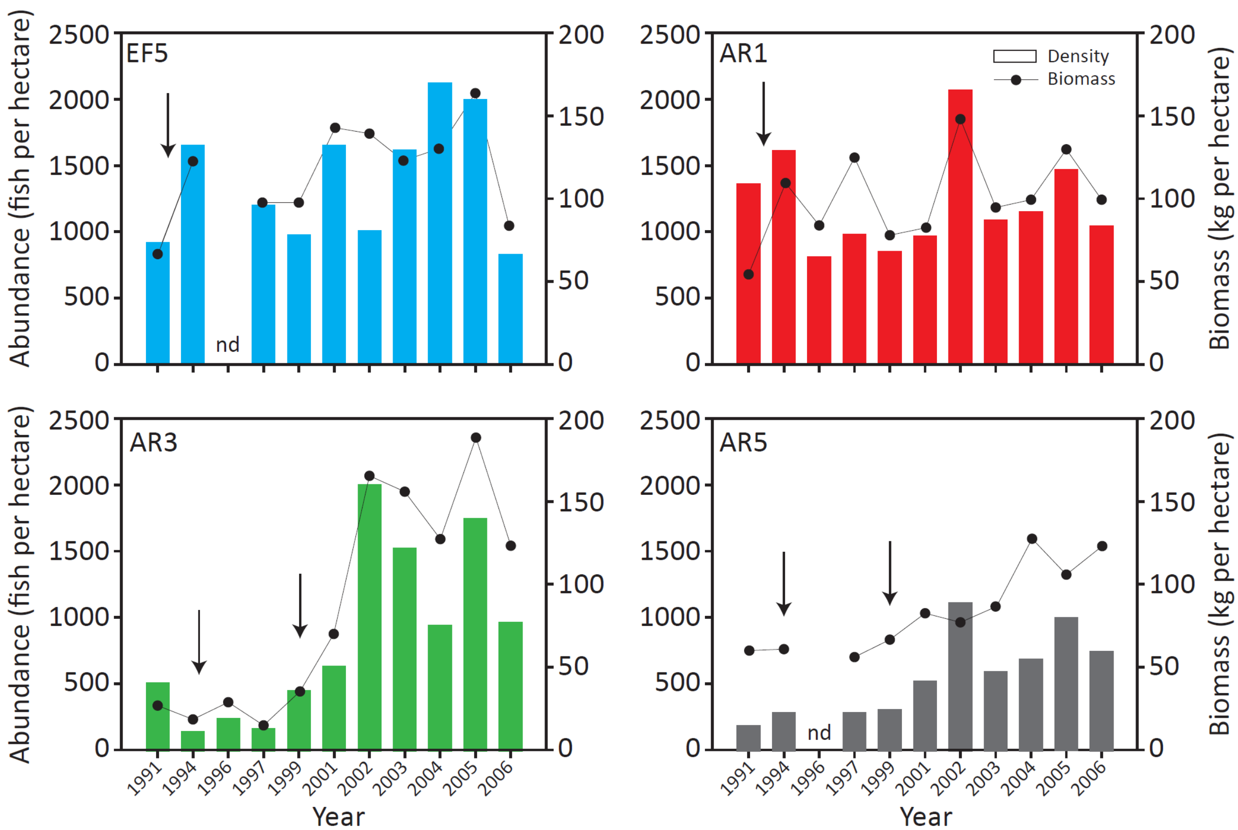

Figure 9.12. Variations in Brown Trout Abundance and Biomass in the Arkansas River from Sites Upstream (EF5, AR1) and Downstream (AR3, AR5) from California Gulch. Arrows indicate the completion of major restoration projects. Restoration events upstream are shown by solid arrows, and downstream events are dashed. Upstream events include the start of operation of the Leadville Mine Tunnel and Yak Tunnel treatment plans, and downstream events are tailings removals. Modified from Clements et al. (2010).

Eagle River

Macroinvertebrate and fish data from 1990 to 2005 for six sites in the Eagle River in the vicinity of the Eagle Mine are described and presented by Woodling et al. (2005). Macroinvertebrate data include the abundance and number of taxa. Fish data include abundance and size. Macroinvertebrates at Sites 3 and 4 showed the most severe and persistent impacts from MIW, whereas Sites 5 and 6, downstream from the area of surface disturbance, exhibited steady improvement after the start of water treatment in 1991 (Figures 9.13 and 9.14). The brown trout data have similar trends (Figure 9.15). The reader is referred to Woodling et al. (2005) for more detailed discussion of their biologic data.

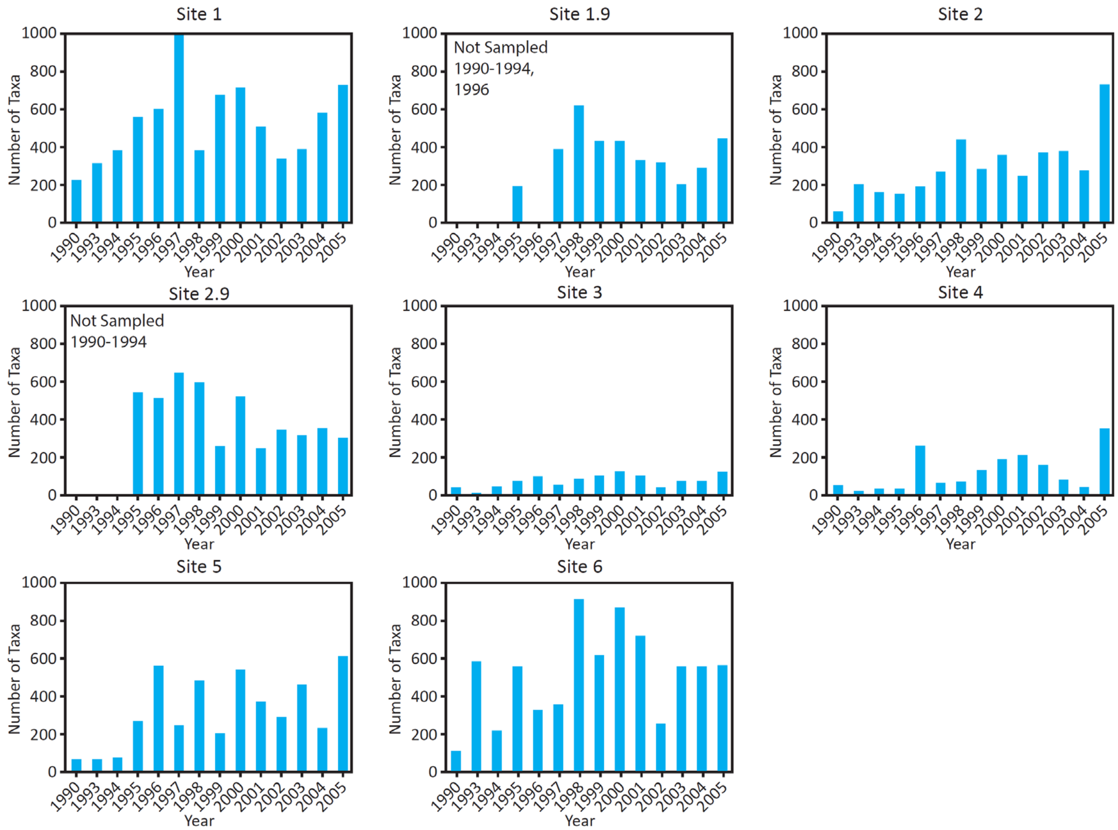

Figure 9.13. Variations in Macroinvertebrate Abundance in the Eagle River. Significant influx of MIW occurred downstream of Site 2, and surface disturbance related to the mine was noted upstream from Site 5. Modified from Woodling et al. (2005).

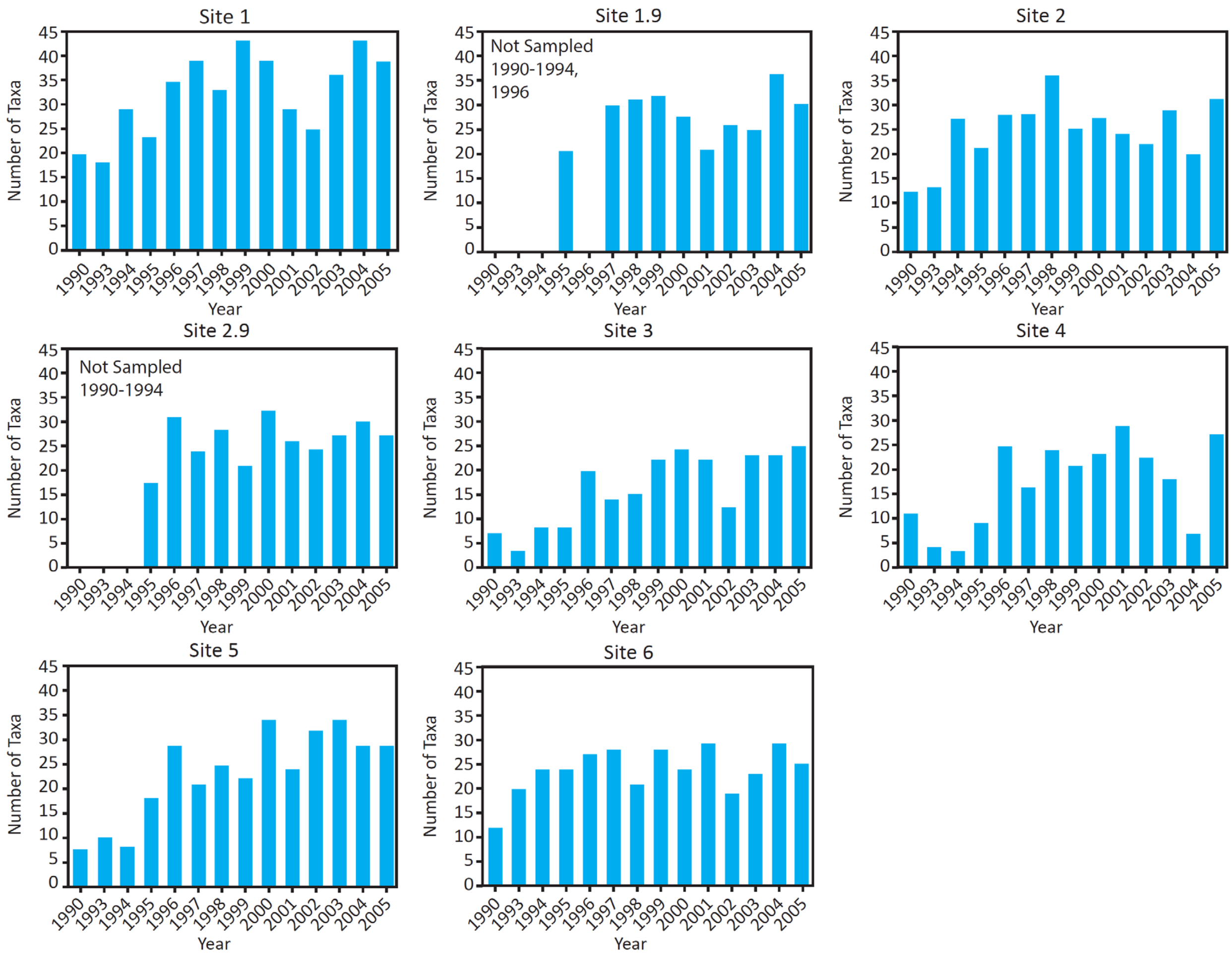

Figure 9.14. Variations in Number of Macroinvertebrate Taxa in the Eagle River. Significant influx of MIW occurred downstream from Site 2, and surface disturbance related to the mine was noted upstream from Site 5. Modified from Woodling et al. (2005).

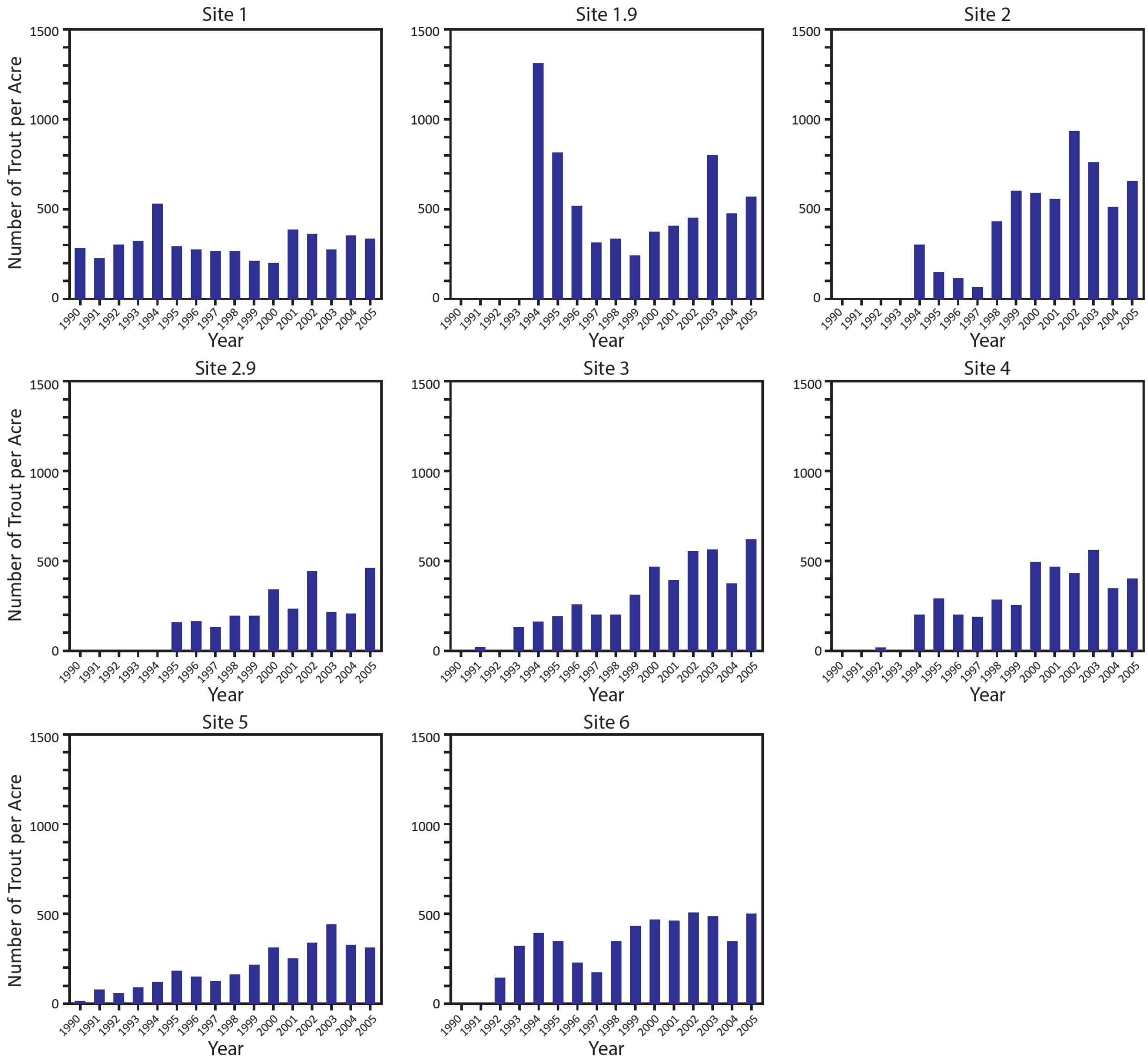

Figure 9.15. Variations in Brown Trout Abundance in the Eagle River in the Vicinity of the Eagle Mine with 95% Confidence Intervals. Significant influx of MIW occurred downstream from Site 2, and surface disturbance related to the mine was noted upstream from Site 5. Modified from Woodling et al. (2005).

Discussion

Geo-environmental setting

Geo-environmental models have value for understanding environmental risks shared by mineral deposits of similar types, such as the carbonate replacement sulfide deposits of the Leadville and Gilman districts in central Colorado. The shared features include the nature of the ores, the nature of the host rocks, the chemical behavior of the elements found in the ores and solid mines wastes, the mining methods, the ore-processing methods, and the waste management practices. The environmental characteristics of the abandoned mines in the Leadville and Gilman districts are influenced by the regional geologic characteristics of the watershed, especially the carbonate rocks. The Manitou Dolomite, the Leadville Dolomite, and the Dyer Dolomite are important rock units within the watersheds of both mining districts. The latter two represent the most important ore hosts in the area. The effect of these dolomite units is to support near neutral pH, high alkalinity, and high hardness in the watersheds that historically have been affected by MIW. It is used in the following discussion because it is more general than the term “acid mine drainage” and avoids confusion surrounding acid drainage that becomes neutralized yet still carries potentially toxic concentrations of metals.

The primary ores are massive sulfide accumulations, dominated by pyrite, iron-rich sphalerite, and galena with lesser amounts of chalcopyrite, important accessory minerals include argentite, electrum, and tetrahedrite-tennantite (Thompson and Arehart, 1990). The pyrite, which ended up in waste rock and mill tailings, embodies the considerable bulk of the acid-generating potential of the mine waste. Iron from remnant sphalerite in the mine waste also contributes to its acid-generating potential. Ultimately, the acid-generating potential of the mine waste depends on the balance between the acid-generating potential from the pyrite and the acid-neutralizing potential from the carbonate minerals derived from the host rock. Acid generated from pyrite oxidation enhances the liberation of metals, such as iron, zinc, cadmium, copper, and lead, from mine wastes. The carbonate host rocks represent acid-neutralizing potential for their solid mine wastes (unprocessed waste rock or mill tailings). However, the ubiquitous silicification associated with mineralization served to dilute the acid-neutralizing potential of the mined rock and processed mill tailings.

