VALUATION OF ECOSYSTEM SERVICES IMPACTED BY MINE SITE POLLUTION

by

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A thesis submitted to the Faculty and the Board of Trustees of the Colorado School of Mines in partial fulfillment of the requirements for the degree of Doctor of Philosophy (Mineral and Energy Economics).

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ABSTRACT

While pollution from mine sites is well regulated in developed countries, regulatory agencies are currently trying to value its environmental impact to gain a better understanding of the trade-offs between mineral development and environmental quality. Ecosystem service valuation helps to incorporate the value of ecosystem services into decisions regarding abandoned mine lands, legacy sites, operating mines, proposed mine development, and mine closure. Benefit transfer provides an attractive ecosystem service valuation method that applies valuation results, from the environmental valuation literature, to sites in need of valuation estimates. Therefore, this dissertation constructs a benefit transfer model to value mine site pollution and applies it to mine sites around the world. The results illuminate the benefits of a Superfund remediation and the impacts of artisanal and small-scale mining.

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ACKNOWLEDGMENTS

I would like to thank Dr. Rod Eggert for his steady hand in guiding me to completion of the REServ project and my dissertation. Dr. Eggert served as a consistent mentor and was generous with his time - even when he had little of it.

In my first year, Dr. Dan Kaffine listened to my interest in environmental valuation of acid mine drainage and recruited me to work on the REServ project - which hooked me on the PhD. For this, and his help through the project, I am grateful.

I would like to thank Carol Russell for her dedication to overcoming obstacles during our work on the REServ project. Carol fought for the information that the project needed and provided me with a better understanding of how regulatory bodies function. I am similarly grateful to Dr. Bob Seal for having a personal interest in my well-being that extended beyond our working relationship. I thank Bob for partnering with me to finish the REServ project.

I would like to thank Dr. Ed Balistreri for taking me on as an orphaned advisee and Dr. Harrison Fell for serving as a thesis committee member.

I am grateful to Dr. Hugh Miller for his extensive review of my dissertation before the defense. Dr. Miller's experienced perspective and attention to my work improved it dramatically.

I would like to thank Dr. Scott Houser for serving as an informal and impartial mentor while I was in MinEc.

Finally, I would like to thank Dr. Graham Davis for the consistent provision of exemplary advice. I rarely followed it, but always saw its value after the fact.

"Teacher, tender, comrade, wife,

A fellow-farer true through life."

-Robert Louis Stevenson

This dissertation is dedicated to my loving wife and my supportive parents. Let's be honest, we did this together.

This work is also dedicated to my daughter and her future siblings. I did not think it was possible to love my wife any more than I already did.

Then she turned our marriage into a family.

CHAPTER 1

A SUMMARY OF THE ENVIRONMENTAL VALUATION LITERATURE AS IT PERTAINS TO MINE SITE POLLUTION

The main objective of this dissertation is to construct an ecosystem service valuation model for ecosystem services related to mine sites. The results of environmental valuation studies are employed to construct this model. 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 legacy site remediation and the impacts of gold production on neighboring communities and ecosystems.

An additional objective of this dissertation is to improve the application of environmental valuation study results to unstudied sites. Transferring existing study estimates to unstudied sites is still an art - rather than a science - and improvements are made in the transfer of valuation results. The final objective is exploration of the relationship between spot and futures prices of copper. The debate regarding speculators' ability to influence today's spot prices via speculation in futures markets hinges on the link between the two prices. Exploration of this relationship illuminates this debate.

The main contribution of this dissertation is the construction, and application, of an ecosystem service valuation model for environmental impacts related to mine sites. This dissertation synthesizes a wide body of environmental valuation literature and applies it to specific impacts of mine sites. Legacy site remediation benefits are calculated for the California Gulch Superfund site and are compared to available estimates of remediation expenditures to provide a post-mortum cost-benefit analysis. Gold mining impacts (per ounce) are calculated for publicly-traded company projects, small-scale mining operations, and artisanal mining sites to contrast the relative impact of each form of mining.

An additional contribution is the analysis of benefit transfer error measurement and its sensitivity to outlying result estimates for a recreational angler's willingness to pay to catch an additional fish. The final contribution is the determination that spot and futures prices move in lock-step during rare market periods of strong contango. This indicates that speculators can directly affect affect spot prices via participation in futures markets.

1.1 Dissertation Introduction

Modern mining policy in the United States can be traced to the California Gold Rush of 1849, which triggered a period of westward migration. During the gold rush, mining had little legal structure and claim disputes were often settled by force. Over time, local miners developed common procedures for staking claims and claim disputes were settled by miners' courts (Clay & Wright, 2005). From 1848 to 1872, miners occupied federal lands and extracted what they could. There was no defined policy regarding their industry but local procedures became systematized as miners tramped from one district to another (Clay & Wright, 2005). The 1872 General Mining Law was the first comprehensive and lasting piece of legislation regarding mining. As incentive to move west the 1872 Mining Law gave settlers the right to buy federal land cheaply and the right to mine without payment of royalties on production. "All valuable mineral deposits in lands belonging to the United States government are hereby declared to be free and open to exploration or purchase, by citizens of the United States and those who have declared their intentions to become such." The land became private property as long as it was purchased at five dollars an acre and the owner spent one-hundred dollars a year in labor or improvement costs[1].

Few changes were made to mining policy in the United States over the following years. In fact, the 1947 Mineral Materials Act and the 1954 Multiple Mineral Use Act are the next major pieces of legislation that affect mining. The Mineral Materials Act dealt with

[1] Today, the land patent process - which transitions the land from public ownership to private ownership has been suspended. Further, while the Federal Government only receives five dollars an acre, the process required to develop a mineral deposit on public land (proving discovery of mineralization, permitting, etc.) ends up being approximately thirty thousand dollars per acre in rural Nevada (Personal Correspondence with Hugh Miller)

the extraction of sand, gravel, and other common materials that were not covered by the 1872 Mining Law. It also reaffirmed the sale of public land for mineral exploitation without the imposition of a royalty. The Multiple Mineral Use Act merely established that multiple minerals could be mined at one location.

The 1872 Mining Law governs mining in the United States today. To many, this is proof that the mining industry has captured the legislative process intended to govern it. However, as the Baby Boomer generation emerged from college in the late 1960's and 1970's, the environmental movement became a powerful force. Legislators passed novel environmental legislation to address pollution of air, water, and soil.

The 1969 National Environmental Policy Act (NEPA) was the first significant environmental reform to hit the mining industry. As the beginning of a wave of environmental legislation signed into law by Richard Nixon, NEPA required federal agencies to complete an Environmental Impact Statement (EIS) for "major federal actions significantly affecting the human environment." As a result, the Bureau of Land Management and the United States Forest Service (the agencies responsible for overseeing mining operations on federal lands) were forced to introduce an environmental component into the permitting process. NEPA required consideration of alternatives, as well as public comment and review of permitting activities (Council, 1999, pp. 41). The newly minted Environmental Protection Agency (EPA) spent its first years convincing industrial producers that it had the power to regulate their effluents and cleaning up the most egregious examples of pollution. Environmental Assessments (EA) and EIS became common place in permitting and regulatory decisions. The main concern during these early years was preservation of human health. Developed societies feared that they were poisoning themselves (Carson, 1962).

The 1972 Clean Water Act (CWA) and 1974 Safe Drinking Water Act (SWDA) aimed to protect public health by regulating drinking water and its sources. CWA required that all pollution discharges have a water permit. This meant that the mine permitting system had to incorporate a CWA requirement for all operations that could potentially discharge into an aquatic environment. Like NEPA, this act was not targeted at the mining sector, but had a pronounced effect. The SWDA authorized the EPA to set safe drinking water standards for both natural and man made contaminants. These standards become contentious for the mining industry because they were set without consideration for ambient standards of water in the natural environment. The mining industry operated in geologies with naturally high concentrations of heavy metals (i.e. contaminants) in the water. Mining companies insisted that they could not be expected to clean water beyond the ambient quality that occurred naturally.

The next evolution of mining policy was the Federal Land Policy and Management Act of 1976. This act updated the procedural context governing the 1872 Mining Law and required the Bureau of Land Management to ensure that federal land had multiple uses so that federal resources would meet both present and future needs. The multiple use requirement meant that mine operators had to reclaim mined land in a way that would allow for a specific use after reclamation was complete.

The 1977 Surface Mining Control and Reclamation Act (SMCRA) was the first law that directly targeted the mining industry in 30 years. SMCRA was intended to ensure reclamation of surface coal mines. It created the Office of Surface Mining (OSM), which oversaw coal reclamation projects and permitting of coal operations on public lands. SMCRA allowed the public to petition OSM to make specific lands off-limits to surface mining if the operation would have proven ill effects on areas downstream. Finally, SMCRA created a coal production tax to service an Abandoned Mine Land fund for abandoned coal mines.

The idealism of the 1960's and 1970's faded in the 1980's and 1990's as the cost and complexity of environmental regulation became apparent. Congress passed the 1980 Comprehensive Environmental Response, Compensation and Liability Act (Superfund) and the 1986 Superfund Amendments and Reauthorization Act (SARA) in an attempt to fast track cleanup of hazardous sites. The idea was to fund immediate cleanup and recover costs later from Potentially Responsible Parties (PRP's). The Superfund legislation made liability at

Superfund sites joint and several - which meant that every party involved at the site was potentially liable for antecedent liabilities. Sites that posed the greatest threat to human health or the environment were proposed for the National Priorities List (NPL). An index was created to prioritize the sites on the NPL.

Naturally, PRPs fought the notion that they should pay remediation costs for activities that were not previously illegal. Prolonged legal battles meant that a good portion of Superfund dollars were spent in court instead of at contaminated sites (Revesz & Stewart, 1995). In addition to payment of remediation costs, PRPs could also be sued for damages to natural resources - which could be two or three times the cost of cleanup. While Superfund cleaned up many toxic sites around the nation, it proved to be an inefficient system. One perverse incentive of Superfund was that anyone who attempted to remediate a contaminated site inherited all of the site's environmental liability.

The liability provisions of Superfund created three distinct types of contaminated mine sites; legacy sites, private abandoned mine lands, and public abandoned mine lands. Legacy sites are often comprised of historic mining districts that one, or more, mining companies are cleaning up as PRPs. Private abandoned mine lands are comprised of abandoned mining operations that are privately held property. The owner of the land is responsible for effluent which exceeds clean water standards. Public abandoned mine lands are comprised of abandoned mining operations on public lands. The federal government is largely responsible for these lands. Groups have stepped forward to voluntarily remediate public abandoned mine lands. However, Superfund's liability provisions and provisions in the Clean Water Act would make them liable for all environmental liability at the site. Further, if these 'Good Samaritans' try to clean up a site and fail, they are liable for future effluents from the site. The lack of 'Good Samaritan' legislation protecting voluntary cleanup serves as a major barrier to the abandoned mine lands issue.

Today, it is well-established that industries: need permits to discharge any effluent that exceeds environmental standards, must successfully execute an EIS for new developments, are liable for any damages they impose on natural resources, must post a bond to ensure payment of remediation activities, and inherit all environmental liability from contaminated sites that they purchase. Regulatory standards for pollution effluent are set relative to their impact on human health, ecosystems, and habitat - regardless of ambient pollution levels. In short, industries in the United States now operate within a robust and comprehensive environmental regulatory regime. Environmental damage affecting human health is rare and the focus is now on protecting local and global ecosystems.

The majority of developed nations around the world have similarly robust environmental regulatory regimes for mining. Within these jurisdictions, companies are expected to be sophisticated in their operations and capable of abiding by the rules. If they are not, the regulator often will stop their production by revoking the mining permit. However, in developing nations, environmental regulation may be less robust. In these settings, the type of mine operator greatly influences environmental and social outcomes.

For example, publicly-traded mining companies have strong financial incentives to address labor, environmental and community concerns. Publicly-traded mining company employees are well trained and valuable (Garcia *et al.*, 2001). Accidents are taken seriously and safety is a primary concern. For publicly-traded companies, financial incentives to protect environmental quality come in the form of market devaluation of equity resulting from a spill, the maintenance of reputational capital in negotiating future projects, government fines, natural resource damage assessments, and potential loss of operating permit. In addition to financial incentives to protect the environment, publicly-traded companies also have financial incentives to gain community acceptance of the company's project (known as social license to operate). Prevention of delays, reduced security costs, and the maintenance of reputational capital in negotiating future projects are all financially beneficial results of social license to operate (Davis & Franks, 2011). In short, the mandate that publicly-traded companies must maximize shareholder value, encourages them to address social and environmental concerns before they can affect the financial performance of the company. While there are many examples of publicly-traded mining companies struggling to cope with community or environmental problems, these are the exceptions rather than the rule.

In contrast to publicly-traded companies, private mining companies have fewer financial incentives to address environmental and community concerns. First, private companies are not subject to equity devaluation resulting from negative environmental or social publicity. While private companies are largely subject to the same fines, damage assessments, loss of operating permit, and loss of reputational capital - protection from public equity markets changes the incentive structure drastically. Second, some private mining companies have no intention of developing new projects. This insulates them from the need to maintain a favorable reputation with future communities or governments. Third, in the developing world, some private mining companies are owned by political insiders who are not subject to fines, assessments, or other regulatory actions. Private companies held under such conditions have done tremendous damage in developing countries (Broad & Cavanagh, 1993) and ought not be confused with publicly-traded mining companies.

Similar to private companies, state-owned enterprises (SOEs) that engage in mining do not have the financial incentive to address environmental and community concerns that publicly-traded mining companies do. Although many SOE's are as sophisticated as publicly-traded companies, SOEs are insulated from public equity markets, may not need to maintain a favorable reputation to negotiate development of future projects, and may not be held to the same regulatory standards as publicly-traded mining companies.

Small-scale mining (SSM) consists of mining operations with a relatively small capacity (McMahon, 1999; Veiga *et al.*, 2004). Generally, SSM operations are not as sophisticated as operations conducted by publicly-traded companies. The mineral resource is rarely delineated prior to extraction. While machinery is often employed in resource extraction, these operations primarily depend on manpower. Mercury and cyanide are often applied to gold ore during mineral processing. SSM operations theoretically have financial incentives to address environmental concerns because their operating permits are dependent upon meeting environmental requirements. However, SSM operations are so small and diffuse that they are difficult to regulate. Environmental damage from SSM is often significant (McMahon, 1999; Rodrigues-Filho *et al.*, 2004; Veiga *et al.*, 2004). SSM operations rarely require skilled labor and working conditions are often dangerous. Workers are injured and sometimes killed. Injury compensation mechanisms are rare and other non-skilled laborers fill the now vacant position so SSM operations are less concerned with labor conditions. Finally, SSM operations have little financial incentive to address concerns from neighboring communities. Conflicts of interest are generally ignored because the communities do not have the power to terminate SSM operations.

Artisanal mining is mining conducted as a subsistence activity. It is characterized by small capacity, use of manual labor, and poor land tenure. Artisanal mining is often illegal and only represents approximately 2% of annual global gold production. The financial incentives of artisanal mining encourage self-endangerment, marginalization of non-miners, and environmental destruction (McMahon, 1999; Veiga *et al.*, 2004). Technical expertise is minimal and working conditions are often dangerous. After an area has been worked over, high grade ore may be left in particularly dangerous areas to access. Artisanal miners disregard community and environmental concerns partially because they have no legal right to work the land. Their goal is often to get the gold before they get caught.

1.1.1 Mercury Use in Small-Scale and Artisanal Mining

Mercury is a potent toxin and its concentration in animal tissues has rapidly increased over the last two centuries (UNEP, 2013, pp.29) - see Figure Figure 1.1. To understand mercury's toxicity, one must understand its forms and their absorption pathways into the human body. Elemental mercury (Hg^0) is liquid at room temperature, but evaporates easily with minor heat increases (Park & Zheng, 2012). Absorption of Hg^0 in humans primarily occurs by inhalation of vapor (Park & Zheng, 2012; UNEP, 2013). Due to its lipid solubility, approximately 80% of inhaled Hg^0 vapor is quickly absorbed and distributed to the brain and kidneys (Park & Zheng, 2012). Acute exposure to Hg^0 can lead to severe lung damage and even death (Park & Zheng, 2012). Chronic exposure to Hg^0 vapor causes tremors, trembling of the hands, narrowing of vision, shyness, excessive emotionality, gingivitis and excessive salivation (Bose-O'Reilly *et al.*, 2008; Park & Zheng, 2012; Veiga *et al.*, 2004). Artisanal mining techniques typically employ Hg^0 to extract gold from its ore (Lacerda, 1997; Veiga *et al.*, 2004). A range of 67% - 82% of this Hg^0 is lost to the atmosphere after its use (Lacerda, 1997). Vapor is often inhaled by miners who heat the gold/mercury amalgam to release gold (Lacerda, 1997; Swain *et al.*, 2007; UNEP, 2013; Veiga *et al.*, 2004).

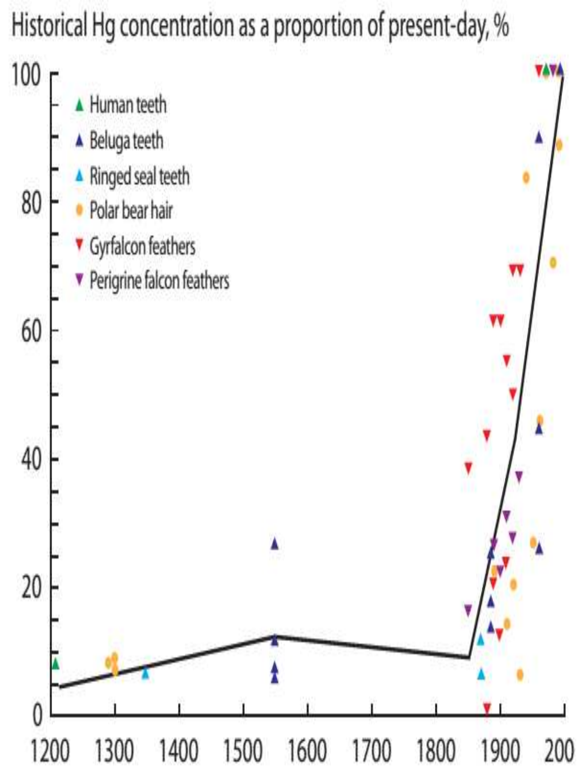

Figure 1.1: Historical Total Mercury Concentration in Animal Teeth, Hair, and Feathers as a Proportion of Present-Day,% from UNEP (2013)

Hg^0 vapor travels into the atmosphere and resides there for up to two years - which allows for global distribution of local Hg^0 pollution (Seigneur *et al.*, 2004; Spadaro & Rabl, 2008; Swain *et al.*, 2007). Eventually, Hg^0 vapor is deposited into aquatic sediments where approximately 2% is transformed by microorganisms into methylmercury (MeHg) - a toxic organic mercury compound (Mergler *et al.*, 2007). MeHg bio-accumulates and concentrates in the aquatic food chain to produce dangerous concentrations in large predatory fish that are consumed by humans (Hylander & Goodsite, 2006; Mergler *et al.*, 2007; UNEP, 2013) - see Figure Figure 1.2. Human exposure to MeHg is driven by consumption of contaminated fish and shellfish (Hylander & Goodsite, 2006; Mergler *et al.*, 2007; Park & Zheng, 2012; UNEP, 2013). Once MeHg is consumed, it impacts the central nervous system and heart (Ekino *et al.*, 2007; Mcalpine & Araki, 1958; Mergler *et al.*, 2007). The developing brain of a human fetus is particularly susceptible to MeHg because it has not yet developed the blood-brain barrier (Ekino *et al.*, 2007; Mcalpine & Araki, 1958; Mergler *et al.*, 2007). By consuming contaminated fish multiple times a week, a pregnant woman can unknowingly harm the development of her baby's brain (Ekino *et al.*, 2007; Mcalpine & Araki, 1958; Mergler *et al.*, 2007). Reduction of IQ, as a result of fetal MeHg poisoning, is one of the best studied impacts of MeHg (Mergler *et al.*, 2007). In adults, their is evidence of an association between MeHg ingestion and cardiovascular disease, acute myocardial infarction, high blood pressure, and other heart issues (Mergler *et al.*, 2007). Chronic exposure to MeHg often results in symptoms similar to those for chronic exposure to Hg^0 vapor (Mergler *et al.*, 2007).

Sundseth *et al.* (2010) point out that, "[m]ost of the anthropogenic mercury emitted to the atmosphere originates from mineral processing undertaken at high temperatures, such as combustion of fossil fuels, roasting and smelting of non-ferrous metal ores, coke production and iron and steel foundries, as well as kilns operations in cement industry." This makes mercury emissions an important issue for the the extractive industries. Anthropogenic emissions of mercury represent about 30% of total annual mercury emissions. 'Re-emission' of mercury from soil and the ocean represent 60% and natural emissions represent just 10% (UNEP, 2013, pp. i). Artisanal/small-scale gold mining make up 37% of anthropogenic emissions and the burning of coal represents 24% (UNEP, 2013) - see Figure Figure 1.3.

To begin to address the extractive industry's mercury issue, Lacerda (1997) estimates total historical global mercury emissions from gold mining. To do this, Lacerda (1997) uses

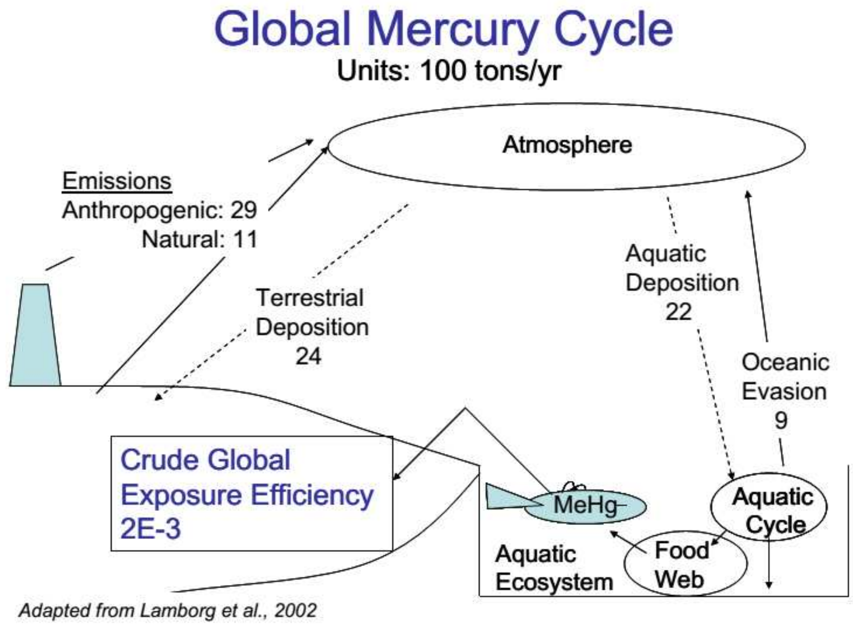

Figure 1.2: Global Mercury Cycle from Rice & Hammitt (2005)

historical records of gold production and known ratios of gold production to mercury loss for the type of gold processing employed. The estimate for total historical emissions is 300,000 metric tons. Lacerda (1997) also estimates annual global mercury emissions from artisanal gold mining to be about 460 metric tons per year. Presumably, the gold price spike up to 2010 account for the near 100% increase in artisanal mercury emissions between 1997 and 2010.

Similarly, McMahon (1999) study the environmental impacts of artisanal, small, and medium mining in the countries of Bolivia, Chile, and Peru. McMahon (1999) conclude that artisanal mining has an enormous impact upon the environment through the use of mercury. Gold recovery is rather low (35-60%) from the grinding and mercury amalgamation process. Therefore, the waste is often taken to cyanidation plants that reprocess the waste. The introduction of cyanide increases the likelihood that 2% of the Hg^0 remaining in the waste will methylate. Communities up to 100km downstream are affected by Hg^0 from artisanal operations (McMahon, 1999). Nonetheless, McMahon (1999) note that public

Sector	Emission (range), tonnes*	%**
By-product or unintentional emissions		
Fossil fuel burning		
Coal burning (all uses)	474 (304 - 678)	24
Oil and natural gas burning	9.9 (4.5 – 16.3)	1
Mining, smelting, & production of metals		
Primary production of ferrous metals	45.5 (20.5 – 241)	2
Primary production of non-ferrous metals (Al, Cu, Pb, Zn)	193 (82 – 660)	10
Large-scale gold production	97.3 (0.7 – 247)	5
Mine production of mercury	11.7 (6.9 – 17.8)	<1
Cement production	173 (65.5 - 646)	9
Oil refining	16 (7.3 - 26.4)	1
Contaminated sites	82.5 (70 - 95)	4
Intentional uses		
Artisanal and small-scale gold mining	727 (410 – 1040)	37
Chlor-alkali industry	28.4 (10.2 – 54.7)	1
Consumer product waste	95.6 (23.7 – 330)	5
Cremation (dental amalgam)	3.6 (0.9 - 11.9)	<1
Grand Total	**1960 (1010 – 4070)**	**100**

Figure 1.3: 2010 Estimated Anthropogenic Mercury Emission Sources from UNEP (2013)

concern generally focused on new projects being developed by large companies.

Veiga *et al.* (2004) is the first study of artisanal gold mining to quantify the impact of artisanal mercury emissions on the health of surrounding communities. This work was conducted by the World Bank, the United Nations Environment Program, the United Nations Development Program, and the United Nations Industrial Development Organization under the umbrella of the Global Mercury Project (Veiga *et al.*, 2004). The Global Mercury Project was conducted from 2002 through 2007 to demonstrate how to overcome barriers to the adoption of pollution prevention measures from artisanal and small-scale mining. The site data for the following analysis are primarily derived from this work.

While there are many additional studies on artisanal mining, small-scale mining, and the use of mercury in mineral processing - a full-scale literature review is not necessary here.

1.2 Dissertation Roadmap

This dissertation is composed of seven chapters. The first five chapters deal with environmental valuation of mine site pollution. Chapter 1 evaluates the capacity of the environmental valuation literature to value mine site pollution. Chapter 1's contribution is a synthesis and summary of environmental valuation methods, primary environmental valuation studies, and environmental valuation meta-analyses that are applicable to mine site pollution. It is shown that the environmental valuation literature is broadly capable of estimating appropriate ballpark values for mine site pollution. However, large gaps exist between scientific understanding of pollution and economists' ability to value that pollution.

