Mining Activities and Associated Environmental Impacts in Arid Climates: A Literature Review

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Abstract

Mining operations have released measurable levels of geogenic trace metals (e.g. Cd, Cr, Pb), metalloids (e.g. As, Se), and anthropogenic chemicals (e.g. CN^-, Hg) into surrounding sediments. Abandoned mining sites in hyperarid climates has not been the focus of much research compared to wet and temperate areas. Research has focused on historical mining sites in semiarid and wetter regions in the United States, south pacific and Europe. Those areas have obvious risks associated with them including aqueous phase mobilization as a result of abundant precipitation. However, many mining areas in the American Southwest and aboard are located in hyperarid regions and viewed as not having a potential for mobilization of contaminants. Seasonal storm events can mobilize sediments containing contaminates beyond a small, localized area and into the wider environment. Literature indicates that arid and hyperarid mining regions have not been studied as extensively as those in wetter climates.

Keywords: mining, trace metals, metalloids, cyanide, migration, arid climates, sediments

1. Introduction

Storage and mobilization of contaminants at abandoned mining sites in arid to hyperarid climates are key areas of inquiry to furthering our understanding of the potential pollution, ecological and human impacts caused by such sites. Mining tends to release geogenic metals (e.g. Ag, Cd, Cr, Pb, Hg), metalloids (e.g. As, Se), and anthropogenic chemicals (e.g. CN^-, Hg) into sediments that can be mobilized beyond a small, localized area, and into the surrounding environments (Purves, 1985; Sims, 2010, 2011; Miranda-Aviles et al., 2012; Sims et al., 2013). These contaminates have been the focus of interest due to their long residence time in the environment and can be toxic to biota (Khalid et al., 1981; Frignana & Bellucci, 2004; Hutchinson & Ellison, 1992).

During the past 20 years mine research has focused on historical mining sites in semiarid and wetter regions in the United States (i.e. California, Colorado, Idaho, Utah), south pacific (i.e. Australia) and Europe. These areas have obvious risks associated with them including more abundant precipitation. However, in arid and hyperarid regions, environmental issues and contamination related to historical mines have not been studied as extensively as those in wetter climates. Mine sites in arid regions are not seen as an immediate or potential risk to the environment due to their location (Ross, 2008).

Anthropogenic and geogenic inputs of trace elements to surficial sediments can impact a greater area than previously considered (Lottermoser & Ashley, 2005; Lottermoser et al., 1999; Coulthard & Macklin, 2003; Mackenzie & Pulford, 2002). Abandoned mines in temperate climates have been a focus for the United States Environmental Protection Agency since the mid 1980s (USEPA, 1995). There has been a great interest in the impacts of abandoned mining operation in semi-temperate climates in areas of Northern California and Nevada, Colorado, Montana, and Idaho (e.g. Engle et al., 2001; Horowitz et al., 1993).

Mining and milling activities are known to concentrate trace elements of geogenic origin into waste materials (Besser & Leib, 2000; Berger et al., 2000; Blowes et al., 1994). Historical mining operations used CN^- to remove precious metals from floated slurries of water and crushed ore, and Hg to form an amalgam with other precious metals so that they could be removed (Bonzongo et al., 1996; Churchill, 1999). Milling processes tend to

concentrate naturally occurring elements (e.g. As, Pb, Cr, Cd) during the process by separating the precious metals from the ore and concentrating residual trace elements in the tailings (Prusty et al., 1994). Subsequent disposal of mine waste containing the concentrated geogenic elements (e.g. Ag, As, Cd, Cr, Pb, Se) can greatly add to the anthropogenic contaminants burden to the surrounding area (Nicholson et. al., 2003; Miranda-Aviles et al., 2012).

It has been suggested that mines in hyperarid climates are of little threat to the wider environment because of a lack of precipitation (Ross, 2008). In hyperarid environments however, sediment redistribution from storm events can be the dominant form of transport for mining wastes (Reheis, 2006). Studies have suggested, for example, that contamination from mining activities related to dust (i.e. wind blown and fly ash) has the potential to impact the wider environment many kilometers from the source (Davis & White, 1981; Camm et al., 2003; 2004; Petaloti et al., 2006).