The approach used to develop a mine also influences its environmental attributes. The mining methods, ore-processing methods, and waste management practices utilized significantly influenced environmental risks associated with mining in the Leadville and Gilman districts. The dipping tabular nature of the ore bodies in both districts was conducive to underground mining for most deposits. Underground mining minimizes the volume of waste rock generated, making either hand-sorted, low grade waste for historical mining or mill tailings for more recent mining the predominant solid mine waste materials. The differences in physical properties of coarse-grained hand sorted waste and fine-grained mill tailings make their environmental behavior distinct from one another. Water that has interacted with mine workings, waste rock, or mill tailings (MIW) can become acidic due to the oxidative weathering of pyrite and can leach significant quantities of metals, other trace elements, and sulfate. The coarse-grain size of hand sorted waste (or waste rock) results in a fairly oxygenated, unsaturated waste pile – an ideal environment for sulfide oxidation (Amos et al., 2015). In contrast, the fine-grain size of mill tailings can limit the access of oxygen, and the retaining structures of tailings storage facilities can maintain a saturated condition within much of the pile, which additionally can limit the access of oxygen (Lindsay et al., 2015). The fine-grained nature of mill tailings also makes them more prone to erosion, transport, and redeposition at low-gradient sites downstream where they can act as additional sources of contamination. Historical mining operations typically employed waste management practices that ignored environmental mitigation strategies, whereas mining operations permitted after 1970 were required to have environmental mitigation incorporated into mine plans. The only mine in the study to fall in the latter category is the Black Cloud Mine (1971 – 1999) in the Leadville district.

An additional, somewhat unique, feature of mining in the Leadville district was the construction of tunnels that consolidated access to numerous small mines on the mountainside. The tunnels facilitated haulage of ore and also served to drain water from the mines. Today, the tunnels are major sources of acid mine drainage. The Leadville Mine Drainage Tunnel extends southeastward from the Arkansas River valley north of the town of Leadville. It surfaces in the watershed of the East Fork of the Arkansas River. The Yak Tunnel extends from California Gulch northeastward beneath Breece Hill and surfaces in the California Gulch watershed. Both are sites of active water treatment systems today.

In historical mining districts where mining has occurred over the span of a century, such as Leadville and Gilman, mining, ore processing, and waste management practices typically evolve over time due to technological advancements and increased regulation. The nature of environmental risks will vary accordingly because this evolution in practices will affect the nature of mine waste. The geo-environmental landscape of a historical mining district will, therefore, be an agglomeration of these effects.

The regional geologic setting influences the watershed scale chemistry of surface water receiving MIW. As mentioned, carbonate rocks, such as those found in central Colorado, serve to elevate the pH, alkalinity, and hardness of rivers and streams. Neutralization of metal-rich acid drainage due to mixing with larger, higher alkalinity streams under oxygenated conditions should result in the precipitation of most of the dissolved iron, but should leave appreciable amounts of zinc and cadmium in solution (Figure 10.1). In contrast, lead has limited solubility in sulfate and carbonate rich waters due to the low solubility of anglesite and cerussite, respectively (Figures 10.2 and 10.3). Therefore, lead is considered to be of limited concern in the aqueous phase, but does pose significant risks to humans in particulate form. The primary particulate forms of concern are waste rock, and particularly mill tailings. The fine-grained nature of mill tailings makes them more amenable to transport away from the site of initial disposal by wind or water erosion. Trace metals may also be attenuated in aqueous settings by sorption onto hydrous ferric oxides (Figure 10.4). Lead and copper are effectively removed at low pH (< 5.5), but significant amounts of zinc and cadmium may persist to higher pH values. Therefore, neutralization of acid drainage may effectively remove dissolved lead and copper in addition to dissolved iron, but zinc and cadmium may persistent downstream.

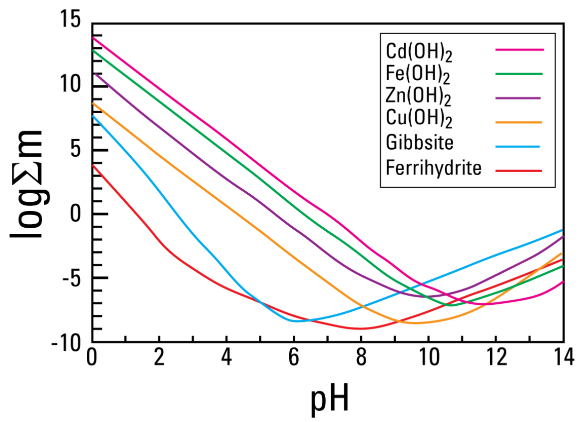

Figure 10.1. Solubility of Various Metals as a Function of pH. Note low solubility of ferrihydrite ($Fe[OH_3]$) compared to the hydroxides of zinc, copper, and cadmium.. Modified from Nordstrom and Alpers (1999).

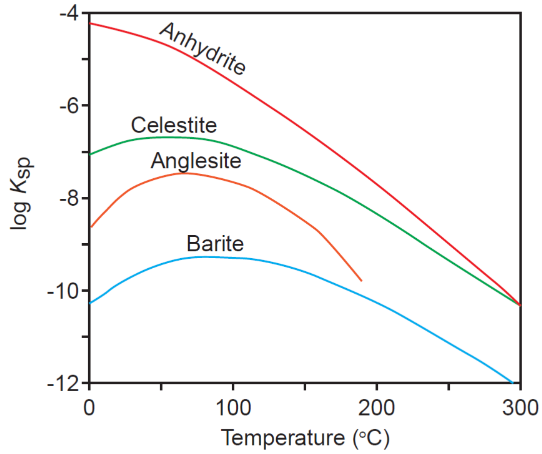

Figure 10.2. Logarithm of the Solubility Product (K_{sp}) of Selected Sulfate Minerals as a Function of Temperature. Note the low solubility of anglesite ($PbSO_4$). Modified from Rimstidt (1997).

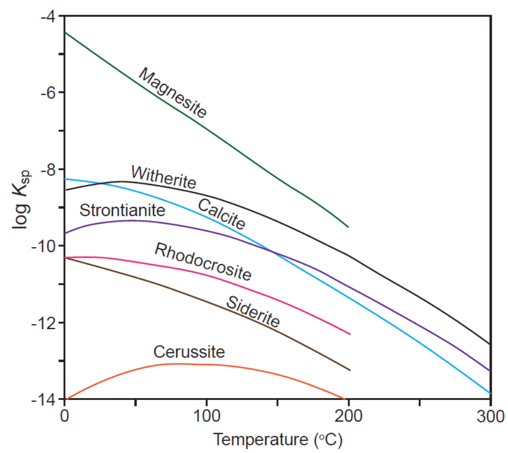

Figure 10.3. Logarithm of the Solubility Product (K_{sp}) of Selected Carbonate Minerals as a Function of Temperature. Note the low solubility of cerussite ($PbCO_3$). Modified from Rimstidt (1997).

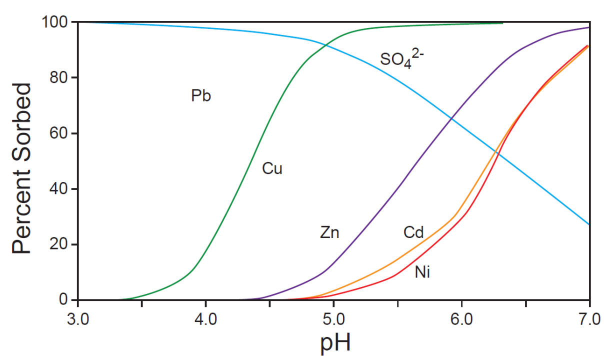

Figure 10.4. Model Sorption Curves for Selected Metals and Sulfate on Hydrous Ferric Oxide. Modified from Smith (1999).

The fate of the dissolved metals and other trace elements depends on the chemical characteristics of the MIW and the receiving water body. The neutralization that commonly accompanies mixing of acidic water with higher alkalinity receiving waters can partially or totally remove some metals from solution. The behavior of metals during neutralization varies on an element by element basis and is also influenced by other elements, especially iron, in the MIW. Hydrated ferric oxides that can precipitate as a result of neutralization can act as strong sorbents of trace metals such as Cu, Pb, Zn, and Cd. The sorption of these metals varies as a function of pH with the onset of removal with increasing pH starting in the order of Pb, Cu, Zn, and Cd.

Aquatic Setting

History of aquatic ecological recovery

The toxicity of metals in MIW to aquatic organisms depends on a number of factors. Precipitation and sorption during the mixing of MIW with a receiving water body can reduce metal concentrations in water. Dilution both due to mixing with receiving water bodies and the influx of clean tributaries downstream serves to decrease metal concentrations. Decreases in metal concentrations due to precipitation, sorption, and dilution all serve to reduce toxicity. Toxicity to aquatic organisms also varies as a function of water chemistry. Water hardness is important in mitigating the toxic effects of metals.

The effect of metals in MIW on fish populations is complex. Not only can the dissolved metals directly affect the health of fish, but they can also affect aquatic macroinvertebrates that serve as important food sources for fish. The precipitation of hydrated ferric oxides at the mixing zone of MIW with receiving streams can cement cobbles and gravels, resulting in a degraded habitat for aquatic macroinvertebrates. Surface-water impacts in the vicinity of the Leadville and Gilman mining districts have been extensive. One of the most extensive early reports was written by LaBounty et al. (1975). Their study included investigations of surface water chemistry, sediment chemistry, macroinvertebrates, and fish collected from April through November 1974 in the upper Arkansas River. They documented extensive impairment of the aquatic ecosystem due to MIW.

The recovery of a watershed during and after remediation can be equally complex. Sources of contamination must be identified and properly addressed. Common sources of MIW include drainage from mine workings and leachate from solid waste piles. Drainage from mine workings represents a particularly intractable challenge because source control is often difficult or impossible. Instead, treatment, either passively or actively, is required. Solid mine waste typically acts as a long-term source of contaminants for surface waters and groundwater. Mill tailing impoundments can be prone to large-scale erosion, which can lead to downstream transport and deposition. Fluvial tailings depositions can act as long-term sources of contamination to surface waters that must be identified and addressed during remediation.

The recovery of fish populations following remediation involves a number of factors. Contaminant sources must be addressed, including solid materials, such as tailings, that may have been transported downstream and redeposited on stream banks. Water quality must return to tolerable conditions for both the fish and the macroinvertebrates that serve as food sources for the fish. Suitable habitat must be available for both fish and macroinvertebrates. Finally, adequate time must be provided for both the macroinvertebrate and fish populations to rebound.

The abundance and diversity of fish and aquatic macroinvertebrates have shown steady increases since the start of remediation in both watersheds. Clements et al. (2010) documented increases in macroinvertebrates and fish in the upper Arkansas River in the period from 1989 to 2006 once remediation began. They reported rapid response of macroinvertebrates to improvements in water quality, reflecting their resilience to chemical stressors. This observation suggests that a food source was not a limiting factor in trout recovery. Policky (2016) reported increased numbers of trout over 14 in long in the upper Arkansas River near Leadville beginning around 2002. The upper Arkansas River is predominantly a wild brown trout fishery, although rainbow trout have been historically stocked mostly in lakes (Policky, 2016). Brown trout constitutes over 75% of the trout community. Woodling et al. (2005) reported similar changes in the Eagle River for macroinvertebrates and fish after remediation began in the period from 1990 to 2005.

Uncertainty in inter-annual healthy populations

In the Arkansas River watershed, significant correlations of biologic measures of macroinvertebrate and fish populations were found with level of exceedance of water-quality criteria for Zn, Cd, and Cu, location, and sampling date. Broader aspects of the water chemistry (temperature, pH, specific conductance, and alkalinity) and physical hydrology (stream depth, velocity, and discharge) did not improve correlations (Clements et al., 2010). For brown trout density and biomass, the highest correlation coefficients (r^2) achieved were 0.54 and 0.51, respectively. In other words, approximately 50% of the variance in brown trout population can be related to the factors outlined above.

In the Eagle River watershed, logarithmic biologic measures of fish populations correlated significantly with level of exceedance of water-quality criteria for Zn, Cd, and Cu that varied by sample site, but most strongly at the three sites immediately downstream from the Eagle mine but before the confluence with Gore Creek. The coefficients of correlation (r^2) varied by site and ranged from 0.03 to 0.86 (Woodling et al., 2005). Correlations of the number of 1-year old brown trout with maximum flow from the preceding year were weak at best (Woodling et al., 2005). As with the Arkansas River, a significant amount of the variation cannot be described by differences in water quality and discharge.

For both watersheds, correlations among fish metrics and water-quality and physical hydrology parameters generally yielded correlation coefficients (r^2) that typically ranged between 0.50 and 0.70, but locally reached lows of 0.03 and highs of 0.86 (Woodling et al., 2005, Clements et al., 2010). In other words, a significant portion of the variance is not described by the parameters considered by these studies. High variance in trout populations from year to year is not unique to watersheds that have been impacted by mine drainage. Dauwalter et al. (2009) examined temporal variations in trout populations in North America through literature review. They found that coefficients of variation for trout populations in healthy streams averaged 49 ± 27% (range 15 to 108%) over time for all ages of trout. For brown trout in Colorado, they reported coefficients of variation for abundance that ranged from 15% for year 2+ trout to 82% for year-1 trout in Little Beaver Creek. The coefficients of variation for brown trout in South Saint Vrain Creek had a smaller range for various age groups from 17 to 49%. Thus, the high variance observed in the Arkansas River and Eagle River watersheds is characteristic of trout populations in general, and not necessarily attributable to the effects of mine drainage and its remediation.

The rapid recovery of macroinvertebrates in the Arkansas River watershed, described by Clements et al. (2010), suggests that the availability of food sources was not the limiting factor. The rapid neutralization of acid-mine drainage upon mixing with higher alkalinity receiving waters and subsequent precipitation of hydrated ferric oxides near the sites of mixing should limit physical impairment of aquatic habitats to the immediate site of mixing with limited downstream impacts. Therefore, the recovery of the brown trout population should be considered in terms of population growth through "normal" breeding and reproduction starting at the remediation of water quality.

Modeling population growth

The literature on ecological population dynamics was reviewed to evaluate the role of time in ecological recovery. Several models to describe population growth, particularly for fisheries, were identified. The models vary in their complexity and can incorporate a number of variables and other factors including growth, cooperation and competition within a species, interactions with other species, age-class structure, and limiting physical characteristics of the habitat (Berryman, 2003). Available models, particularly for brown trout, fall into two main categories: population ecology models that seek to describe population dynamics and genetics, and population distribution models that seek to describe habitat and spatial characteristics (Frank et al., 2011). Population dynamics models are most appropriate for our ecosystem services valuation exercise.

The logistic function (Verhulst, 1838) is considered to be an overly general, simplistic model for describing ecological population dynamics (Turchin, 2001, Frank et al., 2011). More complex population dynamic models, such as those based on the Leslie matrix (Leslie, 1945), require a complex set of input parameters, including age-specific data on survival, fecundity, population structure, and physical characteristics of the hydrologic setting (Sabaton et al., 1997, Gouraud et al., 2001), all of which are beyond the scope of currently available data for this study. Despite its simplistic approach to predicting population growth, the logistic function should be adequate for the purposes of the present study because of the large coefficients of variations found for healthy trout populations as described by Dauwalter et al. (2009). Clearly, if more sophisticated population growth models are available for a watershed, they should be used instead of or in addition to a logistic model.

The logistic function requires minimal inputs: a starting population, a carrying capacity for the site, and a growth rate (Table 9.1), as described by Equation 10.1 below:

$$N(t) = \frac{\mathrm{N}_o \mathrm{K}}{\mathrm{N}_o + (\mathrm{K} - \mathrm{N}_o)e^{-rt}} \qquad (10.1)$$

where N(t) is the population at time t, N_o is the initial population, K is the carrying capacity of the system, r is the rate of population growth, and t is time.

Site AR3 was used to calibrate the model for the entire watershed because that site displayed the greatest and simplest increase in population after the last remediation event upstream from the site – the remediation of the California Gulch tailings (Figure 8.3). The carrying capacity for the study area was defined as the combined average population at site AR3 from 2002 to 2006. For the Arkansas River, the starting population for each site was chosen as the population in 1991 (Clements et al., 2010). The simulations began at the end of the last remediation event upstream from the site. For EF5 and AR1, this event was the commissioning of the Leadville mine drainage tunnel treatment system in 1992. For AR3 and AR5, the event was the California Gulch tailings remediation, which was completed in 1999. Therefore, the only differences among the models for individual sites are the starting populations and the recovery time (Table 10.1).

Table 10.1. Input Parameters Used for Trout Population Modeling of the Arkansas River Watershed

Site	Initial Population (number of fish)	Carrying Capacity (number of fish)	Growth Rate (% growth per year)
EF5	900	1456	0.944
AR1	1360	1456	0.944
AR3	279	1456	0.944
AR5	261	1456	0.944

The results of these simulations are presented in Figure 10.2 shown with fish population data from those sites. The heavy green line represents the prediction for each site. The thin green lines and thin black lines represent 20 and 50% variance, respectively. The 20% value approximates the "best case" for observed Colorado brown trout coefficients of variations from Dauwalter et al. (2009), and the 50% value reflects the approximate coefficient of variation for all trout in their study.