Chapter 2 outlines the important components of mine site pollution using an ecosystem service framework and constructs a benefit transfer model to value several relevant ecosystem services. Chapter 2's contribution is the creation of an ecosystem service valuation model for fish population, municipal water, and aquatic habitat. This model could be augmented by future research on air quality, soil quality, groundwater, irrigation water, livestock watering, and noise pollution.

Chapter 3 applies the ecosystem service benefit transfer model to the remediation of mine site pollution in the Upper Arkansas River between Leadville and Canyon City, Colorado. Chapter 3 contributes a quantification of the benefits of remediating legacy lands around Leadville in Colorado. Poor data availability limit monetary valuation to improved recreational fishing and aquatic habitat.

Chapter 4 departs from the ecosystem service framework. Nonetheless, benefit transfer is employed to compare the social and environmental impact of gold mining in various contexts. Chapter 4 contributes a method of comparing the impacts of dissimilar mining operations. This approach could be useful to international aid agencies in categorizing and prioritizing artisanal gold mining sites.

Chapter 5 is significantly different from Chapters 1 through 4. Chapter 5 explores the inclusion of study methodology variables in meta-regression models used for benefit transfer. The measure of benefit transfer error and the treatment of outliers both prove to be important components of this analysis.

Chapter 6, is entirely different from the previous chapters. It uses correlation analysis to evaluate the ability of speculators to influence spot prices via the futures market. The contribution is the empirical result that spot and futures prices move in lock-step during periods of strong contango. This indicates that speculators can influence spot prices via the futures market during these rare periods. However, it is also found that spot and futures prices are highly correlated during weak contango and backwardation.

Chapter 7 concludes with a summary of each chapter's methodology, results, and implications.

1.3 Chapter Introduction

Abandoned mines are an unsavory component of the American West (EPA, 2004). Piles of mine waste are common along Interstate 70, west of Denver. They begin at the mouths of historic mining tunnels and run down the slope into the rivers below. Their orange-yellow tinge gives away their acidic, heavy-metal content.

These polluted slopes are dwarfed by large legacy mine sites such as Leadville, CO and Couer d'Alene, ID. Mining and smelting occurred for a hundred years without concern for, or understanding of, environmental degradation. At these sites, heavy metals contaminated air, water, and soil. Contact with contaminated soils raises lead levels in childrens' blood which causes health problems that impose health care costs and lost earnings (EPA, 1998). When acid generating waste comes into contact with oxygen and water, the resulting acid mine drainage mobilizes heavy metals in stream systems which are toxic to fish (EPA, 2002). Consumption of contaminated fish impairs human health (EPA, 1998) and fish kill imposes costs on fisherman and tourism industries (Morey *et al.*, 2002). In short, many facets of surrounding ecosystems are impaired by mine site pollution and this impairment has an impact on the surrounding community. Economists often refer to these impacts more broadly as *costs*. Such impacts are not costs in the financial sense - where dollars are spent out of pocket. Rather, they are costs that someone in the society must bear.

The legacy of abandoned mines inherited by the West fosters today's opposition to new mine development. Abandoned mines and legacy sites are cited by NGOs as evidence that the mining industry is not a responsible partner. While modern environmental regulation minimizes the possibility of legacy sites on the scale of Leadville or Couer d'Alene, the connection is still made between historic pollution and expectations of pollution at proposed sites (Gestring, 2012)[2]. EIS, required by NEPA, may quantify social and environmental impacts - but it does not *value* them.

Federal and state agencies make decisions regarding the cleanup of legacy and abandoned mine sites based on NPL raking and State AML priority processes. Politically, these decisions are made more difficult by the fact that the costs are well known, but the benefits are not. In other words, the financial and economic *costs* of remediation line up closely. A project

[2]This dissertation is derived from work related to a joint grant from the US Environmental Protection Agency (EPA) and the US Geological Survey (USGS) - known as the Regional Ecosystem Services Research Project (REServ Project). The purpose of the REServ project was to provide a framework to balance the need for metals with their potential impacts to the environment. To achieve this purpose EPA intended to provide mine site data to the USGS as inputs of a geo-environmental model (GEM) that would predict potential changes in ecosystem services from remediation of legacy sites or development of proposed sites.

planner can estimate the cost of labor, equipment rental, and other necessities within a reasonable range. However, the financial and economic *benefits* of remediation do not line up at all. A remediated site produces water of higher quality, fish habitat, and healthy soil - but no dollars. The *financial* benefits are zero, even though *economic* benefits accrue to surrounding populations.

In contrast to remediation cost estimation, an estimate of remediation benefits requires a multi-step, cross-disciplinary approach. First, an ecologist must determine how the remediation will likely affect the surrounding environment. Second, the ecologist must quantify changes in the environment in units that an economist can understand. Finally, the economist must determine how much the human population values the predicted changes in environmental services.

This is difficult for the ecologist because of the complex interactions among ecosystem processes. It is also difficult for the economist because the services in question are not traded in a market and determining their value requires special techniques. Finally, communication between the ecologist and the economist requires a common understanding, a common language and common units. These difficulties result in the absence of an expected benefit value for remediation. Politically, this may be partially responsible for the poor funding of abandoned mine remediation in the United States and the absence of Good Samaritan legislation.

As 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 neighboring communities are hard to value[3]. The absence of a tangible expected value for the impact of

[3]Many within the mining industry are wary of predictions regarding the value of environmental damage. They oppose such an exercise because the United States has a sophisticated regulatory environment for mining and environmental standards generally prohibit the release of pollution which would damage the environment. Additionally, financial costs of environmental damage are often imposed on the company by a regulator in the form of a fine, a natural resource damage payment, or a mandated remediation when a spill does occur (Gestring, 2012). However, it is possible that environmental valuation could be used to the mining industry's advantage. For example, one could compare the losses of salmon population estimated by EPA/Earthworks/Trout Unlimited in dollar terms with the benefits that would be reaped by the project. Such a social cost-benefit exercise may turn out in the industry's favor - even using fish population loss estimates from its detractors.

environmental change may heighten opposition to proposed mines. If local communities had a prediction of environmental change and its associated losses, a more constructive dialogue may be possible regarding proposed mines.

As a disclaimer, the following environmental valuation research does not align with any existing regulatory procedure regarding the valuation of environmental goods in relation to mine projects. The Natural Resource Damage Assessment framework (laid out by the Department of the Interior and the National Oceanic and Atmospheric Administration under CERCLA) has moved away from traditional economic estimates of welfare damages (Roach & Wade, 2006, pp. 423). Environmental impact studies, which are required during permitting, do not value the ecosystem services in question. If anything, this work could be used to estimate use and non-use value for services lost between the time of spill and remediation - in the case where the natural resource can not be restored or replaced (Ofiara, 2002). Additionally, this work could be paired with an EIS to conduct a social cost-benefit analysis of a proposed mine.

Economic research informs environmental decisions through non-market valuation, which allows a more direct comparison of costs and benefits relating to environmental change (Carson *et al.*, 1992; Costanza *et al.*, 1998). Governments can incorporate these comparisons into their decision making process to achieve more efficient allocation of scarce resources. The goal of this chapter 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 dollar terms.

1.4 A Brief Explanation of Environmental Valuation Techniques

Environmental valuation allows humans to incorporate the effects of human activity into the decision-making process by assigning a monetary value to environmental goods and services that humans use. The absence of environmental valuation often results in a de-facto value of zero for the environment in a cost benefit analysis (Hanemann, 1994).

Via environmental valuation, economists estimate the benefit from directly using an environmental service, such as catching a fish. Value derived from use is known as *use value*. But, economists can also estimate the intrinsic value that humans place on environmental services. Value not derived from use of the environmental good is 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, and while they are sometimes excluded or ignored when valuing environmental services, they are an important component that should be considered.

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 service. But, it is often the least desirable method because cost does not necessarily equate with value (Loomis, 2000). For example, a rather valuable environmental service (such as water of drinkable quality) may be relatively inexpensive to replace. In this case, the replacement cost represents a minimum value of benefits that the environmental service provides. A contrasting example is the replacement cost of a fish population in an area sparsely inhabited by humans. Remediating a fishery is expensive, but no one will use it once it is repaired. The replacement cost is higher than the value to humans. 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. And 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.

Stated preference methods gather environmental service values through surveys that detail hypothetical changes in non-market services (e.g. 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 & 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 study methods are available for valuing environmental services. The final methods of valuation are covered in detail in the following section. These methods can be employed when a sufficient number of ecosystem service valuations have been conducted at sites similar to the new site where an economist is attempting to value the same ecosystem service.

1.5 Benefit Transfer, Meta-Analysis, and Meta-Regression Model 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 nature, primary valuations this option is expensive. Scientists must gather information on the water, soil, geology, flora, and fauna to estimate the natural impacts of the project. Then a professional surveying firm must be hired to survey a sample of the relevant population and economists must be hired to make sense of the data in terms of individuals' willingness to pay to avoid a negative change in environmental quality. Budgets for primary valuations can be in the tens of millions of dollars. The large number of mine sites that could be studied precludes primary valuation of many of them, let alone all. In the 1990's, 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 & Hoehn (2006) explain 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 & 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).

Initially, benefit transfer consisted of: 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 (income, study site characteristics, region, etc.) on the valuation result (willingness to pay) that primary studies had generated. Using this information, they constructed a benefit transfer function with the intent of further improving the accuracy of benefit transfer (Loomis, 1992).

1.5.1 Benefit Transfer Issues

A brief description of the main issues related to benefit transfer provides the full context for the benefit transfer process.

1) A benefit transfer can only be as good as the primary study (Bingham *et al.*, 1992), so the quality of primary studies must be thoroughly evaluated. Nelson & Kennedy (2009) enumerates criteria by which primary valuation quality may be judged.

2) The study site and policy site must correspond to one another in relation to population and site characteristics. The more differences between sites, the less likely the benefit transfer is to be valid (Boyle *et al.*, 2010).

3) Boyle *et al.* (2009) note that non-random sorting of populations between study and policy sites could invalidate benefit transfers. For example, transferring benefits from a study site with random fish kills to a policy site with frequent fish kills may not be appropriate because the policy site population self-selected an area with few fish. In other words, the study site population likely values fish more highly than the policy site population and a benefit transfer may overstate the value of loss at the policy site (holding all other characteristics equal).

4) Richardson *et al.* (2014) highlight three important issues in the aggregation of individual benefit estimates to total benefit estimates. Benefit transfers must take non-constant marginal values, geographic scale (or market definition), and substitutability into account during aggregation. Of these three issues, market definition has the largest impact on total benefit estimates (Bateman *et al.*, 2006; Bergstrom & Taylor, 2006; Smith, 1993).

1.5.2 Meta-Analysis

As more benefit value estimates emerged for the same environmental service, it became clear that these estimates were seldom of the same magnitude. Some even had conflicting signs. While 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 & Huang, 1995; Smith & Kaoru, 1990; Woodward & Wui, 2001).

Meta-analysis is simply a quantitative analysis of valuation analyses. It strives to make sense of disparate primary study results and the factors that cause their variation. 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 & Kennedy, 2009). Usually these determinants include population income, population demographics, primary study site characteristics, study method, pollutant type, and publication method (Bergstrom & Taylor, 2006; Navrud & Ready, 2007). Second, the determinants of variation are used to regress a summary estimate of the benefit value (which is known as the *effect size*) (Nelson & Kennedy, 2009).

1.5.3 Meta-Regression Model Benefit Transfer

When the authors of primary studies provide enough information, experts using meta-analysis can address benefit transfer issues more directly. The quality of primary studies can be evaluated by comparing sample size, elicitation format, model construction, etc. with similar studies in the field. Variation in valuation results generated by non-random sorting in populations can be filtered out through inclusion of population variables such as: if the respondent was a local, if her parents were locals, income, environmental persuasion, race, and other demographic characteristics. Variation in results generated by differing site characteristics could also be filtered out using site context variables such as: accessibility, congestion, view, and geographic region. While meta-analysis does not solve all benefit transfer problems, it helps to address many of them (Smith & Pattanayak, 2002).

A meta-analysis function is regressed on the modeled determinants of in-sample variation - with the primary studies' characteristics and valuation results serving as the data. 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 & Loomis, 2000; Shrestha & 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.

1.5.4 Meta-Regression Model Benefit Transfer Issues

Although meta-regression model benefit transfer addresses many benefit transfer issues, it has its own set of challenges (Boyle *et al.*, 2010). These issues are briefly covered below to provide full context for the practice of meta-regression model benefit transfer. It is worth noting that these issues are geared towards the reduction of error generated by meta-regression model benefit transfer.

Temporal trends are often tested for in meta-analysis. Benefit values may increase over time as people gain more appreciation for the value of environmental services. Significant time trends have not been identified to date. The literature also warns of correlation problems between observations produced by the same study or author. However, meta-analyses testing for correlation find that models assuming observational independence have the best fit (Bateman & Jones, 2003; Rosenberger *et al.*, 2000; Vista & Rosenberger, 2013).

Boyle *et al.* (2009) argue that a valuation study subject's utility function (which is what generates the subject's benefit value) must be separable in unobservable characteristics and that the meta-regression model must be correctly specified. These common economic assumptions mean that unobservable characteristics do not effect the benefit value and that the meta-regression model controls for the major determinants of benefit value variation. Model specification test are often conducted to identify the best model and adhere as closely as possible to these assumptions.

The benefit value estimate results discussed in this research are measures of economic welfare. Economic welfare can be measured holding *income* constant or holding *utility* constant. Marshallian surplus (income constant) is often the measure used in hedonic models, travel cost models, and averting behavior valuations. Hicksian surplus (utility constant) is the measure used in contingent valuation and choice model analysis. Accounting for the difference between Marshallian and Hicksian surplus is important in meta-regression model benefit transfer because they produce different results. Many have argued that meta-analyses should only evaluate studies with one measure of welfare (Boyle *et al.*, 2010; Nelson & Kennedy, 2009; Smith & Pattanayak, 2002). Others argue that meta-analyses must simply control for the difference in measurement (Johnston *et al.*, 2003, 2006; Rosenberger & Loomis, 2000; Shrestha & Loomis, 2001).

Publication selection bias is a bias created within the valuation literature through the conscious and unconscious decisions of researchers, editors, and research funding groups regarding what valuation research is conducted and published. Many authors have evaluated publication selection bias with the purpose of identifying its existence, determining its magnitude/sign, and rectifying the bias for the purpose of benefit transfer (Rosenberger & Phipps, 2007; Rosenberger & Stanley, 2006; Stanley, 2005, 2008). Many of the methods of correcting publication selection bias require data and unpublished studies that are not available for the scope of this dissertation.

Meta-regression model benefit transfer provides higher benefit transfer accuracy when the study site and policy site are dissimilar - which is most cases (Kaul *et al.*, 2013; Loomis & Rosenberger, 2006; Moeltner *et al.*, 2007; Richardson *et al.*, 2014; Smith & Pattanayak, 2002). However, there is no consensus protocol for appropriate meta-regression model benefit transfer (Johnston & Rosenberger, 2010; Kaul *et al.*, 2013). Model specification is of crucial importance, but Nelson & Kennedy (2009) highlight the fact that there is still minimal theoretical backing for the determinants used in environmental valuation meta-regression model benefit transfer. Economic theory anticipates variation in non-market valuation results due to factors such as socio-economic indicators, site specific factors, site context, etc. However, meta-analyses provide empirical evidence that variation in non-market valuation results can also be explained by how the study was conducted (Johnston *et al.*, 2003, 2005; Nelson & Kennedy, 2009; Shrestha & Loomis, 2001). These sources of variation are known as *study methodology variables* and economic theory has difficulty with their explanation of result variation (Johnston *et al.*, 2006).

Methodological variables describe how the original study was conducted: contingent valuation/hedonic/travel cost/replacement cost method, phone/in-person/mail survey, econometric model chosen for benefit summary estimation, etc. An unstudied policy site will (by definition) not have any values to apply to the methodological variables. The dilemma is that meta-regression model benefit transfer controls for methodological variation by estimating the parameters associated with methodological variables - but it is unclear as to what value they should receive when applied to the policy site.

This situation sparked a debate regarding the proper method for conducting meta-regression model benefit transfer (Bergstrom & Taylor, 2006; Smith & Pattanayak, 2002). One side argued for a parsimonious, theoretically consistent meta-regression model benefit transfer function (Moeltner *et al.*, 2007). The other side argued for a less parsimonious model that includes all relevant variables (Johnston *et al.*, 2005). The less parsimonious approach is dominating the literature due to a 2006-2010 flood of empirical papers which show that methodological differences can help in explaining WTP result variation between studies (Johnston *et al.*, 2006; Rosenberger & Phipps, 2007; Stapler & Johnston, 2009).

Another issue raised by the literature is convergent validity. Convergent validity is the notion that studies measuring the same benefit value ought to achieve similar results. In the context of benefit transfer, convergent validity means that a benefit transfer from a study site converges with the policy site's WTP. In practice, convergent validity is measured by conducting valuation analyses at multiple sites, transferring values across sites, and comparing the transferred values with the estimated value for the site. For the sake of this discussion, the primary valuation WTP result is assumed to be its true value - even though there is likely some error in this value. Several studies have evaluated the convergent validity of benefit transfer and yielded promising results (Boyle *et al.*, 2010; Downing & Ozuna, 1996; Kaul *et al.*, 2013; Rosenberger & Phipps, 2007; Rosenberger & Loomis, 2000; Rosenberger & Stanley, 2006).

Within the literature, there is no consensus protocol for *testing* convergent validity in benefit transfer (Boyle *et al.*, 2010, pp. 20). The two most common methods are: 1) evaluating the percentage of absolute difference between the transferred value and the actual value (benefit transfer error) and 2) testing the equality of meta-regression model benefit transfer parameters. While testing the equality of parameters appears to be more rigorous (Boyle *et al.*, 2010), the percentage of absolute benefit transfer error seems to be a more relevant and accessible method (Kaul *et al.*, 2013). Two studies that summarize the convergent validity literature for benefit transfer error yield incredibly low errors. Boyle *et al.* (2010) find an average absolute transfer error in the range of 32%-39% while Kaul *et al.* (2013) found median absolute benefit transfer error of 39%.

In conclusion, the general environmental valuation literature is comprised of many techniques to estimate the value of non-market environmental goods. These techniques produce primary valuation studies - the results of which can be transferred to unstudied sites via benefit transfer. now the focus turns from the broad environmental valuation literature to environmental valuation literature that has been applied to mine sites.

1.6 A Summary of the Mine Site Environmental Valuation Literature

The number of environmental valuation studies geared towards abandoned, proposed, or operating mine sites is rather small. Section 1.6 is dedicated to highlighting and summarizing each of these studies. The purpose of this review is to discover material for the construction of a mine site pollution benefit transfer model. However, as will be explained, many difficulties arise.

1.6.1 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) employ 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) provides 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, geo-chemistry, ecology, and lithology. This is because the authors accounted for particularities 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 hardrock mining policy site. Information can be obtained from this study but one would have to delve deeply into the basic components (water treatment, fish replacement, etc) to approximate transferable benefits.

Similarly, Michael & Pearce (1989) conduct a social cost-benefit-analysis of a remediation project in Northwestern England that reclaims a large abandoned coal field. A residential area was built around coal spoil heaps which caught fire, collapsed into gardens, blew dust, and contained mine shafts. The coal field was turned into an agricultural area with forested footpaths and soccer fields. Michael & Pearce (1989) employ 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) propose a framework for valuing remediation benefits, but do not produce a benefit value estimate. Nonetheless, Button *et al.* (1999) is important because it highlights the complexities, synergies, and difficulties associated with remediation of mine site pollution. For example, Button *et al.* (1999) note 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) mention 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) discuss the difficulty and cost of collecting site specific economic valuation information and suggest benefit transfer as the best alternative.

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

Damigos & Kaliampakos (2003a) and Damigos & Kaliampakos (2003b) are derived from the same contingent valuation of the proposed remediation of an abandoned rock quarry in Athens. 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 & Messner (2009) both apply a rigorous contingent valuation to a remediation scheme in East Germany that converts open pit coal mines into recreational lake parks. It is easy to imagine using these studies as study sites for open pit, hardrock mines in the United States that could economically be turned into lake parks. Such a possibility would depend on the site specific hydrology and rock chemistry. But, a lake for recreation would likely provide more economic benefits than a bare, abandoned open pit.

Mendes *et al.* (2007) work to create 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 are 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) employ a hedonic study in the Cheat River watershed of West Virginia and show 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. 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, hardrock 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) use conjoint analysis to estimate what the public would be willing to accept for reducing various bauxite mine remediation schemes around Perth, Australia. The focus is on timing and reductions in plant species, richness, wildlife habitat, and bird populations. Like many of the previous studies, it is difficult to map Burton *et al.* (2012) into a benefit transfer. The benefits/losses being estimated are particularly site specific and the policy site would have to be an excellent match for a valid benefit transfer.

While not a remediation scheme, Neelawala *et al.* (2013) use 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 are willing to pay $13,703 (2013$) to be one kilometer farther from the pollution source when they are within a four kilometer radius (Neelawala *et al.*, 2013).

1.6.2 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 & Dubourg (1993) takes a hypothetical and expert opinion approach to a social cost-benefit analysis of a proposed coal strip mine in North Staffordshire, England. Trigg & Dubourg (1993) survey 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 NOAA contingent valuation standards and it would be dubious to transfer the results.

Damigos & Kaliampakos (2006) is a social cost-benefit analysis of a proposed open pit gold mine in Greece - which was funded by the project's owner. Damigos & Kaliampakos (2006) indiscriminately use results from many of the previously mentioned studies for benefit transfer (Damigos & Kaliampakos, 2003a; Randall *et al.*, 1978; Trigg & Dubourg, 1993). While Damigos & Kaliampakos (2006) provides a framework for social cost-benefit analysis of proposed mining projects, it does 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 & Kaliampakos (2006) employ the 30% property value reduction from Trigg & Dubourg (1993). Considering that Trigg & Dubourg (1993) do not identify which environmental goods (view, air quality, congestion, etc.) are responsible for the decrease in property value, it seems difficult for Damigos & Kaliampakos (2006) to directly transfer these values. Each benefit transfer within Damigos & Kaliampakos (2006) is of this nature. The study site is related, but it is unclear how similar it is to the policy site or how valid the benefit transfer is. UNALDI *et al.* (2011) is a study that essentially duplicates the efforts of Damigos & Kaliampakos (2006), but for a 'generic' gold deposit in Turkey.

In short, the social cost-benefit-analyses of proposed mine sites are of low quality and should not be used.

1.6.3 Natural Resource Damage Assessments Related to Mine Sites

The final source 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 & Schulze, 1985) and the defendant (Ward *et al.*, 1992) yield completely different estimates. The plaintiff's expert witness conducts a contingent valuation at the local, county, and state level where the focus of the survey is the subject's willingness to pay to cleanup the entire site (Rowe & Schulze, 1985). Once the value is elicited, the survey asks respondents what percentage of their willingness to pay value is represented by use value, non-use value, and existence value (Rowe & Schulze, 1985). The defendant's expert witness, on the other hand, estimates replacement cost values to cover contaminated areas with top soil and to redrill wells for residential water (Ward *et al.*, 1992). These replacement values have 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 (Council., 2005; EPA, 2002). Similarly, a preliminary estimate of damages at Leadville (Incorporated, 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 Incorporated (2006) (as with all habitat equivalency analyses) is that its economic assessment of damages fails to incorporate how the changes in these ecosystem services affect human well being. Instead, Incorporated (2006) calculates damages using abatement costs required to return the ecosystem services to their baseline condition. As a result, HEA is unlikely to result in efficient outcomes because there is no balancing of costs and benefits of remediation. HEA is 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 & Schulze (1985) values the whole site, rather than ecosystem services. Ward *et al.* (1992) estimates replacement values, which are the minimum value for the ecosystem service in question. The Coeur D'Alene, Idaho NRDA does not estimate any benefits. And the habitat equivalency assessment (for Leadville) is a replacement cost approach, not a technique for the optimization of social resources. An ideal literature would provide numerous environmental valuation studies of mine site pollution. This literature review demonstrates the limited number of studies on the subject.

1.7 The Ecosystem Service Framework

A significant limitation of the mine site valuation literature is that few studies specify exactly which environmental/ecosystem services are being valued. Valuation is conducted at the level of an entire site which makes it difficult to determine what is being valued or how to transfer the valuation results. The ecosystem service framework for environmental valuation addresses these difficulties by breaking site level valuations into each component of the ecosystem that enters human utility functions to create benefit. Ecosystem service valuation not only delineates the exact services to be valued, but it also facilitates future benefit transfer by reducing the requirements for site correspondence.

Take the contending Eagle Mine NRDA valuations for example. The Eagle Mine polluted soil, air, and water. The polluted soil could impact human health via ingestion of heavy metals and ecosystem function via heavy metal contamination. Polluted air could affect human health via inhalation of wind-blown soil. And polluted water could affect human health via drinking water or contact during boating. Instead of valuing each of these components separately, the plaintiff's expert valuation is conducted at the site level. Unless mine site contamination and context are the same at another site - which is not likely - the results of this study cannot be transferred accurately. On the other hand, the defendant's expert valuation estimates the cost to limit exposure to pollution from the Eagle Mine. Cost estimation misses the point of valuation by confusing the concepts of cost and value. At best, the cost of replacement provides a floor for the value of an ecosystem service. At worst, it confuses the issue entirely.

In contrast to the Eagle Mine NRDAs stands the valuation of the BP Oil Spill (Board *et al.*, 2013). Board *et al.* (2013) breaks 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 & 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 has been developed in the field of ecological economics and focuses on valuing only the *final outputs* of an ecosystem. This remedy is known as the *ecosystem service framework*.

1.7.1 Ecosystem Service Definition

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 recycling, regulation of pH balance, and maintenance of biological diversity. By contrast, an ecosystem service is the result of ecosystem processes and it sustains or enhances human life. Within the ecosystem service framework, "Ecosystem services emanate from a functioning ecosystem... They enter the [human's] utility function either directly, or along with other produced goods as inputs in a production process resulting in consumable goods" (Brown *et al.*, 2007). In this way, ecosystem services link the functioning of ecosystems to human welfare.