Although it is well documented that mining contamination in wetter climates has the possibility of migrating long distances by water and other transport mechanisms, there has been significantly less research in arid and hyperarid regions. Investigations of mine waste in wet temperate to wet climates are important however, it is clear that similar methods in hyperarid climates would not provided similar data. In mining regions with minute precipitation, contaminants migration occurs with sediment transport during storm events, not overland flow or groundwater movement. Thus it is imperative that research be conducted in these arid and hyperarid regions where significant amounts of mining and milling have occurred to evaluate the environmental distribution of contaminants to better understand the processes.

This paper examines literature concerning transport and distribution of trace elements in arid and hyperarid environments to assess the impact to the wider environment as compared to sites located in wetter regions. In order to fully evaluate relevant processes this review examines trace metals, metalloids, and anthropogenic inputs with wind transport of contaminates; transport of contaminates in wet climates; speciation and chemical behavior of trace metals and contaminates; siltation transport; potential for flora, fauna and human health affects; uptake and effects on flora; uptake and effects on fauna and on human health.

2. Trace Metals, Metalloids, and Anthropogenic Inputs

Historic mining operations in the American West date back to the middle 1800s with some sites having continuous operations into the mid-1950s (Greene, 1975). Many of these mines are small excavations that transported their ore to milling facilities in the area for processing. Some mining operations had their own milling systems adjacent to the mine, many within large ephemeral wash or stream systems (Greene, 1975). Historic milling operations employed the use of CN^-, Hg, and other chemicals to extract metals from ore however, this process extracted both previous and geogenic metals by concentration techniques. Once tailings were processed, wastes containing high levels of concentrated geogenic trace elements (e.g. As, Ag, Cr, Cd, Hg, Pb, Se) and anthropogenic contaminants (e.g. CN^- and Hg) were released into nearby areas for disposal.

Geogenic metals (e.g., Cr, Pb) and metalloids (e.g., As, Se) are those contained within rock, ore and sediments and include major crustal metals such as Al, Fe, and Mn. It is known that mining and milling activities concentrate these metals into waste materials and potentially causing environmental impacts at a later time (Besser & Leib, 2000; Berger et al., 2000; Blowes et al., 1994). On the other hand historical mining operations used CN^- to remove precious metals from floated slurries of water and crushed ore, and Hg to form an amalgam with other precious metals so that they could be removed (Bermejo et al., 2003; Bonzongo et al., 1996; Churchill, 1999). Disposal of mine waste containing these geogenic elements (e.g. Ag, As, Cd, Cr, Pb, Se) can greatly add to the anthropogenic inputs (CN^-, Hg) to the surrounding area.

Historical mining activities are known for their adverse impacts on the local environments, from esthetic to environmental. Typically, acid mine drainage (AMD) is a principal source of environmental impact; however, in an arid climate, a major potential source of contaminants is from mine tailings and waste rock where geogenic and anthropogenic sources are enriched in mine wastes and distributed across the landscape by the operators (Earman, 1996; Lu et al., 2012; Miranda-Aviles et al., 2012). In hyperarid regions wastes are generally left to the elements as it has been suggested that there is little risk due to a lack of precipitation (Ross, 2008).

Tailings of small particle sizes containing contaminants from milling processes are readily transported by wind and in over-land flow during storm events. Crushed ores are processed to particle sizes < 1 mm prior to flotation and extraction. After flotation, spent tailings waste was usually pumped to areas near a mill site such as ephemeral washes and basins. The transport of this material is limited to aeolian transport, aqueous, chemical behavior, flora and fauna and siltation/sediment transport during storm events. In arid regions, transport is limited to aeolian and sediment transport during storm events.

Cyanide and Hg are anthropogenic contaminants in mining areas that have been enriched in wastes and released to surrounding environments at a number of mining sites throughout the world (e.g. Engle et al., 2001; Besser et al., 2008; Yeddou et al., 2010). Most studies involving contaminated milling wastes are primarily in temperate climates where more rainfall occurs, and transport of anthropogenically derived CN^- and Hg is facilitated by both sediment and aqueous phase transport (Navarro et al., 2008).