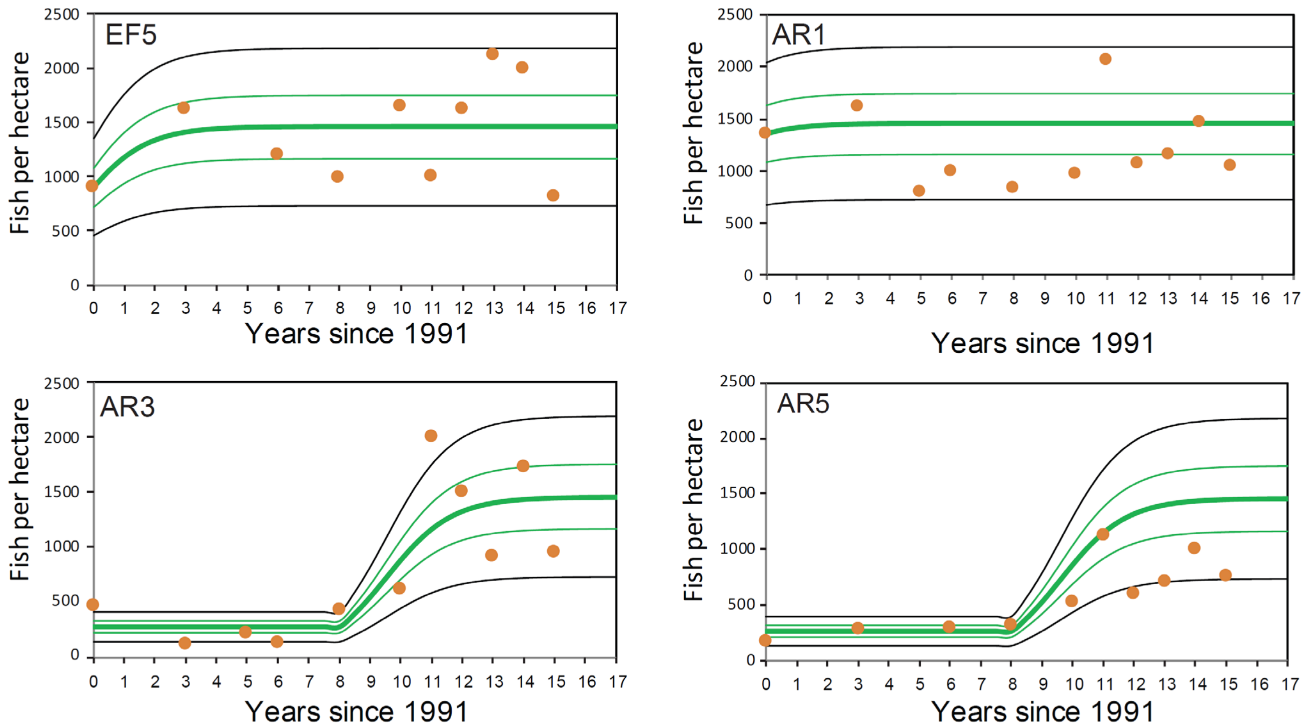

Figure 10.2. Comparison of Fish Population Data with Ecological Population Growth Models. The heavy green lines represent the population predictions, the thin green lines represent a variance of 20%, and the thin black lines represent 50% variance. Arkansas River: Site EF5, Site AR1, Site AR3, and Site AR5.

In general, the data for Sites EF5, AR1, and AR3 appear to scatter randomly about the prediction within the amount of variance reported by Dauwalter et al. (2009). (Note that site AR3 was used to calibrate the model for use in making predictions at the other sites.) In contrast, site AR5 appears to fall short of the predictions. Walton-Day and Mills (2015) described ongoing drainage from the Dinero Tunnel on Lake Fork, a tributary of the Arkansas River (Figure 8.3), which appears to have continued to suppress recovery in this part of the river.

The simulations for the Eagle River watershed assumed that the carrying capacity and growth rate used in the Arkansas River watershed (Site AR3) would be applicable to the Eagle River watershed given similar stream and habitat conditions. The simulations began at the nominal end of the remediation at the Eagle mine in 2001 (Table 8.4). Input for the Eagle River simulations are summarized in Table 10.2.

Table 10.2. Input Parameters Used for Trout Population Modeling of the Eagle River Watershed

Site	Initial Population (number of fish)	Carrying Capacity (number of fish)	Growth Rate (% growth per year)
Site 1	786	1456	0.944

Site	Initial Population (number of fish)	Carrying Capacity (number of fish)	Growth Rate (% growth per year)
Site 1.9	1340	1456	0.944
Site 2.9	418	1456	0.944
Site 3	508	1456	0.944
Site 4	582	1456	0.944
Site 5	379	1456	0.944
Site 6	803	1456	0.944

In general, the data for Sites 1, 1.9, 3, 4, 5, and 6 are within the plus or minus 50% prediction for population growth, consistent with the amount of variance reported by Dauwalter et al. (2009) shown in Figure 10.4. Site 2.9 is the sole site for which population measurements fell below this predicted range. Site 1, the upstream site, has had zinc concentrations that have been consistently below the acute toxicity guideline (Figure 9.4). Site 2.9 has exhibited decreases in zinc concentrations over time, yet it is the site that has consistently exceeded the acute toxicity guideline by the greatest amount. This consistently high exceedance suggest that a source of zinc must remain in this reach because the proximal sites both upstream and downstream exceed the guideline by lesser amounts. The reach between Site 2.9 and Site 3 is known as the North Property area (OU1, Table 8.4). A remedial investigation, released in 2006, of the North Property area documented remnants of solid mine waste and contaminated seepage from waste piles in this area (ERM, 2006). Even though most of the sites fall with the uncertainty range in the predicted population growth, the majority of the sites (Sites 1, 29, 5, and 6) appear to be biased toward the lower half of the predicted range (Figure 10.4). This low bias may indicate that the carrying capacity of this stream may be lower than that of the Arkansas River that was used in the model. The discharge in the two watersheds is similar with base flow typically ranging between 10 and 20 cfs and peak flow between 200 and 1000 cfs (Figures 8.4 and 8.7, respectively). Thus, differences in carrying capacity may be due to differences in the physical habitat or food sources.

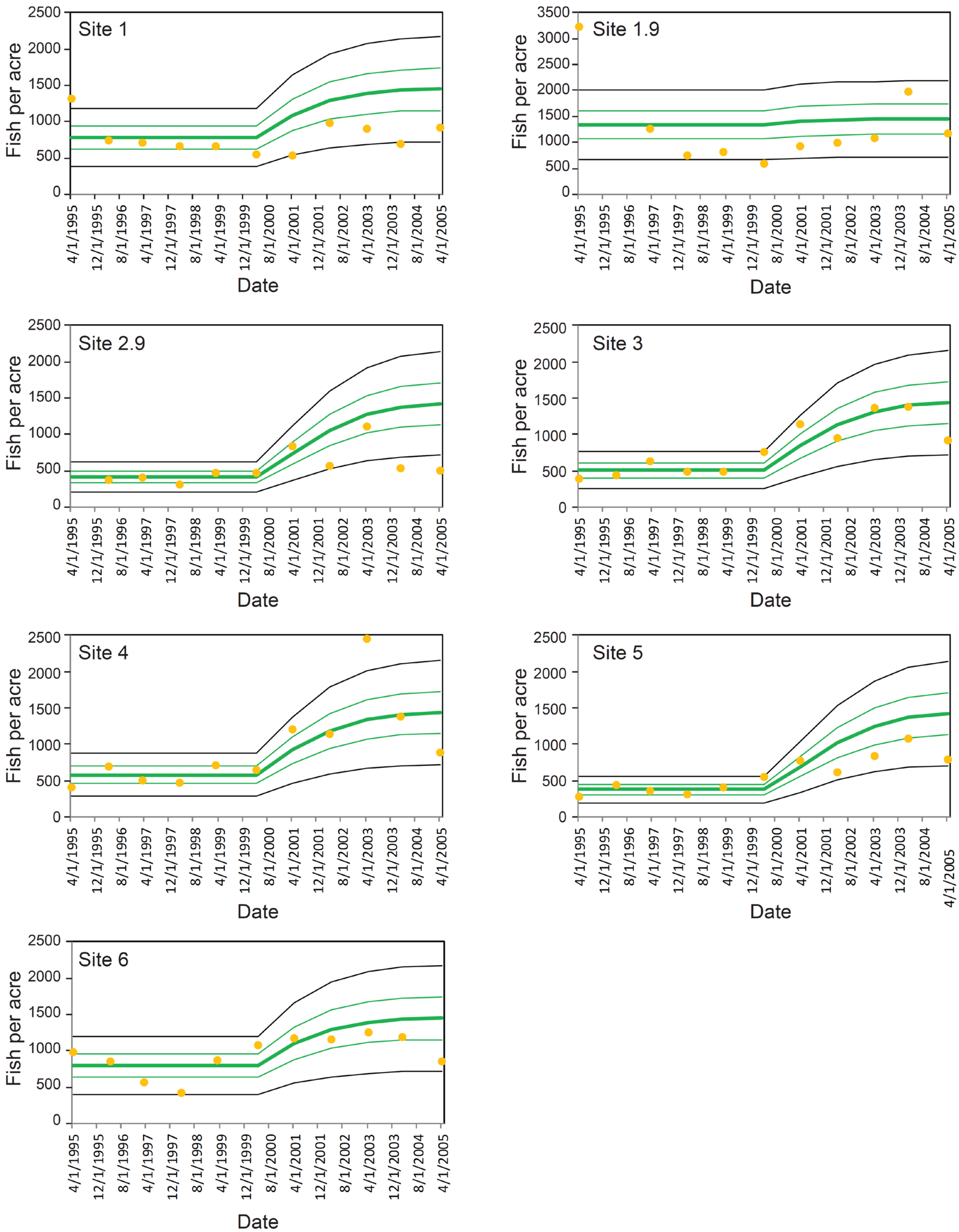

Figure 10.4. Comparison of Fish Population Data with Ecological Population Growth Models. The

heavy green lines represent the population predictions, the thin green lines represent a variance of 20%, and the thin black lines represent 50% variance. Eagle River: Site 1, Site 1.9, Site 2.9, Site 3, Site 4, Site 5, and Site 6.

Effects of residual wastes

Both the Arkansas River and the Eagle River watersheds have trout populations that locally deviate from the predicted values. For both watersheds, the areas of suppressed fish populations can be linked to areas that contain either residual mine waste that was not removed during remedial, contained seepage from waste piles, or fluvial tailings that have not been remediated. In the Arkansas River watershed, the Dinero Tunnel continues to discharge contaminated drainage (Walton-Day and Mills, 2015). In other words, deviations from predicted populations appear to be an indicator of the continued presence of solid or aqueous contamination.

Link between fish population growth predictions and estimates of their value

The recreational angling valuation literature currently provides three methods to link changes in catchable target fish population to estimates of the resulting increase in net economic value, as discussed in the endpoint problem and benefit transfer sections (Sections 6 and 7). The first method is to estimate (or assume) the proportion of increased catchable fish population that will be caught by anglers and multiply this percentage by the value of increased catch (Johnson et al., 1995, Mazzota et al., 2015). The second method is to econometrically estimate the impact that increased fish population has on the number of fish caught, while controlling for other variables such as skill, equipment, species, and angling time (Loomis and Ng, 2009, Ng, 2011). The final method (which is an extension of the second method) estimates the impact of increased catchable population on catch rate and then the impact of the new catch rate on the number of angling days/trips taken (Loomis and Ng, 2009, Ng, 2011). The resulting increase in net economic value is divided by the increase in the catchable fish population to achieve a net economic value for the change in catchable fish population (Loomis and Ng, 2009, Ng, 2011).

The following ecosystem service valuations cannot employ the second method because there are no data available to control for other explanatory variables. For the Upper Arkansas River, there is only *one point* in time and space (sampling site AR-5 in River Reach 1 for the year 2008) where predictions regarding fish population and fish catch coincide. This data point is applied to each of the four sampling sites to allow use of the first (proportional) method above. The final method (WTP per catchable fish) is used to evaluate the sensitivity of the valuation exercise to the proportional linkage assumption made in the first method. For the Eagle River, only the final method of applying WTP per catchable fish is employed.

Ecosystem service valuation

Four sampling sites in the Upper Arkansas River

This section values the geo-environmental model's trout population growth predictions for the Arkansas River watershed (Table 10.1). This valuation estimates the change in net economic value resulting from the initial catchable population's increase to the catchable carrying capacity. It is important to note that this section does not value the full extent of catchable trout population recovery resulting from the California Gulch remediation. Instead, it focuses on the areas surrounding the four sampling sites where the geo-environmental model had sufficient data to model changes in fish populations. (For a broader ecosystem service valuation of the remediation (without the aid of geo-environmental modeling), please see Appendix C. The relative scale of the fishery that was impacted by remediation of the California Gulch Superfund site and the area where sufficient data were available for evaluation are shown in Figure 10.5.

Upper Arkansas River Watershed

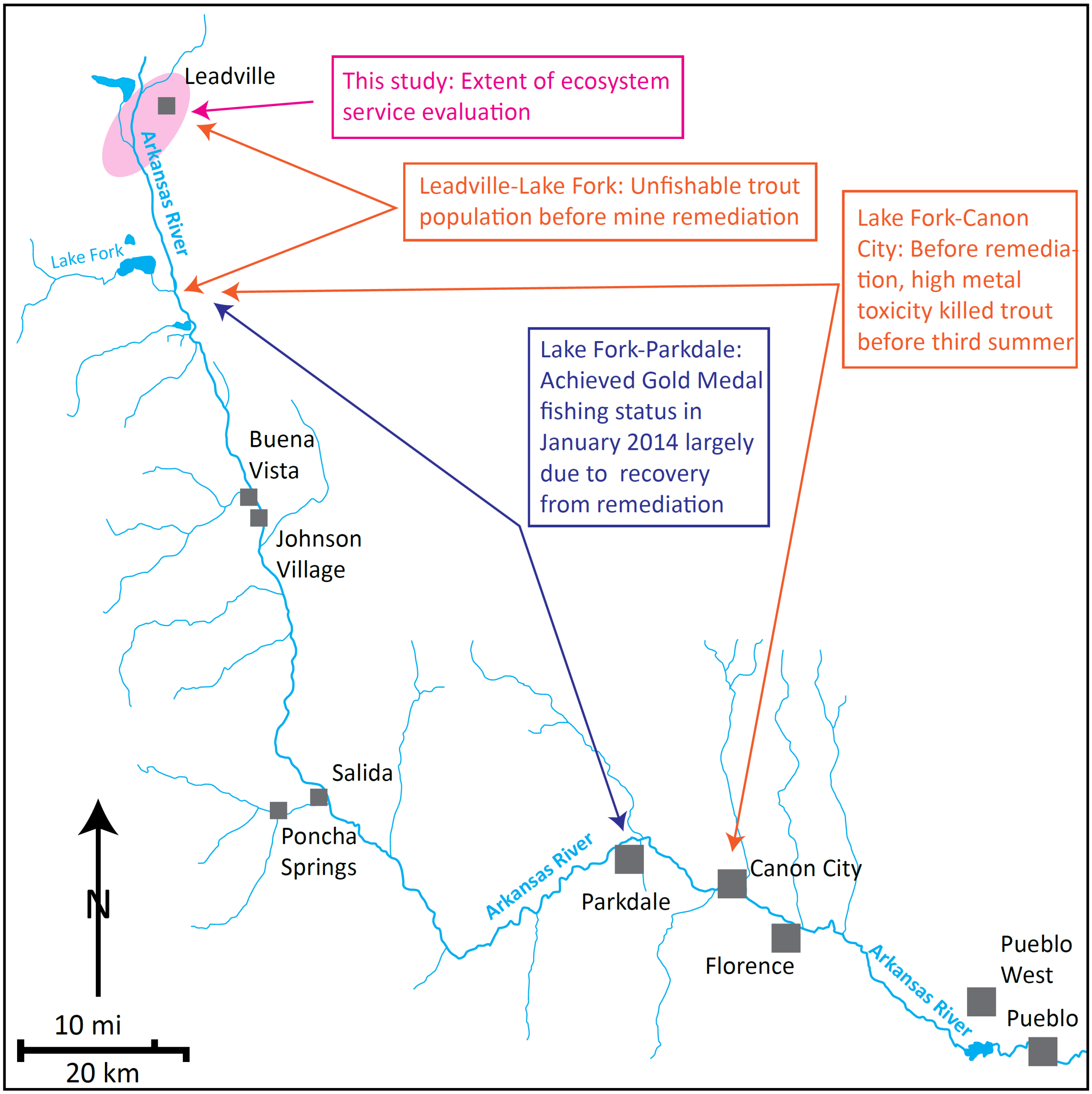

Figure 10.5: Upper Arkansas River: Trout and Mine Pollution Overview

Figure 10.6 illustrates the location of the four sampling sites. The river reaches labeled 1 through 11 represent reaches of the Upper Arkansas River where Colorado Parks and Wildlife (CPW) has estimated angling hours, angling days, and fish caught by conducting creel surveys at particular locations (Policky, 2013). CPW has also extrapolated the localized creel survey results to broader river reaches (Policky, 2013). The angling information collected by the creel surveys is useful for linking changes in fish population to changes in benefits derived from humans, namely anglers. The river reaches labeled EF (East Fork), -1, and 0 represent reaches that have not been surveyed by CPW, but correspond to sampling sites EF-5, AR-1, and AR-5, respectively.