A definition of ecosystem services is required that satisfies the underlying ecological science *and* the requirements of economic application. Ecologists historically defined ecosystem services as the benefits that humans derive from ecosystems - which encompasses ecosystem processes and ecosystem services (Wallace, 2007). This definition tends to double count the benefits that humans derive from the ecosystem. In other words, the value of the function and the value of the service are both counted.

Economists, on the other hand, define ecosystem services as the *final* components of ecosystem processes that are consumed by humans to create human benefit (Boyd & Banzhaf, 2007; Boyd, 2007; Fisher *et al.*, 2008). Examples of ecosystem services under this definition are target fish populations for recreational angling, water quality at municipal water intakes, and air quality.

This definition has three implications as described by Boyd & Banzhaf (2007). First, ecosystem services are directly enjoyed or valued by humans. Therefore, natural land cover is only an ecosystem service if it can be accessed and enjoyed by humans. Second, ecosystem services are measurable physical components, which can be paired with a value. For example, the number of acres of land cover is measurable and the value associated with it can be measured in dollars per acre. Third, an ecosystem service is not the benefit received by humans. Rather, it is a subset of inputs that create utility and can potentially augment human welfare. The following illustrates this concept:

> *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 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 (Boyd & Banzhaf, 2007).*

Thus, the ecosystem service is the portion of a person's utility that is attributable to the ecosystem itself (target fish population, surroundings, and water body) rather than from other determinants (inputs) of utility, such as, fishing tackle, a boat, or road access. The angler can *substitute* these other inputs for the ecosystem service to obtain the same level of utility and vice versa (e.g. hike to a spot with less road access, but better surroundings). This is an important distinction to ensure that the services themselves are being valued (and not the end utility) for the purposes of a valuation study. Additionally, ecosystem services are benefit specific. For example the quality of the natural land cover in a viewshed provides an aesthetic benefit to anglers, but is inconsequential to the commercial fishermen. Therefore the quality of the surrounding nature is an ecosystem service to the anglers, but not for the commercial fishermen. This definition avoids double counting by focusing on the final specific service of the ecosystem from which humans derive a benefit - rather than focusing on the general benefit derived or on intermediate components.

In conclusion, an ecosystem service is a measurable quantity, which represents the final component of ecosystem processes that is consumed by humans. This quantity is paired with a value per unit - which yields a value for the ecosystem service. Ecosystem services provide an avenue for more precise valuation of environmental quality because they are quantifiable and changes in their quantity affect humans.

1.7.2 The Endpoint Problem

While the above definition of ecosystem services appears rather tight, problems persist in linking natural and social science models (Kontogianni *et al.*, 2010). Boyd (2007) explain, "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." For example, in the REServ Project the endpoint for recreational angling valuation is fish population. Natural scientists at the USGS have used water quality data to model changes in the ecosystem service endpoint - fish population. The endpoint problem is that anglers have no value for the fish population itself (Personal communication with Dr. Boyd). Instead, they value the fish they catch or the day they spent attempting to catch fish. Therefore, economic valuation studies of fishing have focused on the value of an angling-day or the willingness to pay to catch an additional fish. Put simply, the endpoint problem here is that natural scientists provide estimates of fish population which social scientists can not value[4].

The question can be stated as follows, "What is the dollar benefit of doubling the fish population via improvements in water quality and habitat?" To answer this question one would have to isolate the various inputs to fish caught - such as; fish population, fishing capital, fishing skill, labor, and time. To date, such a study has not been conducted (Personal correspondence with Dr. James Boyd).

1.8 Selection of Ecosystem Services to Value Via a Benefit Transfer Model

The mine site environmental valuation literature supports the combination of new and existing data to value environmental damage related to mining. In fact, several authors highlight it as the only possibility,

> *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, pp.464).*

The first step in taking this approach is the selection of ecosystem services.

1.8.1 Selection Criteria

There are three main criteria for the selection of ecosystem services to model in this context[5]. The first is that natural scientists must be able to model the ecosystem service in relation to underlying ecosystem processes and produce an output that the economist can

[4] It is acknowledged that a model could be created to predict changes in the number of fish caught as a result of changes in fish population and that such a model could apply the valuation of fish caught to the fish population. However, such a model would be predicting information that is already available and the valuation would still be based on fish caught, not the fish population

[5] Mine site impact assessments would identify ecosystem services impacted by specific mines. However, this section refers to ecosystem services impacted more generally by mine site pollution.

value. The second criteria is that there must be sufficient data and valuation studies available for the ecosystem service. Finally, the ecosystem service must encapsulate a significant portion of costs or benefits related to mine site pollution.

Guidance is provided by Button *et al.* (1999), "In terms of AMD remediation, the importance of restoring water quality, the restoration of scenic beauty and the reintroduction of fish stock emerge as key issues" (Button *et al.*, 1999). The three ecosystem services that match these criteria and follow this guidance are: target fish populations for recreational angling, household and municipal water intake quality, and acres of natural land cover altered by mining.

1.8.2 Selected Ecosystem Services

The ecosystem's production of fish for recreational angling is the first ecosystem service to be valued by the benefit transfer model. Fish population can increase in a stream segment as a result of mine site pollution remediation. For example, capping of mine waste can reduce heavy metal concentration downstream. Alternatively, mine site development can decrease fish population by altering water flow, reducing habitat, or accidentally releasing toxic effluent.

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 & Ng, 2009; Vaughan & Russell, 1982). While the target fish population is required for the value of an angling day, the WTP for an angling day does not specifically inform the value of the fish population. Therefore, these studies can only be employed under an ecosystem service framework for areas where target fish populations come into (or go out of) existence due to the proposed project. In other words, if the target fish population was previously zero (during which time no angling benefits accrued to society) and a project brought the target fish population into existence (after which angling benefits accrue to society), then the ecosystem service framework allows for the improvement in the stream system to be valued at the endpoint of the target fish population via WTP per angling day. However, if a target population already exists, an increase in the target population may not be reflected by a larger WTP per angling day and a different approach is required.

For non-zero fish populations, the ecosystem service units provided for this analysis are percentage changes in population. Within the valuation literature, two studies address this issue. Loomis & Richardson (2008) provide a WTP based on percentage changes in salmon and steelhead populations and EPA (2006) provide a sound meta-analysis of WTP to catch an additional fish.

Municipal and household intake water quality refers to the quality of groundwater or surface water brought into the treatment system[6]. In this context, quality is thought of 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, it's value falls to zero.

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 can not 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 Mt. Polley tailings storage facility in British Columbia released twenty-five million cubic meters of water and tailings into Hazeltine Creek, Quesnel Lake, and Quesnel River. A drinking water ban was imposed on approximately 150 households for nine days.

Natural land cover refers to forests, meadows, lake surface area, riparian habitat, etc. that humans use for various activities. Abandoned mine site remediation often converts mine waste (tailings) acreage to meadow, forest, or riparian acreage. Mineral development often turns meadow, forest, or riparian acreage into tailing pile acreage, open-pit acreage, or acres of subsidence from underground mining.

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[6] For municipalities, the ecosystem service is the quality of raw water that enters their treatment system. For households, the ecosystem service is the quality of well-water being drawn into the treatment system.

The ecosystem service units for natural land cover are in acres. Damages from mining regarding land cover can be estimated by natural scientists using the grade and tonnage of the deposit. The result of the estimate is a model of how many acres will be covered with tailings and where these acres of tailings will be placed. The valuation literature provides studies for acres of national wildlife refuge (Ingraham & Foster, 2008), acres of terrestrial wildlife habitat (Kroeger *et al.*, 2008), acres of riparian habitat (Holmes *et al.*, 2004), acres of forest land (Kramer *et al.*, 2003), and acres of wetland (Woodward & Wui, 2001).

1.8.3 Additional Ecosystem Services

Additional ecosystem services could have been modeled for valuation of mine site pollution. They were not modeled because data concerning biological, geological, hydrological, and spatial characteristics of intended study sites proved elusive. This prevented the modeling of changes in additional ecosystem services - which encouraged a focus on the three ecosystem services with the most impact on total value (Button *et al.*, 1999, pp.465-466). Others have noted similar data difficulties when attempting to value ecosystem services related to mine site pollution (Burton *et al.*, 2012; Button *et al.*, 1999; Williamson *et al.*, 2008). The most valuable ecosystem services that were not modeled are: soil quality for human health, air quality for human health, and scenic viewsheds.

1.9 Gaps in the Literature and Conclusion

The first gap in the literature is the lack of a benefit transfer tool capable of capturing the majority of value related to changes in mine site pollution. Use of the ecosystem service framework will help in constructing a scientifically rigorous benefit transfer model. The second gap in the literature is the application of a mine site pollution benefit transfer tool. Identification and valuation of mine site data is proven to be difficult by the existing literature. Successful application of a benefit transfer model would illuminate viable options for exploiting existing mine site data.

CHAPTER 2
A BENEFIT TRANSFER MODEL FOR VALUATION OF ECOSYSTEM SERVICES IMPACTED BY MINE SITE POLLUTION

The goal of this chapter is to apply the environmental valuation literature to ecosystem services that are affected by mine site pollution. The purpose of this is to construct a model to measure 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[7]. The second concept is that ecosystems are made up of interactive processes between minerals and organic material 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.

2.1 Benefit Transfer Models for Selected Ecosystem Services

Combining these three concepts allows the environmental valuation literature 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 illuminate environmental costs and benefits of changing mine site pollution.

2.1.1 Fish Population

To value changes in target fish population for recreational angling, results of a meta-analysis[8] are employed (EPA, 2006). EPA (2006) estimates a fisherman's willingness to pay

[7]Or, conversely, the amount they would need to be compensated to accept the change.
[8]Again, meta-analysis is a statistical analysis of existing valuation studies.

to catch an additional fish with the purpose of valuing changes in fish population due to new regulations for manufacturing plants that draw water from streams, rivers, lakes, and oceans in the United States. EPA (2006) is employed for benefit transfer by using its statistical synthesis of the fish valuation literature and applying the results to unstudied sites. The transfer of EPA (2006) provide estimates for the value of trout in Central Colorado. The rest of this subsection focuses on the methodology used by EPA (2006) and their findings. For more detail, please refer to EPA (2006) and Johnston *et al.* (2006).

EPA (2006) employs a meta-analysis of the fish valuation literature to estimate the willingness to pay to catch a fish. EPA (2006) use estimates from original fish valuation studies as data points for regression analysis to estimate a value per fish that can be transferred to new sites. EPA (2006) conducts an extensive literature review of fish valuation studies in the published economic literature, academic dissertations, and conference presentations. From this universe of studies, EPA (2006) select 48 studies that provide marginal values of catching an additional fish. The 48 studies vary in aspects such as study methodology (i.e. stated vs. revealed preference), elicitation method (i.e. phone interview, survey, in person interview, etc.), fish species, study location, study date, baseline catch rate, and population sample characteristics. Each of the 48 studies contains multiple estimates, with varied sample characteristics and model specifications being the reason for more than one estimate per study. These original study values provide EPA (2006) with 391 estimates of willingness to pay to catch an additional fish.

To estimate the marginal value of catching an additional fish, EPA (2006) estimate a regression on the 391 willingness to pay estimates. This regression estimates the influence of primary study variables described above such as: baseline catch rate, species, angler income, and study methodology. The result is an estimation of the influence these variables have on willingness to pay per fish and a meta-regression function that embodies the influence of these variables. EPA (2006) then use the resulting meta-regression function to transfer their results. This involves setting the function variables so that they correspond to the new site in question. In turn, this predicts anglers' willingness to pay to catch an additional fish. The main shortcomings of this approach are: 1) that the marginal WTP per fish are constant and 2) that WTP to catch a fish is still one step removed from valuing the fish population itself (the endpoint problem).

Table Table 2.1 shows how species in the primary valuation studies were aggregated for the meta-analysis. Table Table 2.2 reports the marginal value per fish results of EPA (2006), updated to 2013 dollars using the Bureau of Labor Statistics CPI Inflation Calculator. These estimates represent mean value estimates and comprise the fish portion of the benefit transfer model for mine site pollution. Table Table 2.2 also provides the standard error of these mean value estimates in parentheses.

These mean values fit the ecosystem services approach, they can be transferred to a variety of species in regions throughout the United States, and they are the result of a comprehensive analysis of the fish valuation literature by EPA (2006). The difficulty in constructing a benefit transfer model is in finding an appropriate study that is of high quality. EPA (2006) is such a study.

While EPA (2006) provides a rigorous valuation study, it is useful to compare the meta-analysis results with the underlying valuation study results. Table Table 2.3 shows a representative sample of valuation studies from the economic literature that apply directly to trout in Colorado.

Table Table 2.3 indicates that the value of an additional trout caught is within the range of \$0.77 to \$4.16, with a study weighted average of \$2.28. The recommended value of \$2.94 from EPA (2006) falls within this range.

2.1.2 Household and Municipal Intake Water Quality: A Binomial Approach

Periodically, metal contamination makes water unusable. When water quality becomes so low that it is no longer treatable, it is best to value the lost benefits of water service[9]. Es-

[9]Replacement cost is another potential valuation method. But, as described in 1.4, the replacement cost represents a lower bound of the value. Additionally, site correspondence is difficult for replacement cost - making it a less transferable method.

Table 2.1: Aggregate Fish Species Groups from EPA (2006)

Aggregate Group	Number of Observations	Species Included[a]
Big game	30	Billfish family, dogfish, rays, sharks, skates, sturgeon, swordfish, tarpon family, tuna, other big game
Small game	74	Barracuda, bluefish, bonito, cobia, dolly garden, dolphinfish, jacks, mackerel, red drum, seatrout, striped bass, weakfish, other small game
Flatfish	46	Halibut, sand dab, summer flounder, winter flounder, other flatfish
Other saltwater	89	Banded drum, black drum, chubby, cod family, cow cod, 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	Northern pike, walleye
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]Some 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 were assigned to the aggregate species group with which they shared the most species.
[b]Rainbow trout in the Great Lakes were classified 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 same species *Oncorhynchus mykiss*

Table 2.2: Marginal Recreational Value per Fish by Region and Species from EPA (2006) (with Krinsky-Robb 5% and 95% Confidence Intervals)[a,b]

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
	($4.12-$13.76)	($1.95-$19.14)	($2.06-$17.94)	($2.42-$14.31)	($2.53-$13.31)		($1.47-$20.74)
Flatfish	$10.14	$6.19	$5.83	$5.83			
	($4.85-$20.89)	($3.59-$10.73)	($3.45-$9.95)	($3.59-$9.47)			
Other saltwater	$3.07	$3.10	$3.03	$2.96	$2.89		
	($1.62-$5.86)	($1.62-$5.94)	($1.65-$5.60)	($1.83-$4.82)	($1.80-$4.65)		
Salmon						$13.78	
						($10.38-$18.29)	
Walleye/pike						$4.27	$4.25
						($2.61-$7.02)	($2.28-$8.03)
Bass						$8.89	$9.36
						($6.04-$13.12)	($5.49-$15.98)
Panfish			$1.10			$1.38	$1.10
			($0.59-$2.01)			($0.91-$2.12)	($0.59-$2.01)
Trout						$9.79	$2.94
						($7.24-$13.31)	($1.50-$4.46)
Unidentified	$3.22	$3.12	$3.37	$2.97	$3.80	$6.46	$2.32
	($1.69-$6.17)	($1.63-$5.99)	($1.71-$6.88)	($1.84-$4.87)	($2.02-$7.33)	($4.43-$9.47)	($1.29-$4.14)

[a]All values are converted to 2013$

[b]The 5% and 95% Confidence Intervals in parentheses are based on results of the Krinsky and Robb (1986) approach (USEPA, 2006).

Table 2.3: Trout Specific Values Corresponding to Eagle and Leadville in Colorado

Author and Year	State/Region	Study Methodology	Type of Trout	Marginal Value per Fish[a]
Boyle, Roach, and Waddington (1998)	FWS Mountain Trout	CV	General	$4.16
Johnson (1989)	CO	CV	Rainbow	$3.27
Johnson (1989)	CO	CV	General	$1.10
Johnson (1989)	CO	CV	General	$1.44
Johnson (1989)	CO	CV	General	$2.04
Johnson (1989)	CO	CV	General	$2.19
Johnson et al. (1995)	CO	CV	General	$3.72
Johnson et al. (1995)	CO	CV	General	$2.03
Johnson et al. (1995)	CO	CV	General	$1.85
Johnson et al. (1995)	CO	CV	General	$1.71
Johnson et al. (1995)	CO	CV	General	$1.55
Johnson et al. (1995)	CO	CV	General	$1.42
Johnson et al. (1995)	CO	CV	General	$1.28
Johnson et al. (1995)	CO	CV	General	$1.14
Johnson et al. (1995)	CO	CV	General	$2.18
Johnson et al. (1995)	CO	CV	General	$0.90
Johnson et al. (1995)	CO	CV	General	$0.69
Johnson et al. (1995)	CO	CV	General	$2.30
Johnson et al. (1995)	CO	CV	General	$1.71
Johnson et al. (1995)	CO	CV	General	$1.38
Johnson et al. (1995)	CO	CV	General	$1.14
Johnson et al. (1995)	CO	CV	General	$0.98
Johnson et al. (1995)	CO	CV	General	$0.87
Johnson et al. (1995)	CO	CV	General	$0.77
Johnson et al. (1995)	CO	CV	General	$1.02
Vaughan and Russell (1982)	USA	Travel Cost	General	$1.44
Overall Average				$1.70
Weighted Average (by study)				$2.28

[a]All values are in 2013$

CV refers to contingent valuation which is a stated preference method.

timating 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 is the method used by Aubuchon & Morley (2013).

In the past, FEMA used Dalhuisen *et al.* (2003), a meta-analysis of studies that estimated the price elasticity of demand for water, to evaluate the benefits of creating more secure water supplies for municipalities and households. Aubuchon & Morley (2013) built on FEMA's method of using price elasticity of demand estimates for water to value water service. In doing so, they created a benefit transfer tool that estimates the cost of losing water service for residential customers and businesses. Results for U.S. residential customers based on per capita per day (PCPD) consumption are presented in Table Table 2.4.

Table 2.4: Impact of Municipal Water Loss on Residential Consumers, per Capita per Day (PCPD) from Aubuchon & Morley (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)*
[b]All values are converted to 2013\$

The values in the left hand column of Table Table 2.4 represent different assumptions about the amount of water that is consumed per capita per day (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 Aubuchon & Morley (2013)'s 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, the benefit loss that people suffer from losing service is reduced. However, for the purposes of the benefit transfer model, the Aubuchon & Morley (2013) recommended value of $158 (2013$) will be used. This is the average value of the three population weighted estimates.

In addition to the economic cost to residential consumers, the loss of service for municipal water also includes business losses. Aubuchon & Morley (2013) calculate business losses as the forgone business value due to service disruption by using resilience factors from two studies, one from the Applied Technology Council *et al.* (1991) and the other by Chang *et al.* (2002). The resilience factor is the percent of capacity at which an industry could operate in the absence of water.

Using these resilience factors, Aubuchon & Morley (2013) calculate the economic loss using Equation 2.1.

$$\frac{1}{365 * population} \sum_{i=1}^{n} GDP_i * (1 - r_i) \tag{2.1}$$

Equation 2.1 calculates a per capita daily loss for industry i, with an industry specific resilience factor and then sums across all industry. Aubuchon & Morley (2013) go on and calculate a state level and a population weighted per capita per day business economic loss (see Aubuchon & Morley (2013) for specific formulas). Their results for U.S. total, state level, and population weighted per capita per day business economic loss are shown in Table Table 2.5 for both sets of resilience factors. Using this process, Aubuchon & Morley (2013) recommend the value of $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[10].

Combining the PCPD business loss and residential loss, Aubuchon & Morley (2013) recommend a total economic impact of $215 per person per day for loss of municipal water. This is simply the sum of the two prior recommended values. Table Table 2.6 summarizes the total business and residential economic impact and provides summary statistics.

[10]The population-weighted estimates are considered by Aubuchon & Morley (2013) to be most accurate.

Table 2.5: Impact of Municipal Water Loss on Business Economic Activity, per Capita per Day (PCPD), from Aubuchon & Morley (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**	

[a]All values are converted to 2013$

Table 2.6: Total Economic Impact of Municipal Water Loss, Per Capita Per Day (PCPD) from Aubuchon & Morley (2013)[a]

	Per Capita Per Day Consumption (gal):	**Elasticity of Demand:**		
		-0.41	-0.35	-0.26
Current Value	**Recommended Value=$215**			
Mean:		$93	$135	$1,016
Median:		$96	$114	$458
Std Dev:		$21	$53	$912
Min:		$66	$82	$309
Max:		$128	$219	$2,287

[a] U.S. and State Level (GDP and Consumption) Totals
[b]All values are converted to 2013$

2.1.3 Natural Land Cover

Incorporation of the value of natural land cover, in an ecosystem service framework, must be dealt with carefully. Depending upon the benefit being quantified, natural land cover can serve as an ecosystem process or an ecosystem service. For example, when evaluating the use benefits of hiking, acres of forest are an ecosystem service. The ecosystem service is paired with other inputs of human utility to create a benefit. On the other hand, when valuing the use benefits of wildlife viewing, acres of terrestrial habitat are an ecosystem process. If the value of terrestrial habitat is incorporated, the value of the wildlife population should not also be counted. Prevention of double counting requires attention to this concept.

Following this logic, the value that anglers hold for aquatic habitat ought not be added to the value they hold for fish caught. That 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 ought to 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) Loomis & Richardson (2008) is employed. Loomis & Richardson (2008) rely upon a meta-analysis of aquatic habitat valuations, Johnston *et al.* (2005), to create a benefit transfer tool for the non-use valuation of aquatic habitat.

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

Loomis & Richardson (2008) focus on the non-use portion of Johnston *et al.* (2005) and construct a user-friendly benefit transfer model for the USFS. An example for this benefit transfer model is provided in Table Table 2.7 for the Upper Arkansas River in Central

Colorado. These values are calculated using the median household income for Colorado in 2006 and final values are updated to 2013 dollars. The baseline and increase in water quality figures in Table Table 2.7 are derived from the water quality ladder provided in Johnston *et al.* (2005). Figure Figure 2.1 shows the water quality ladder.

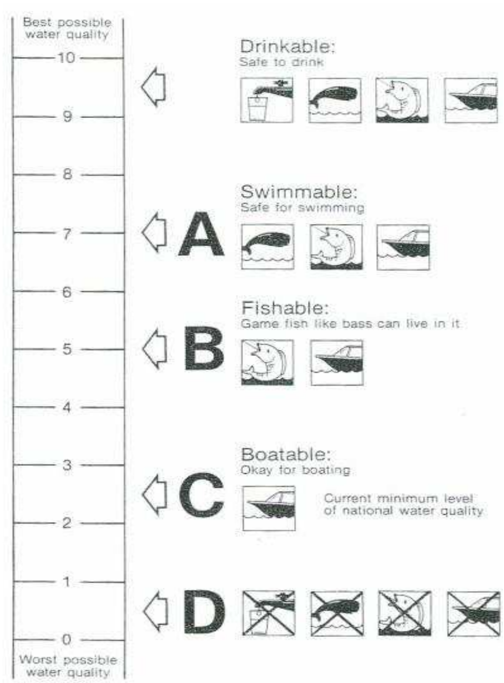


Figure 2.1: Water Quality Ladder from Johnston *et al.* (2005)

When the Loomis & Richardson (2008) benefit transfer model is calibrated to reflect the changes in Colorado the result is an aquatic habitat non-use value of $27.68 (2013$) per household. Note that the endpoint problem is an issue for valuing aquatic habitat non-use 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

Table 2.7: Non-use Values of Aquatic Habitat for Upper Arkansas River, per Household per Year, from Loomis & 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 & Richardson (2008).

with the simplicity of the RFF 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.

2.1.4 Selected Ecosystem Services Benefit Transfer Model Conclusion

The mine site pollution benefit transfer model is summarized for Central Colorado in Table Table 2.8. The model values fish population, household and municipal intake water quality, and acres of natural land cover by transferring estimated values from meta-analytic studies. By translating changes in ecosystem services to monetary values of cost and benefit, decision makers can compare apples to apples when making decisions relating to ecosystem services.

Table 2.8: Summary Table of Benefit Transfer Model Values for Central Colorado (2013\$)

	Central Colorado	Units
Fish Population		
Trout	\$2.94	\$/Fish
Bass	\$9.36	
Water Quality		
Total Economic Impact	\$215.00	\$/PCPD
Natural Land Cover		
Aquatic Habitat Non Use Value	\$13.51	\$/Household

2.2 Benefit Transfer Modeling of Additional Ecosystem Services: Where to Begin

Additional ecosystem services could be valued in an economic model of mine site pollution. However, inaccessible natural science data, lack of predictions for changes in ecosystem services, insufficient time, and/or relatively small additional value prevented their inclusion in the model. Nonetheless, it is worth highlighting where one would begin in the construction of a benefit transfer model for several of these ecosystem services.

2.2.1 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. Historic mine tailings have been scattered by wind and water over the decades. Historic smelting operations did not capture pollutants from their smoke stacks so they precipitated onto the surrounding communities[11]. 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, CO who had backyard soil lead levels greater than 500ppm were 8.4 times more likely to have blood lead levels at or above 10 ug/dL (CDH, 1990, pp.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 childrens' blood lead level.

Elevated blood lead levels in children have been linked to IQ loss, ADHD, 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

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

a start in modeling the value of soil quality improvements at a legacy site. Table Table 2.9 uses Leadville as an example of this process.

Table 2.9: Impact of Elevated Blood Lead Levels in Children from Leadville, CO (2013$)

Blood Lead (ug\dL)	**Medical Cost**[a]	**IQ Loss per ug\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)

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 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 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 versus 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 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 & Krouskas, 2012).

2.2.2 Air Quality for Human Health: Particulate Matter

Air quality can also be affected at mine sites by suspended particulate matter. Smith & Huang (1995) use 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, is a good place to start (Smith & Huang, 1995). Similarly, Vassanadumrongdee *et al.* (2004) provide a starting place for valuing the short-term health effects of air pollution such as coughing, congestion, asthma attacks, etc.