In a study by Ismail et al. (2009), it was shown that the mobility of CN^- rich sediments is also controlled by pH of the surrounding sediments and the availability of metals such as Fe, Cu, and Zn to form complexes with CN^-. For example, between sediment pH 6 and 8, CN^- complexes of Zn are stable and less soluble, whereas, in an environment where the pH is > 8, Zn/CN^- complexes were found to be more soluble (Rennert & Mansfeldt, 2002).

Jambor et al. (2009) examined the solubility of CN^- from metal rich sediments with similar results to Rennert and Mansfeldt (2002) and Ismail et al. (2009). Jambor et al. (2009) illustrated that in metal rich sediments, CN^- complexes of Fe and Cu were less soluble from sediments, whereas, CN^- complexes of Ni and Zn are more soluble at pH values greater than 8. Thus, the dissolution of CN^- salts and metal complexes from sediments, followed by aqueous transport during storm events, was likely to occur in addition to transport with suspended sediment (Jambor et al., 2009; Yngard et al., 2007).

The transport of Hg from mining areas is most the result of trapped elemental Hg (Gemici & Oyman, 2003). Craw (2005) reported that Hg is strongly adsorbed to sediment particles and also to organic matter. Furthermore, Lechler et al. (1997) reported that surface contamination of Hg around mine sites is most likely as elemental Hg, as a result of its release from the Hg-Au amalgam used in the mining processing.

It has been shown that the release of anthropogenic contaminants such as CN^- and Hg in arid and semi-arid climates can impact the surrounding areas to a greater extent than previously believed (Navarro et al., 2008; Figueroa et al., 2008). The purpose of examining CN^- and Hg in an arid environment can determine the present-day distribution of these mining wastes in arid environments where their potential environmental impacts are not yet fully understood as compared to those in wet climates.

3. Wind Transport of Contaminants

There are few mechanisms of transport that impact the area adjacent to and downgradient/downwind from mining sites in arid climates. One such mechanism is wind (aeolian) dispersion. Aeolian dispersion has been well documented concerning transport of contaminated materials great distances from sources, impacting areas not directly affected by mining activities (e.g. Camm et al., 2003, 2004; Petaloti et al., 2006; Lopez et al., 2005; Yadav & Rajamani, 2006).

Aeolian dispersion can be one of the more dominant forms of contaminant transport of mining wastes in arid climates. Many of the environmental impacts associated with mine wastes can be attributed to the wind-blown transport of fine tailings to the surrounding areas (Davies, 1976; Reheis, 2006; Conko et al., 2013). A number of researchers showed that contamination from mining activities is related to dust, processed tailings, and fly ash, with the potential to impact the environment many kilometers down-wind from the source (Davis & White, 1981; Camm et al., 2003, 2004; Petaloti et al., 2006). For example, Davis and White (1981) have extensively investigated transport of particulate matter (PM) by wind. They showed that wind-blown mineral particles containing Pb, and other mine related wastes, travel great distances from the source. They further noted that particulates < 2 mm and > 2 μm in size are susceptible to aeolian dispersion, and the particles are evenly distributed directionally along wind patterns.

Meteorological data are important when performing such aeolian transport studies to understand the number of windy days, the wind velocity, and ground conditions, including grain size and land cover. Davis and White (1981) used this approach in a study to illustrate the dispersion of contaminated waste at a site in Western Wales. This study looked at 193 days of force 4 winds in a calendar year. Their findings demonstrated that Pb contaminated sediment, found down gradient, originated from the mining activities 1.8 km up gradient from the sampling area. The study of directional dispersion of wind-blown contaminated materials can provide information on the extent of environmental contamination from source areas, even in areas where transport is limited due to a lack of precipitation.