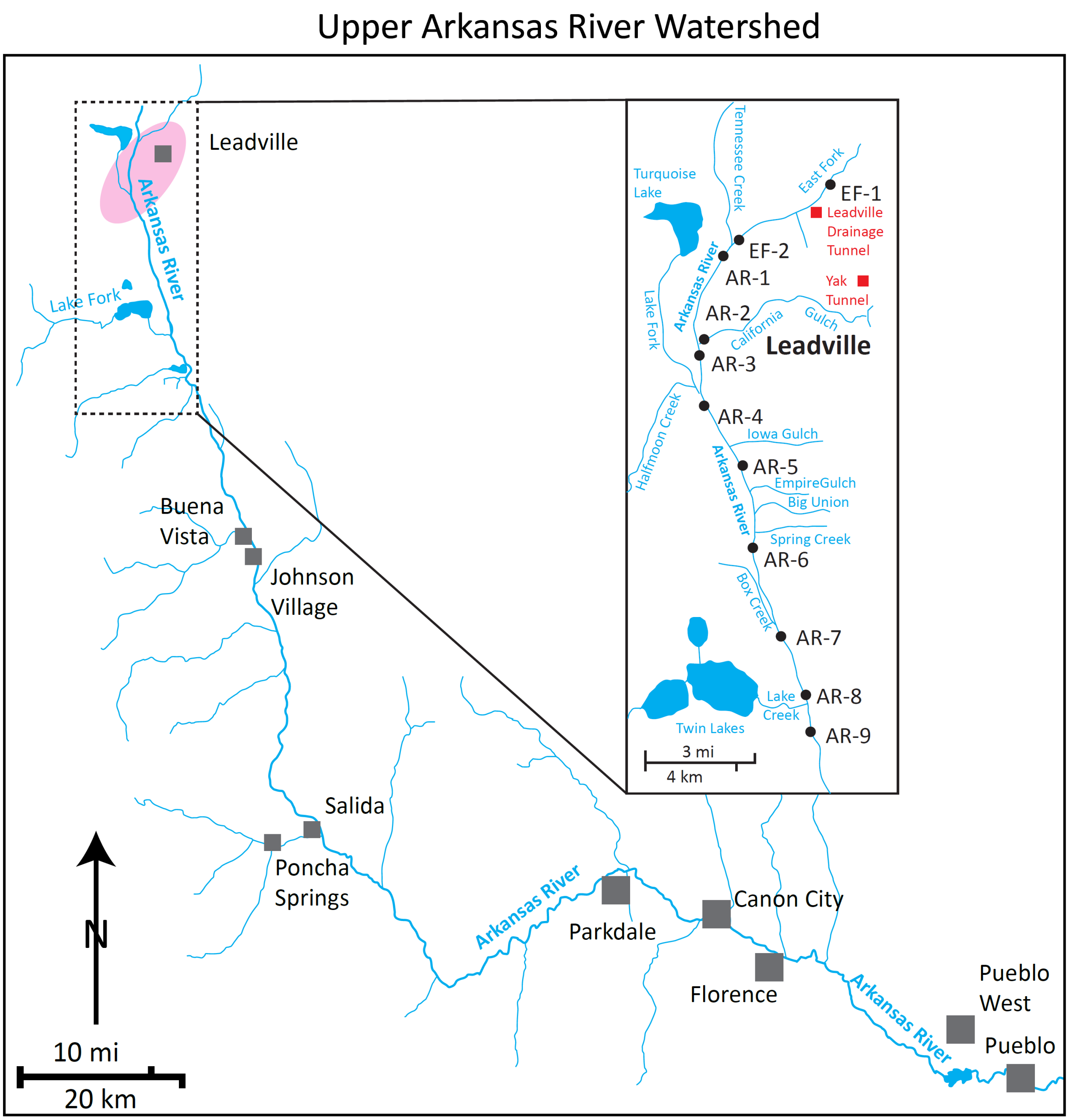

Figure 10.6: Upper Arkansas River: Colorado Parks and Wildlife River Segments and EPA Sampling Sites

Table 10.3 provides detailed information on the size of creel census areas (fishing access points where anglers were surveyed) and their respective river reaches. River reach 1 is the only reach relevant for this analysis, but Table 10.3 describes the relationship between creel census areas and broader river reaches for the Upper Arkansas River all the way down to Parkdale to provide a broader context for this study.

Table 10.3. Colorado Parks and Wildlife River Reaches and Corresponding Creel Census Areas

River Reach	River Reach Name	Miles in River Reach	Creel Census Area	Creel Census Miles
EF	EF-1 - Confluence	2.2		NA
-1	Confluence – California Gulch	3.8		NA
0	California Gulch – Crystal Lakes	2.8		NA
1	Crystal Lakes – Kobe Bridge	5.1	Highway 24 – Kobe	3.2
2	Kobe Bridge – Lake Creek	4.2	Kobe – Two Bit Gulch	2.2
3	Lake Creek – Otero Bridge	10.6	Ball Town – Granite	2.6
4	Otero Bridge – Highway 285 Bridge	9	Otero Bridge – Railroad Bridge	3.1
5	Highway 285 Bridge – Ruby Mountain	6	Big Bend – F Street	5.7
6	Ruby Mountain – Stone Bridge	11.2	Big Bend – F Street	5.7
7	Stone Bridge – Stockyard Bridge	10.9	Big Bend – F Street	5.7
8	Stockyard Bridge – Howard Bridge	11.4	Stockyard Bridge – Badger Creek	5.9
9	Howard Bridge – Lazy J	8.3	Big Cottonwood Creek – Lone Pine	3
10	Lazy J – Texas Creek	12	Big Cottonwood Creek – Lone Pine	3
11	Texas Creek – Parkdale	13.3	Big Cottonwood Creek – Lone Pine	3

Data and methods

Two approaches are employed in this benefit transfer valuation. Both estimate the change in annual net economic benefit as a result of the change in catchable brown trout population from initial to carrying capacity.

The first approach is straightforward because the value estimate employed from the economic literature directly values the endpoint in question – catchable fish population. Modeled fish population estimates (see Figure 10.2) are multiplied by the estimated hectares in each reach[5] and estimated WTP for catchable trout ($0.60 in 2013 dollars from Table 7.9) from Loomis and Ng (2009). The results of this approach are presented in Table 10.4.

[5] Estimated using Google Earth map software.

Table 10.4. Value of Catchable Fish Population, estimated via the WTP for Catchable Fish Approach (2013$)

River Reach	Hectares in Reach	Catchable Fish Population Estimate		Catchable Fish Population Value Estimate (2013$)	
		1995	2008	1995	2008
EF	2.2	1,436	1,456	$1,900	$1,900
-1	4.7	1,454	1,456	$4,100	$4,100
0	4.1	279	1,455	$700	$3,600
1	8.7	261	1,455	$1,400	$7,600
Total	19.6	3,430	5,822	$8,100	$17,200

The second approach requires more complex calculations because the benefit transfer estimate from the literature values fish caught, which is one step removed from the endpoint of catchable fish population. This subtle difference requires the use of creel census data that have been extrapolated by an expert from the creel survey site to the broader river reach. This set of data is used to estimate the percentage of catchable brown trout population that gets caught. The percentage estimate serves as the link between the WTP per fish caught and the ecosystem service. Calculation of the catch percentage requires data on the number of days that anglers spent fishing and their daily catch rate to estimate the number of fish caught for a relevant time and location within the sampling area. Two sets of data points fulfilling this purpose are provided by CPW creel censuses conducted along the Upper Arkansas River between Crystal Lakes and Kobe Bridge for the years 1995 and 2008 (Policky, 2012, Policky, 2013).

Because creel census areas are not representative of their broader river reaches, expert knowledge is required to extrapolate creel census area estimates to broader river reach estimates. Policky (2013) provides an expert extrapolation of creel census angling days to river reach angling days for each river segment in the year 2012. Equation 10.2 combined with Table 10.5 illustrate how the extrapolation factor was calculated for river reach 1 in 2012. This extrapolation factor is then applied to creel census data for the relevant years of 1995 and 2008:

$$ExtrapolationFactor = 1 + \frac{2012ReachDayEstimate - 2012CreelDayEstimate}{2012CreelDayEstimate}$$

Table 10.5. CPW Creel Census and River Reach Angling Day Estimates Used to Calculate Extrapolation Factor

River Reach Number	Creel Census Area Angling Days	CPW Estimate of Angling Days per River Reach	Angling Day Extrapolation Factor
	2012	2012	
1	3,600	7,200	1.99

The extrapolation factor from Table 10.5 is multiplied by creel census angling hours (Policky, 2012, Policky, 2013) to estimate river reach angling hours in Table 10.6, which are then multiplied by the average hourly catch rate to estimate the number of fish caught in Table 10.6.

Table 10.6: Resulting Angling Hour, Fish Catch, and Value of Fish Catch Estimates

River Reach		River Reach Angling Hours	Average Hourly Catch Rate		Estimated Fish Catch	
			1995	2008	1995	2008
1	-	4,700	NA	0.9	0	4,322

To estimate the percentage of the fish population caught by anglers in river reach 1, the number of fish caught is divided by the product of the number of hectares in the reach and the fish population estimate for AR-5 (Table 10.7). River reach 1 is the only location where trout population estimates and creel data coincide.

Table 10.7: Calculation of Percentage of Catchable Fish Population that is Actually Caught

River Reach	Hectares in Reach	Catchable Fish Population		Actual Fish Catch		Percentage of Catchable Fish Actually Caught	
		1995	2008	1995	2008	1995	2008
1	8.7	2,281	12,717	0	4,322	0%	34%

The percentages estimated in Table 10.7 for river reach 1 are applied to reaches 0, -1, and EF as well. Table 9.8 depicts how the predicted catchable fish population estimates are combined with these percentage-catch estimates and WTP per fish caught ($2.94) estimates from USEPA (2006a) to estimate the value of the catchable fish population.

Table 10.8: Fish Population, Catch, Catch Value, and Catchable Population Value Estimates

River Reach	Hectares in Reach	Fish Population Estimate		Fish Catch Estimate		Fish Catch Value Estimate		Catchable Fish Population Value Estimate	
		1995	2008	1995	2008	1995	2008	1995	2008
EF	2.2	3,099	3,143	0	1,068	$0	$3,140	$0	$1,067
-1	4.7	6,785	6,785	0	2,306	$0	$6,780	$0	$2,304
0	4.1	1,113	5,995	0	2,038	$0	$5,991	$0	$2,036
1	8.7	2,281	12,717	0	4,322	$0	$12,707	$0	$4,319
Total	19.6	13,277	28,639	-	9,734	$0	$28,617	$0	$9,726

Discussion and Conclusions

The changes in net economic value resulting from the WTP for catchable fish approach and from the 'percentage fish caught' approach are almost identical - $9,217 ($17,184 - $7,966 in Table 10.4) and $9,726 ($9,726 - $0 in Table 9.8) per year, respectively. In comparison, the total documented costs for clean up at Leadville exceed $138,000,000 (Table C.9). The first approach relies on WTP for catchable fish estimates from one study (Loomis and Ng, 2009), whereas the second approach relies on a meta-analysis of WTP for fish caught estimates (USEPA, 2006a). The first approach is appealing because of its simplicity, but future research must work to develop the economic valuation literature regarding WTP for catchable fish. The second approach is appealing because the WTP estimate from USEPA (2006a) is grounded in a robust valuation literature search. However, the results are sensitive to the accuracy of creel surveys, the extrapolation of these survey results to wider river reaches, and, above all, the percentage of catchable fish in the population that are actually caught.

The valuation exercise in this section successfully links the geo-environmental ecosystem service outputs to estimates of their value. However, the limited nature of available data limited this linked natural and social science exercise to a small portion of the whole fishery that was improved as a result of the California Gulch Superfund remediation. Appendix C provides an attempt to more broadly characterize fishery benefits of remediation in the absence of geo-environmental model outputs.

Six sampling sites in the Eagle River

This brief section values the model's trout population growth predictions for the sampling sites on the Eagle River (Table 10.2). It is important to note that this section focuses narrowly on the seven sampling sites where the model had sufficient data to model changes in fish populations. No connection could be made between creel survey data and relevant river reaches, so the change in net economic value estimates below cannot be extrapolated beyond their units of dollars per hectare. Therefore, the information presented in Table 10.9 is best viewed as an untested hypothesis about the value of this recovering ecosystem service in the Eagle River watershed. Future creel and economic surveys will be needed to evaluate these predictions.

Table 10.9. Trout Population Characteristics and Estimated Values for Eagle River Watershed

Site	Initial Population (trout/ha)	Carrying Capacity (trout/ha)	Initial Population Value Estimate ($/ha)	Carrying Capacity Population Value Estimate ($/ha)	Change in Net Economic Value ($/ha)
Site 1	786	1456	$472	$874	$402
Site 1.9	1340	1456	$804	$874	$70
Site 2.9	418	1456	$251	$874	$623
Site 3	508	1456	$305	$874	$569
Site 4	582	1456	$349	$874	$524
Site 5	379	1456	$227	$874	$646
Site 6	803	1456	$482	$874	$392

Recommendatons

This study has pioneered the integration of: 1. a geologically-based environmental assessment of remediation of abandoned mines sites dominated by MIW, 2. the progress of associated in-stream biologic recovery as a result of remediation, and 3. the potential economic impacts of that remediation and biologic recovery due to sport fishing. The study has relied on data from existing sources to conduct this exercise. The study has also utilized existing models in disparate fields such as geochemistry, hydrology, population ecology, and economics to form this integrated approach. As originally envisioned, this study planned to conduct similar assessments for, 1. mining districts that developed a variety of mineral deposit types having different environmental contaminants, and 2. several ecosystem services. Ultimately, the scope of the study was limited by the availability of adequate data from abandoned mine sites and their watersheds, and by the availability of suitable geochemical, ecological, and economic models.

In the course of the geochemical and hydrological modeling in this study, the greatest limitations were found in the availability of data. For the sites used in the study, minimum requirements included pH, hardness, and trace metals, specifically Zn, Cd, and Cu. A broader range of water quality parameters would have enabled more comprehensive modeling of source, transport, and fate processes than was afforded by the available data.

The modeling of the trout population ecology used an admittedly simplistic approach to model trout population growth, the logistic function, yet that level of sophistication was compatible with the level of data available (Frank et al., 2011). With the current model, the input parameters included a starting population, a growth rate, and the carrying capacity of the stream. The starting population was easily obtained from the available data. However, the growth rate and carrying capacity required a number of assumptions. A better estimate of the carrying capacity may be obtainable from investigations in nearby streams in adjacent watersheds unaffected by MIW. The growth rate was estimated based on the fastest and most systematic increase in population in the Arkansas River watershed. A more accurate growth rate for recovering natural populations will likely require extensive literature review of numerous case studies. Despite the simplistic approach embodied in the logistic function, the uncertainties seem to fit within the uncertainties observed in healthy natural populations year to year (Dauwater et al., 2009). More sophisticated population dynamics modeling requires information such as age class structure of populations and spawning efficiency, among others, that are beyond the scope of the current study (Frank et al., 2011).

Conclusions

Decisions regarding mine site remediation can be clouded by the fact that its costs are easily estimated but its benefits are not. Remediation costs for labor, equipment rental, and materials can all be estimated within a reasonable range. However, benefits such as water of higher quality, improved fish habitat, or healthy soil generate no revenue. For proposed mines, the opposite is true. The *benefits* are well summarized in discounted cash flows, while the *costs* of impact to the environment and communities are less clear. A goal of this project was to explore a possible framework to begin to remedy this problem.

Recent mine spills and mine development controversies reveal the importance of three ecosystem services modeled via benefit transfer valuation (catchable fish population in Section 6.1, drinking water in Appendix B, and aquatic habitat in Appendix B). Concerns about the Mount Polley mine tailings spill in August 2014 (British Columbia, Canada) focused on salmon killed (or diverted from spawning habitat), household drinking water intake quality, and impacts to aquatic habitat. The ecosystem service valuation models presented in Section 6.1 and Appendix B are capable of valuing much of the ecosystem damage from this spill if ecologists are able to model resulting changes to ecosystem services. Similarly, public concern about development of the contentious Pebble project in southwestern Alaska focuses on possible impacts to salmon populations and salmon habitat. In short, mine site pollution commonly affects water and fish. Estimation of the value of related ecosystem services captures major components of the value of mine site pollution, while abiding by the restrictive ecosystem service framework.

In contrast, the August 2015 Gold King mine spill in southwestern Colorado demonstratred the shortcoming of current scientific and economic understanding when it comes to modeling the long-term ecosystem service values impacted by a mine spill. First, the Animas River was not a pristine watershed before the Gold King mine briefly turned it yellow. Any attempt to study the ecosystem service impacts of the spill would first have to disentangle the impacts of background levels of contamination from the impacts of contaminants introduced by the spill. Assuming that this could be done, ecologists would then have to trace the impact of the spill's contaminant through the ecosystem until they were able to quantify changes to an ecosystem service that economists could value.

For example, one concern related to the Gold King spill was the long-term contamination of river bed sediment. This contamination could conceivably make its way into plants/micro-organisms and work its way up the food chain until it had an impact on an ecosystem service, such as fish population, bird population, or wildlife population. If it were scientifically possible to trace the contaminants through the ecosystem, then economists would have to figure out how to value these ecosystem service changes. Huntable bird and wild life populations do not have meta-analyses that have valued WTP to shoot another bird/deer, as USEPA (2006a) has done for recreational angling. Existing estimates of WTP for a day of hunting do not relate directly enough to the ecosystem services in question.

As for the ecosystem services mentioned in Appendix A, provision of ecosystem service changes, additional time, and additional funding could allow value estimation and incorporation into the benefit transfer model. Groundwater contamination, water supply reliability, and irrigation water are three of the most important ecosystem services that were not valued. Such research would also be useful for decisions regarding unconventional oil and gas development.

Future research emanating from this analysis calls for a full-scale characterization of ecosystem service impacts from acid mine drainage in the Animas watershed and predictive modeling of ecosystem service impacts from large, open-pit tailings storage facility failures. The Animas is representative of many watersheds impacted by acid mine drainage in the western United States. Characterization of the impacts could lead to estimation of benefits from remediation. While remediation is mainly driven by regulatory concerns, it is possible that defensible remediation benefit estimates could channel more resources towards acid mine drainage remediation.