2.2.3 View from a Residence

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/industrial parks/ets have various negative impacts on property value.

2.2.4 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 & Richardson (2008) employ a wetland valuation meta-analysis (Brander & Florax, 2007) for use in benefit transfer. Brander & Florax (2007) analyze European and North American wetland valuation studies that focus on flood prevention, water quality, water quantity, fishing, birdwatching, habitat and storm drainage. Loomis & Richardson (2008) built a meta-regression model benefit transfer function that can be used to value wetlands based on their size, location, and the ecosystem services that they provide.

Similarly, Ingraham & Foster (2008) conduct 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 & Foster (2008) conduct this evaluation for five separate land classifications: open water, forest, shrubland, grasslands and wetlands. Ingraham & Foster (2008) appear to estimate per acre valuations of the land classifications, but they do not explicitly enumerate the values.

Finally, Borisova-Kidder (2006) provides a valuation study for terrestrial open space and habitat. Borisova-Kidder (2006) uses 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 are too disparate and the sample size is 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.

2.2.5 Drinking Water and Groundwater: An Incremental Approach

Drinking water can be affected by excessive runoff at legacy sites or spills from operating mines. Görlach & Interwies (2003) summarize the drinking water literature up to 2003. Much of this literature is comprised of averting behavior studies that evaluate the costs associated with poor drinking water quality (Abdalla, 1990; Abdalla *et al.*, 1992; Collins & Steinback, 1993; Harrington *et al.*, 1989; Laughland *et al.*, 1993). Although these studies are for pollutants unrelated to mining, they apply 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 focus on safe/unsafe drinking water and a group that focus 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 ought to be taken into account. But, these costs do not equate to the value of intake water quality because costs depend on more than the ecosystem service of water quality, e.g., 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 heavy 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 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 & Interwies (2003) summarize 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) conduct meta-analyses of the contingent valuations regarding groundwater. Both Boyle *et al.* (1994) and Poe *et al.* (2001) conclude 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.

2.2.6 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 & Mjelde (2000), Koss & 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 encapsulated 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).

2.3 Summary and Conclusion

The previous chapter constructs a benefit transfer model for the ecosystem services of fish population, municipal intake water quality, and aquatic habitat. While this appears to be a rather limited set of ecosystem services, recent mine spills and mine development controversies reveal the importance of these three services to valuation of mine site pollution. For example, concerns about the massive Mt. Polley Mine tailings spill in August of 2014 focused on household drinking water intake quality, salmon killed (or diverted) by the spill, and impacts to salmon spawning habitat. The ecosystem service model above is capable of valuing the vast majority of 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 Alaska focus on possible impacts to salmon populations and salmon habitat. Finally, the primary *downstream ecological*[12] concern of the Samarco spill in November of 2015 has been the loss of subsistence fishing due to sedimentation. In short, mine site pollution chiefly affects water and fish. Estimation of the value of related ecosystem

[12] Headlines focused upon the loss of life and destruction of a local village that resulted from the spill. Outside of these important non-ecosystem service concerns, the primary concern has been fish.

services captures major components of the value of mine site pollution, while abiding by the restrictive ecosystem service framework.

In contrast to Mt. Polley, Pebble, and Samarco's demonstration of the utility of the mine site pollution benefit transfer model, the Gold King Mine spill in August of 2015 shows the inability of current scientific and economic modeling to value a mine spill. First, the Animas River was not a pristine watershed before the Gold King mine briefly turned it yellow. Any attempt to assess EPA for natural resource damages would first have to disentangle the impacts of background levels of contamination from the impacts of contaminants introduced by the spill[13]. 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, the current public concern related to the Gold King spill is the long-term contamination of river bed sediment. This contamination could conceivably work 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 all of this were scientifically possible, then economists would have to figure out how to value these ecosystem servcie changes. While fish population is more straight forward, bird and wildlife population is less so. Huntable bird and wild life populations do not have a study akin to EPA (2006) that values WTP to shoot another bird/deer. Existing estimates of WTP for a day of hunting do not relate directly enough to the ecosystem services in question.

To conclude this discussion of ground-truthing the benefit transfer model, it is worth comparing the Gold King spill to periodic contamination events in the Arkansas River from the Yak Tunnel in the 1980's. Both watersheds have high levels of contamination from historic mine site pollution. The Navajo Nation downstream of Gold King closed off irrigation ditches during the spill in the same manner as ranchers on the Arkansas River when the Yak Tunnel 'burped'. Events such as the Gold King spill and the conflicts in Peru over irrigation

[13]Ironically, PRP's of the California Gulch Superfund site made the same argument to the EPA 30 years before this spill.

water show the need for future research on the value of clean irrigation water.

As for the ecosystem services mentioned in Chapter 2, but not modeled; provision of ecosystem service changes, additional time, and additional funding could allow value estimation and incorporation into the benefit transfer model. Groundwater contamination, supply reliability, and irrigation water are the three 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 chapter calls for a full scale characterization of natural resource impacts from acid mine drainage in the Animas watershed and predictive modeling of natural resource damages from large, open-pit tailings storage facility failures. The Animas is representative of many watershed 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 concers, it is possible that defensible remediation benefit estimates could kick-start Good Samaritan legislation and channel more resources towards acid mine drainage remediation.

Predictive modeling of natural resource damages 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 defensible amounts for environmental liability bonding. Finally, insurance markets are unwilling to cover low frequency, high magnitude events like Mt. Polley and Samarco. If they could forecast natural resource damage magnitudes and spill frequency, maybe they would be more likely to insure such events. Insurance would spread the risks more likely and could provide a market mechanism to regulate these catastrophic failures.

The Office of the President recently mandated that all federal agencies begin incorporating the value of ecosystem services into federal decisions (OEP, 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.

CHAPTER 3

VALUATION OF A TRANSFORMATION IN THE UPPER ARKANSAS RIVER FISHERY

Thirty years ago, Leadville was the center of a fierce battle between the EPA, the mining industry, the state of Colorado, and local residents over the questions of: "Who should pay to cleanup historic mining waste?" and "How clean is 'clean'?" After 30 years of legal battles, a couple hundred million dollars of environmental remediation, and designation of the Upper Arkansas River as a Gold Medal fishery - the question remains, "Was it worth it?" To illuminate this question, Chapter 3 applies the mine site pollution benefit transfer model to the recovery of the Arkansas River headwaters recreational fishery - see Figure Figure 3.1.

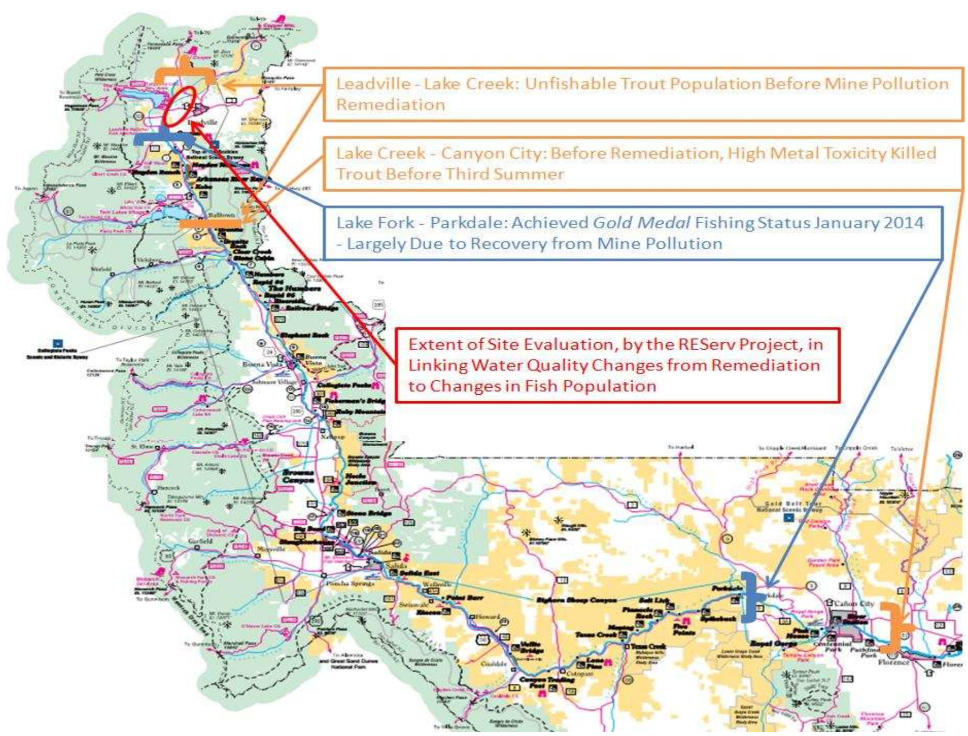


Figure 3.1: Upper Arkansas River: Trout and Mine Pollution Overview

3.1 Background

Before mining came to the area, greenback cutthroat trout and yellowfin cutthroat trout were documented in the Arkansas River headwaters by the U.S. Fish Commission (Behnke, 2010, pp.198,201)[14]. Between 1870 and 1900, non-native rainbow, brook, and brown trout were introduced to the Upper Arkansas River and out-competed the cutthroat trout (Nehring & Policky, 2003). Two centuries of mining around Leadville, created mine site pollution sources that contaminated the Arkansas River headwaters with heavy metals as far downstream as Cañon City. This pollution reduced the fish population between Leadville and Lake Creek to a level that was unfishable before the remediation (Clements *et al.*, 2010; Policky, 2012, 2013). Farther downstream, between Lake Creek and Cañon City, fish did not live beyond their third year due to chronic toxicity of heavy metals (Clements *et al.*, 2010; Policky, 2012, 2013).

In the mid-1980's the EPA grouped the mine site pollution sources around Leadville into the California Gulch Superfund site. The local mining industry and local residents - proud of their mining heritage - insisted that the town was safe to live in and that existing environmental contamination of the Arkansas River was the natural result of providing minerals to society (Klucas, 2004). The EPA and the state argued that residents were at risk to their health and that mining was responsible for pollution of the Arkansas River for 100 miles downstream (Klucas, 2004). The site was remediated over the next twenty-five years and the flow of mine site pollution was reduced dramatically. One benefit of this remediation was the improvement of the Upper Arkansas River fishery. Water treatment, capping of tailings, fish stocking, and in-stream habitat improvements elevated water quality and the trout population. The remediation caused a transformation in the fishery which culminated in 2014 when it achieved the highest fishery designation possible - Gold Medal Water. Aquatic habitat along the entire Upper Arkansas River was improved. Of all the remediation carried

[14] It has been argued that fish did not exist prior to mining in the Upper Arkansas River. This assertion can not be substantiated by the literature available.

out at the California Gulch Superfund site, the only ecosystem service improvement with enough usable data to estimate benefits of remediation is the recovery of trout populations in the Upper Arkansas River.

Evaluation of heavy metals pollution, and its effect on the aquatic ecosystem, has been conducted for over thirty years (Bergstedt *et al.*, 2005; Clements, 1991; Clements *et al.*, 2010; Nehring & Policky, 2003; Policky, 2012, 2013; Roline & Boehmke, 1981). Clements (1991) concludes that heavy metals from mine site pollution reduced species richness, reduced abundance, and caused a shift in community composition from metal-sensitive species to metal-tolerant ones. For example, brown trout were the dominant trout species in the Upper Arkansas because they are more resistant to chronic heavy metal pollution than cutthroat, rainbow, and brook trout. Clements (1991) also notes that metal concentrations in organisms were elevated for ten miles downstream - even when metal concentrations in the water had fallen.

Nehring & Policky (2003) present a synopsis of 17 years of Colorado Parks and Wildlife electrofishing efforts to determine how trout populations were responding to heavy metal pollution abatement. Nehring & Policky (2003) conclude that chronic heavy metals pollution during the 1980's and 1990's had negative impacts on the fish populations - and that an increase in fish populations occurred during the years 2001 and 2002. Bergstedt *et al.* (2005) also quantify brown trout and invertebrate population responses to remediation. Bergstedt *et al.* (2005) argue that these populations stop declining in 1994 and that population improvements became comparable to reference levels upstream of the California Gulch confluence beginning in 2002. Bergstedt *et al.* (2005) assert that, "These data indicate reclamation efforts have improved conditions for the aquatic biota in the Arkansas River downstream of California Gulch." Clements *et al.* (2010) echo this general sentiment. However, Policky (2012) and Policky (2013) argue that the 2001-2003 spike in population was due to favorable low-flow conditions for trout spawning. An ensuing population decline makes the gains between 1995 and 2008 relatively flat. Policky (2012) and Policky (2013) point to 2010 as the period when improvements in the trout population could be fully attributed to California Gulch remediation.

3.2 Purpose and Scope

The purpose of this analysis is to compare the value of remediation expenditures with the value of ecosystem service improvements from remediation. The scope of this analysis covers improvements in fish population from the confluence of California Gulch to Cañon City.

A perfect ecosystem service valuation for the recovery of the fishery would first calculate the increase in fish population that is directly attributable to the remediation. Then it would translate the attributable population increase to an increase in the number of fish caught by anglers as a result of the remediation. Due to insufficient usable data, neither of these tasks was completed successfully. Therefore, an important assumption required by this analysis is that the increase in catch is entirely due to the remediation. While this assumption may not be perfectly true, it is certainly defensible based upon previous analyses of the impact of heavy metals on the Upper Arkansas River (Clements *et al.*, 2010; Policky, 2012, 2013). The ecosystem service framework remains intact because there is no double counting and no question about what component of the ecosystem is being valued.

3.3 Data and Method

The main data relied upon by this analysis were collected from Colorado Parks & Wildlife (CPW) creel censuses conducted along the river from California Gulch to Cañon City 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. Figure Figure 3.2 and Table Table 3.1 provide detailed information on the size of creel census areas and their respective river reaches.

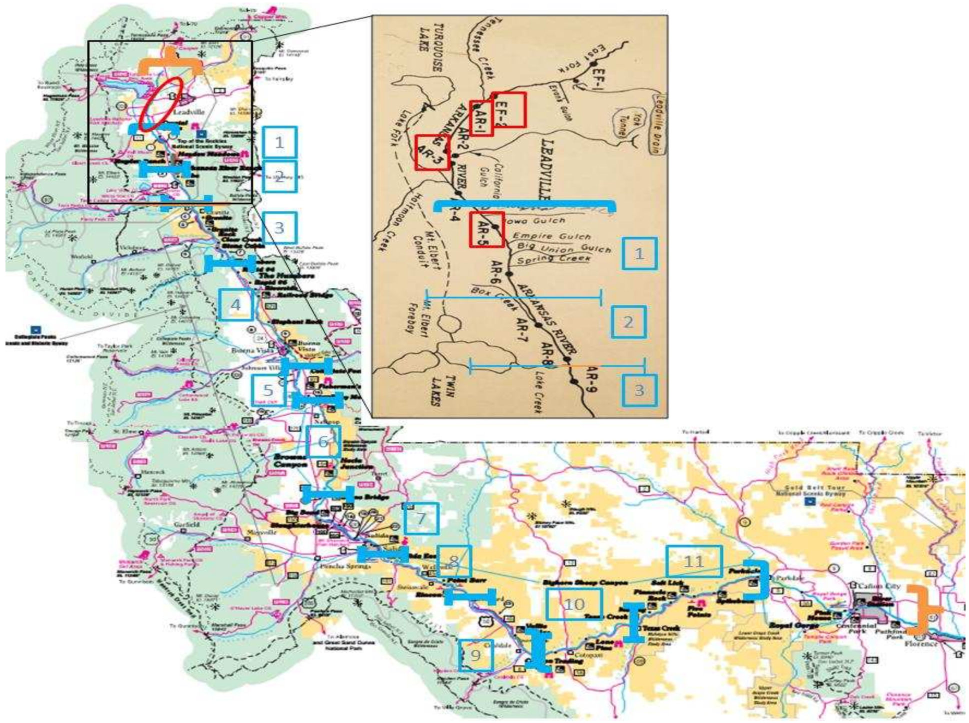


Figure 3.2: Upper Arkansas River: Colorado Parks and Wildlife River Segments and EPA Sampling Sites

Table 3.1: 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
1	Chrystal Lakes - Kobe Bridge	5.1	Hwy 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 - Hwy 285 Bridge	9	Otero Bridge - Railroad Bridge	3.1
5	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 (Personal correspondence, Andrew Treble, CPW). 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 3.1 and Table Table 3.2 show how these factors are calculated.

$$ExtrapolationFactor = 1 + \frac{2012ReachDayEstimate - 2012CreelDayEstimate}{2012CreelDayEstimate} \tag{3.1}$$

Table 3.2: 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**	

The extrapolation factors from Table Table 3.2 are important because they allow extrapolation of creel census area estimates to a larger and more useful scale. The extrapolation factors are 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) - see Table Table 3.3.

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 - see Table Table 3.5. Then, the estimated catch is multiplied by the

Table 3.3: 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	10,800
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**

WTP to catch an additional fish - \$2.94 from Table Table 2.2. See Table Table 3.5 for results in the section below.

The use of an ecosystem services approach for 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 & Ng (2009). This 'angler-centric' approach combines estimates of the number of in-state and out-of-state angling days with the value of in-state (\$72.72) and out-of-state (\$128.13) angling days to achieve a valuation of the change in angling over the study period - see Equation 3.2. Table Table 3.6, in the following section provides the results of this analysis.

$$\begin{aligned} ValueofAnglingEstimate = (ResidentAnglingDays * ResidentWTPperDay) + \\ (NonResidentAnglingDays * WTPperDay) \end{aligned} \quad (3.2)$$

3.3.1 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 upon 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 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 ought to be captured when possible.

Loomis & Richardson (2008) provides a straight forward benefit transfer of non-use value associated with aquatic habitat. To achieve this benefit transfer, the annual non-use value of $29.95 per year is combined with the number of households living in the counties that straddle this section of the Arkansas River. As reflected in Figure Figure 2.1 and Table Table 3.4, the water quality in the Upper Arkansas River moved up two rungs on the 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 heavy metal toxicity.

Table 3.4: Non-use Values of Aquatic Habitat for Central Colorado, per Household per Year, from Loomis & Richardson (2008)

Baseline Water Quality	**Increase in Water Quality**	**Less than 50% Fish Population Change**	**More than 50% Fish Population Change**
4	2	$13.51	**$29.95**[a]

[a]Recommended value for Central Colorado because two rivers went from 'unfishable' to 'fishable'.
All values are in $2013.
Source: (Loomis & Richardson 2008).

Multiplying the recommended value of $29.95 from Table Table 2.7 by the number of households in the three counties that encompass this watershed provides an estimate of the non-use value provided by the improvement in the Upper Arkansas River's aquatic habitat[15]. County household data is provided for 2010 by the U.S. Census Bureau. Table Table 3.8 details the population and annual non-use aquatic habitat values.

[15]While it could be argued that the appropriate market scope for non-use benefits is the state of Colorado, this analysis limits the scope to the relevant counties for a more conservative approach

3.4 Results

Table 3.5: Fish-Centric Valuation Results (2013$)

River Reach Number	Average Hourly Catch Rate			Estimated Number of Fish Caught			Estimated Value of Fish Caught		
	1995	**2008**	**2012**	**1995**	**2008**	**2012**	**1995**	**2008**	**2012**
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 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) indicate that the fishery was slow to improve until 2008. But, 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 thirteen years to double, but doubled again just four years afterward.

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 - $5,630,000 to $5,667,000. Although less pronounced, the bump in annual value between 2008 and 2012 is also reflected in this analysis - $5,667,000 to $8,207,000. 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 - see Table Table 3.7.

Between 1995 and 2012, non-use value of aquatic habitat increases by $834,000 per year. This dollar figure can be thought of as the value that residents place on the transformation

Table 3.6: Angler-Centric Valuation Results ($2013)

River Reach Number	**Estimated River Reach Angling Days**			**Estimated Value of Angling Days**		
	1995	**2008**	**2012**	**1995**	**2008**	**2012**
1	0	2,000	7,200	$0	$160,000	$584,000
2	1,100	1,500	2,400	$87,000	$123,000	$196,000
3	5,000	3,500	10,800	$412,000	$288,000	$878,000
4	2,000	3,200	4,300	$164,000	$260,000	$355,000
5	900	2,300	3,500	$72,000	$187,000	$283,000
6	3,200	8,400	12,700	$262,000	$686,000	$1,038,000
7	2,600	6,900	10,400	$215,000	$563,000	$852,000
8	7,200	6,700	11,600	$591,000	$545,000	$950,000
9	8,500	5,600	9,300	$696,000	$454,000	$760,000
10	12,800	8,300	14,000	$1,044,000	$681,000	$1,140,000
11	13,100	8,600	14,300	$1,072,000	$699,000	$1,171,000
Total	**56,500**	**56,900**	**100,600**	**$4,615,000**	**$4,645,000**	**$8,207,000**

Table 3.7: Comparison of Valuation Results ($2013)

	1995	**2008**	**2012**	**Annual Value of Improvement**
Fish-Centric	$252,000	$522,000	$773,000	$521,000
Angler-Centric	$4,615,000	$4,645,000	$8,207,000	$3,592,000

of the Arkansas River into a Gold Medal Fishery.

Table 3.8: 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
Freemont	16,900	$507,000
Total	**27,800**	**$834,000**

If one assumes that the improvement in the fish-centric valuation between 1995 to 2012 is *due to the remediation*, then the result is a benefit to society of $521,000 a year. The models of trout population improvement from AR3 and AR5 suggest that this assumption holds for the first and second river stretches. Policky (2012) and Policky (2013) suggest that this assumption also holds for the remaining river stretches. Using similar logic for the angler-centric approach, the result is a benefit of $3,600,000 a year.

From a net present value standpoint, the fish-centric value of fishery improvement from 1995 to 2012 (using a three percent discount rate) is $13,401,000 for a 50 year time frame and $16,458,000 for a 100 year time frame - see Table Table 3.9. The angler-centric net present value of fishery improvement from 1995 to 2012 (using the same discount rate) is $92,426,000 for a 50 year time frame and $113,509,000 for a 100 year time frame. Finally, the non-use aquatic habitat net present value is $21,461,000 for a 50 year time frame and $26,356,000 for a 100 year time frame.

Table 3.9: 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	$521,000	$13,400,000	$16,500,000
Angler-Centric	$3,592,000	$92,400,000	$113,500,000
Non-Use Value	$834,000	$21,500,000	$26,400,000

These benefits can be compared to the expenditures made during the remediation. Table Table 3.10 details the expenditures, purpose, and funding source for expenditures found in the public record for the California Gulch Superfund cleanup. The total of \$138,810,000 is a minimum estimate because many large expenditures were made privately by PRPs and are not available to the public. Requests to the PRPs for this information have not been answered to date.

Table 3.10: 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 Yr Field Demo, Biosolids/Lime in Soil	ASARCO/Resurrection	\$6,850,000
2001	OU1 23 Yrs of YWTP Costs	ASARCO/Resurrection	\$21,180,000
2012	OU1 Costs Associated w/ Black Cloud Mine	Resurrection Mining	\$5,070,000
1994	OU2 Malta Gulch Removal Actions (1995 - 1996)	Hecla Mining	\$1,070,000
2001	OU2 15 Yrs of Monitoring	Hecla Mining	\$790,000
	OU3 Denver and Rio Grande Slag Piles	Union Pacific	???
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	OU55 Yrs of Monitoring Costs for AV/CZL Site	ASARCO	\$120,000
2001	OU5 Institutional Control Costs for EGWA	ASARCO	\$40,000
2001	OU5 5 Yrs 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 Yr NPV of Costs	EPA	\$490,000
	OU7 Remedial Costs	ASARCO/Resurrection	???
2001	OU7 14 Yrs 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	Ressurection Mining	\$940,000
2001	OU8 14 Yrs of Monitoring Costs	Ressurection Mining	\$100,000
2001	OU9 Lead Program Costs Over 12 Yrs	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 30Yr NPV of Costs	Resurrection Mining	\$3,690,000
2001	OU10 16 Yrs of Additional Monitoring Costs	Resurrection Mining	\$250,000
2005	OU11 Combined Capital and Operting Costs	ASARCO/Resurrection	\$6,220,000
2012	OU11 Combined Capital and Operting Costs	ASARCO/Resurrection	\$15,870,000
2009	OU12 Institutional Control Monitoring Costs	EPA	\$1,370,000
2012	OU12 3 Yrs of Monitoring Costs	EPA	\$640,000
2012	OU12 3 Yrs of Enforcement Costs	EPA	\$150,000
Total			**\$138,810,000**

3.5 Discussion, Implications and Conclusion

Thirty years after the California Gulch Superfund designation, the regulations of Superfund and the Clean Water Act have provided the answers to the questions, "Who should pay to cleanup historical mining waste?" and "How clean is 'clean'?" Large companies that own some portion of the site will pay and 'clean' is clean enough for ecosystems to provide functional habitat.

The results of this analysis indicate that remediation costs exceeded \$150 million (2013\$) and that the net present value of benefits (@ 3% over 100 years) from the improvement of the fishery are, at most, \$140 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. Therefore, on their face, these results suggest that, "No, the remediation was not worth it." However, this analysis reinforces the importance of the first two questions about Colorado's natural resources - especially now that Silverton is considering Superfund designation after the Gold King Mine spill. Anyone familiar with the Superfund battles of the '80's must wonder if we could now achieve similar benefits with fewer costs. If so, it may be in everyone's interest to dedicate more resources to Colorado's abandoned mine lands problem.

A concern raised by this analysis is the discrepancy between the fish-centric and angler-centric valuations. This difference is the result of differing valuation approaches in the studies from which the benefit estimates are transferred. The fish-centric approach transfers values from Johnston *et al.* (2006), which is a meta-analysis of studies that provide marginal values of catching an additional fish. This approach 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 the angler as a result of her fishing. On the other hand, the angler-centric approach transfers values from

Loomis & Ng (2009), which uses the economic benefits generated by the angler to estimate the willingness to pay values of $72.75 per resident angling day and $128.13 per non-resident angling day. While Loomis & Ng (2009)'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.

This discussion encompasses the endpoint 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, angler days, etc. Future research from ecological economists ought to isolate the portion of angling-day value that comes from catching each marginal fish.