Work by Davis and White (1981) implies that the dispersion of contaminated materials will not only impact sediments and waters around a site but also has the potential of affecting the food-chain. To illustrate further, it has been demonstrated that produce grown (e.g. lettuce) down wind from a source of mine waste in the U.K. has been impacted with Pb above natural background levels (Davis & White, 1981). Wind-blown material from

mining areas therefore, can constitute a direct environmental hazard through both inhalation and ingestion of foods by humans and animals in the vicinity of such sites or, beyond (Davis & White, 1981; Camm et al., 2003).

By measuring metals or metalloids such as As, wastes from abandoned mines can be characterized and mapped (Camm et al., 2003, 2004; Conko et al., 2013). They demonstrated that contaminated sediments can be tracked down wind from a known source. Their research showed that As contaminated dust from milling facilities can be transported up to 2 kilometers downwind of source area and ultimately impacting a wider area (Camm et al., 2003, 2004; Davis & White, 1981).

Aeolian dispersion of As at milling facilities also presents a potential environmental hazard. The process of removing metals from ore involves heating in a calciner system which produces contaminated fly ash. Sediments at processing facilities have been found to contain As ranging between 200 and 3,325 mg kg^{-1}, and residue directly around a calciner can be 12% As by weight (Camm et al., 2004). The dominant form of As found at calciner facilities is As-Fe oxides in a range of textural wastes. Particulate matter ≤ 20 μm (PM_{20}) containing As-Fe oxides, with a good portion of this material in the PM_{10} size fraction, are known to be a potential inhalation exposure hazard (Lopez et al., 2005; Yadav & Rajamani, 2006). The study of PM fractions at mine and mill sites provides a baseline for long-term dispersal of materials containing potentially hazardous materials, and, the potential exposure to humans in relation to locations of milling facilities.

Similar findings regarding transport of trace metals (e.g. Pb, Cd, Cu, Cr, and Zn) in emissions from industrial regions of China have been studied by researcher (e.g. Kim et al., 1998; Hsu et al., 2005; Petaloti et al., 2006). Hsu et al. (2005) found PM_{10} containing Pb, Cd, Cu, Cr, and Zn transported from industrial areas in China to as far away as the United States. This is possible, according to their findings, because, during the winter, particulate matter will adsorb Pb and Cd and be transported by winds blowing from China to Taiwan and beyond (Hsu et al., 2005; Kim et al., 1998). In contrast, metal-containing particulate matter originating from mining sites has a similar but limited range of transport (Petaloti et al., 2006; Tasdemir et al., 2006).

Particle size fractionation has been used to investigate land use and fluvial deposition of contaminated mine waste (Horowitz & Stephens, 2008; Horowitz, 1993, 1988, 2008). Horowitz (2008) and others used air elutriation to evaluate each fraction of contaminated sediment and found that the finer aeolian fraction contained 10 times more metals than the larger fractions (Callender, 1988; Lu et al., 2012). When using this method of evaluating the lighter fractions, it was obvious that contaminants associated with the smaller particles can be transported farther (compared to shorter transport for larger size particulates) and deposited by aeolian processes in areas where there is no anthropogenic activities (Horowitz & Stephens, 2008; Lu et al., 2012). While their study of aeolian transport is extremely valuable in arid regions it was not seen as useful for this study due to the top layer (10-20 mm) of the tailings having a hard crust as a result of evaporation, evaporation in arid regions bring what little dissolved salts to the surface that produces a thin but stable crust trapping sediments beneath (Slowey et al., 2007a). However, studies by Mackenzie and Pulford (2002) of milling and calciner facilities in Scotland showed that wind distribution can transport materials from source areas to areas not directly impacted by industrialization. While literature has shown that wind dispersion has been studied at great lengths around industrial areas, studies do not show that it has been a focus in arid and hyperarid mining regions because of location of sites to population centers.

4. Transport of Contaminates in Wet Climates

Surface water influence on transport of contaminated sediments is a focus of much research. Mine sites located in areas of high precipitation are investigated for overland flow as a major means of transport, and the conveyance of contaminated materials in streams, rivers and subsurface is also a means of transport. Surface water and storm runoff as means of transport are the pathways by which contaminated sediments can move beyond their site of origin. Although it is known that overland flow and aeolian transport are major processes conveying contamination from a mine site, overland flow is typically ignored in arid regions because of the lack of precipitation and resulting infrequency of such an event, not to mention the location of much of these sites.