Predictive modeling of ecosystem service impacts from worst-case scenario failures may help engineers of future mines to more accurately assess financial risks during design trade-off studies. From a regulatory perspective, it would also provide more defensible amounts for environmental liability bonding. In addition, insurance markets are currently unwilling to cover low frequency, high consequence events like the tailing spills at Mount Polley and Samarco. If they could forecast ecosystem service impact values and spill frequency, maybe they would be more likely to insure such events. Insurance would spread the risks more widely and could provide a market mechanism to help regulate these catastrophic failures.

Finally, the Office of the President recently mandated that all federal agencies begin incorporating the value of ecosystem services into federal decisions (EOP, 2015). This chapter represents a robust analysis of ecosystem service valuation for federal decisions related to the extractive sector. Future research ought to be conducted to incorporate this model into the decision-making frameworks of relevant federal agencies.

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Appendices

Appendix A: Review of Environmental (and Ecosystem Service) Valuation Literature Pertaining to Mine Site Pollution

To employ the benefit transfer valuation technique, primary valuation studies must be located that correspond to the 'policy' site in question. Given that the goal of this analysis is to value changes in ecosystem services related to mine sites, the following section reviews the environmental valuation literature that relates to mine site pollution. As described below, the number of environmental valuation studies geared towards abandoned, proposed, or operating mine sites is rather small. Each study from this narrow literature resource is highlighted and summarized below. The purpose of this review is to uncover potential material for the construction of a mine site pollution benefit transfer model.

Social Cost-Benefit-Analyses of Mine Remediation Schemes

A common theme of this small body of literature is social cost-benefit-analysis of various remediation schemes. Randall et al. (1978) employed water treatment costs, fish restocking costs, government established per-day recreation values, and visual disamenity valuation to estimate the benefits of proposed tightening of state and federal regulations regarding the reclamation of surface coal mines. While Randall et al. (1978) provided an instructive framework for (and application of) environmental valuation of mine site pollution, it would be difficult to transfer the benefit results to a policy site that did not have the same topography, geology, hydrology, geochemistry, and ecology. This difficulty is because the authors accounted for specific details in slope, sediment loading, host-rock generated acid mine drainage, recreational characteristics of the area, and the visual impacts of surface mining of coal. A benefit transfer practitioner would be hard pressed to demonstrate site correspondence between this study site and a Central Colorado hard rock mining site. Information can be obtained from this study, but one would have to delve deeply into the basic components such as mine waste management, water treatment, and fish replacement practices to approximate transferable benefits.

Similarly, Michael and Pearce (1989) conducted a social cost-benefit-analysis of a remediation project in northwestern England that reclaimed a large abandoned coal field. A residential area was built around coal spoil heaps that caught fire, collapsed into gardens, blew dust, and which also included mine shafts. The coal field was turned into an agricultural area with forested footpaths and soccer fields. Michael and Pearce (1989) employed a contingent valuation to ascertain the total economic value that the average household placed on the remediation. As with Randall et al. (1978), this study is instructive. However, its results could only be transferred to an equally flammable and dangerous policy site.

Button et al. (1999) proposed a framework for valuing remediation benefits, but did not produce a benefit value estimate. Nonetheless, Button et al. (1999) is an important literature source because it highlights the complexities, synergies, and difficulties associated with remediation of mine site pollution. For example, Button et al. (1999) noted that the sum of the benefits from remediating many sources of pollution within a watershed will be larger than remediating the parts. Also, Button et al. (1999) mentioned that remediation can be coupled with heritage activities (such as highlighting a region's mining history) to augment the benefits of remediation. Finally, Button et al. (1999) discussed the difficulty and cost of collecting site-specific economic valuation information and suggested benefit transfer as the best alternative.

Farber and Griner (2000) employed a conjoint analysis, in conjunction with a random utility model, to value various combinations of stream quality improvements. The two study sites were in western Pennsylvania and could support fishing, boating, and hiking. Depending on policy site correspondence (for stream quality improvement, recreational characteristics, and population), Farber and Griner (2000) may be useful in benefit transfer for mine site pollution affecting stream quality.

The studies of Damigos and Kaliampakos (2003a) and Damigos and Kaliampakos (2003b) were derived from the same contingent valuation of the proposed remediation of an abandoned rock quarry in Athens, Greece. This contingent valuation does not appear to be of sufficient quality to transfer the benefit value. Also, it would appear to only correspond to other urban quarry sites.

Ahlheim et al. (2004) and Lienhoop and Messner (2009) both applied a rigorous contingent valuation to a remediation scheme in East Germany that converted open pit coal mines into recreational lake parks. It is easy to imagine using these studies as study sites for open-pit hard rock mines in the United States that could economically be turned into lake parks. Such a possibility would depend on site-specific hydrology and rock chemistry. However, a lake for recreation would likely provide more economic benefits than a bare, abandoned open pit.

Mendes et al. (2007) created a framework to value the non-market economic benefits of remediating an open pit copper-silver-gold mine and smelter site in Portugal. Mendes et al. (2007) set the stage for a contingent valuation at the site, but like Button et al. (1999) they were unable to achieve a benefit value result. Nonetheless, Mendes et al. (2007) is a good example of how much preparation must be conducted for primary valuation and of how prediction of physical impacts of the remediation is required to conduct a valuation.

Williamson et al. (2008) employed a hedonic study in the Cheat River watershed of West Virginia and showed that being within a quarter mile of an acid mine drainage impaired stream reduces home property value by $8,525 (2013$). Each of the studies above highlights the difficulty of employing mine site valuations in an ecosystem service framework. The sites are valued wholesale, and the value of a particular ecosystem service cannot be parsed from others. The study by Randall et al. (1978) is the only exception to this rule. Site correspondence requirements pose another major challenge because of the differences between coal sites, hard rock sites, and remediation schemes. In other words, these studies are only useful for benefit transfer at sites that correspond to the site and context in question.

Burton et al. (2012) used conjoint analysis to estimate what the public would be willing to accept for reducing various bauxite mine remediation schemes around Perth, Australia. The focus was on timing and reductions in plant species, richness, wildlife habitat, and bird populations. Like many of the previous studies, it is difficult to translate Burton et al. (2012) into a benefit transfer. The benefits/losses being estimated were particularly site-specific, and the policy site would have to have been an excellent match for a valid benefit transfer.

While not a remediation scheme, Neelawala et al. (2013) used a hedonic property value model to estimate the marginal willingness to pay to be farther from mining and smelting operations in Queensland, Australia. The result is that households were willing to pay $13,703 (2013$) to be one km farther from the pollution source when they are within a 4-km radius (Neelawala et al., 2013).

In summary, valuable information can be gleaned from these social cost-benefit analyses of mine site remediation, but a benefit transfer model for the purpose of this analysis cannot be built on these studies. The studies by Farber and Griner (2000) and Williamson et al. (2008) may prove useful for benefit transfer of the value of various stream-quality improvements, although it is not clear exactly which ecosystem/environmental services are being valued. The studies by Ahlheim et al. (2004) and Lienhoop and Messner (2009) could provide benchmarks for the value of turning open-pit mines into recreational lakes, but that is not common practice in the United States.

Social Cost-Benefit-Analyses of Proposed Mine Sites

The second common theme within this literature is social cost-benefit analysis of proposed mine sites. Trigg and Dubourg (1993) took a hypothetical and expert opinion approach to a social cost-benefit analysis of a proposed coal strip mine in North Staffordshire, England. Trigg and Dubourg (1993) surveyed real estate agents for their expert opinion on how much property values would decrease if the proposed mine was developed. The average estimate was roughly 30 %. This approach is certainly not up to National Oceanic and Atmospheric Administration (NOAA) contingent valuation standards, and it would be dubious to transfer the results.

The study by Damigos and Kaliampakos (2006) was a social cost-benefit analysis of a proposed open pit gold mine in Greece, which was funded by the project's owner. Damigos and Kaliampakos (2006) indiscriminantly used results from many of the previously mentioned studies for benefit transfer (Damigos and Kaliampakos, 2003a, Randall et al., 1978, Trigg and Dubourg, 1993). While Damigos and Kaliampakos (2006) provided a framework for social cost-benefit analysis of proposed mining projects, it did not appear that the policy site (an open-pit gold mine in Greece) corresponds to the study sites from which the benefits are transferred (coal fields and urban quarries). For example, Damigos and Kaliampakos (2006) employed the 30% property value reduction from Trigg and Dubourg (1993). Considering that Trigg and Dubourg (1993) did not identify which environmental goods (for example, view, air quality, and congestion) were responsible for the decrease in property value, it seems difficult for the approach of Damigos and Kaliampakos (2006) to directly transfer these values. Each benefit transfer within Damigos and Kaliampakos (2006) was of this nature. The study site was related, but it is unclear how similar it was to the policy site or how valid the benefit transfer was. The research by Unaldi et al. (2011) was a study that essentially duplicated the efforts of Damigos and Kaliampakos (2006), but it was for a generic gold deposit in Turkey. In short, the social cost-benefit-analyses of proposed mine sites were of low quality and should not be used.

Natural Resource Damage Assessments Related to Mine Sites

The final sources of valuation studies related to mining are natural resource damage assessments (NRDA). Estimates of economic benefit values are sparse in the NRDA literature. EPA's CERCLA cleanup of the Eagle Mine site in Gilman, Colorado prompted litigation regarding cleanup costs, liability, and monetary compensation. Competing expert witness valuation analyses from the trustee (Rowe and Schulze, 1985) and the defendant (Ward and others, 1992) yielded completely different estimates. The plaintiff's expert witness conducted a contingent valuation at the local, county, and state level where the focus of the survey was the subject's willingness to pay to clean up the entire site (Rowe and Schulze, 1985). Once the value was elicited, the survey asked respondents what percentage of their willingness to pay value was represented by use value, non-use value, and existence value (Rowe and Schulze, 1985). The defendant's expert witness, on the other hand, estimated replacement cost values to cover contaminated areas with topsoil and to redrill wells for residential water (Ward et al., 1992). These replacement values had little to do with the benefits that society has foregone as a result of pollution, or society's willingness to pay to clean up the site.

The Coeur D'Alene, Idaho NRDA provided many cost estimates, but no benefits were estimated (USEPA, 2002). Similarly, a preliminary estimate of damages at Leadville (IEc., 2006) employed a *habitat equivalency analysis* (HEA) that valued damage by the cost that would be required to repair the habitat to its original condition. The main shortcoming of IEc. (2006) (as with all habitat equivalency analyses) was that its economic assessment of damages failed to incorporate how the changes in these ecosystem services affected human well-being. Instead, IEc. (2006) calculated damages using abatement costs required to return the ecosystem services to their baseline condition. As a result, HEA was unlikely to result in efficient outcomes because there was no balancing of costs and benefits of remediation. HEA was also the preferred approach for the Holden Mine site in Washington, the Southeast Missouri Lead Mining District, and the Blackbird Mine in Idaho.

Like many of the valuations above, these natural resource damage assessments are not useful for benefit transfer. The contingent valuation from Rowe and Schulze (1985) valued the whole site, rather than ecosystem services. Ward et al. (1992) estimated replacement values, which are the minimum value for the ecosystem service in question. The Coeur D'Alene, Idaho NRDA did not estimate any benefits. The habitat equivalency assessment for Leadville was a replacement cost approach, not a technique for the optimization of social resources. An ideal literature portfolio would provide numerous environmental valuation studies of mine site pollution. This literature review demonstrates the limited number of studies on the subject.

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Appendix B: Starting Points for Benefit Transfer Modeling of Additional Ecosystem Services Impacted by Mine Site Pollution

Soil Quality for Human Health: Lead

Soil quality at legacy mine sites has often been contaminated by pollutants in mine tailings and from smelting operations. Historical mine tailings have been scattered by wind and water over the decades. Many historical smelting operations did not capture pollutants from their smoke stacks, so they dispersed onto the surrounding communities[6]. When soil is contaminated with a pollutant such as lead, it can become a pathway to children via unintentional soil ingestion. For example, EPA models concluded that children in Leadville, Colorado who had backyard soil lead levels greater than 500 ppm were 8.4 times more likely to have blood lead levels at or above 10 µg/dL (CDH, 1990, p. 33), which causes health defects and triggers intervention (Gould et al., 2009). When tested, these models over-estimated the connection between backyard soil lead level and blood levels. Nonetheless, one can imagine that high levels of lead in the surrounding environment could conceivably raise blood lead levels. To deal with this problem, Leadville initiated a voluntary program called *Kid's First* to monitor children's blood lead level.

Elevated blood lead levels in children have been linked to IQ loss, attention deficit hyperactivity disorder, the need for special education, and criminal behavior (Gould et al., 2009). Gould et al. (2009) have summarized the medical literature on the effects of elevated blood lead level and the monetization of their effects. Pairing Gould et al. (2009) with population blood lead level data provides a start in modeling the value of soil quality improvements at a legacy site. Table B.1 uses Leadville as an example of this process.

Several problems with this approach are enumerated below. First, the lead contamination in the children may have nothing to do with lead from the surrounding environment. Blood lead levels for children in Leadville were often on par with blood lead levels for children in inner cities. The argument was made by Leadville residents (and the Potentially Responsible Parties [PRPs]) that high blood lead levels could have been the result of lead paint. Second, a molybdenum mine that employed many Leadville residents closed at the same time that EPA designated the area as a Superfund site. This mine closure resulted in significant emigration from the town, especially for families employed by the mine. Many children from Leadville were never included in the baseline population. Third, it is not possible to determine how much of the improvement in blood lead level was due to remediation vs. behavioral changes. For example, if children washed their hands after playing outside and before eating, that may have been just as effective as physically removing the mine waste and tailings. In other words, there is no direct link between a change in

[6] Soil quality contamination at prospective mine sites is expected to be less pronounced due to modern operators active management of tailings and smelting operations.

the ecosystem service and the impact to be valued. Finally, the medical literature appears divided on the issue of whether lead in soil contributes to lead absorption by children (Kimbrough and Krouskas, 2012).

Table B.1. Impact of Elevated Blood Lead Levels in Children from Leadville, Colorado (2013$)

Blood Lead (µg/dL)	Medical Cost[a]	IQ Loss per µg/dL[a]	Lost Earnings per Child[a]	Cost of Special Education[a]	Number of Children (1991)[b]	1991 Cohort Total
2-10	$0	0.513	$63,608	$0	285	$18,100,000
10-15	$86	0.19	$49,080	$0	25	$1,200,000
15-20	$86	0.19	$68,712	$0	4	$300,000
20-25	$1,400	0.11	$51,147	$49,823	0	$0
25-45	$1,400	0.11	$79,562	$49,823	0	$0
45-70	$1,549	0.11	$130,709	$49,823	0	$0
> 70	$3,995	0.11	$181,856	$49,823	0	$0
Total					314	$19,600,000

Source: [a] Gould et al. (2009), [b] EPA (1996)

Air Quality for Human Health: Particulate Matter

Air quality can also be affected at mine sites by suspended particulate matter. Smith and Huang (1995) used a hedonic property value model to estimate the value of changes in total suspended particulate matter. The median value of $52.39 (2013$) per household to reduce total suspended particulates by 1 mg/m^3 was estimated (Smith and Huang 1995). Similarly, Vassanadumrongdee et al. (2004) provided a starting place for valuing the short-term health effects of air pollution such as coughing, congestion, and asthma attacks.

Air Quality for Human Health: Mercury Emissions

A wide review of the literature suggests that IQ loss due to the consumption of mercury-contaminated fish is the only properly monetized damage estimate relating to mercury emissions (Sundseth et al., 2010). In the early 2000's, regulations were proposed to require coal-fired power plants in the United States to abate mercury emissions from burning coal. These regulations spawned attempts to weigh the costs and benefits of mercury emission abatement. Several studies were conducted to map the chain of mercury emission, mercury deposition, conversion to methylmercury, methylmercury bioaccumulation, consumption of contaminated fish, ensuing impact on fetal cognitive functioning, and loss of IQ (Hylander and Goodsite, 2006, Mergler et al., 2007, Rice and Hammitt, 2005, Seigneur et al., 2004, Spadaro and Rabl, 2008, Sundseth et al., 2010, Swain et al., 2007, Trasande et al., 2005, UNEP, 2013). These research efforts linked atmospheric, oceanic, chemical, biological, and economic models - a titanic task fraught with complexity. In short, these analyses focused on the roughly 2 % of elemental mercury (Hg°) that becomes methylated, is ingested from fish fillets[7], and affects the fetal nervous system.

From this literature, Trasande et al. (2005), Rice and Hammitt (2005), and Spadaro and Rabl (2008) provided estimates of the translation from a quantity of mercury emission to a dollar amount of lost earnings due to fetal IQ loss. The goal of Trasande et al. (2005) was to estimate the economic costs of fetal neurodevelopmental impacts attributable to mercury emissions from American power plants. To achieve this, Trasande et al. (2005) combined an environmentally attributable fraction (EAF) model with national blood mercury prevalence data from the Centers for Disease Control and Prevention. They found that between 316,588 and 637,233 children each year have cord blood levels greater than the 5.8 μg/L level associated with loss of IQ. The 5.8 μg/L of cord blood level serves as the neurotoxicity threshold for all estimates from that study.

Trasande et al. (2005) estimated damages to the American economy due to IQ loss in an annual birth cohort from mercury deposited in the United States from three sources. First, global anthropogenic emissions are assumed to deposit 87,000 kg of mercury in the United States. Assumptions regarding cord/maternal Hg blood level ratios and linear/logarithmic IQ decrements produced a range of estimated damages from $2.9 billion (B) to $59.2B (2013$). Within this range, Trasande et al. (2005) recommended a cord/maternal ratio of 1.7 and a logarithmic model that resulted in a recommended value of $11.8B (2013$) for damages from mercury deposited in the United States from global anthropogenic sources. Second, American anthropogenic emissions were assumed to deposit 52,200 kg of mercury in the United States. The range of estimated damages is $0.5B-$21.4B (2013$), and the recommended value is $4.2B (2013$). Finally, estimates from American anthropogenic emissions from coal-fired power plants provided a range $0.1B-$8.8B and a recommended value of $1.8B. Averaging the low, recommended, and high estimates of damages per kg results in estimates of $11,000, $60,000, and $193,000, respectively.