CHAPTER 4

HOW DIRTY IS 'DIRTY' GOLD?: VALUATION OF MINING'S SOCIAL AND ENVIRONMENTAL IMPACTS

The production of every unit of metal has *some* impact upon neighboring communities and ecosystems. Gold has recently been singled out as a particularly useless and harmful metal by the *No Dirty Gold* campaign (Earthworks, 2014). *No Dirty Gold* focuses on open pit mines operated by publicly-traded mining companies, but characterizes all gold mining as dirty (Earthworks, 2014). The following analysis parses this claim by employing environmental valuation to estimate the impact of producing one troy ounce of gold on laborers, communities, and the environment[16]. In other words, this analysis quantifies environmental damage from mining to ask, "How dirty is 'dirty' gold?" Open pit mines in Nevada and British Columbia are compared to sites with low levels of production in developing nations (small-scale mining) and sites where mining is conducted as a subsistence activity (artisanal mining). Data availability limits the scope of this analysis to thirteen sites with the goal of comparing the magnitude of impact from open pit gold production in well-regulated jurisdictions to artisanal and small-scale mining in areas with little rule of law[17].

The environmental valuation methodology of benefit transfer is used to apply existing environmental valuation literature estimates to the sites in question. The primary components of impact valued by this analysis are, methylmercury (MeHg) induced fetal IQ loss, MeHg intoxication, fatalities, and injuries. Site-specific information on mercury emission, gold production, MeHg intoxication, fatalities, and injuries related to gold mining at each site are provided by the *Global Mercury Project*, a master's thesis on artisanal and small-

[16] One troy ounce is about twice the amount of gold in a typical wedding ring.

[17] As noted in Section 1.1, artisanal and small-scale mining operations are entirely different from operations owned by publicly-traded mining companies. With respect to the incentives for social and environmental responsibility, these operations could not be more different. The comparison of socio-environmental impacts between these operations serves to reinforce these differences and may guide the prioritization of resources dedicated to reducing the impacts of gold mining globally.

scale mining in China (Gunson, 2004), evaluations of artisanal mining in Peru (Iramina *et al.*, 2014; Kuramoto, 2002; McMahon, 1999), and regulatory filings in British Columbia (BCME, 2014) and Nevada (NMCP, 2015).

4.1 Method and Data

This analysis uses estimates from the environmental valuation literature to assign monetary value to social and environmental impacts of mining. This method, known as benefit transfer, is the practice of applying the result of an existing environmental service valuation to a site with a similar environmental service and context (Bingham *et al.*, 1992; Loomis, 1992; Wilson & Hoehn, 2006). If there are two identical populations and contexts then the value of the environmental service should be the same for both sites - and this value should be transferable between them. The need for environmental 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 new locations (Wilson & 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).

Within the context of benefit transfer, there is the problem that environmental valuation studies (that value the same environmental service) seldom have the same magnitude. This problem is addressed by statistically evaluating the relevant environmental valuation literature to estimate the 'true' value that the literature is signaling (Boyle *et al.*, 1994; Carson *et al.*, 1996; Smith & Huang, 1995; Smith & Kaoru, 1990; Woodward & Wui, 2001). This 'study of studies' is known as meta-analysis. A meta-analysis function regresses the primary valuation study results on explanatory variables - such as site, study, and population characteristics of the primary valuations. 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-analytic benefit transfer (Kirchhoff, 1998; Rosenberger & Loomis, 2000; Shrestha & Loomis, 2001) which reduces prediction error when using benefit transfer for environmental valuation (Kaul *et al.*, 2013).

To estimate the value of social and environmental impacts from mining, this analysis draws from valuation literature studies that value MeHg induced fetal IQ loss, MeHg intoxication, injuries, and fatalities. Subsections 4.1.1, 4.1.2, 4.1.3, and 4.1.4 elaborate on the benefit transfer process that generates the value estimates in Table Table 4.1 for each of these components. It is acknowledged that the use of benefit transfer in the context of this analysis has substantial limitations. Specifically, many of the benefit transfer model values being transferred ought to be scaled up or down based on income. Given that the goal is a straight forward comparison of human suffering due to environmental pollution, deflation of values based on subsistence income does not seem appropriate. Therefore, benefit estimates are not scaled based on income.

Table 4.1: Summary of Benefit Transfer Model Values

Damages to Global Economy Due to IQ Loss from 1kg Hg Air Release	\$53,000
Settlement Value of a Case of MeHg Intoxication[1]	\$34,000
Cost of a Case of Myocardial Infarction Due to MeHg Intoxication[2]	\$69,000
Value of a Statistical Non-Fatal Injury[3]	\$62,000
Value of a Statistical Life[3]	\$8,500,000

Source: 1) Veiga *et al.* (2004), 2) Rice & Hammitt (2005), 3) Viscusi & Aldy (2003)

4.1.1 Valuation Studies for Mercury Emissions

A wide review of the literature suggests that IQ loss - due to the consumption of MeHg 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 MeHg, MeHg bio-accumulation, consumption of contaminated fish, ensuing impact on fetal cognitive functioning and loss of IQ (Hylander & Goodsite, 2006; Mergler *et al.*, 2007; Rice & Hammitt, 2005; Seigneur *et al.*, 2004; Spadaro & 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 focus on the 2% of elemental mercury (Hg^0) that becomes methylated, is ingested from fish fillets[18], and affects the fetal nervous system. Please see Section 1.2.2 for an extended discussion of mercury toxicity, methylation, absorption, and transfer pathways.

From this literature, Trasande *et al.* (2005), Rice & Hammitt (2005), and Spadaro & Rabl (2008) provide 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) is to estimate the economic costs of fetal neurodevelopmental impacts attributable to mercury emissions from American power plants. To achieve this, Trasande *et al.* (2005) combine an environmentally attributable fraction (EAF) model with national blood mercury prevalence data from the Centers for Disease Control and Prevention. Trasande *et al.* (2005) find 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. 5.8 μg per liter of cord blood serves as the neurotoxicity threshold for all estimates from this study.

Trasande *et al.* (2005) estimate 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 produce a range of estimated damages from \$2.9B to \$59.2B (2013\$). Within this range, Trasande *et al.* (2005) recommend a cord/maternal ratio of 1.7 and a logarithmic model that result in a recommended value of \$11.8B (2013\$) for damages from mercury deposited in the United States from global anthropogenic sources. Second, American anthro-

[18]While MeHg primarily concentrates in fish organs, MeHg 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 MeHg concentrations higher than the EPA human-health criterion (Scudder, 2010). Additionally, the mean MeHg concentration of skinless-fillets from the 59 fish sampled in basins with gold mining exceeded the EPA human-health criterion (Scudder, 2010, pp.12).

pogenic emissions are 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, American anthropogenic emissions from coal fired power plants provide an estimate 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 & Hammitt (2005) estimate the economic benefits of greater control of mercury emissions from coal-fired power plants in the United States. Mercury emissions reduction is assumed to have a linear and proportional decrease in MeHg concentrations in fish. Changes in deposition rates are based on regional deposition modeling from the EPA's analysis of the Clear Skies Initiative, under which power plants reduce mercury emissions from 49,000kg/year to either 26,000kg/year or 15,000kg/year. Human exposure to MeHg is modeled through commercial and non-commercial harvest of fish. Rice & Hammitt (2005) use dose-response functions from recent MeHg epidemiological studies and data on fish consumption from the FDA to estimate damages of mercury deposition. The estimates provided by Rice & 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 assumes a neurotoxicity threshold of maternal MeHg intake greater than 0.1 μg/kg of fish per day. The second estimate of \$26.9B assumes that there is no neurotoxicity threshold. Dividing these estimates by the assumed 124,300 kilograms deposited in the United States by all sources yields damages estimates of \$33,000 and \$208,000 - respectively.

Finally, Spadaro & Rabl (2008) use worldwide *average* MeHg doses from fish to calculate global damages from total (anthropogenic and non-anthropogenic) emissions. Spadaro & Rabl (2008) define a comprehensive transfer factor for ingestion of MeHg 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 MeHg 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 & Rabl (2008) scale damages based on income. For these reasons, the estimates from Spadaro & Rabl (2008) are not incorporated in the averages for the recommended value in Table Table 4.2.

The IQ loss estimates from Trasande *et al.* (2005) and Rice & Hammitt (2005) are normalized to reflect the same value per IQ point Spadaro & Rabl (2008). Then the estimates are inflated to 2013 dollars using the CPI. One of the most influential factors in the calculation of lost earnings from MeHg 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 Table 4.2 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 1kg 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 & 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 & 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.

4.1.2 Local Economic Impact of Methylmercury Intoxication

The neurological disorders associated with severe MeHg intoxication were first popularized in the 1950's by an extreme contamination event around Minamata Bay on the island of

Table 4.2: Valuation of Environmental Damage Due to 1kg 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 United States from Anthropogenic Sources (kg)	48,000
Damages to American Economy Due to IQ Loss in Annual Birth Cohort[a] (2013$)	$1.8B
Lost Lifetime Earnings Due to IQ Loss from 1kg Hg Air Release (2013$)	**$27,000***
Trasande *et al.* (2005) Cost of American Anthropogenic Hg Emissions Deposited in US	
Mercury Deposited in United States from Anthropogenic Sources (kg)	52,200
Damages to American Economy Due to IQ Loss in Annual Birth Cohort[a] (2013$)	$4.2B
Lost Lifetime Earnings Due to IQ Loss from 1kg Hg Air Release (2013$)	**$57,000***
Trasande *et al.* (2005) Cost of Global Anthropogenic Hg Emissions Deposited in US	
Mercury Deposited in United States from Anthropogenic Sources (kg)	87,000
Damages to American Economy Due to IQ Loss in Annual Birth Cohort[a] (2013$)	$11.8B
Lost Lifetime Earnings Due to IQ Loss from 1kg Hg Air Release (2013$)	**$97,000***
Rice & Hammitt (2005) Cost of Global Anthropogenic Hg Emissions Deposited in US	
Mercury Deposited in United States from Anthropogenic Sources (kg)	124,300
Damages to American Economy Due to IQ Loss in Annual Birth Cohort[a] (2013$)	$4.2B
Lost Lifetime Earnings Due to IQ Loss from 1kg Hg Air Release (2013$)	**$33,000***
Rice & Hammitt (2005) Cost Estimate *Without* Neurotoxicity Threshold	
Mercury Deposited in United States from Anthropogenic Sources (kg)	124,300
Damages to American Economy Due to IQ Loss in Annual Birth Cohort[b] (2013$)	$26.9B
Lost Lifetime Earnings Due to IQ Loss from 1kg Hg Air Release (2013$)	**$208,000**
Trasande *et al.* (2005) Cost Estimate with Alternative Linear Model	
Mercury Deposited in United States from Anthropogenic Sources (kg)	87,000
Economy Wide Damages Due to IQ Loss in Annual Birth Cohort[a] (2013$)	$44.5B
Lost Lifetime Earnings Due to IQ Loss from 1kg Hg Air Release** (2013$)	**$366,000****
Global Estimate from Spadaro & Rabl (2008)	
Lost Lifetime Earnings to Global Economy Due to IQ Loss in Annual Birth Cohort[a] (2013$)	**$1,818****
Lost Lifetime Earnings to Global Economy Due to IQ Loss in Annual Birth Cohort[b] (2013$)	**$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.

Kyushu, Japan. A chemical factory plant dumped waste containing MeHg chloride directly into Minamata Bay and Minamata River for nearly two decades. The residents, being in a coastal area, consumed fish and shellfish as a large part of their diet. By 1997, seventeen thousand people applied to be officially recognized as victims of severe MeHg intoxication - also known as Minamata Disease (Veiga *et al.*, 2004, pp.136). By 1999, 11,235 people had been paid $34,000 (2014$) each by the company as compensation (Veiga *et al.*, 2004, pp.136). This compensation figure is surprisingly low - especially given that 1,289 of these patients died by 1998.

Table Table 4.3 provides reference levels for total mercury in the human body. While these reference levels are for *total* mercury, they guide identification of severe MeHg intoxication (Ekino *et al.*, 2007; Park & Zheng, 2012; Veiga *et al.*, 2004), and will guide the application of MeHg intoxication valuation.

Table 4.3: Reference Levels for Total Mercury in the Human Body

Sample Type	**Total Mercury**	**Unit of Measurement**	**Level**	**Description**
Urine	5 μ g	per g creatinine	Alert	Cap when not occupationally exposed to MeHg
Urine	20 μ g	per g creatinine	Action	Removal from MeHg pollution source required
Urine	50 μ g	per g creatinine	Maximum	Max level recommended by WHO
Urine	100 μ g	per g creatinine	Critical	High probability of severe MeHg intoxication
Hair	1-2 μ g	per g of hair	Normal	Normal MeHg level in hair
Hair	5-10 μ g	per g of hair	Fetal Guideline	Upper guideline for pregnant women
Hair	20 μ g	per g of hair	Fetal Hazard	Hazardous effects on fetus are likely
Hair	50 μ g	per g of hair	General Hazard	High probability of severe MeHg intoxication
Blood	200 μ g	per L of blood	General Hazard	High probability of severe MeHg intoxication

Source: Ekino *et al.* (2007); Veiga *et al.* (2004)

Even though $34,000 represents a lower bound for the value of severe MeHg intoxication, it is the only relevant estimate provided by the literature. Therefore, the value of $34,000 will be applied to cases where total mercury exceeds 100 μ g per g creatinine in urine, 50 μ g per g of hair, or 200 μ g per L of blood. Future research could provide a more accurate figure by exploring healthcare costs, costs of pain and suffering, and lost productivity associated with severe MeHg intoxication.

Due to the dearth of such literature, it is worth mentioning a health impact valuation that was later called into question by the medical literature. In addition to the work that Rice & Hammitt (2005) conducted on fetal neurotoxicity, Rice & Hammitt (2005) also attempted a valuation of myocardial infarction - which is the death of heart tissue due to lack of oxygen from MeHg poisoning. Rice & Hammitt (2005) cite a portion of the medical literature that identifies a link between MeHg intoxication and myocardial infarction. To value this link, Rice & Hammitt (2005) analyze valuation literature regarding myocardial infarction health costs and lost productivity due to myocardial infarction. They estimate a value of $69,000 (2014$) per case.

However, upon publication of Rice & Hammitt (2005), scientists from the Brookhaven National Laboratory published a review of the medical literature regarding the proposed connection between myocardial infarction and MeHg intoxication (Lipfert & Sullivan, 2005). Lipfert & Sullivan (2005) show that the medical literature concludes that such a link does not exist and accuse Rice & Hammitt (2005) of cherry-picking from this literature to bolster their valuation. The myocardial valuation from Rice & Hammitt (2005) is worth mentioning in this analysis, in case the medical literature changes its mind.

4.1.3 Value of a Statistical Injury

As one might expect, artisanal and small-scale mining are risky. Miners are injured when underground rock or open-pit slopes do not stay where they are supposed to. To get an accurate picture of the full impact of these activities, health care costs and lost productivity from injuries must be added to the previous environmental damage calculations. To estimate this impact, this analysis employs a meta-analysis of thirty-nine studies of the implicit value of a statistical injury (Viscusi & Aldy, 2003). The primary studies analyzed by Viscusi & Aldy (2003) evaluate the risk premium that workers demand for bearing nonfatal job risks - such as serious bodily injury. Thirty-one of these studies are conducted within the United States. The remaining nine are conducted in Canada, the UK, India, and Taiwan. The majority of studies produce estimates that range from $20,000 to $70,000 (2000$). This analysis employs the average of this range for the value of a statistical injury of $45,000 (In year 2000 dollars (2000$)) - or $62,000 (In year 2014 dollars (2014$)).

4.1.4 Value of a Statistical Life

Further, many artisanal and small-scale miners are killed in their occupation. A value for their untimely death is required to fully assess the impact of artisanal and small-scale mining. Again, a meta-analysis by Viscusi & Aldy (2003) is employed. For this portion, the primary studies analyzed in Viscusi & Aldy (2003) evaluate the implicit value of a life. Using differing probabilities of mortality and the compensation required, a figure can be calculated for the value that laborers implicitly set for their own lives. Viscusi & Aldy (2003) analyze sixty studies of mortality risk premiums from ten countries and estimate the value of a statistical life to be $8,500,000 (2014$). This may seem like a repulsive practice, but it is commonplace (EPA, 2010; Viscusi & Aldy, 2003). It is used to justify everything from transportation improvements to the removal of lead from common products. VSL results are surprisingly similar to lifetime earnings calculations.

4.1.5 Method Conclusion and Benefit Transfer Model Template

In conclusion, the benefit transfer model - derived from the literature above - consists of four components; IQ loss due to airborne emission of one kilogram of mercury, MeHg intoxication, non-fatal injury, and fatality. The value of $53,000 for IQ loss due to emission of one kilogram of mercury represents the average of three estimates from Trasande *et al.* (2005) and one estimate from Rice & Hammitt (2005) - each of which incorporate neurotoxicity thresholds. Where airborne mercury emissions are available, the value of $53,000 for IQ loss is applied directly. Where mercury emissions are on land or in water (instead of airborne), Lacerda (1997) estimates that 68%-82% of mercury emissions eventually become airborne. Therefore, this analysis applies the average of this range, 75%, to translate estimates of mercury released into the environment into airborne mercury emissions.

The value of \$34,000 for MeHg intoxication is derived from the Minamata settlement and is applied to individuals with total mercury that exceeds 100 μ g per g creatinine in urine, 50 μ g per g of hair, or 200 μ g per L of blood. For the value of a non-fatal injury, \$62,000 is recommended by the statistical injury valuation meta-analysis conducted by Viscusi & Aldy (2003). Similarly, the value of \$8.5 million is recommended by Viscusi & Aldy (2003) as the result of a meta-analysis of studies on the value of a statistical life. Table Table 4.4 illustrates the template by which these values are applied to the thirteen sites in this analysis.

Table 4.4: Template for Valuation of Environmental Damage per Ounce

Damages Related to Fatalities and Injuries	
Value of a Casualty[1]	\$8.5M
Number of Casualties[2]	Insert Site Value
Total Liability Due to Casualties	**\$XXXM**
Value of Non-Fatal Injury[1]	\$62,000
Number of Non-Fatal Injuries[2]	Insert Site Value
Total Liability Due to Injuries	**\$XXX,000**
Global Damages from Airborne Mercury Emissions Causing IQ Loss	
Global Loss in Lifetime Earnings Due to 1kg of Mercury Emissions	\$53,000
Region's Portion of Airborn Mercury Emissions (kg per year)[2]	Insert Site Value
Total Liability Due to IQ Loss	**\$XXXM**
Local Damages from Methylmercury Intoxication	
Settlement Value of a Case of MeHg Intoxication[3]	\$34,000
Number of Cases of MeHg Intoxication[2a]	Insert Site Value
Total Liability Due to MeHg Intoxication	**\$XXXM**
Overall Total Liability	**\$XXXM**
Region's Production of Gold (troy ounces per year)[2]	Insert Site Value
Environmental Damage per Troy Ounce	**\$XXX**

Source: 1) Viscusi & Aldy (2003), 2) Study with Site Data, 3) Veiga & Gunson (2004)
Note: a) Assumes those with total mercury exceeds 100 μ g per g creatinine in urine, 50 μ g per g of hair, or 200 μ g per L of blood suffer Minamata disease.

4.2 Data

The dataset required for this analysis is comprised of site-specific characteristics on the socio-environmental impact of gold mining that map into Table Table 4.4. Site data for artisanal and small-scale mining sites in Venezuela (Veiga *et al.*, 2005; Veiga & Gunson, 2004), Brazil (Rodrigues-Filho *et al.*, 2004), Indonesia (Darmutji, 2003; Veiga, 2003), Zimbabwe (Shoko & Veiga, 2004), and Tanzania (Appleton *et al.*, 2004; Tesha, 2003) come from the *Global Mercury Project* (Veiga *et al.*, 2004). *The Global Mercury Project* was a joint venture between the World Bank, the United Nations Environment Program, and the United Nations Development Program conducted from 2002 through 2007. The primary goal was to overcome barriers to the adoption of mercury pollution prevention measures in artisanal and small-scale mining. However, site data was also collected on Hg^0 (elemental mercury) emissions, gold production, total mercury levels in sampled populations, injuries, and fatalities.

Site data for artisanal and small-scale mining in China are pulled from Gunson (2004), which is a mining engineering master's thesis on mining techniques in China. The focus of Gunson (2004) is the amount of Hg^0 released into the environment. No data were collected on total mercury levels, injuries, or fatalities.

Site data for artisanal and small-scale mining sites in Peru come from three evaluations of mining in Peru (Iramina *et al.*, 2014; Kuramoto, 2002; McMahon, 1999). McMahon (1999) is a technical paper published by the World Bank. The main focus is to compare the environmental impacts of artisanal, small, and medium scale mining in Bolivia, Chile, and Peru. In the process, McMahon (1999)provide estimates of Hg^0 emissions from underground artisanal mining in the Ica-Arequipa region of Peru. Kuramoto (2002) is a technical paper published by the International Institute for Environment and Development. Kuramoto (2002) provides data on gold production in the Ica-Arequipa region as well as Hg^0 emissions in Ica-Arequipa and Madre de Dios. Finally, Iramina *et al.* (2014) provides data on fatal accidents in Peru. No data were available in Peru to estimate MeHg intoxication or injuries.

Site-specific data regarding the Mt. Polley tailings storage facility spill were pulled from public sources where available. For example, a drinking water ban was imposed for nine days and annual tourism revenues from fishing in Quesnel Lake and River were estimated to be approximately $6 million, and the salmon run was estimated to be approximately 1 million salmon. In the absence of reliable estimates, tourism revenue loss, fish catch, and fish losses were estimated. The goal is to create a worst-case scenario of site specific characteristics that could be compared to the everyday operations of artisanal miners[19]. Finally, the site specific Hg^0 emissions for gold ore smelters in Nevada were collected from the Nevada Mercury Control Program Annual Emissions Report.

4.3 Results

Table Table 4.5 provides an overview of the benefit transfer valuation results for the thirteen sites. The first two estimates represent open pit mines in Nevada before the state initiated a mercury control program. The next two estimates represent the same mines, 8 years later, after the mercury control program successfully reduced mercury emissions from gold mines. These four estimates are in chronological order to show the damage reduction caused by the mercury control program. The fifth estimate represents the Mt. Polley tailings storage facility failure from 2014. The final eight estimates are small-scale and artisanal sites ranked in ascending order by damage estimate.

Mercury exists naturally in the ore at the two Nevada sites. This mercury was being emitted into the atmosphere during the smelting process. In 2006, the state of Nevada imposed mercury controls on smelting operations and mercury emissions were cut drastically. Mt. Polley, another open pit mine owned by a publicly-traded company, represents a significant mine design failure. The Mt. Polley tailings spill primarily affected drinking water, salmon, and salmon habitat. The estimate of damage per ounce for the publicly-traded company open pits range from less than$1 to $35.

The remaining ten sites represent artisanal and small-scale mining sites with enough information to conduct an impact valuation. The six small-scale and artisanal sites where miners concentrate the ore before applying elemental mercury (Hg^0) for amalgamation have damages estimates of $1,100 to $4,500. These estimates vary based on the level of gold ore concentration, the amount of Hg^0 applied to the concentrate, the gold ore grade, and the

[19]This exercise is akin to financial analysis conducted immediately after the spill with the goal of estimating potential environmental liability. It does not appear that the province of British Columbia, or the Canadian federal government, will not bring a natural resource damages case against Imperial.

level of gold production. For example, when the gold within the ore is concentrated more heavily before the Hg^0 is added for amalgamation, less mercury will be lost and the damage estimate will be lower. Similarly, if excess Hg^0 is captured and re-used, there will be less mercury loss and the damage estimate will be lower. Finally, the higher the gold grade and gold production, the more diluted the damage estimates will become.

It is no coincidence that the remaining four sites, where Hg^0 is applied to the whole ore during amalgamation, have the highest impact values (\$6,400 to \$23,000). Applying Hg^0 to the whole ore causes increased Hg^0 loss to the environment, which drives up the impact value. However, El Callao, Talawaan, and Ica-Arequipa also have fatalities that are factored into the damage estimate. In addition to whole ore amalgamation these damage estimates are inflated by the value of statistical lives that are lost at these sites. The damage estimate at El Callao is the lowest of this group because the site has high ore grades that increase gold production and diminish the per ounce damage estimate. Finally, the damage estimate for China is the highest because of a whole ore amalgamation and milling process that loses exceptional amounts of Hg^0 per ounce of gold produced.

4.3.1 Nevada

Gold Quarry is an open pit gold mine, owned by a publicly-traded mining company - Newmont Mining. It is located in the state of Nevada and has few neighbors. There were no fatalities, injuries, or cases of MeHg intoxication resulting from the Gold Quarry mine in 2005 or 2013. The primary impact to the environment resulted from airborne mercury emissions due to smelting of Gold Quarry ore. In 2005, 329.4 kg were emitted. This number was reduced to 48.3 kg in 2013. Table Table 4.6 shows that the environmental damage per ounce was \$35 and \$5 for the years 2005 and 2013 respectively. These values are on the high end for operations owned by publicly-traded companies in this analysis.

The Phoenix open pit gold mine is also owned by Newmont Mining in the state of Nevada. Similar to Gold Quarry, the main impact from the Phoenix mine is the mercury released during the smelting process. Mercury emissions at the Phoenix mine were 1.2kg in 2005

Table 4.5: Damage Estimate Overview for the Study Sites

Site Name	Mining Type	Hg^0 Applied to Whole Ore?	Annual Production (Ounces)	Damage Estimate ($ per oz)
Gold Quarry, NV 2005	Publicly-Traded Company Open Pit	No	500,000	$35
Pheonix, NV 2005	Publicly-Traded Company Open Pit	No	220,000	<$1
Gold Quarry, NV 2013	Publicly-Traded Company Open Pit	No	500,000	$5
Pheonix, NV 2013	Publicly-Traded Company Open Pit	No	220,000	<$1
Mt. Polley Mine, BC	Publicly-Traded Company Open Pit	No		$8
Kadoma, Zimbabwe	Small-Scale Underground Mining	No	140,000	$1,100
Creporizinho, Brazil	Artisanal Underground Mining	No	19,300	$2,500
Galangan, Indonesia	Artisanal Mining with Water Jets	No	20,100	$3,000
Sao Chico, Brazil	Artisanal Mining with Raft Dredges	No	390	$3,300
Rawamagasa, Tanzania	Small-Scale Underground Mining	No	400	$4,100
Madre de Dios, Peru	Small-Scale Mining	No	354,000	$4,500
El Callao, Venezuela	Small-Scale Undergrond Mining	Yes	48,000	$6,400
Talawaan, Indonesia	Artisanal Underground Mining	Yes	177,000	$12,000
Ica-Arequipa, Peru	Artisanal Underground Mining	Yes	36,000	$19,000
China	Artisanal and Small-Scale Mining	Yes	400,000	$23,000

The third column refers to whether, or not, elemental mercury (Hg^0) is applied to the whole ore for amalgamation, or just a concentrate of the whole ore.