In a similar study, Rice and Hammitt (2005) estimated the economic benefits of greater control of mercury emissions from coal-fired power plants in the United States. Mercury emissions reduction was assumed to have a linear and proportional decrease in methylmercury

[7] While methylmercury primarily concentrates in fish organs, methylmercury concentrations in the muscle tissue are approximately 50% of liver concentrations (Oliveira Ribeiro et al., 1999). The USGS found that 27% of fish sampled in US streams had skinless-fillet methylmercury concentrations higher than the EPA human-health criterion (Scudder, 2010). Additionally, the mean methylmercury concentration of skinless-fillets from the 59 fish sampled in basins with gold mining exceeded the EPA human-health criterion (Scudder, 2010, p.12).

concentrations in fish. Changes in deposition rates were based on regional deposition modeling from the EPA's analysis of the Clear Skies Initiative, under which power plants reduced mercury emissions from 49,000 kg/year to either 26,000 kg/year or 15,000 kg/year. Human exposure to methylmercury was modeled through commercial and non-commercial harvest of fish. Rice and Hammitt (2005) used dose-response functions from recent methylmercury epidemiological studies and data on fish consumption from the FDA to estimate damages of mercury deposition. The estimates provided by Rice and Hammitt (2005) that are useful for this analysis are two estimates of damages to the American economy due to IQ loss in an annual birth cohort from mercury deposited in the United States by all sources. The first estimate of $4.2B assumed a neurotoxicity threshold of maternal methylmercury intake greater than 0.1 μg/kg of fish per day. The second estimate of $26.9B assumed that there was no neurotoxicity threshold. Dividing these estimates by the assumed 124,300 kg deposited in the United States by all sources yields damages estimates of $33,000 and $208,000, respectively.

Finally, Spadaro and Rabl (2008) used worldwide *average* methylmercury doses from fish to calculate global damages from total (anthropogenic and non-anthropogenic) emissions. Spadaro and Rabl (2008) defined a comprehensive transfer factor for ingestion of methylmercury as a ratio of global average dose rate (2.4 μg/day) and global emission rate (6,000t/year). The immediate problem with this approach is that methylmercury damages primarily come from the high doses ingested by those consuming large amounts of fish. Using a global average dose rate smooths the high doses out across the population to the point where they appear to have no effect. Additionally, Spadaro and Rabl (2008) scaled damages based on income. For these reasons, the estimates from Spadaro and Rabl (2008) are not incorporated in the averages for the recommended value in Table B.2.

The IQ loss estimates from Trasande et al. (2005) and Rice and Hammitt (2005) were normalized to reflect the same value per IQ point of Spadaro and Rabl (2008). The estimates were then inflated to 2013$ using the CPI. One of the most influential factors in the calculation of lost earnings from methylmercury poisoning is the incorporation of a neurotoxicity threshold. The neurotoxicity threshold reduces the estimate of lost earnings by ruling out the large contingent of infants who have trace amounts of MeHg in their blood. Current scientific understanding indicates that a neurotoxicity threshold exists. Therefore, the recommended value is an average of the non-outlier estimates that have a neurotoxicity threshold. Table B.2 details the valuations and their conversion into lost IQ per kilogram of mercury released. The recommended value for lost lifetime earnings due to IQ loss from 1 kg of vaporized mercury is $53,000.

The main weakness of this approach is the assumption of constant marginal impacts from each kilogram of mercury emitted into the air. By dividing the economy-wide damage estimates from Trasande et al. (2005) and Rice and Hammitt (2005) by the kilograms of mercury deposited, this analysis assumes that the effect of each kilogram of mercury deposited is a linear function. Future research needs to be conducted in the same vein as Spadaro and Rabl (2008) to more accurately estimate the marginal impacts from each kilogram of mercury emitted into the air. In absence of such research, this analysis provides a strong starting point for comparing the environmental impacts of various forms of gold mining.

Table B.2: Valuation of Environmental Damage Due to 1 kg Release of Mercury into the Atmosphere

Trasande et al. (2005) Cost of American Anthropogenic Coal Power Plant Hg Emissions Deposited in US	
Mercury deposited in the United States from anthropogenic sources (kg)	48,000
Damages to American economy due to IQ loss in annual birth cohort[a] (2013$)	$1.8 billion
Lost lifetime earnings due to IQ loss from 1 kg Hg air release (2013$)	$27,000*
Trasande et al. (2005) Cost of American Anthropogenic Hg Emissions Deposited in US	
Mercury deposited in United States from anthrogenic sources (kg)	52,200
Damages to American economy due to IQ loss in annual birth cohort[a] (2013$)	$4.2 billion
Lost lifetime earnings due to IQ loss from 1 kg Hg air release (2013$)	$57,000*
Trasande et al. (2005) Cost of Global Anthropogenic Hg Emissions Deposited in the US	
Mercury deposited in United States from anthrogenic sources (kg)	87,000
Damages to American economy due to IQ loss in annual birth cohort[a] (2013$)	$11.8 billion
Lost lifetime earnings due to IQ loss from 1 kg Hg air release (2013$)	$97,000*
Rice and Hammitt (2005) Cost of Global Anthropogenic Hg Emissions Deposited in the US	
Mercury deposited in United States from anthrogenic sources (kg)	124,300
Damages to American economy due to IQ loss in annual birth cohort[a] (2013$)	$4.2 billion
Lost lifetime earnings due to IQ loss from 1 kg Hg air release (2013$)	$33,000*
Rice and Hammitt (2005) Cost Estimate *Without* Neurotoxicity Threshold	
Mercury deposited in United States from anthrogenic sources (kg)	124,300
Damages to American economy due to IQ loss in annual birth cohort[a] (2013$)	$26.9 billion
Lost lifetime earnings due to IQ loss from 1 kg Hg air release (2013$)	$208,000
Trasande et al. (2005) Cost Estimate with Alternative Linear Model	
Mercury deposited in United States from anthrogenic sources (kg)	87,000
Damages to American economy due to IQ loss in annual birth cohort[a] (2013$)	$44.5 billion
Lost lifetime earnings due to IQ loss from 1 kg Hg air release (2013$)	$366,000**
Global Estimate from Spadaro and Rabl (2008)	
Lost lifetime earnings to global economy due to IQ loss in annual cohort[a] (2013$)	$1,818**
Lost lifetime earnings to global economy due to IQ loss in annual cohort[b] (2013$)	$4,056
Recommended Value: Average of non-outlier threshold estimates	$53,000***
Average of all estimates	$127,000
Average of all threshold estimates	$115,000

[a] Indicates a neurotoxicity threshold is assumed
[b] Indicates no neurotoxicity threshold is assumed
* Indicates the value is included in the average of non-outlier threshold estimates
** Indicates an outlier value

The following example provides a concrete context for the model above. Gold Quarry is an open pit gold mine in Nevada. The primary impact to the environment resulted from airborne mercury emissions due to smelting of Gold Quarry ore. In 2006, the EPA and the state of Nevada instituted a mercury emissions control program. Before the program (in 2005) 329.4 kg were emitted from the Gold Quarry smelter. This number was reduced to 48.3 kg in 2013. Table B.3 shows a valuation of the environmental damage per troy ounce of gold produced. This value was $35 and $5 for the years 2005 and 2013, respectively.

Table B.3. Valuation of Environmental Damage per Ounce from the Gold Quarry Mine in Nevada

Environmental Damage per Ounce Calculation	
Gold Quarry Mine 2005	
Loss in American lifetime earnings due to 1 kg of mercury emissions (2014$)	$53,000
Gold Quarry's airborne mercury emissions in 2005 (kg)[a]	329.4
Gold Quarry's portion of damages due to IQ loss in 2005 (2014$)	$17,500,000
Gold Quarry's Gold equivalent production in 2005 (Troy oz)[b]	500,000
Environmental damage per ounce (2005)	$35
Gold Quarry Mine 2013	
Gold Quarry's airborne mercury emissions in 2013 (kg)[a]	48.3
Gold Quarry's portion of damages due to IQ loss in 2013 (2014$)	$2,500,000
Gold Quarry's Gold equivalent production in 2013 (Troy oz)[b]	500,000
Environmental damage per ounce (2005)	$5
Source: [a] NMCP Annual Emissions Reporting, [b] Nevada Division of Minerals	

View from a Residence

In addition to the economic cost to residential consumers, the loss of service for municipal water also includes business losses. Aubuchon and Morley (2013) calculated business losses as the forgone business value, measured by industry level Gross Domestic Value, due to service disruption. To do this, they took into consideration the variation in operating capacity among industries in the event of the loss of water. For example, if water services were disrupted health care and social assistance would shut down, whereas transportation and warehousing are resilient and could continue to operate at almost the same capacity. Aubuchon and Morley (2013) used resilience factors for each industry from two studies, one from the Applied Technology Council (1991) and the other from Chang et al. (2002). A resilience factor is the percent of capacity an industry could operate in the absence of water.

Using these resilience factors, Aubuchon and Morley (2013) calculated the economic loss using equation B.6.1:

$$\frac{1}{365 * population}\sum_{i=1}^{n} GDP_i * (1 - r_i) \qquad \text{(B.6.1)}$$

This formula calculates a per capita daily loss for industry *i,* with an industry specific resilience factor (r) and then sums across all industry. Aubuchon and Morley (2013) went on and calculated a state level and a population weighted per capita per day business economic loss (see Aubuchon and Morley [2013 Viewsheds can be affected negatively by proposed mines or positively by the remediation of legacy sites. These changes often affect the value of residential property that has a view of them. A benefit transfer model could be constructed by beginning with the literature reviews in Bourassa et al. (2003) and Walls et al. (2015). General conclusions are that the effect of access to the scenic area must be parsed from the effect of the view of the scenic area, that a mountain view increases the value of a property by 6%, that a forest view increases the value of a property by 5%, and that a view of roads, railways, and industrial parks have various negative impacts on property value.

Wetland, Open Water, Shrubland, Grassland, and Terrestrial Habitat

The following examples of natural land cover valuation require careful attention to the issue of double-counting. Loomis and Richardson (2008) employed a wetland valuation meta-analysis (Brander and Florax, 2007) for use in benefit transfer. Brander and Florax (2007) analyzed European and North American wetland valuation studies that focused on flood prevention, water quality, water quantity, fishing, birdwatching, habitat, and storm drainage. Loomis and Richardson (2008) built a meta-regression model benefit transfer function that was used to value wetlands based on their size, location, and the ecosystem services that they provided.

Similarly, Ingraham and Foster (2008) conducted a meta-analysis of valuation studies on the indirect uses for natural land cover. Examples of such uses are carbon sequestration, disturbance prevention, freshwater regulation and supply, habitat provision, and nutrient removal and waste assimilation. Ingraham and Foster (2008) conducted this evaluation for five separate land classifications: open water, forest, shrubland, grasslands, and wetlands. Ingraham and Foster (2008) appeared to estimate per acre valuations of the land classifications, but they did not explicitly enumerate the values.

Finally, Borisova-Kidder (2006) provided a valuation study for terrestrial open space and habitat. Borisova-Kidder (2006) used 11 studies with 23 observations to conduct a meta-analysis of the literature valuing terrestrial open space and habitat. The primary studies evaluated in this meta-analysis were too disparate and the sample size too small to be incorporated in the current analysis. Nonetheless, many mining sites affect terrestrial habitat and this study should be highlighted as a good place to start for a per acre value of terrestrial habitat.

Municipal and Household Intake Water Quality

Intake water quality refers to the quality of groundwater or surface water brought into a water system. For municipalities, the ecosystem service is the quality of raw water that enters the treatment system. For households, the ecosystem service is the quality of well-water being drawn into the home. In this context, quality is considered in terms of acceptability for use (NRC, 1997, pp. 31). Therefore, the ecosystem service has value if the water is of high enough quality that it can be treated for use. When the quality is too low for the water to be treated, its value falls to zero. Aubuchon and Morley (2013) supported this binomial endpoint with a thorough ecosystem service valuation. In contrast, the valuation literature was mute regarding other possible methods of ecosystem service valuation for drinking water, such as changes in water treatment cost as a function of continuous changes in contaminant concentrations.

Municipal and household intake water quality is valued to account for the impact of mine pollution events that temporarily make intake water quality so poor that it cannot be treated. Such events have occurred at mine sites. For example, in 2014 a chemical used to wash coal was accidentally spilled into the Elk River in West Virginia. Water treatment plants were overwhelmed and 300,000 residents in nine counties were left without potable water. Similarly, an impoundment failure at the Mount Polley tailings storage facility in British Columbia released 25 million m^3 of water and tailings into Hazeltine Creek, Quesnel Lake, and Quesnel River. A drinking water ban was imposed on approximately 150 households for 9 days.

Valuing household/municipal water quality first requires determining whether the water is suitable for household/municipal use and then values the benefits (costs) of having (losing) that ecosystem service. Contamination of household/municipal water from mining often involves an increased risk to high concentrations of toxic elements, for example, West Virginia's coal separation chemical spill of January 2014 (Plumber, 2014). Because contamination makes water unusable, as opposed to merely raising treatment costs, the value of lost service is the best approach. Estimating the foregone value due to water service disruption involves three steps: 1) estimate benefit loss for residential consumers, 2) estimate benefit loss from the affected business and commercial consumers, and 3) add these values together for a total economic loss. This method was used by Aubuchon and Morley (2013), which this project will follow for water service valuation. The rest of this section describes the details of the method used by Aubuchon and Morley (2013).

This valuation technique requires an initial estimation of household demand for water at various prices. Once a demand curve has been estimated, the area under the curve provides the willingness to pay for household water service for a given quantity of water.

In the past, FEMA used a meta-analysis of studies that estimated the price elasticity of demand for water (Dalhuisen et al., 2003) to evaluate the benefits of creating more secure water supplies for municipalities and households. Aubuchon and Morley (2013) built on FEMA's method to create a benefit transfer tool that estimates the cost of losing water service for residential customers and businesses. Their results for U.S. residential customers based on per capita per day (PCPD) consumption are presented in Table B.6.5.

Table B.6.5. Impact to Residential Consumers, PCPD (2013$[a])

	Elasticity of Demand		
Per Capita Per Day Consumption (gal)	-0.41	-0.35	-0.26
172	$57	$147	$2,248
98	$24	$40	$266
Population Weighted	$26	$47	$402
Current Value	Recommended Value = $158		

[a] Includes the FEMA cost for Basic Water Requirements (6.6 gal @ $1.85/gal); Source: Aubuchon and Morley (2013)

The values in the left column of B.6.5 represent different assumptions about the amount of water that is consumed PCPD. The first row, 172 gallons PCPD, is an estimate used by FEMA. The second row, 98 gallons PCPD, comes from an estimate from the USGS, and the third row is a state level population weighted PCPD. The fourth row is a recommended value. Three estimates of the cost of losing service PCPD are given for each consumption level based on different assumptions of demand price elasticity. These elasticity values are represented in the top row. Higher per capita consumption rates result in higher losses. Additionally, when demand is more elastic (flatter demand curve), the benefit loss that people suffer from losing service is reduced. However, for the purposes of this study the author's recommended value of $158 (2013$) will be used. This is the average value of the three population weighted estimates.

Their results for U.S. total, state level, and population weighted per capita per day business economic loss are shown in Table B.6.6, converted to 2013$, for both sets of resilience factors. Using this process, the authors' recommended value is $57 per person per day for loss of water for business uses. This is the average of the two estimates in the population-weighted column.[8]

Table B.6.6. Impact to Business Economic Activity, PCPD (2013$)

	U.S. Total	State Mean	State Mean, Population Weighted
ATC-25(1991) Resilience Factors	$43	$42	$43
Chang et al. (2002) Resilience Factors	$71	$70	$71
Current Value	Recommended Value = $57		

Source: Aubuchon and Morley (2013)

Combining the PCPD business loss and the residential PCPD economic loss, the authors recommended a total economic impact of $215 per person per day for loss of municipal water.

[8] The population-weighted estimates were considered by Aubuchon and Morley (2013) to be the most accurate approach. See their paper for explicit formulas.

This value is simply the sum of the two prior recommended values. Table B.6.7 summarizes the possible combinations of total economic impact for business and residential based on various assumptions of water consumption, resilience factors, and demand elasticity. Additional summary statistics are provided at the end of Table B.6.7.

Table B.6.7. Total Economic Impact, PCPD (2013$[a])

	PCPD Consumption (gal)	Elasticity of Demand		
		-0.41	-0.35	-0.26
U.S. Total, ATC-25	172	$100	$191	$2,258
	98	$67	$84	$310
	USGS, Population Weighted	$69	$90	$444
U.S. Total, Chang et al. (2002)	172	$128	$219	$2,287
	98	$95	$112	$338
	USGS, Population Weighted	$97	$118	$473
State Mean, ATC-25	172	$98	$190	$2,257
	98	$66	$82	$309
	USGS, Population Weighted	$68	$88	$442
State Mean, Chang et al. (2002)	172	$127	$217	$2,257
	98	$94	$111	$337
	USGS, Population Weighted	$96	$117	$472
State Mean, Population Weighted, ATC-25	172	$99	$191	$2,258
	98	$67	$83	$310
	USGS, Population Weighted	$69	$89	$443
State Mean, Population Weighted, Chang et al. (2002)	172	$128	$218	$2,286
	98	$95	$112	$338
	USGS, Population Weighted	$97	$118	$473
Current Value	Recommended Value = $215			
Mean		$93	$135	$1,016
Median		$96	$114	$458
Standard Deviation		$21	$53	$912
Minimum		$66	$82	$309
Maximum		$128	$219	$2,287

[a] U.S. and State Level (GDP and Consumption) Totals
Source: Aubuchon and Morley (2013)

Additional components of household/municipal water use value are supply reliability and probability of contamination. Supply reliability refers to the value tied to the consistency of being able to turn on a sink and have running water. Examples of the valuation of supply reliability are Howe et al. (1994), Griffin and Mjelde (2000), Koss and Khawaja (2001), and Thorvaldson et al. (2010). Probability of contamination refers to the value of reducing the chance of spoiling the water beyond a potable quality. Work on the value of changes in the probability of contamination is best encapsulated by Poe et al. (2001). Although these two issues are included in the value of a clean water ecosystem service, they are a second order consideration when compared to complete water disruption and the current literature does not support their use in benefit transfer (Poe et al. 2001).

Drinking Water and Groundwater: An Incremental Approach

Drinking water can be affected by excessive runoff at legacy sites or spills from operating mines. Görlach and Interwies (2003) summarized the drinking water literature up to 2003. Much of this literature comprised averting behavior studies that evaluate the costs associated with poor drinking water quality (Abdalla, 1990, Abdalla et al., 1992, Collins and Steinback, 1993, Harrington et al., 1989, Laughland et al., 1993). Although these studies are for pollutants unrelated to mining, they applied to the broad issue of contamination. More recent studies relating to drinking water have been conducted in Brazil, Pakistan, and Nicaragua (Casey et al., 2006, Khan et al., 2010, Vásquez et al., 2012). Construction of a benefit transfer model to value drinking water quality would separate the studies into a group that focuses on safe/unsafe drinking water and a group that focuses on percentage changes in quality.

Metal contamination of household and municipal water by mine sites can often be treated. In this case, the additional costs of treatment should be taken into account. However, these costs do not equate to the value of intake water quality because costs depend on more than the ecosystem service of water quality, for example, water treatment plant specifications. This analysis is unable to connect increased metal contamination with increased treatment cost due to several factors. First, water chemistry and metal contamination have complex interactions, so straightforward economic analyses of treatments costs for various contaminants could not be found. Second, operating mine sites must adhere to regulations concerning the quality of the water that they discharge. This limitation prevents situations where a municipality or household routinely incurs additional treatment costs as a result of mining operations. Conversely, households and municipalities are unlikely to locate water intakes in streams polluted by abandoned mine runoff. Therefore, cases where treatment costs are reduced as a result of an abandoned mine cleanup have proven elusive. This dearth of information on treatment costs encourages the approach employed above.

Groundwater can also be affected by legacy sites and operating mines. Görlach and Interwies (2003) summarized the valuation literature regarding ground drinking water. This body of literature is composed of contingent valuation studies, avoided treatment cost studies, and replacement cost studies. Boyle et al. (1994) and Poe et al. (2001) conducted meta-analyses of the contingent valuations regarding groundwater. Both Boyle et al. (1994) and Poe et al. (2001) concluded that the groundwater contingent valuation literature produces defensible values.

However, both recommend against using the meta-analysis for benefit transfer. A groundwater valuation benefit transfer model, which is desperately needed in the unconventional oil and gas development debate, could be constructed by gathering groundwater contingent valuation studies conducted after Poe et al. (2001) and then updating the analysis using updated techniques for benefit transfer error reduction.

Household and Municipal Water: Supply Reliability

Additional components of household and municipal water use value are supply reliability and probability of contamination. Supply reliability refers to the value of having a consistent supply of water. Starting places for the valuation of supply reliability are Howe et al. (1994), Griffin and Mjelde (2000), Koss and Khawaja (2001), and Thorvaldson et al. (2010). Probability of contamination refers to the possibility of having a water supply contaminated to the point that the water is unusable. Work on the value of changes in the probability of contamination is best captured in Poe et al. (2001). Although these two issues are included in the value of a clean water ecosystem service, they are a second-order consideration when compared to complete water disruption and the current literature does not support their use in benefit transfer (Poe et al., 2001).

Non-Use Value of Acquatic Habitat

Aquatic habitat refers to the form of natural land cover that serves as habitat for aquatic organisms. Johnston et al. (2005), as well as Loomis and Richardson (2008), supported the endpoint of aquatic habitat at the level of a watershed or lake. Incorporation of the value of aquatic habitat in an ecosystem service framework must be dealt with carefully. The value that anglers hold for aquatic habitat should not be added to the value they hold for fish caught. That combination would be double counting of use-value. However, non-users hold value for fish and their aquatic habitat as well. This value is significant and should be incorporated in regard to fishery improvements (or degradations). To capture non-use value associated with aquatic habitat, a study sponsored by the US Forest Service (USFS) by Loomis and Richardson (2008) is employed. Loomis and Richardson (2008) relied on a meta-analysis of aquatic habitat valuations by Johnston et al. (2005) to create a benefit transfer tool for the non-use valuation of aquatic habitat.

Johnston et al. (2005) surveyed the aquatic habitat valuation literature searching for studies in the United States that: 1) contained both use and non-use value, 2) valued changes in water quality affecting aquatic habitat, 3) used academically accepted methodologies, and 4) provided sufficient information on the resource, context, and study attributes. Of 300 relevant studies, Johnston et al. (2005) selected 34 as meta-data, which provide a total of 81 observations. Johnston et al. (2005) regressed these 81 observations to determine the influence of relevant variables and illuminate the magnitudes of use and non-use value of aquatic habitat.

Loomis and Richardson (2008) focused on the non-use portion of Johnston et al. (2005) and constructed a user-friendly benefit transfer model for the USFS. An example for this benefit transfer model is provided in Table B.6.8 for the Upper Arkansas River in Central Colorado. These values were calculated using the median household income for Colorado in 2006, and final values were updated to 2013 dollars. The baseline and increase in water quality figures in Table 7.9 are derived from the Vaughan (1981) water quality ladder, more popularly known as the Resources for the Future (RFF) water quality ladder.

Figure B.6.3. Illustrative Water Quality Ladder. Modified from Vaughan (1981).

Table B.6.8. Non-Use Values of Aquatic Habitat for Upper Arkansas River, per Household per Year, (From Loomis and Richardson, 2008)

Baseline Water Quality	Increase in Water Quality	Less than 50% Fish Population Change	More than 50% Fish Population Change
4	2	\$12.48	\$27.68[a]

[a] Recommended value because the Upper Arkansas River went from “unfishable” to “fishable”.
All values are in 2013\$
Source: Loomis and Richardson (2008)

When the Loomis and Richardson (2008) benefit transfer model was calibrated to reflect the changes in Colorado, the result was an aquatic habitat non-use value of $27.68 (2013$) per household. Note that the endpoint problem is an issue for valuing aquatic habitat nonuse value. Economists conduct the valuation at the scale of a single river, multiple rivers, a single lake, or multiple lakes. Environmental scientists do not conduct aquatic habitat improvements at the scale of an entire river or lake. Instead, it is more likely to be by the river mile, wetland acre, or acre of surface water. Additionally, ecologists likely do not agree with the simplicity of the water quality ladder used as the basis for the example. For the example above, the endpoint problem is partially alleviated because remediation was conducted at the scale of a river and because there is no double counting of non-use value.

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Appendix C: A Valuation of the Upper Arkansas River Fishery Recovery from Leadville to Cañon City – Providing Context for the Selection of Ecosystem Service Endpoints

A century's worth of legacy mine sites around Leadville created mine site pollution sources that contaminated the Arkansas River headwaters with metals as far downstream as Cañon City. Before remediation, mine pollution reduced the fish population between Leadville and Lake Creek to a level that was unfishable (Clements et al., 2010, Policky, 2012, Policky, 2013). Farther downstream, between Lake Creek and Cañon City, fish did not live beyond their third year due to chronic toxicity of metals (Clements et al., 2010, Policky, 2012, Policky, 2013). The California Gulch Superfund site remediation has been credited with a transformation in this fishery that culminated in 2014 when it achieved the highest fishery designation possible - Gold Medal Water.

This appendix applies the benefit transfer model components of WTP for fish caught, WTP for angling days, and non-use value for aquatic habitat to the Upper Arkansas River Fishery in Colorado. This exercise does not represent linked natural and social science because the data are solely derived from economic studies, creel surveys, and demographic information. Instead, the purpose is to circumvent data availability issues, which prevented geo-environmental modeling of aquatic habitat and trout populations on the full scale of impacts, to quantify a significant portion of the benefits produced by remediation. The relative scale of the fishery that was impacted by remediation of the California Gulch Superfund site and the area where sufficient data were available to conduct geo-environmental modeling of contaminant source processes is shown in Figure C.1.

Upper Arkansas River Watershed

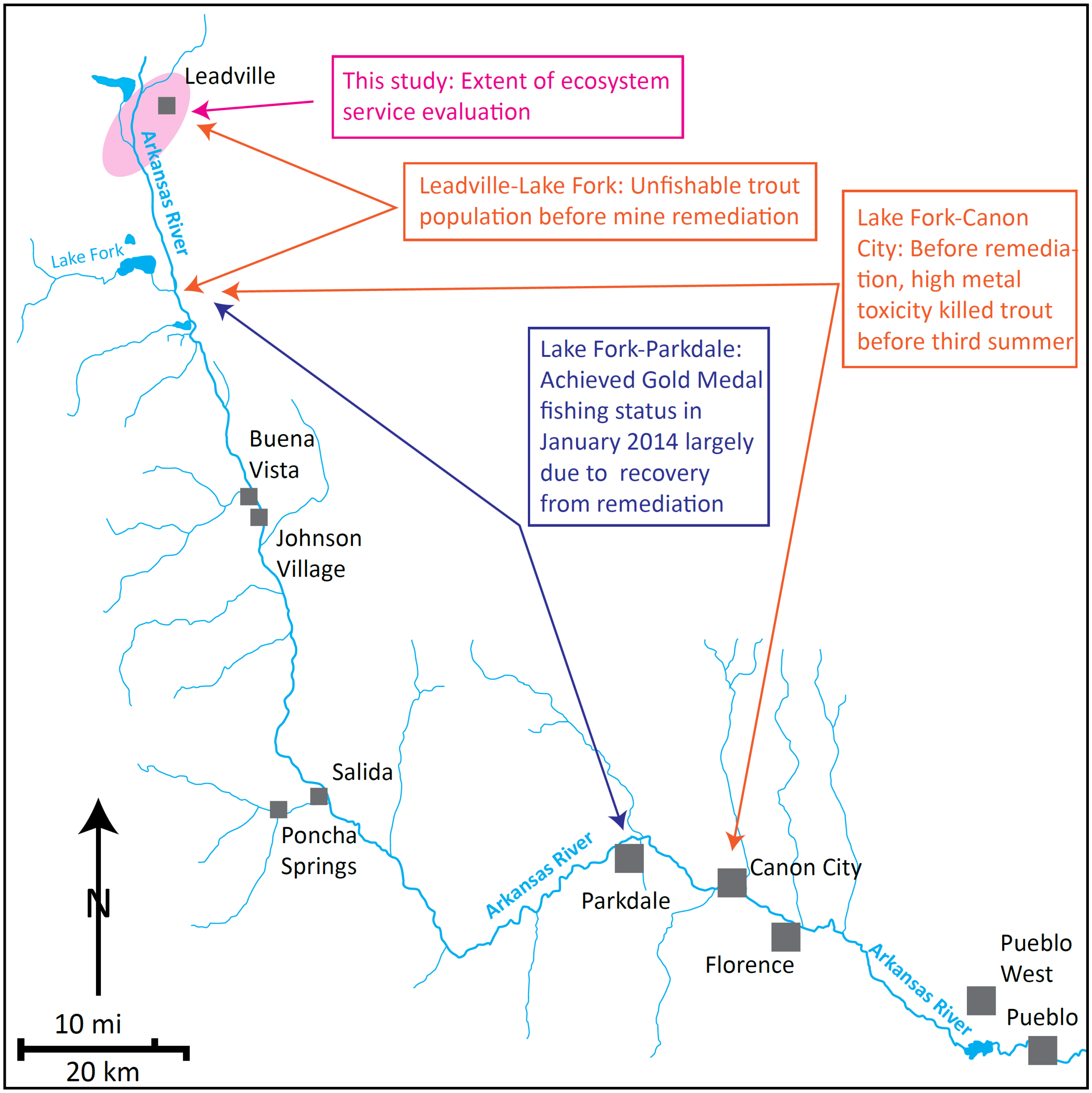

Figure C.1. Upper Arkansas River: Trout and Mine Pollution Overview

Purpose and Scope

The purpose of this analysis is to improve the understanding of the value of ecosystem improvements due to the Superfund remediation. The scope covers improvements in fish population on the Upper Arkansas River from the East Fork of the Arkansas River down to Parkdale (Figure C.2). Because of the lack of comprehensive information and the resultant inability to model the links among remedial action, water quality improvement, and fish population in the framework of climate variability, an important assumption required by this analysis is that the increase in trout catch is entirely due to the remediation. Although other factors surely contribute to the increase in trout catch, such as increasing numbers of anglers in Colorado, this assumption is supported by previous analyses of the impact of trace metals on the Upper Arkansas River (Clements et al., 2010, Policky, 2012, Policky, 2013).

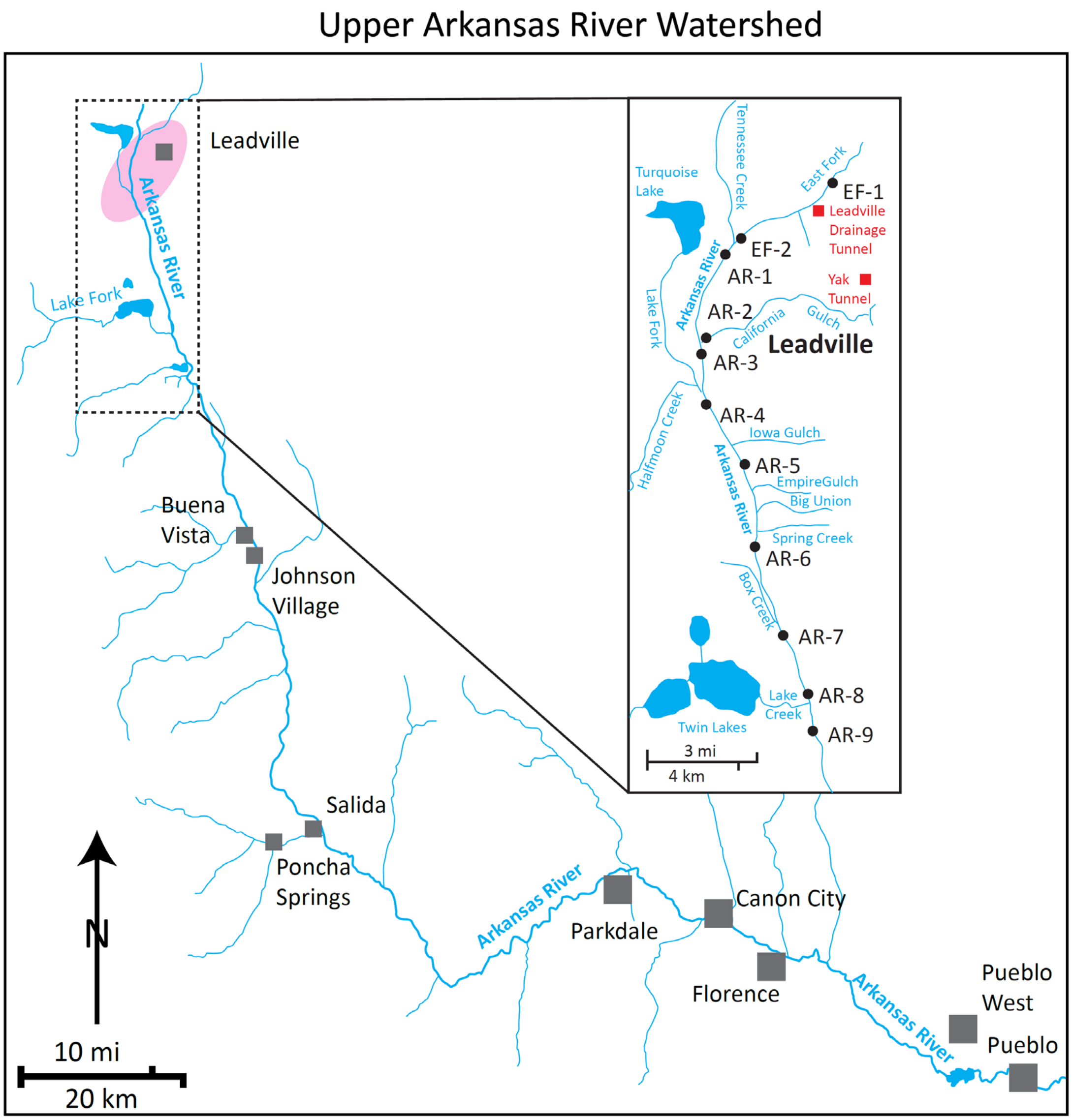

Figure C.2. Upper Arkansas River: Colorado Parks and Wildlife River Segments and EPA Sampling Sites

Data and Method

The main data relied on by this analysis were collected from Colorado Parks and Wildlife (CPW) creel censuses conducted along the river from Crystal Lakes to Parkdale for the years 1995, 2008, and 2012. CPW collects information on creel census area angling hours, angling days, hourly catch rate, and proportion of anglers who are from out-of-state. The specific sites surveyed by the creel censuses are small and rarely represent the larger river segments that they are located within. Table C.1 provides detailed information on the size of creel census areas and their respective river reaches.