Table 4.6: 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 1kg of Mercury Emissions (2014$)	$53,000
Gold Quarry's Airborne Mercury Emissions in 2005 (kg)[1]	329.4
Gold Quarry's Portion of Damages Due to IQ Loss in 2005 (2014$)	$17,500,000
Gold Quarry's Gold Equivalent Production in 2005 (Troy Oz)	500,000
Environmental Damage per Ounce (2005)	**$35**
Gold Quarry Mine 2013	
Gold Quarry's Airborne Mercury Emissions in 2013 (kg)[1]	48.3
Gold Quarry's Portion of Damages Due to IQ Loss in 2013 (2014$)	$2,500,000
Gold Quarry's Gold Equivalent Production in 2013 (Troy Oz)[2]	500,000
Environmental Damage per Ounce (2013)	**$5**

Source: 1) NMCP Annual Emissions Reporting 2) Nevada Division of Minerals and 0.3 kgs in 2013. Table Table 4.7 shows that the environmental damage per ounce was \$0.28 and \$0.07 for the years 2005 and 2013 respectively. These values represent the lowest environmental impact in this analysis.

Table 4.7: Valuation of Environmental Damage per Ounce from the Phoenix Gold Mine in Nevada

Environmental Damage per Ounce Calculation	
Pheonix Mine 2005	
Loss in American Lifetime Earnings Due to 1kg of Mercury Emissions (2014\$)	\$53,000
Phoenix's Airborne Mercury Emissions in 2005 (kg)[1]	1.2
Phoenix's Portion of Damages Due to IQ Loss in 2005 (2014\$)	\$61,000
Phoenix's Gold Equivalent Production in 2005 (Troy Oz)	220,000
Environmental Damage per Ounce (2005)	**\$0.28**
Pheonix Mine 2013	
Phoenix's Airborne Mercury Emissions in 2013 (kg)[1]	0.3
Phoenix's Portion of Damages Due to IQ Loss in 2013 (2014\$)	\$14,000
Phoenix's Gold Equivalent Production in 2013 (Troy Oz)[2]	220,000
Environmental Damage per Ounce (2013)	**\$0.07**
Source: 1) NMCP Annual Emissions Reporting, 2) Nevada Division of Minerals	

4.3.2 Mount Polley

The Mt. Polley open pit gold mine is owned by a publicly-traded mining company - Imperial Metals - and resides in the province of British Columbia. Quesnel Lake and the small town of Likely, British Columbia are the mine's neighbors.

In the early hours of August 4th, 2014 twenty-five million cubic meters of mine tailings, impoundment water, and dam material roared through a section of forest and into a salmon run of about one million salmon. This disaster was caused by the failure of the tailings storage facility. A drinking water ban for nearby residents was put in place due to heavy metals and other toxins from the mine waste. The ban lasted nine days and affected approximately 150 households. Quesnel Lake and Quesnel River, which were the main repositories of the spilled material, suffered abrupt drops in their usual tourism revenue from fishing - which is estimated to be \$6,000,000. This loss of revenue provides a straightforward valuation of immediate fishing losses due to human perception of the spill. In the absence of reliable estimates, this analysis assumes revenue was cut by 50% in 2014 and that there will be a 10% recovery every year.

There does not appear to be any immediate fish kill. The sheer volume of water diluted the waste to a level that did not inflict immediate damage. The main concerns are metal toxicity in salmon and the degradation of salmon spawning habitat. This analysis assumes that their is a 10% loss in salmon population for the next ten years and that 10% of the lost salmon would have been caught. Value estimates for the loss of salmon catch and for the drinking water ban are pulled from Section 2.1.1 and Section 2.1.2, respectively. The estimated environmental damage from the spill is in the range of $11,700,000. Dividing this by the 1,390,753 ounces of gold equivalent produced over time, the Mount Polley ounce damage is $8 - see Table Table 4.8. While this value is on the high end of operations owned by publicly-traded companies, it is miniscule in comparison to the small-scale and artisanal mining sites.

4.3.3 Kadoma, Zimbabwe

The Global Mercury Project investigation site of Kadoma is dominated by artisanal gold panning. While some shallow shafts are dug, 75% of the 20,000 artisanal miners in Kadoma pan for gold (Shoko & Veiga, 2004). Ore is ground in stamp mills and Hg^0 is applied to the whole ore during amalgamation on copper plates (Shoko & Veiga, 2004). Less than 30% of gold is recovered during amalgamation. Once the miners have left, the millers put the tailings in cyanidation vats to extract the remaining gold.

No fatalities or injuries are recorded at the site, even though it is mentioned that underground shafts often collapse during the rainy season. Similarly, no cases of MeHg intoxication are recorded. Hg^0 release is the only impact that can be valued. Table Table 4.9 indicates that the damage per ounce at Kadoma is $1,100. This value is low for small-scale and artisanal mining sites, but is much higher than the damage values for operations owned by publicly-traded mining companies. Future research at Kadoma ought to quantify fatalities, injuries, and cases of MeHg intoxication related to mining. Further, valuation of sexually

Table 4.8: Valuation of Environmental Damage per Ounce from the Mount Polley Tailings Dam Failure in British Columbia

Environmental Damage per Ounce Calculation	
Damages for Drinking Water Ban	
Value of Damage per Household per Day	$158
Households Affected	150
Days	9
Total Liability	$213,300
Damages to Tourism	
Estimated Tourism Revenue per Year	$6,000,000
Assume revenue cut 50% in 2014	$3,000,000
Assume 10% Recovery Each Year Thereafter	$6,000,000
Total Liability	$9,000,000
Damages to Salmon Population	
Estimated Chinook Salmon Population per Year	1,000
Estimated Chinook Salmon Population Loss for Next Ten Years	10%
Estimated Portion of Chinook Salmon Population Loss that Would Have Been Caught	10%
Total Chinook Salmon Catch Loss	100
Average Value per Fish of Chinook Salmon Catch Loss	$40
Total Liability from Chinook Salmon Loss	$4,000
Estimated Sockeye Salmon Population per Year	1,500,000
Estimated Sockeye Salmon Population Loss for Next Ten Years	10%
Estimated Portion of Sockeye Salmon Population Loss that Would Have Been Caught	10%
Total Sockeye Salmon Catch Loss	150,000
Average Value per Fish of Sockeye Salmon Catch Loss	$16.50
Total Liability from Sockeye Salmon Loss	$2,475,000
Total Liability from Salmon Loss	$2,479,000
Total Liability from Spill	**$11,692,300**
Cumulative Production of Gold Equivalent Troy Ounces	1,390,753
Environmental Liability per Ounce	**$8**

transmitted diseases from prostitution, decreased productivity from poor school attendance by child miner's, and alcoholism would provide interesting components for the valuation of social impacts from artisanal mining.

Table 4.9: Valuation of Environmental Damage per Ounce from Artisanal Mining in Kadoma, Zimbabwe

Kadoma, Zimbabwe	
Global Damages from Airborne Mercury Emissions Causing IQ Loss	
Global Loss in Lifetime Earnings Due to 1kg of Mercury Emissions	\$53,000
Kadoma's Portion of Airborn Mercury Emissions (kg per year)[1]	3,000
Overall Total Liability	**\$159M**
Region's Production of Gold (troy ounces per year)[1]	140,000
Environmental Damage per Troy Ounce	**\$1,100**
Source: 1) Shoko & Veiga (2004)	

4.3.4 Creporizinho, Brazil

The Global Mercury Project site near Creporizinho consists of 300 artisanal miners working underground mine shafts. Once the gold ore is extracted, it is taken to a hammer mill where the ore is ground. The ground ore is washed over a sluice box with a carpet on the bottom to catch the gold concentrate. Hg^0 is added to the concentrate to produce a gold/Hg^0 amalgam. This amalgam is placed in a piece of cloth, which is twisted to remove excess Hg^0. Finally, a blow torch is used to vaporize the Hg^0 in the amalgam. The result is a gold bead with approximately 10% Hg^0 remaining.

No information is provided for fatalities, injuries, or cases of MeHg intoxication at Creporizinho so Hg^0 emissions serve as the main impact to value. Table Table 4.10 indicates that the damage per ounce at the study site near Creporizinho is \$2,500. This relatively low value for small-scale and artisanal mining sites is primarily due to the concentration of gold ore before the introduction of Hg^0.

Table 4.10: Valuation of Environmental Damage per Ounce from Artisanal Mining in the Rainforest of Creporizinho, Brazil

Creporizinho, Brazil	
Global Damages from Artisinal Airborne Emissions Causing IQ Loss	
Global Loss in Lifetime Earnings Due to 1kg of Mercury Emissions	\$53,000
Region's Portion of Airborne Mercury Emissions (kg per year)[1]	900
Overall Total Liability	**\$47.7M**
Region's Production of Gold (troy ounces per year)	19,290
Environmental Damage per Troy Ounce	**\$2,500**
Source: 1) Rodrigues-Filho *et al.* (2004)	

4.3.5 Galangan, Indonesia

The Global Mercury Project investigation site of Galangan consists of artisanal mining with water jets. The water is sprayed to wash gold-bearing sand over a sluice box. The concentrate at the bottom of the sluice box is then panned to remove additional waste material. The panned concentrate is combined with Hg^0 and stirred by hand in a bucket. Miners do not fire the amalgam. Instead, they sell it to nearby gold shops.

In Galangan, Hg^0 emission is the only impact that can be valued - see Table Table 4.11. Given the Hg^0 emissions and gold production for the area, the impact per ounce is \$3,000 - which is still on the low end of the artisanal and small-scale mining sites. Concentration of gold ore before application of Hg^0 helps to lower the environmental impact.

Table 4.11: Valuation of Environmental Damage per Ounce from Artisanal Mining in the Rainforest of Galangan, Indonesia

Galangan, Indonesia	
Global Damages from Artisinal Airborne Emissions Causing IQ Loss	
Global Loss in Lifetime Earnings Due to 1kg of Mercury Emissions	\$53,000
Portion of Airborne Mercury Emissions (kg per year)[1]	1,125
Overall Total Liability	**\$59.6M**
Region's Production of Gold (troy ounces per year)[1]	20,100
Environmental Damage per Troy Ounce	**\$3,000**
Source: 1) Darmutji (2003)	

4.3.6 Sao Chico, Brazil

In the Global Mercury Project investigation site of Sao Chico, artisanal miners bring sediment and gold ore up from the riverbed through a dredge on a river raft. The water and sediment is run over a sluice box with carpet on the bottom to catch fine gold. At the end of each shift, the concentrate from the bottom of the sluice box is shaken from the carpet into an on-board blender. A heavy dose of Hg^0 is added to ensure maximum gold capture. The amalgam is retrieved and cooked to vaporize the Hg^0 (Rodrigues-Filho *et al.*, 2004). Hg^0 that does not amalgamate is thrown overboard and creates hot-spots in the river.

The damage figure, from Table Table 4.12, for Sao Chico is \$3,300, which is in the lower end for artisanal sites. Rodrigues-Filho *et al.* (2004) conduct a thorough health assessment and do not find individuals with total mercury values above the 100 μg per g creatinine threshold. However, there are many individuals with values between 20 and 100. Future research could be conducted to determine the health impacts of these lower values and their costs.

Table 4.12: Valuation of Environmental Damage per Ounce from Small-Scale Placer Mining in Sao Chico, Brazil

Sao Chico, Brazil	
Global Damages from Artisinal Airborne Emissions Causing IQ Loss	
Global Loss in Lifetime Earnings Due to 1kg of Mercury Emissions	\$53,000
Region's Portion of Airborne Mercury Emissions (kg per year[1]	24
Overall Total Liability	**\$1.3M**
Region's Production of Gold (troy ounces per year)[1]	390
Environmental Damage per Troy Ounce	**\$3,300**
Source: 1) Rodrigues-Filho *et al.* (2004)	

4.3.7 Rawamagasa, Tanzania

The Global Mercury Project site of Rwamagasa is an area with small-scale underground mining. Tunnels are narrow, steep, and curved. Ore is crushed with sledge hammers, put in diesel-powered steel ball mill, and then run through a sluice box. The concentrate from the sluice box is then pressed with Hg^0 by hand for approximately two hours. The resulting amalgam is fired over charcoal stoves or bonfires, which heat the amalgam poorly. As a result, the final gold bead often contains upwards of 20% Hg^0.

The damage figure, from Table Table 4.13, for Rwamagasa is $4,100, which is mid-range for artisanal sites that concentrate ore before applying Hg^0. The loss of Hg^0 during the hand-pressing stage is likely responsible for this higher value.

Table 4.13: Valuation of Environmental Damage per Ounce from Small-Scale Mining in Rwamagasa, Tanzania

Rwamagasa, Tanzania	
Global Damages from Airborne Mercury Emissions Causing IQ Loss	
Global Loss in Lifetime Earnings Due to 1kg of Mercury Emissions	$53,000
Rwamagasa's Portion of Airborne Mercury Emissions (kg per year)[1]	30
Total Liability Due to IQ Loss	**$2M**
Region's Production of Gold (troy ounces per year)[1]	400
Environmental Damage per Troy Ounce	**$4,100**
Source: 1) Tesha (2003)	

4.3.8 Madre de Dios, Peru

In Peru, Amazonian headwaters have eroded gold deposits and disseminated the gold in the meandering stream beds of the Madre de Dios region. To access this disseminated gold, small-scale miners slash and burn the overlying forest. Then front end loading equipment is used to grab the gravely gold ore and place it on an iron grid washing platform. The nearest stream is diverted to wash the lighter material off the platform as the lime and ore fall into a sluice box below. The concentrate is mixed with copious amounts of Hg^0 in a can and shaken vigorously. The resulting amalgam is heated in the kitchen of the miners' makeshift settlements to produce a gold bead.

Information was only available for Hg^0 emissions and the damage per ounce is $4,500 - see Table Table 4.14. The damage figure for Madre de Dios is in the upper range for artisanal sites that concentrate ore before applying Hg^0. The use of excessive Hg^0 is likely the cause of this higher estimate. Future valuation research at this site could focus on the social problems associated with small-scale mining in the region. The expansion of so-called 'canteens' in mining boom-towns has generated adolescent prostitution, street violence, family violence and robbery Kuramoto (2002). Police have insufficient resources to control these boom-towns and rarely enter them. Assault and homicide are frequent events.

Table 4.14: Valuation of Environmental Damage per Ounce from Small-Scale Mining in Madre De Dios, Peru

Madre de Dios, Peru	
Global Damages from Airborne Mercury Emissions Causing IQ Loss	
Global Loss in Lifetime Earnings Due to 1kg of Mercury Emissions	$53,000
Region's Portion of Airborn Mercury Emissions (kg per year)[1]	30,000
Overall Total Liability	**$1.59B**
Region's Production of Gold (troy ounces per year)[1]	354,000
Environmental Damage per Troy Ounce	**$4,500**
Source: 1) Kuramoto (2002)	

4.3.9 El Callao, Venezuela

The Global Mercury Project site of El Callao - in Bolivar State, Venezuela - contains a block of legal mining concessions that are rented to small-scale mining groups. High gold grades (12 to 20 grams of gold per tonne) result in high annual production of almost 50,000 ounces per year. Miners pull ore from shafts approximately 60 meters deep and transport the ore via truck to one of the thirty processing centers - known as molinos (Veiga *et al.*, 2005). The molinos crush and grind the ore. Hg^0 is then applied to the whole ground ore. Amalgamation takes place on copper-amalgamating plates, which lose large amounts of Hg^0 (Veiga *et al.*, 2005).

El Callao has some of the highest levels of Hg^0 intoxication documented by the Global Mercury Project Veiga *et al.* (2004). Veiga *et al.* (2005) explain that 260 miners and millers have a level of μg per g creatinine in urine over the threshold for MeHg intoxication. Because mercury intoxication via inhalation has similar health impacts as MeHg intoxication, these 260 miners and millers are included in the valuation under MeHg intoxication. Further, sixteen millers have levels of mercury intoxication that are so high, they are dying from the mercury they inhale (1,221-3,260 μg per g creatinine). These millers are included in the valuation as fatalities.

The environmental damage from a ounce of gold produced in El Callao is \$6,400 - see Table Table 4.15. This value is the lowest of sites where Hg^0 is applied to the whole ore because of the exceptional grade of gold in the rock. This high grade increases the production, which lowers the per ounce damage.

Table 4.15: Valuation of Environmental Damage per Ounce from Small-Scale Mining in El Vallao, Venezuela

El Callao, Venezuela	
Damages from Extreme Mercury Inhalation	
Value of a Casualty[1]	\$8.5M
Number of Casualties[2]	16
Total Liability Due to Casualties	**\$139M**
Global Damages from Airborne Mercury Emissions Causing IQ Loss	
Global Loss in Lifetime Earnings Due to 1kg of Mercury Emissions	\$53,000
Region's Portion of Airborn Mercury Emissions (kg per year)[2]	3,000
Total Liability Due to IQ Loss	**\$159M**
Local Damages from Methylmercury Intoxication	
Settlement Value of a Case of MeHg Intoxication[3]	\$34,000
Number of Cases of MeHg Intoxication[2a]	260
Total Liability Due to MeHg Intoxication	**\$8.8M**
Overall Total Liability	**\$306.8M**
Region's Production of Gold (troy ounces per year)[2]	48,000
Environmental Damage per Troy Ounce	**\$6,400**

Source: 1) Viscusi & Aldy (2003), 2) Veiga *et al.* (2005), 3) Veiga & Gunson (2004)
Note: a) Assumes the population in which total mercury exceeds 100 μ g per g creatinine in urine, 50 μ g per g of hair, or 200 μ g per L of blood suffer Minamata disease.

4.3.10 Talawaan, Indonesia

Informal mining activity in the Global Mercury Project site of Talawaan has taken place where formal mining companies have explored and then vacated due to disputes with the Indonesian government. Vertical shafts are dug with broad hoes up to thirty meters deep. Processing of the ore is done via a ball mill where Hg^0 is introduced to the whole ore.

Tunnel and slope failures occur three to five times a year, killing and maiming informal miners (Darmutji, 2003). This analysis assumes that one out of five of these failures result in a fatality and four out of five result in non-fatal injuries. The environmental damage per ounce produced in Talawaan is $12,000 - see Table Table 4.16. The magnitude of this value is high due to the injuries, fatalities, and introduction of Hg^0 to the whole ore - which increases the amount released into the environment.

Table 4.16: Valuation of Environmental Damage per Ounce from Underground Artisanal Mining in Talawaan, Indonesia

Talawaan, Indonesia	
Damages from Underground Rock Failure	
Value of a Casualty[1]	$8.5M
Average Annual Number of Casualties[2]	1
Total Liability Due to Casualties	**$8.5M**
Value of Non-Fatal Injury[1]	$62,000
Number of Non-Fatal Injuries[2]	4
Total Liability Due to Injuries	**$248,000**
Global Damages from Airborne Mercury Emissions Causing IQ Loss	
Global Loss in Lifetime Earnings Due to 1kg of Mercury Emissions	$53,000
Portion of Airborn Mercury Emissions (kg per year)[2]	41,250
Total Liability Due to IQ Loss	**$2.2B**
Overall Total Liability	**$2.2B**
Region's Production of Gold (troy ounces per year)[2]	177,000
Environmental Damage per Troy Ounce	**$12,000**
Source: 1) Viscusi & Aldy (2003), 2) Darmutji (2003)	

4.3.11 Ica-Arequipa Region, Peru

Artisanal underground mining in the Ica-Arequipa region is characterized by the use of a manual drill and dynamite to remove the rock adjacent to the gold vein. After the blast, the vein remains in place and the miner eases high grade ore from it onto a cloth on the ground (McMahon, 1999). The miner selects the pieces that look to be the highest grade and carries them out in a basket. Once the artisanal miner has brought his ore to the surface, it is taken to a quimbalete, which is a huge mortar and pestle that grinds the ore in the presence of Hg^0.

Iramina *et al.* (2014) estimate the annual number of deaths, due to rock failure, at 1.25 between the years 2000 and 2010. Given the region's fatalities, Hg^0 emissions, and production, the damage per ounce is \$19,000 - see Table Table 4.17. Fatalities, addition of Hg^0 to the whole ore, and low production numbers make this estimate the second highest of the analysis. Given the high levels of Hg^0 released into the surrounding environment, a health study of the surrounding population warrants future research.

Table 4.17: Valuation of Environmental Damage per Ounce from Underground Artisanal Mining in Ica-Arequipa, Peru

Ica-Arequipa, Peru	
Local Damages from Underground Rock Failure	
Value of a Casualty[1]	\$8.5M
Average Annual Number of Casualties[2]	1.25
Total Liability Due to Casualties	**\$10.6M**
Global Damages from Airborne Mercury Emissions Causing IQ Loss	
Global Loss in Lifetime Earnings Due to 1kg of Mercury Emissions	\$53,000
Region's Portion of Airborn Mercury Emissions (kg per year)[3]	12,600
Rudimentary Mining's Portion of Damages Due to IQ Loss	**\$667.8M**
Overall Total Liability	**\$678.4M**
Region's Production of Gold (troy ounces per year)[4]	36,000
Environmental Damage per Ounce	**\$19,000**
Source: 1) Viscusi & Aldy (2003), 2) Iramina *et al.* (2014), 3) McMahon (1999), 4) Kuramoto (2002)	

4.3.12 Aggregated Artisanal and Small-Scale Mining in China

In contrast to the sites previously evaluated, Section 4.3.12 evaluates environmental damage in an aggregated context. While this approach is less precise, it is important to highlight the volume of Hg^0 emissions coming from China. Using estimates of gold production, production methods, and the ratio of Hg^0 loss associated with each production method, Gunson (2004) estimates that 240,000 kilograms of Hg^0 are released by artisanal and small-scale miners in China every year. Gunson (2004) also estimates that 70% of this release is due the use of Muller Mills, which apply Hg^0 to the whole ore while it is being ground. The \$23,000 estimate, from Table Table 4.18, of damage per ounce of gold production does not include any fatalities, injuries, or MeHg intoxication. Instead, it is the highest estimate due to the fact that Muller Mills apply Hg^0 to *all* material that is mined. Gunson (2004) argues that the use of gravity concentration, prior to application of Hg^0, would greatly reduce Hg^0 emissions in China.

Table 4.18: Valuation of Environmental Damage per Ounce from Artisanal and Small-Scale Mining in China

All of China	
Global Damages from Artisinal Airborne Emissions Causing IQ Loss	
Global Loss in Lifetime Earnings Due to 1kg of Mercury Emissions	\$53,000
Portion of Airborne Mercury Emissions (kg per year)[1]	170,800
Rudimentary Mining's Portion of Damages Due to IQ Loss	\$9.05B
Region's Production of Gold (troy ounces per year)	400,000
Environmental Damage per Troy Ounce	**\$23,000**
Source: 1) Gunson (2004)	

4.4 Discussion, Implications, and Conclusion

The primary results of this analysis are as follows: 1) environmental damage values in Nevada dropped by approximately 75% after a mercury emissions control program was implemented by the state, 2) the tailings storage facility failure at Mt. Polley produced a damage value of approximately \$8 per ounce, 3) artisanal and small-scale sites that concentrate ore before applying mercury for amalgamation have damage values from \$1,100 to \$4,500 per ounce, and 4) artisanal and small-scale miners that apply mercury to the whole ore are the same miners who are being injured and killed - producing damage values from \$6,400 to \$23,000. Figure Figure 4.1 provides a visual overview of the results.

In Nevada, the results indicate that damage values were relatively low to begin with. However, NGOs and regulators identified a problem with mercury emissions from gold mining in the state. In 2006 a robust regulatory framework was initiated to address the problem. In response, the company acted on its financial incentives to comply with new regulations and engaged its technical specialists to solve the problem. Eight years later mercury emissions were reduced significantly.

Figure 4.1: Environmental Valuation of Damage per Ounce of Gold Produced at Various Sites

The Mt. Polley tailings storage facility failure was a rare event in mining that could not have occurred at a worse time for the one and a half million sockeye salmon moving up the river system for their annual spawn. Nonetheless, assuming high estimates for salmon and tourism losses, the damage per ounce of gold equivalent produced is still two orders of magnitude lower than the lowest small-scale or artisanal operation.

The only damage that is valued at Kadoma, Creporizinho, Galangan, Sao Chico, and Madre de Dios is for mercury emissions. Given that each of these operations concentrates the ore before amalgamation, these damage values were on the low end for artisanal and small-scale operations. On the other hand, operations at El Callao, Talawan, and Ica-Arequipa amalgamate the whole ore and cause injuries and fatalities. The damage figures for these sites are the worst - with the exception of the excessive mercury loss from Muller milling in China.

The results of this analysis indicate that the 'dirtiest' gold is produced via artisanal and small-scale mining. These damages are primarily due to the use of Hg^0 in mineral processing and fatalities from cave-ins or excessive Hg^0 inhalation. Even though open-pit gold mines in well-regulated jurisdictions move twenty tons of rock to get an ounce ounce of gold, they have a small socio-environmental impact ($<$\$1 - \$35) relative to artisanal mining operations (\$1,100-\$23,000).

Such a direct comparison of mining operations has not been conducted before and this analysis suggests that resources dedicated towards the minimization of social and environmental impact from mining ought to be directed towards the complicated issues of artisanal and small-scale mining.