Table C.1. Colorado Parks and Wildlife River Reaches and Corresponding Creel Census Areas

River Reach	River Reach Name	Miles in River Reach	Creel Census Area	Creel Census Miles
EF	EF-1 - Confluence	2.2		NA
-1	Confluence – California Gulch	3.8		NA
0	California Gulch – Chrystal Lakes	2.8		NA
1	Chrystal Lakes – Kobe Bridge	5.1	Highway 24 – Kobe	3.2
2	Kobe Bridge – Lake Creek	4.2	Kobe – Two Bit Gulch	2.2
3	Lake Creek – Otero Bridge	10.6	Ball Town – Granite	2.6
4	Otero Bridge – Highway 285 Bridge	9	Otero Bridge – Railroad Bridge	3.1
5	Highway 285 Bridge – Ruby Mountain	6	Big Bend – F Street	5.7
6	Ruby Mountain – Stone Bridge	11.2	Big Bend – F Street	5.7
7	Stone Bridge – Stockyard Bridge	10.9	Big Bend – F Street	5.7
8	Stockyard Bridge – Howard Bridge	11.4	Stockyard Bridge – Badger Creek	5.9
9	Howard Bridge – Lazy J	8.3	Big Cottonwood Creek – Lone Pine	3
10	Lazy J – Texas Creek	12	Big Cottonwood Creek – Lone Pine	3
11	Texas Creek – Parkdale	13.3	Big Cottonwood Creek – Lone Pine	3

Because creel census areas are not representative of river reach, expert knowledge is required to extrapolate creel census area estimates to broader river reach estimates. Policky (2013) provided an expert extrapolation of creel census angling days to river reach angling days for each river segment in the year 2012. How these factors were calculated is illustrated in Equation C.1 and Table C.2.

$$ExtrapolationFactor = 1 + \frac{2012ReachDayEstimate - 2012CreelDayEstimate}{2012CreelDayEstimate}$$

The extrapolation factors from Table C.2 are important because they allow extrapolation of creel census area estimates to a larger and more useful scale. The extrapolation factors are multiplied by creel census estimates to estimate river reach angling hours (for 1995, 2008, and 2012) and river reach angling days (for 1995 and 2008, Table C.3).

Table C.2: CPW Creel Census and River Reach Angling Day Estimates Used to Calculate Extrapolation Factor

River Reach Number	Creel Census Area Angling Days	CPW Estimate of Angling Days per River Reach	Angling Day Extrapolation Factor
	2012	2012	
1	3,600	7,200	1.99
2	1,600	2,400	1.53
3	2,700	10,800	4.02
4	1,600	4,300	2.78
5	5,600	3,500	0.63
6	5,600	12,700	2.29
7	5,600	10,400	1.88
8	8,100	11,600	1.43
9	5,100	9,300	1.83
10	5,100	14,000	2.75
11	5,100	14,400	2.82
Total	49,700	100,600	

Table C.3: Resulting Angling Hour and Day Estimates per River Reach

River Reach Number	Estimated River Reach Angling Hours			Estimated River Reach Angling Days		
	1995	2008	2012	1995	2008	2012
1		4,750	16,600		2,000	7,200
2	2,400	3,700	5,500	1,100	1,500	2,400
3	5,000	7,200	19,900	5,000	3,500	2,400
4	2,700	4,100	5,600	2,000	3,200	4,300
5	900	5,500	7,800	900	2,300	3,500
6	3,200	20,300	28,500	3,200	8,400	12,700
7	2,700	16,600	23,400	2,600	6,900	10,400
8	9,500	21,300	31,300	7,200	6,700	11,600
9	10,400	12,400	20,000	8,500	5,560	9,300
10	15,600	18,700	29,900	12,800	8,300	14,000
11	16,000	19,200	30,800	13,100	8,600	14,300
Total	68,400	133,800	219,200	56,600	56,900	100,600

Estimates of the number of hours fished for each river segment in 1995, 2008, and 2012 are paired with average hourly catch rates from Policky (2012) to estimate the total catch for each river segment (Table C.4). Next, the estimated catch is multiplied by the WTP to catch an additional fish - $2.94 from Table C.5.

To estimate the number of fish caught in the river segments above the highest CPW creel sites (Segments 0, -1, and EF), this analysis applies an estimate of the percentage of the fish population caught by anglers in Segment 1 (34%) to the fish population estimates from sample sites AR-3 (Segment 0), AR-1 (Segment -1), and EF-5 (Segment EF). The results of this application can be seen in Table C.6.

The use of an ecosystem service approach to value recreational fishing is rather novel. Typically, it is the `fishing day' that is valued to estimate benefits of recreational fisheries. Therefore, to put the fish-centric valuation approach in full context, it is compared to a similar valuation using the value of an angling day estimated by Loomis and Richardson (2008). This `angler-centric' approach combines estimates of the number of angling days with the value of an angling day ($67.91) to achieve a valuation of the change in angling over the study period. Table C.6 in the following section provides the results of this analysis.

Non-Use Value of Aquatic Habitat Improvement

Finally, the value of fishery improvements to non-fisherman is estimated through non-use value that households place on healthy aquatic habitat. People who do not fish still accrue benefits from the remediation of the Upper Arkansas River fishery. This may be as simple as the pride a resident feels from knowing that the Upper Arkansas River is now a Gold Medal trout fishery, rather than a periodic conduit for acid-mine drainage. As mentioned previously, this sentiment is referred to by economists as non-use value. It often has great weight in non-market valuation and should be captured when possible.

Loomis and Richardson (2008) provided a straight-forward benefit transfer of non-use value associated with aquatic habitat. To achieve this benefit transfer, the annual non-use value of $27.68 per year is combined with the number of households living in the counties that straddle this section of the Arkansas River. As reflected in Table C.7, the water quality in the Upper Arkansas River moved up two rungs on the Resources for the Future (RFF) water quality ladder from the fourth rung. The resulting improvement in fish population was greater than 50% because fish could not live beyond their third year due to chronic metal toxicity.

Multiplying the recommended value of $27.68 from Table 6.9 by the number of households in the three counties that encompass this watershed provides an estimate of the non-use value created by the improvement in the Upper Arkansas River's aquatic habitat. County household data were provided for 2010 by the U.S. Census Bureau. Table 9.8 details the population and annual non-use aquatic habitat values.

Results

The fish-centric valuation approach resulted in a trebling of the annual value of fishing from 1995 to 2012 - $252,000 to $773,000 respectively. Remediation of the California Gulch site was ongoing between 1995 and 2008, but was largely completed by 2009. Policky (2012) and Policky (2013) indicated that the fishery was slow to improve until 2008. However, between 2008 and 2014, the fishery improved rapidly. The results of the fish-centric valuation reflect this sentiment. The original annual value ($252,000) took 13 years to double, but doubled again just 4 years afterward.

Table C.4. Fish-Centric Valuation Results (2013$)

River Reach Number	Average Hourly Catch Rate			Estimated Number of Fish Caught per Year			Estimated Value of Fish Caught per Year		
	1995	2008	2012	1995	2008	2012	1995	2008	2012
EF	NA	NA	NA	1,100	1,100		$2,700	$2,700	
-1	NA	NA	NA	2,306	2,300		$5,800	$5,700	
0	NA	NA	NA	400	2,000		$1,000	$5,000	
1	1	0.91	1.2	0	4,300	19,900	$0	$12,700	$58,400
2	1	0.91	1.2	2,400	3,300	6,600	$7,200	$9,800	$19,300
3	1	0.91	1.2	5,000	6,600	23,900	$14,800	$19,400	$70,200
4	1	0.91	1.2	2,700	3,800	6,800	$7,800	$11,100	$19,900
5	1.3	1.4	1.2	1,200	7,700	9,300	$3,400	$22,800	$27,400
6	1.3	1.4	1.2	4,200	28,400	34,200	$12,400	$83,400	$100,600
7	1.3	1.4	1.2	3,500	23,300	28,100	$10,200	$68,500	$82,500
8	1.3	1.4	1.2	12,300	29,900	37,600	$36,200	$87,800	$110,400
9	1.3	1.4	1.2	13,500	17,400	23,900	$39,700	$51,200	$70,400
10	1.3	1.4	1.2	20,300	26,100	35,900	$59,600	$76,800	$105,600
11	1.3	1.4	1.2	20,800	26,800	36,900	$61,200	$78,900	$108,500
Total				85,900	177,700	263,000	$252,500	$522,300	$773,300

The angler-centric valuation approach resulted in a less pronounced change in the annual value of fishing. The annual value remained essentially unchanged between 1995 and 2008 - $3.83M to $3.871M. The bump in annual value between 2008 and 2012 is also reflected in this analysis - $3.871M to $6.825M. One important result of this comparison is that the fish-centric approach results in annual values that are approximately one-tenth the annual value of the angler-centric approach. This is an interesting result given the discussion above about the portion of total angling value that is represented by the number of fish caught. Note that the angler-centric valuation does not include Segments 0, -1, or EF because no estimates are available for the number of angling days in these stretches and because no proxy could be found for these estimates.

Table C.5. Angler-Centric Valuation Results (2013$)

River Reach Number	Estimated River Reach Angling Days per Year			Estimated Value of Angling Days per Year		
	1995	2008	2012	1995	2008	2012
1	0	2,000	7,200	$0	$135,800	$489,000
2	1,100	1,500	2,400	$74,700	$101,900	$163,000
3	5,000	3,500	10,800	$339,600	$237,700	$733,400
4	2,000	3,200	4,300	$135,800	$217,300	$292,000

River Reach Number	Estimated River Reach Angling Days per Year			Estimated Value of Angling Days per Year		
5	900	2,300	3,500	$61,100	$156,200	$237,700
6	3,200	8,400	12,700	$217,300	$570,400	$862,500
7	2,600	6,900	10,400	$176,600	$468,600	$706,300
8	7,200	6,700	11,600	$489,000	$455,000	$787,800
9	8,500	5,600	9,300	$577,200	$380,300	$631,600
10	12,800	8,300	14,000	$869,200	$563,700	$950,700
11	13,100	8,600	14,300	$889,600	$584,000	$971,100
Total	56,400	57,000	100,500	$3,830,100	$3,870,900	$6,825,100

Table C.6. Comparison of Valuation Results (2013$)

	1995	2008	2012	Increase in Annual Value from 1995 to 2012
Fish-centric	$252,000	$522,000	$773,000	$521,000
Angler-centric	$3,830,000	$3,871,000	$6,825,000	$2,995,000

Between 1995 and 2012, non-use value of aquatic habitat increased by $834,000 per year. This dollar figure can be thought of as the value that residents place on the transformation of the Arkansas River into a Gold Medal Fishery.

Table C.7. Valuation of Aquatic Habitat for Upper Arkansas River

County	Number of Households	Annual Non-Use Aquatic Habitat Value
Lake	3,100	$92,000
Chaffee	7,800	$235,000
Fremont	16,900	$507,000
Total	27,800	$834,000

The geo-environmental models of trout population improvement from AR3 and AR5 suggest that remediation improved the fish populations in the first and second river stretches. Policky (2012) and Policky (2013) suggest that the remediation also improved the other river stretches as well. Extending this assumption to the fish-centric valuation, the benefit of the remediation pertains to all river segments, increasing from $252,000 per year in 1995 to $773,000 in 2012—resulting in an additional benefit of $521,000 per year (see Table C.6). Extending this assumption to the angler-centric valuation, the benefit of the remediation increases from $3,830,000 per year in 1995 to $6,825,000 in 2012—resulting in an additional benefit of $2,995,000 per year (see Table C.6).

From a net present value standpoint, the fish-centric value of fishery improvement from 1995 to 2012 (using a 3% discount rate) is $13,401,000 for a 50-year time frame and $16,458,000 for a 100-year time frame. The angler-centric net present value of fishery improvement from 1995 to 2012 (using the same discount rate) is $77,100,000 for a 50-year time frame and $94,600,000 for a 100-year time frame. Finally, the non-use aquatic habitat net present value is $21,500,000 for a 50-year time frame and $26,400,000 for a 100-year time frame.

Table C.8. Net Present Value of Fishery Improvements at 3% over 50 and 100 Years (2013$)

	Annual Value of Improvement	NPV 50 Years 3% Discount	NPV 100 Years 3% Discount
Fish-Centric	$520,000	$13,400,000	$16,500,000
Angler-Centric	$2,995,000	$77,100,000	$94,600,000
Aquatic Habitat	$834,000	$21,500,000	$26,400,000

These benefits can be compared to the expenditures made during the remediation. Table C.9 details the expenditures, purpose, and funding source for expenditures found in the public record for the California Gulch Superfund cleanup.

Discussion, Implications and Conclusion

The results of this analysis indicate that the net present value of benefits (at 3% over 100 years) from the improvement of the fishery are, at most, $94.6 million plus $26.4 million equals $121 million (2013$). Although additional benefits accrued to society from the remediation, such as reduction of blood lead level in children, increased quality of irrigation water, and greater municipal water supply reliability, the causal link is more dubious and valuation of the benefits is not possible due to poor availability of usable data. To compare the benefits that have been estimated for the remediation of this fishery, Table C.9 below details the expenditures that could be found for the California Gulch Superfund remediation.

Table C.9. Expenditures Located for the California Gulch Superfund Site

Year	Purpose	Funding Source	Expenditure (2013$)
1988	Yak Tunnel Plug/Treatment Plant	ASARCO/Resurrection	$29,490,000
1988	Annual Yak Tunnel O&M costs (1988-1992)	ASARCO/Resurrection	$4,530,000
1999	1-year field demonstration, biosolids/lime in soil	ASARCO/Resurrection	$6,850,000
2001	OU1 23 years of YWTP Costs	ASARCO/Resurrection	$21,180,000
2012	OU1 Costs associated with Black Cloud Mine	Resurrection Mining	$5,070,000
1994	OU2 Malta Gulch removal actions (1995-1996)	Hecla Mining	$1,070,000
2001	OU2 15 years of monitoring	Hecla Mining	$790,000
	OU3 Denver and Rio Grande slag piles	Union Pacific	???

Year	Purpose	Funding Source	Expenditure (2013$)
1998	OU4 NPV of removal costs	Resurrection Mining	$5,830,000
2001	OU4 Erosion control/inspection	Resurrection Mining	$580,000
2001	OU5 AV/CZL and EGWA remediation costs	ASARCO	$4,280,000
2001	OU5 5 years of monitoring costs for AV/CZL site	ASARCO	$120,000
2001	OU5 Institutional control costs for EGWA	ASARCO	$40,000
2001	OU5 5 years of monitoring costs for EGWA site	ASARCO	$20,000
	OU6 Removal action costs (1995-2001, '05, '08, '11	EPA	???
2010	OU6 Stray Horse Gulch Waste Rock Repository	EPA	$19,230,000
2010	OU6 100 years NPV of costs	EPA	$490,000
	OU7 Remedial costs	ASARCO/Resurrection	???
2001	OU7 14 years of monitoring costs	ASARCO/Resurrection	$1,570,000
	OU8 1995/1998 Oregon Gulch tailing removal	Resurrection Mining	???
2001	OU8 Fluvial tailings removal	Resurrection Mining	$1,300,000
2001	OU8 Stream sediment remediation costs	Resurrection Mining	$940,000
2001	OU8 14 years of monitoring costs	Resurrection Mining	$100,000
2001	OU9 Lead program costs over 12 years	ASARCO/Resurrection	$6,370,000
2012	OU9 Annual costs for Phase 2 of lead program	ASARCO/Resurrection	$760,000
	OU10 Cost of remedial actions	Resurrection Mining	???
1997	OU10 30 years NPV of costs	Resurrection Mining	$3,690,000
2001	OU10 16 years of additional monitoring costs	Resurrection Mining	$250,000
2005	OU11 Combined capital and operational costs	ASARCO/Resurrection	$6,220,000
2012	OU11 Combined capital and operational costs	ASARCO/Resurrection	$15,870,000
2009	OU12 Institutional control monitoring costs	EPA	$1,370,000
2012	OU12 3 years of monitoring costs	EPA	$640,000
2012	OU12 3 years of enforcement costs	EPA	$150,000
Total			$138,810,000

These expenditures do not represent the full tally for this remediation effort because several large expenditures could not be estimated. Nevertheless, the largest estimate of benefits that were estimated (NPV over 100 years at 3%) do not quite cover this cost estimate. Future research could address this question by locating the missing expenditures and estimating additional benefits related to the remediation.

The fish-centric recreational angling valuation approach [using USEPA (2006)] is valuable because of its explicit focus on fish caught - the closest possible endpoint to the fish population. However, the marginal value of $2.94 per fish does not include the economic benefits generated by anglers as a result of their fishing. On the other hand, the angler-centric approach transfers values from Loomis and Richardson (2008), which used the economic benefits generated by the angler to estimate the willingness to pay a value of $67.91 per angling day. While Loomis and Richardson (2008)'s values may paint a clearer picture of the economic benefits of fishing, they have more to do with the joy of a family fishing trip than with an increase in fish population. However, the desirability of a specific reach of a stream as a destination for a family fishing trip relies on its reputation as a source of abundant and large game fish.

This discussion encompasses the end-point problem that natural and social scientists will continue to work out in relation to ecosystem service valuation (Boyd, 2007). Future research on this issue from the fisheries management side ought to isolate the impact of increasing fish population on the number of fish caught. Data would be required for fish population, fish caught, fishing capital, fishing skill, angler hours, and angler days, among other factors. Future research from ecological economists should isolate the portion of angling-day value that comes from catching each marginal fish.

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