4.4.1 Implications

The production of every ounce of gold has *some* impact on the surrounding community and ecosystem. When considering the permitting of large deposits held by publicly-traded companies in well-regulated jurisdictions, society must consider the fact that such deposits are linked to artisanal and small-scale mining operations by the global price of gold. This link implies a trade-off between formal production and unregulated production that can be formalized by the following question, "How many ounces of artisanal and small-scale production could be prevented by expanding the supply of formal gold production?"

This question can be answered by resolving the following two questions, "What would the supply increase from developing formally-held deposits in well-regulated jurisdictions do to the long term gold price?" and "What is the price elasticity of supply for artisanal and small-scale miners?" The first question is relatively straight-forward, but the second would require production data from groups that focus on artisanal and small-scale mining. Combining production data with price history could illuminate how artisanal and small-scale miners respond to price changes. If successful, this research would allow a comparison of environmental damage from the proposed project with the environmental damage that would occur in its absence.

Additional future research to facilitate monetary comparison of socio-environmental impacts of gold mining ought to focus on social dislocations in artisanal mining areas - such as gambling, drug/alcohol addiction, increased assaults, STD transmission from prostitution, forced labor, and forced migration. This research would provide a more comprehensive picture of the tradeoffs between gold extraction and socio-environmental impacts.

4.4.2 Conclusions

In conclusion, this analysis reveals that gold produced by publicly-traded companies is not nearly as dirty as gold produced from artisanal or small-scale operations. The results argue that society should prefer formal mining in well-regulated jurisdictions - with efficient extraction, minimal environmental damage, an accountable corporation, a competent regulator, and plenty of NGOs to monitor the process. There are approximately twenty million artisanal miners around the world with approximately eighty million dependents (Veiga & Gunson, 2004). If future resources are dedicated to helping the one-hundred million people around the world who are trapped in a cycle of poverty by artisanal mining, this analysis may offer a structure for prioritizing which sites ought to be addressed first.

CHAPTER 5

INCLUSION OF STUDY METHODOLOGY VARIABLES IN META-REGRESSION MODEL BENEFIT TRANSFER: THE INFLUENCE OF OUTLIER TREATMENT AND ERROR MEASUREMENT

Today's environmental valuation literature is characterized by a flood of competing empirical estimates (Heckman, 2001; Smith & Pattanayak, 2002). With the goal of turning information from primary valuation results into knowledge about the value of environmental services, scholars and policy makers have turned to meta-analysis - the statistical evaluation of primary valuation studies (Boyle *et al.*, 1994; Dalhuisen *et al.*, 2003; Johnston *et al.*, 2003, 2006; Poe *et al.*, 2001). Specifically, meta-analysis is used to identify determinants of variation between primary valuation results and to predict the value of environmental services at sites where environmental decisions need to be made. A meta-regression model (MRM) is fit to the primary valuation data to generate parameter estimates for the MRM's determinants. The estimated parameters are then paired with determinant values corresponding to the site in question to achieve a value estimate. This process is known as MRM benefit transfer (MRMBT).

Within the MRMBT literature, there is currently no consensus protocol for: 1) selection of determinants for the MRMBT model, 2) identification and treatment of outlier values, and 3) selection of the yardstick to measure benefit transfer error. Regarding the first point, a debate continues within the MRMBT literature regarding the inclusion of *study methodology* variables that describe *how* the primary valuation study was conducted; such as elicitation methods, valuation techniques, response rates, and other characteristics of the primary valuation study design (Bateman *et al.*, 2011; Bergstrom & Taylor, 2006; Johnston & Rosenberger, 2010; Kaul *et al.*, 2013; Nelson & Kennedy, 2009). Those in favor of excluding study methodology variables from MRMBT argue that economic theory only anticipates variation in primary valuation results due to *core economic variables* such as income, site specific factors, quantity, quality, and substitutes (Bateman *et al.*, 2011; Bergstrom & Taylor, 2006; Moeltner *et al.*, 2007; Smith & Pattanayak, 2002). The *core economic variable* camp believes that study methodology variables add noise when included in the MRM and argue that the unstudied sites requiring benefit transfer - by definition - do not have values for the study methodology variables (Bateman *et al.*, 2011; Bergstrom & Taylor, 2006; Moeltner *et al.*, 2007; Smith & Pattanayak, 2002). The opposing camp argues for the inclusion of study methodology variables on the grounds that these variables can explain in-sample variation between primary valuation results and that these variables may systematically affect benefit transfer values (Johnston & Rosenberger, 2010; Johnston *et al.*, 2005, 2006; Rosenberger & Phipps, 2007; Stapler & Johnston, 2009). This camp argues that the use of additional information, in the form of study methodology variables, ought to improve benefit transfer accuracy.

Regarding the second point, the MRMBT literature does not offer a unified approach regarding the identification and treatment of outliers (Nelson & Kennedy, 2009). A common approach is to rerun the MRMBT without outlying estimates to assess the sensitivity of the model to outliers (Desvousges *et al.*, 1998; Espey & Espey, 2004; Johnson *et al.*, 1997; Murphy *et al.*, 2005; Stapler & Johnston, 2009). Although each of these studies trims outlying data points, each study takes a different approach to identifying the outliers to trim. For example, Johnson *et al.* (1997) and Murphy *et al.* (2005) remove the largest and smallest values (5%), while Stapler & Johnston (2009) remove the 5% of values that generate the largest benefit transfer error.

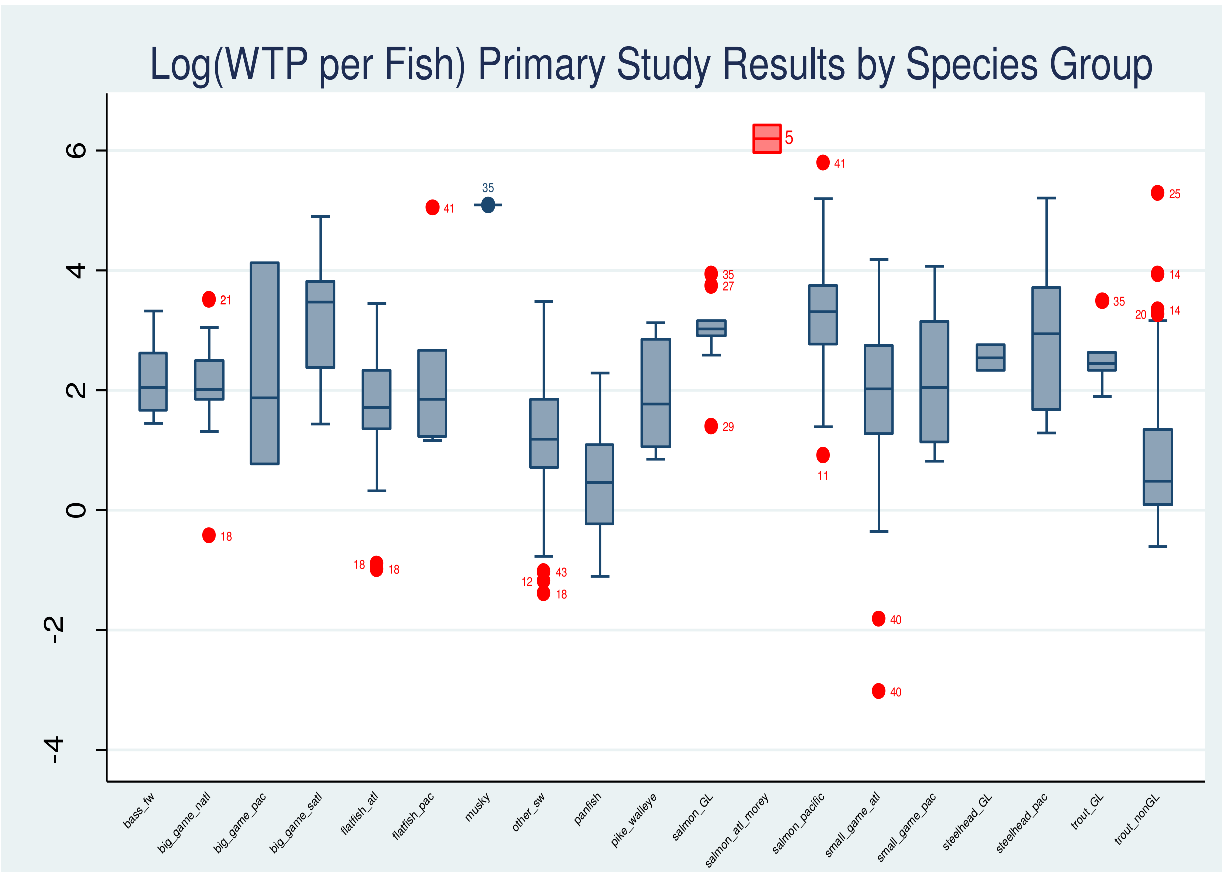
Finally, the MRMBT literature employs three different measures of benefit transfer error; squared error, absolute error, and absolute percentage error - which currently dominates the literature (Bateman *et al.*, 2011; Boyle *et al.*, 2010; Brouwer & Spaninks, 1999; Downing & Ozuna, 1996; Kaul *et al.*, 2013; Loomis, 1992; Rosenberger & Stanley, 2006; Shrestha & Loomis, 2003; Stapler & Johnston, 2009).

These three issues are explored by testing the validity of MRMBT through a repeated 'leave-one-out-then-predict' convergent validity framework (Shrestha & Loomis, 2003; Stapler & Johnston, 2009). This framework begins by leaving one observation (or study) out of the original data set that is used to specify the MRMBT function. Then the specified MRMBT function predicts the dependent variable of the withheld observations using the known values of the observations' determinant variables (Shrestha & Loomis, 2003; Stapler & Johnston, 2009).

In this context, the following note addresses the question: What is the effect on benefit transfer error of removing study methodology variables from the meta-regression model benefit transfer function? This analysis uses fish valuation meta-analysis data from Johnston *et al.* (2006) and Stapler & Johnston (2009) to compare the average benefit transfer error generated by the inclusion and exclusion of study methodology variables. The scope is limited to the effect of study methodology variables on benefit transfer error for this set of fish valuation meta data. A maximum likelihood estimation (MLE) with random effects (RE) model is employed for this analysis.

5.1 Data and Method

This analysis employs data from a meta-analysis of valuation studies concerning fisherman's willingness to pay to catch an additional fish - Johnston *et al.* (2006) and Stapler & Johnston (2009). The dependent variable is the natural log of the marginal value per fish (W_{is}) - where i refers to each observation within study s. This dependent variable represents the valuation result from the primary studies. Figure Figure 5.1 shows box plots of log(WTP per Fish) results by fish species group to identify outlying dependent variable estimates. Thirty seven outliers are identified by the simple examination of box plots in Figure Figure 5.1 (many are hidden in the figure because they overlap). These estimates lie outside of the interval that is one and a half times the inter-quartile range (from 25% to 75%) of estimates for the respective species group.



(Species Groups are defined in Table 5.1. and outliers are identified by study id for reference.)

Figure 5.1: Log(WTP) by Species Group

The core economic variables (X_{si}) are income, number of trips (dummy), the local nature of study respondents (dummy), fish species (dummy), catch rates, age (dummy), and whether respondents fished from the shore (dummy). The study methodology independent variables (Y_{si}) are variables that convey information regarding study type (stated preference, travel cost, or random utility model), elicitation method (mail, phone, in-person), response rate, and study year (non-dummy variable). A full list of variables from Johnston *et al.* (2006) and Stapler & Johnston (2009) is provided in Table Table 5.1. Study method variables are denoted with an (SM). Table Table 5.2 provides descriptive statistics for the data variables. Finally, Table Table 5.3 displays how Johnston *et al.* (2006) mapped individual fish species into its respective group.

5.1.1 Method

Following Johnston *et al.* (2003), Johnston *et al.* (2006), and Stapler & Johnston (2009), this analysis treats the 391 observations, from 48 valuation studies, as unbalanced panel data with random effects. The MRM parameter estimates are generated by a maximum likelihood estimation (MLE) with random effects (RE)[20].

This analysis investigates the effect on benefit transfer error of removing study methodology variables from the MRM that is fit to the meta-analytic data set from Johnston *et al.* (2006) and Stapler & Johnston (2009). To evaluate this effect, a repeated leave-one-out-then-predict convergent validity framework is employed. Formally, the leave-one-out-then-forecast structure from Stapler & Johnston (2009) assumes that the natural log of the marginal value per fish (W_{si}) is a function of the core economic variables X_{si} and the study methodology variables Y_{si}:

$$W_{si} = f(X_{si}, Y_{si}, \beta, \theta) \tag{5.1}$$

[20] The analysis was also conducted with Ordinary Least Squares (OLS), OLS with RE, and OLS with Clustering models to ensure that the analysis is robust to model selection. All of these models produced results similar to those presented below.

Table 5.1: Meta-analysis Variable Descriptions from Stapler & Johnston (2009)

Variable	Description
log_wtp	Natural log of the marginal value per fish.
SP_conjoint^{SM}	Dummy indicating study used conjoint stated preference.
SP_dichot^{SM}	Dummy indicating study used stated preference with dichotomous choice.
TC_individual^{SM}	Dummy indicating study used travel cost method based on number of trips.
TC_zonal^{SM}	Dummy indicating study used a zonal travel cost method.
RUM_nest^{SM}	Dummy indicating study used nested random utility model.
RUM_nonnest^{SM}	Dummy indicating study used non-nested random utility model.
sp_year^{SM}	Year SP study was conducted, subtracted by 1976. If not SP, set to zero.
tc_year^{SM}	Year TC study was conducted, subtracted by 1976. If not TC, set to zero.
RUM_year^{SM}	Year RUM was conducted, subtracted by 1976. If not RUM, set to zero.
sp_mail^{SM}	Dummy indicating SP study administered by mail.
sp_phone^{SM}	Dummy indicating SP study administered by phone.
high_resp_rate^{SM}	Dummy indicating that the sample response rate was greater than 50%.
inc_thou	Household income of survey respondents in 1,000s of 2003$.
age42_down	Dummy indicating mean age of sample was less than 43.
trips19_down	Dummy indicating mean number of fishing trips per year was less than 20.
nonlocal	Dummy indicating that no sample respondents were local residents.
big_game_pac	Dummy indicating Big Game target species in the Pacific.
big_game_natl	Dummy indicating Big Game target species in the North Atlantic.
big_game_satl	Dummy indicating Big Game target species in the South Atlantic.
small_game_pac	Dummy indicating Small Game target species in the Pacific.
small_game_atl	Dummy indicating Small Game target species in the Atlantic.
flatfish_pac	Dummy indicating Flatfish target species in the Pacific.
flatfish_atl	Dummy indicating Flatfish target species in the Atlantic.
other_sw	Dummy indicating Other Saltwater target species.
pike_walleye	Dummy indicating Pike Walleye target species.
bass_fw	Dummy indicating Bass target species.
trout_GL	Dummy indicating Trout target species in the Great Lakes.
trout_nonGL	Dummy indicating Trout target species outside the Great Lakes.
salmon_pacific	Dummy indicating Salmon target species in the Pacific.
salmon_atl_Morey	Dummy indicating Salmon target species in the Atlantic.
salmon_GL	Dummy indicating Salmon target species in the Great Lakes.
steelhead_pac	Dummy indicating Steelhead target species in the Pacific.
steelhead_GL	Dummy indicating Steelhead target species in the Great Lakes.
cr_nonyear	Baseline catch rate for target species in fish/day or fish/trip.
cr_year	Baseline catch rate for target species in fish/year.
catch_year	Dummy indicating study expressed catch rates on a per year basis.
spec_cr	Dummy indicating study presents information on baseline catch.
shore	Dummy indicating all respondents in sample fished from the shore.

(SM) Indicates a study method variable.

Table 5.2: Meta-analysis Variables and Descriptive Statistics from Stapler & Johnston (2009)

Variable	# of Obs	Units	Mean	Standard Deviation	Min	Max
log_wtp	391	ln of (2003$)	1.84	1.32	-3.03	6.42
SP_conjoint^{SM}	391	Binary (1/0)	0.04	0.20	0.00	1.00
SP_dichot^{SM}	391	Binary (1/0)	0.17	0.38	0.00	1.00
TC_individual^{SM}	391	Binary (1/0)	0.11	0.31	0.00	1.00
TC_zonal^{SM}	391	Binary (1/0)	0.04	0.20	0.00	1.00
RUM_nest^{SM}	391	Binary (1/0)	0.24	0.42	0.00	1.00
RUM_nonnest^{SM}	391	Binary (1/0)	0.30	0.46	0.00	1.00
sp_year^{SM}	391	Year Index	4.60	7.36	0.00	25.00
tc_year^{SM}	391	Year Index	0.73	2.19	0.00	18.00
RUM_year^{SM}	391	Year Index	9.37	9.72	0.00	25.00
sp_mail^{SM}	391	Binary (1/0)	0.05	0.22	0.00	1.00
sp_phone^{SM}	391	Binary (1/0)	0.13	0.34	0.00	1.00
high_resp_rate^{SM}	391	Binary (1/0)	0.36	0.48	0.00	1.00
inc_thou	391	1,000's of (2003$)	46.70	10.20	21.99	70.61
age42_down	391	Binary (1/0)	0.10	0.30	0.00	1.00
age43_up	391	Binary (1/0)	0.27	0.45	0.00	1.00
trips19_down	391	Binary (1/0)	0.11	0.31	0.00	1.00
trips20_up	391	Binary (1/0)	0.34	0.47	0.00	1.00
nonlocal	391	Binary (1/0)	0.01	0.07	0.00	1.00
big_game_pac	391	Binary (1/0)	0.01	0.09	0.00	1.00
big_game_natl	391	Binary (1/0)	0.05	0.22	0.00	1.00
big_game_satl	391	Binary (1/0)	0.02	0.14	0.00	1.00
small_game c	391	Binary (1/0)	0.03	0.17	0.00	1.00
small_g _atl	391	Binary (1/0)	0.16	0.37	0.00	1.00
flatfish_pac	391	Binary (1/0)	0.02	0.13	0.00	1.00
flatfish_atl	391	Binary (1/0)	0.10	0.30	0.00	1.00
other_sw	391	Binary (1/0)	0.23	0.42	0.00	1.00
pike_walleye	391	Binary (1/0)	0.03	0.17	0.00	1.00
bass_fw	391	Binary (1/0)	0.04	0.19	0.00	1.00
trout_GL	391	Binary (1/0)	0.01	0.11	0.00	1.00
trout_nonGL	391	Binary (1/0)	0.13	0.33	0.00	1.00
salmon_pacific	391	Binary (1/0)	0.08	0.28	0.00	1.00
salmon_atl_Morey	391	Binary (1/0)	0.01	0.07	0.00	1.00
salmon_GL	391	Binary (1/0)	0.02	0.15	0.00	1.00
steelhead_pac	391	Binary (1/0)	0.04	0.19	0.00	1.00
steelhead_GL	391	Binary (1/0)	0.01	0.07	0.00	1.00
cr_nonyear	391	Fish/Day	1.61	1.99	0.00	14.00
cr_year	391	Fish/Year	1.37	8.58	0.00	67.38
catch_year	391	Binary (1/0)	0.07	0.26	0.00	1.00
spec_cr	391	Binary (1/0)	0.84	0.36	0.00	1.00
shore	391	Binary (1/0)	0.15	0.35	0.00	1.00

(SM) Indicates a study method variable.

Table 5.3: Aggregate Fish Species Groups from EPA (2006)

Aggregate Group	Number of Observations	Species Included[a]
Big game	30	Billfish family, dogfish, rays, sharks, skates, sturgeon, swordfish, tarpon family, tuna, other big game
Small game	74	Barracuda, bluefish, bonito, cobia, dolly garden, dolphinfish, jacks, mackerel, red drum, seatrout, striped bass, weakfish, other small game
Flatfish	46	Halibut, sand dab, summer flounder, winter flounder, other flatfish
Other saltwater	89	Banded drum, black drum, chubby, cod family, cow cod, 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	Northern pike, walleye
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]Some 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 were assigned to the aggregate species group with which they shared the most species.
[b]Rainbow trout in the Great Lakes were classified 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 same species *Oncorhynchus mykiss*

where parameters β correspond to X_{si} and θ correspond to Y_{si}. The parameter estimates are mapped to the corresponding independent variable values of the observation in the hold out sample - which calculates a predicted log(wtp) result ($S_{n,forecast}$) that is compared to the actual result ($S_{n,actual}$) to assess benefit transfer error.

As mentioned above, the benefit transfer literature does not present a unified measure of benefit transfer error. Instead, the analyst chooses one of the following three; absolute percentage error (Equation 5.2),

$$[|e^{S_{n,forecast}} - e^{S_{n,actual}}|/e^{S_{n,actual}}]X100 = AbsolutePercentageError \tag{5.2}$$

squared error (Equation 5.3),

$$(e^{S_{n,forecast}} - e^{S_{n,actual}})^2 = SquaredError \tag{5.3}$$

and absolute error (Equation 5.4).

$$|e^{S_{n,forecast}} - e^{S_{n,actual}}| = AbsoluteError \tag{5.4}$$

The following analysis runs through the convergent validity framework for each measure of benefit transfer error.

To provide context for the benefit transfer error generated by removing study methodology variables from the MRM, two other cases are provided for the treatment of study methodology variables - both of which follow Stapler & Johnston (2009). The first case uses the known values of study methodology variables to predict $S_{n,forecast}$ for the hold-out observation. This is known as the *best case* because it represents the best possible scenario where study methodology variable values can be accurately determined for the site where benefits are being transferred. The second case uses the *mean* value of study methodology variables.

The *best case* scenario (which uses actual study method variable values) is described by Equation 5.5, which is adapted from Stapler & Johnston (2009, pp.232),

$$
\begin{aligned}
\hat{S}_{(1)} = \hat{\alpha}_{(-1)} + \sum_{j=1}^{J} \hat{\beta}_{(-1)} X_{j(1)} + \sum_{j=1}^{J} \hat{\theta}_{(-1)} Y_{j(1)} + e_{s_i} + u_i \\
\ldots \\
\ldots \\
\ldots \\
\hat{S}_{(N)} = \hat{\alpha}_{(-N)} + \sum_{j=1}^{J} \hat{\beta}_{(-N)} X_{j(N)} + \sum_{j=1}^{J} \hat{\theta}_{(-N)} Y_{j(N)} + e_{s_i} + u_i \\
.
\end{aligned}
\tag{5.5}
$$

In contrast, the *mean case* scenario (which uses study method variable value means) is defined by Equation 5.6,

$$
\begin{aligned}
\tilde{S}_{(1)} = \tilde{\alpha}_{(-1)} + \sum_{j=1}^{J} \tilde{\beta}_{j(-1)} X_{j(1)} + \sum_{j=1}^{J} \tilde{\theta}_{j(-1)} \bar{Y}_{j(1)} + e_{s_i} + u_i \\
\ldots \\
\ldots \\
\ldots \\
\tilde{S}_{(N)} = \tilde{\alpha}_{(-N)} + \sum_{j=1}^{J} \tilde{\beta}_{j(-N)} X_{j(N)} + \sum_{j=1}^{J} \tilde{\theta}_{j(-N)} \bar{Y}_{j(N)} + e_{s_i} + u_i \\
.
\end{aligned}
\tag{5.6}
$$

Finally, the *core economic variable* scenario (which removes study method variables) is defined by Equation 5.7,

$$
\begin{aligned}
\breve{S}_{(1)} &= \breve{\alpha}_{(-1)} + \sum_{j=1}^{J} \breve{\beta}_{j(-1)} X_{j(1)} + e_{s_i} + u_i \\
&\cdots \\
&\cdots \\
&\cdots \\
\breve{S}_{(N)} &= \breve{\alpha}_{(-N)} + \sum_{j=1}^{J} \breve{\beta}_{j(-N)} X_{j(N)} + e_{s_i} + u_i \\
&.
\end{aligned}
\tag{5.7}
$$

A priori, econometric theory expects the best case scenario to have the lowest error because it makes the most use of the data available. Similarly, the mean case scenario is expected to have te second lowest error because it uses more data than the core case - even if the study methodology variables are imprecisely measured. The core case is expected to generate the largest errors because it uses the least information.

The baseline parameter results of the MLE with RE model are presented in Table Table 5.4.

5.2 Results

The removal of outliers has a dramatic impact on the largest absolute percentage error results. With outliers, the largest absolute percentage error is over 30,000%. Without outliers, the largest absolute percentage error is just over 800%. Observations generating the worst absolute percentage error tend to be small outlying estimates from low quality primary valuations. Dividing absolute error by such a small estimate, magnifies the absolute percentage error.

Table Table 5.5 shows the impact on absolute percentage error of the three study methodology variable treatments for all observations and three trimmed data samples. When all observations are included, the core case has the lowest errors. This result is troubling, given that an F-Test (with the null hypothesis that all study methodology variables are jointly equal to zero) is rejected at the 95% level. However, as more outliers are removed, the errors

Table 5.4: MLE with RE Regression Results

Variable	Coefficient Estimate	Standard Error
RUM_nest	1.167423	0.648677
RUM_nonnest	1.58754	0.62879
TC_individual	1.007597	0.612675
TC_zonal	1.898762	0.557884
SP_conjoint	-1.18927	0.369713
SP_dichot	-1.00264	0.239867
sp_mail	0.51178	0.349944
sp_phone	1.080134	0.314278
shore	-0.1261	0.151363
RUM_year	0.00328	0.01903
tc_year	-0.03955	0.037276
nonlocal	3.164838	0.621051
high_resp_rate	-0.64841	0.180745
inc_thou	0.003346	0.007655
other_sw	0.440349	0.370279
pike_walleye	0.654782	0.358177
bass_fw	1.40012	0.39466
salmon_GL	1.862599	0.38599
salmon_pacific	2.065517	0.392529
trout_GL	1.477733	0.476602
big_game_natl	1.172706	0.419824
big_game_satl	2.04899	0.479468
big_game_pac	2.014449	0.620961
small_game_pac	1.396005	0.43959
small_game_atl	1.08974	0.378828
flatfish_pac	1.649486	0.489648
flatfish_atl	1.047896	0.380673
steelhead_GL	1.992695	0.683214
steelhead_pac	1.865381	0.419085
spec_cr	0.671214	0.204405
catch_year	1.257381	0.418142
cr_year	-0.05074	0.012952
cr_nonyear	-0.09471	0.031307
age43_up	1.329535	0.196863
age42_down	0.973779	0.247749
trips20_up	-1.12225	0.283653
trips19_down	0.789699	0.200844
trout_nonGL	0.591456	0.341967
salmon_atl_morey	4.979396	0.70454
sp_year	0.081148	0.028509
_cons	-0.98711	0.681308
/sigma_u	4.02E-14	0.145657
/sigma_e	0.831427	0.029732
rho	2.34E-27	1.70E-14

Table 5.5: Average Absolute Percentage Error Results for Trimmed Subsamples

	All Observations		4 Outliers Removed		20 Outliers Removed		All Species Group Outliers Removed	
	Median	Mean	Median	Mean	Median	Mean	Median	Mean
Mean Case	178	574	169	425	65	218	57	70
Best Case	159	483	138	352	56	172	55	67
Core Case	63	125	70	76	59	80	65	72

converge for the three treatments. First, the 1% (four observations) of species group outliers identified above generating the largest absolute percentage error are dropped. Then, the 5% (twenty observations) of species group outliers generating the largest absolute percentage error are dropped. Finally, all species group outliers (thirty-seven observations). Table Table 5.5 shows the impact on absolute percentage error of these four outlier trimming scenarios. The skewed error distributions make the median the best measure of absolute percentage error. As outliers are removed, the best case median absolute percentage error outperforms the other two cases, while the core case eventually generates the largest median absolute percentage error.

As with absolute percentage error, the removal of outliers has a dramatic impact on the square error results. With outliers, the largest square error is over 1.5 million. Without outliers, the largest square error is just over 30,000. Table Table 5.6 shows the impact on square error of the three study methodology variable treatments for all observations and three trimmed data samples. As with absolute percentage error, the 1% (four observations) and 5% (twenty observations) of species group outliers generating the largest square error are dropped. Finally, all species group outliers (thirty-seven observations). When all observations are included, the core case has the lowest errors and the best case has the highest. However, as outliers are removed, the square errors converge. The best case median square error falls and the best case outperforms the other two cases. As with absolute percentage error, the core case eventually generates the largest median square error.

As with the two error measures above, the removal of outliers has a dramatic impact on the largest absolute error results. With outliers, the largest absolute error is around 1,300

Table 5.6: Average Square Error Results for Trimmed Subsamples

	All Observations		4 Outliers Removed		20 Outliers Removed		All Species Group Outliers Removed	
	Median	Mean	Median	Mean	Median	Mean	Median	Mean
Mean Case	82.3	4,248	16.9	421	13.1	403	12.7	372
Best Case	92.1	5,988	11.2	398	10.7	387	10.5	358
Core Case	16.4	1,629	18.9	547	15.8	536	13.0	478

and without outliers the largest absolute error is just around 170. Table Table 5.7 shows the impact on absolute error of the three study methodology variable treatments for all observations and three trimmed data samples. As with the previous two error measurements, when all observations are included, the core case has the lowest errors and as outliers are removed the absolute errors converge. The best case median absolute error falls and outperforms the other two cases.

Table 5.7: Average Absolute Error Results for Trimmed Subsamples

	All Observations		4 Outliers Removed		20 Outliers Removed		All Species Group Outliers Removed	
	Median	Mean	Median	Mean	Median	Mean	Median	Mean
Mean Case	9.1	21.6	4.7	9.4	3.9	9.0	3.6	8.6
Best Case	9.6	23.4	3.1	8.1	3.3	8.1	3.2	8.4
Core Case	3.4	11.6	3.3	9.7	3.4	9.3	3.6	9.7

5.3 Discussion, Implications and Conclusion

The primary result from this analysis is that outliers have a significant impact on all measures of benefit transfer error. This impact is so large that improper conclusions could be made regarding the removal of study methodology variables if the analysis were to be conducted without removing outliers. For each error measure, the core case has the lowest error when all observations are included and the highest error when all species group outliers are removed. While the MRMBT literature widely acknowledges the impact of outliers, this switch (from unexpected to expected results for study methodology variable treatment) is a noteworthy finding for scholars and policy makers that are specifying MRMBT models to minimize benefit transfer error. Future convergent validity tests for benefit transfer ought to identify outliers from the raw data set and determine whether these outliers significantly impact the analysis. A more unified approach to the identification and treatment of outliers would help to advance the MRMBT specification debate.

Despite the MRMBT literature debate regarding proper model specification, this analysis is the first to explicitly evaluate the impact of removing study methodology variables. While this analysis builds on the work of Stapler & Johnston (2009), the author was unable to replicate the absolute percentage error results for the best and mean cases. Presumably, the difference is related to software, random effect treatment, or unpublished data work. Nonetheless, the findings of this analysis are the same as those of Stapler & Johnston (2009) for the best and mean cases - namely that plugging mean values into the study methodology variables for benefit transfer only slightly increases the absolute percentage error, in relation to the best case.

This analysis highlights a drawback of the primary measure of benefit transfer error employed by the benefit transfer literature. Small estimates generate disproportionately large absolute percentage errors and it is unique that this measure is affected by the size of the original estimate. Further, this analysis reveals that if outliers are identified by their impact on error, absolute percentage error leads to identification of different outliers than square and absolute error. Inclusion of an additional measure of error, and a priori identification of outliers, may prevent errant conclusions generated by the intuitively appealing absolute percentage error.

In conclusion, the effect on benefit transfer error of removing study methodology variables from the MRM for this data set depends on removal of outliers and the measure of benefit transfer error. When all observations are included, the core case proves to be the most robust to outliers. However, by identifying outliers with a simple box plot and removing them from the sample, it is shown that study methodology variables can reduce benefit transfer error - even when their mean value is used. Similar analyses ought to be conducted on different meta-analytic data sets to see if this result holds for other meta-regression models used for benefit transfer.

CHAPTER 6

AN EMPIRICAL ANALYSIS OF THE RELATIONSHIP BETWEEN SPOT AND FUTURES PRICES OF COPPER

Over much of the past decade commodity prices have risen, in some cases dramatically, sparking calls for curbs on speculation in commodity markets[21]. Proponents of such measures argue that investor demand has pushed commodity prices up and harmed consumers. Since most investor demand occurs on futures markets, this argument presumes the existence of a close link between changes in futures and spot prices.

Recently in this journal, Tilton *et al.* (2011) examine this mechanism from a conceptual or theoretical perspective[22]. They contend that when the spot and futures markets are in strong contango - that is, when the futures prices are sufficiently above the spot price to cover the costs of storing commodities including the interest cost on the capital this entails - the spot and futures prices will move closely together. If investor demand drives up the three-month futures price, this will encourage investors to buy on the spot market and sell forward on the futures market covering their futures position by storing the quantities they have purchased on the spot market. This inter-temporal arbitrage will continue until the price difference between the spot and futures markets returns to an amount that just covers the storage costs. As a result, they argue that movements up and down in the futures price will cause corresponding shifts in the spot price.

[21] This chapter represents the pre-submission version of a note that Dr. Tilton and I submitted to the journal *Resources Policy*. It was accepted March 14th, 2014. This work originated from a research project in Dr. Tilton's Metal Industries and Markets seminar. I am grateful to Cory Forgrave and Quinn Larwood for their early input on this project and to Phillip Crowson for his comments and assistance (Gulley & Tilton, 2014)

[22] Tilton *et al.* (2011) addresses two issues. The first focuses on the relationship between spot and futures prices during periods of strong contango, weak contango, and backwardation. The second explores the possibility of investor stocks declining even when investor demand is increasing. Östensson (2011) and Östensson (2012) raises various reservations about the analysis surrounding this second issue, to which Tilton *et al.* (2012a) and Tilton *et al.* (2012b) respond. These discussions, however, are not of relevance to the first issue, the topic of interest here.

On the other hand, when the spot and futures markets are in backwardation or weak contango - that is, when the futures prices are either below the spot price or not sufficiently above the spot price to cover storage costs - they contend that the link between spot and futures prices is much weaker (though not entirely absent thanks to the convenience yield of holding physical inventories). This is because inter-temporal arbitrage in the opposite direction is not feasible. One cannot buy physical stocks on futures markets and sell them immediately on the spot market. This leads Tilton *et al.* (2011) to conclude that:

> *. . . a surge in investor demand raising prices on the futures markets will have a direct and comparable effect on the spot market prices when these markets are in strong contango. However, when markets are in weak contango or backwardation, price movements in the futures markets have a much looser effect on spot prices. As a result, changes in investor demand on the futures markets may have little or no influence on spot prices in the absence of a strong contango. Instead, changes in fundamentals (that is, producer supply and consumer demand) and possibly changes in investor demand taking place directly on the spot market largely determine the spot price at such times.*

6.1 Purpose and Scope

This study proposes to provide an empirical test of this hypothesis by examining the relationship between copper spot and futures prices over the period 1994 to 2011. The central question is: Does there in fact exist a high correlation - close to one - between spot and futures prices when the copper market is in strong contango, and a much lower correlation during periods of backwardation and weak contango?

6.2 Data

The data used in the analysis consist of the following. Average daily London Metal Exchange (LME) copper spot prices and futures prices with 3, 15, and 27 month maturities from April 1994 to April 2011. The LME is the source for this information. Average daily

Euro-dollar deposit rates in percent per annum over the same period. The U.S. Federal Reserve Board is the source for this information. LME warehouse storage costs in U.S. cents per ton per day over the same period. The LME is the source of this information.

6.3 Method

From the finance literature[23], we know Equation 6.1 reflects the relationship between the spot price at time t (S_t) for a commodity and its futures price for delivery T months forward (FT_t):

$$FT_t = (S_t + U_t)e^{(r_t - \delta_t)T} \tag{6.1}$$

Where U_t is the unit storage cost at time t, e is the natural logarithm, r_t is the interest rate at time t, and δ_t is the convenience yield at time t. The convenience yield exists for commodities and other assets that are consumed as well as held as investments. It arises because there are benefits to holding inventories other than the expected profits realized from a change in their price over time. In particular, during periods of unanticipated shortages, consuming firms with greater-than-normal inventories can avoid interruptions in their production. The convenience yield varies greatly over time. During normal conditions when supplies appear adequate over the near term, the convenience yield is likely to be positive but quite small. However, when the market anticipates near-term shortages with adequate supplies over the longer term, the convenience yield can be quite large. It is also worth noting that the convenience yield is what ensures that Equation 6.1 is actually an identity. If one assumes that unit storage costs (U_t) are some percentage (μ_t) of the spot price, as is often the case, then Equation 6.1 becomes:

$$FT_t = S_t e^{(r_t + \mu_t - \delta_t)T} \tag{6.2}$$

When markets anticipate that near-term supplies are adequate, the convenience yield, as noted, should be quite small. As a result, $r_t + \mu_t$ will be greater than δ_t; futures prices will exceed the spot price; and the market will be in contango. However, when near-term

[23] This analysis comes from Dahl (forthcoming)

shortages are a concern, δ_t may be greater than $r_t + \mu_t$. In this situation, the spot price will exceed futures prices, and the market will be in backwardation. In their analysis, Tilton *et al.* (2011) assume that the interest rate (r_t) and the storage costs (μ_t) do not vary much from one time period to another. They also assume that, when markets are not concerned about near-term shortages, the convenience yield (δ_t) is quite small and also quite constant over time. In this situation, futures prices closely track the spot price plus the costs of holding stocks, and markets are in what they define as strong contango. When markets are in backwardation or what they call weak contango, this is because the convenience yield is much higher (and no longer negligible compared to the interest rate and storage costs), reflecting concerns about near-term shortages. Such fears, they implicitly assume, are likely to be temporary as additional supplies should eventually become available. As a result, during periods of backwardation and weak contango, the convenience yield is likely to vary considerably over time and the high correlation between spot and futures prices expected during periods of strong contango breaks down.

To test this hypothesis, we conducted the following analysis. First, we estimated the storage costs of a ton of copper by multiplying the number of months (3,15,27) in the contract times the monthly storage costs. Second, we estimated the cost of the capital required to purchase and hold a ton of copper by multiplying the spot price by $e^{r_t T}$, where e is the natural logarithm, (r_t) is the per annum Eurodollar rate, T is the number of months divided by 12 - 3/12, 15/12, or 27/12 - separating the spot and futures prices. Third, we paired spot prices with the 3-month futures price, with the 15-month futures price, and with the 27-month futures prices. Fourth, for each of these three groups of pairs, we separated our price observations into those occurring during periods of strong contango (when the futures price equaled or exceeded the spot price plus the storage and capital costs of holding physical copper) and those occurring during periods of weak contango or backwardation. Fifth, we calculated the simple correlation coefficients between spot and futures prices for each of these six subgroups.

Then, to determine if changes in the behavior of spot and futures prices were occurring over time, we divided our sample into two periods - 1994-2001 and 2002-2011 - and estimated the correlation coefficients for each of the six subgroups for both of these periods. We also estimated the correlation coefficients for spot and futures prices during periods of weak contango and then during periods of backwardation (rather than combining the two) to determine if these two market situations affected prices differently. Finally, we redefined strong contango to include a small convenience yield to see if this altered the findings.

6.4 Results

Table Table 6.1 reports the correlation coefficients between the average daily spot price and futures prices 3, 15, and 27 months forward on the London Metal Exchange over the period April 1994 to April 2011 for periods (a) when the market is in strong contango (assuming a convenience yield of zero) and (b) when the market is in backwardation or weak contango. The reported coefficient during periods of strong contango between the spot and the 3-month futures price is 1.0000. There are no correlations between the spot and the 15- and 27-month futures prices for periods of strong contango because the market was never in strong contango over this period.

When the market is in weak contango or backwardation, the correlation between the spot and 3-month futures prices is slightly less than 1.00 and declines somewhat with 15- and 27-month futures prices. In all cases, the correlations are quite high and close to one.

Table 6.1: Correlation Coefficients Between Average Daily Copper Spot and 3, 15, and 27 Month Futures Prices Assuming Zero Percent Convenience Yield

Correlation Coefficients April 1994 to April 2011 Zero Percent Convenience Yield			
	3 Month	15 Month	27 Month
Strong Contango	1	N/A	N/A
Weak Contango or Backwardation	0.9996	0.9931	0.9828

Table 6.2: Correlation Coefficients Between Average Daily Copper Spot and 3, 15, and 27 Month Futures Prices Assuming Zero Percent Convenience Yield, 1994 to 2001 and 2002 to 2011

Correlation Coefficients Zero Percent Convenience Yield			
1994-2001			
	3 Month	15 Month	27 Month
Strong Contango	1	N/A	N/A
Weak Contango or Backwardation	0.9949	0.9734	0.9515
2002-2011			
	3 Month	15 Month	27 Month
Strong Contango	1	N/A	N/A
Weak Contango or Backwardation	0.9996	0.9914	0.9769

Table Table 6.2 shows how the figures change when the time period is divided into two - the years 1994 to 2001 and the years 2002 to 2011. The results suggest that the correlation between spot and futures prices has been increasing somewhat over time. Otherwise, the figures are quite similar to those in Table Table 6.1.

Table Table 6.3 replicates Table Table 6.1 except that it breaks out periods of weak contango and backwardation rather than combining them. The results indicate that the correlation coefficients between spot and futures prices are slightly lower during periods of backwardation than during periods of weak contango. Again, however, even during periods

Table 6.3: Correlation Coefficients Between Average Daily Copper Spot and 3, 15, and 27 Month Futures Prices Assuming Zero Percent Convenience Yield

Correlation Coefficients April 1994 to April 2011 Zero Percent Convenience Yield			
	3 Month	15 Month	27 Month
Strong	1	N/A	N/A
Weak	1	0.9998	0.9988
Back	0.9996	0.9942	0.9828

Table 6.4: Correlation Coefficients Between Average Daily Copper Spot and 3, 15, and 27 Month Futures Prices Assuming 1.0, 2.5, 5.0, and 10.0 Percent Convenience Yield

	1994-2001			2002-2011		
One Percent Convenience Yield	3 Month	15 Month	27 Month	3 Month	15 Month	27 Month
Strong	0.9997	N/A	N/A	1	N/A	N/A
Weak or Back	0.9915	0.9734	0.9515	0.9996	0.9914	0.9769
Two and a Half Percent Convenience Yield	3 Month	15 Month	27 Month	3 Month	15 Month	27 Month
Strong	0.9997	0.9995	N/A	1	0.9998	N/A
Weak or Back	0.9816	0.9734	0.9515	0.9997	0.9918	0.9769
Five Percent Convenience Yield	3 Month	15 Month	27 Month	3 Month	15 Month	27 Month
Strong	0.9997	0.9883	N/A	1	0.9999	0.9983
Weak or Back	0.9816	0.9669	0.9515	0.9997	0.9919	0.9775
Ten Percent Convenience Yield	3 Month	15 Month	27 Month	3 Month	15 Month	27 Month
Strong	0.9997	0.9865	0.9065	1	0.9999	0.9993
Weak or Back	0.9816	0.9477	0.9457	0.9997	0.9925	0.976

of backwardation, all the coefficients are above 0.9800.

Table Table 6.4 indicates how the correlations in Table Table 6.1 change when strong contango is defined assuming the convenience yield during periods of strong contango is 1, 2.5, 5, or 10 percent (rather than zero). The reported correlations show a slight tendency to decline as the length of the futures price increases and as one moves from periods of strong contango to weak contango or backwardation. Again, however, these reductions are quite modest as all correlations remain quite close to one.

6.5 Conclusion

These results provide some empirical support for the hypothesis advanced by Tilton *et al.* (2011). In particular, as they predict, during periods of strong contango the correlations between spot and futures prices are very high. However, the results provide little support for the second part of their hypothesis - namely, that the correlations between spot and futures prices is much weaker during periods of weak contango and backwardation. For LME copper prices over the 1994-2011 period, these correlations, though slightly lower than those for periods of strong contango, are nevertheless extremely high. These findings suggest that the convenience yield during weak contango and backwardation - periods when concerns about near-term shortages are presumably high - is more stable and varies less from one month to the next than Tilton and his co-authors assume. If true, this finding has an important policy implication. It means that the speculation and investor demand that takes place on futures markets may alter the spot price of commodities not just during periods of strong contango but during periods of backwardation and weak contango as well.

CHAPTER 7
CONCLUSION

The following chapter concludes the presentation of five chapters of environmental valuation for the mining industry and one chapter regarding empirical analysis of commodity prices.

7.1 Chapter 1

Chapter 1 evaluates the capacity of the environmental valuation literature to value mine site pollution. While it is shown that the literature is broadly capable of estimating the appropriate ballpark value, it is also clear that gaps exist between scientific understanding of pollution and economists' ability to translate this understanding into how much the population cares about it. Examples of such gaps are fish population valuation, incremental water quality valuation, and ground water valuation. Scientific principles regarding environmental quality do not map neatly into environmental valuation frameworks. Currently, accurate economic valuation requires a site wide primary valuation that does not clearly delineate which environmental services are being valued.

Future ecosystem service valuations ought to be conducted with meta-analytic benefit transfer in mind. If federal agencies are to successfully incorporate the value of ecosystem services into their decision-making processes, then ecosystem service valuations that they sponsor ought to have the secondary purpose of being useful for benefit transfer and the tertiary purpose of addressing meta-analytic benefit transfer issues. For example, before conducting a primary ecosystem service valuation, federally funded studies ought to evaluate the existing valuation literature via a meta-analysis for benefit transfer. Such an exercise would set expectations for; the most appropriate study methodology, the impact of the selected methodology on the valuation results, market definition, the correct measure of economic value (Hicksian vs. Marshallian), and the valuation results themselves. While conducting the primary valuation, the authors will more fully understand what they must publish for future meta-analysts to make use of their work. Once the primary valuation is complete, the authors may illuminate meta-analytic benefit transfer issues by comparing their new analysis to the a priori expectations of the meta-analysis.

Further, federally funded ecosystem service valuations ought to provide enough information to be replicable. Replicability is a staple of the scientific process, but has yet to become a component of environmental valuations. All that is required to make a study replicable is access to the original data that was compiled and the code of the program used to manipulate the data and generate results. This is especially important when the study is a meta-analytic benefit transfer because the researcher has significant discretion during data gathering, data coding, model selection, and outlier trimming.

7.2 Chapter 2

Chapter 2 outlines the important components of a mine site pollution benefit transfer model; fish population, drinking water, and aquatic habitat. The recent Mount Polley tailings storage facility failure demonstrates the utility of focusing on these three ecosystem services. Drinking water, aquatic habitat, and future fish populations are the primary ecosystem services impacted by the spill. Irrigation water losses represent another important component of value.

For Leadville, soil quality improvements represent a significant benefit of remediation. But, the effect of the remediation on soil quality can not be parsed from other influences. Therefore, soil quality remains a footnote in this analysis. The value of a view is likely an important component of impact valuation for development of new deposits near populated areas. However, future work in this field ought not be too difficult. The literature is well developed and the United Kingdom even has a compensation scheme for such development projects. The white whale of future research for Chapter 2 is groundwater valuation. Developments in this field would greatly illuminate the hydraulic fracturing debate in oil and gas development. A scientifically rigorous cost-benefit analysis of contamination probabilities, contamination plumes, groundwater values, and economic benefit of mineral development would show the magnitude of social value for both the development activity and the surrounding environment. Such an exercise may clarify the public debate and pre-set values for damage compensation that development companies could incorporate into their pollution abatement decisions.

7.3 Chapter 3

The goal of Chapter 3 is to make use of the only analysis provided as a result of the RESErv project. This analysis comprises an evaluation of water quality and fish density at four sampling stations near the confluence of California Gulch and the Upper Arkansas River. As can be seen by the benefit transfer modeling in Chapter 2, the valuation portion of the RESErv project was intended to have broad scope but shallow depth. Fish turned out to be the only ecosystem service for which the opportunity was provided for valuation. This is a shame because the relationship between changes in metal toxicity and fish population can not be modeled precisely enough to make conclusions about effect sizes (Personal correspondence with Barb Horn, Water Quality Specialist, CPW). Further, the relationship between changes in fish population and changes in fish caught has not been firmly established (Personal correspondence with Dr. James Boyd and Dr. Kailin Kroetz). Air quality reduction from mine development, viewshed improvement from remediation, natural forest reduction from mine development, or conversion of an open pit to a recreational lake all represent more straight forward ecosystem services for linking scientific information on mine pollution with environmental valuation.

Nonetheless, Chapter 3 pairs an estimated value for fish caught with Colorado Parks and Wildlife data on the number of fish caught in the Upper Arkansas River between Leadville and Cañyon City. The purpose of Chapter 3 is to draw inferences via valuation on the relationship between improved fish populations and the economic benefit to anglers. It was shown that this fish-centric approach yielded benefit values that are an order of magnitude lower than a human-centric approach - which focuses on the entire value of the fishing trip.

Future research ought to be conducted on this discrepancy.

Chapter 3 also employs Loomis & Richardson (2008)'s model on the non-use value of aquatic habitat to estimate the value that non-users in the area hold for the remediation's improvement of this form of natural land cover. This valuation shows that non-use value associated with aquatic habitat improvements along the Upper Arkansas River ($834,078 per year) is on par with the use value of improvements in fishing along the same stretch ($773,325). EPA (2010) argue that non-use value is a significant portion of total value and these results support that argument in this instance.

7.4 Chapter 4

Chapter 4 departs from the ecosystem service framework and builds off of the Global Mercury Project to compare the environmental impact (per half troy ounce) of various mining contexts. IQ loss, mercury intoxication, injury and death are all considered using relevant valuations from the economic literature. The goal is a straight forward comparison of environmental damage so the valuation estimates are not scaled based on income. Instead, developed world estimates are applied in each context. This may be viewed as a shortcoming of this analysis. However, the benefit of this approach is that it illuminates the enormous disparity between environmental and social damage in various mining contexts. For example, considerable resources are spent in the environmental community to block efforts by multinational companies to develop feasible mega-deposits. An implication of this analysis is that those resources could be employed much more efficiently by continuing the work that has been done on artisanal mining.

Future research is warranted on this topic due to the recent gold price spike. The Global Mercury Project concluded that higher gold prices bring more artisanal miners to the gold fields. Therefore, the work conducted between 2002 and 2007 is in desperate need of an update. Such an update could be conducted with the goal of valuing of social and environmental damage to ensure that the collection of scientific information is clearly communicable to investigators of all backgrounds. Health studies that employ valuation may help to prioritize which sites are in most need of aid.

7.5 Chapter 5

Chapter 5 uses data from EPA (2006), Johnston *et al.* (2006), and Stapler & Johnston (2009) to show that treatment of outliers and the measure of error are crucial components of any analysis addressing the treatment of study methodology variables in meta-regression model benefit transfer. This brief chapter speaks to the model specification debate in the meta-regression model benefit transfer literature by directly addressing the removal of study methodology variables from the meta-regression model. The results indicate that, when outliers are removed, the use of mean values for study methodology variables is superior to the removal of study methodology variables. Similar examinations of meta-analytic data sets will determine whether this result can be generalized outside of the specific data set employed for this analysis.

7.6 Chapter 6

Finally, Chapter 6 employs data on average daily LME copper spot prices, copper futures prices, average daily Euro-dollar deposit rates, and LME warehouse storage costs to conduct a correlation analysis on spot and futures price during three market states. Equation 6.2 is used to pair spot prices with the 3, 15, and 27 month futures prices and determine which market state the pair is in on any given day between April 1994 and April 2011. Simple correlation coefficients are calculated for each group to explore the link between spot and futures prices in each of the three market states. The results provide some support for the hypothesis that spot and futures prices move in lock-step during periods of strong contango. However, spot and futures prices are also highly correlated (with correlation coefficients close to one) during periods of weak contango and backwardation.

7.7 Conclusion

Chapters 1 and 2 evaluate the environmental valuation literature's ability to value mine site pollution. Chapters 3 and 4 explore this ability through site specific benefit transfer valuations. Chapter 5 investigates a method of reducing benefit transfer error in the fish valuation literature that may have implications for other ecosystem and environmental services. Chapter 6 examines the link between spot and futures prices in an attempt to inform the question of when a speculator may be able to influence current spot prices via the futures market.

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