Abandoned mines in temperate climates have been a focus for the United States Environmental Protection Agency (USEPA) since the mid 1980s (USEPA, 1995) and others (e.g. Slowey et al. 2007a, 2007b; Stover, 1996; Lottermoser & Ashley, 2005). During the 1980's, the USEPA investigated abandoned mines in Colorado to determine if there was potential for environmental issues (Stover, 1996). Many of the mines were located next to permanent water sources, such as Chalk Creek, which received contamination from nearby mining sites. Since that time, the USEPA's research has evaluated precipitation and its potential influence on the transport of contamination into surface waters (USEPA, 1995).

Precipitation promotes the movement of trace elements into surface waters in dissolved and particulate forms

and, therefore, can create adverse habitat conditions for fish populations in receiving bodies of water (Stover, 1996). Two sources of mine waste entering the environment are leachates from mine tailing waste and AMD from mine tunnels and shafts (USEPA, 1995). The first source is caused by the leaching of waste materials with high acid-generating potential; second is the surface runoff from contaminated mine sites producing AMD that adversely affect nearby surface waters.

Mines that used gravity separation to remove cinnabar (HgS) from ore for later processing tend to have larger amounts of Hg contamination (Slowey et al., 2007b). Due to environmental issues related to Hg (i.e. metalization), it has been a main focal point for investigation. It has been found that Hg concentration in sediments tends to be higher adjacent to, and down gradient from abandoned mining and milling sites located in either volcanic areas or when Hg was used in processing (Dolenec et al., 2005; Moncur et al., 2005, 2006). High levels of Hg in sediments down gradient from the source area are the direct result of processes of erosion and overland flow as shown by Macklin et al. (2001). Research has shown that in areas where the dominant Hg source is cinnabar, the Au extraction processes alter cinnabar to meta-cinnabar, a more soluble form (Gemici & Oyman, 2003; Moncur et al., 2005). Such a change allows Hg to be more susceptible to physical weathering, permitting an enrichment of Hg in fine-grained sediments and possible metalization later due to environmental effects (Slowey et al., 2007a, 2007b; Kim et al., 2004; Harikumar & Jisha, 2010).

Research shows that higher ratios of Hg between water and sediments are a direct result of a higher solubility of Hg originating from AMD sources adjacent to mines (Nishida et al., 1982; Slowey et al., 2007a, 2007b; Rampe & Runnells, 1989). However, Slowey et al. (2007a) reported that water sources with appreciable amounts of dissolved bicarbonate will buffer the system limiting acid-soluble metals. They reported that AMD receiving waters containing as much as 430 mg L^{-1} $CaCO_3$ can provide sufficient buffering capacity to neutralize acidity that will result in sorption or precipitation of soluble metals. Ultimately, water chemistry controls Hg, and other trace metals, partitioning between the aqueous and particulate phases in an aquatic environment. However, in an arid environment water chemistry is not an issue but, once transported materials can be deposited in an area or location where water can provide an environment where Hg or other trace metals can chemically change.

Release of trace elements into surface sediments and their subsequent transport by storm-water is likely to be influenced by the adsorptive capacity of the sediments, e.g. the texture and the amount of organic matter. For example, Rieuwerta and Farago (1996) found that levels of Pb and Hg decreased in sediments with decreasing organic matter. This work shows that organic matter can control Pb and Hg retention in sediments and subsequent transport. Davis (1976) and others have conducted similar investigations seeking a correlation between base metal mining and pollution in sediment (Rowan et al., 1995; Davis & White, 1981). Davis (1976) found correlations between Pb-Hg and Cu-Zn in sediments with similar organic matter content. It was reported that in areas where base metal mining occurred, high levels of Pb and Hg were more likely to be present (Macklin et al., 1997). In fact, in some historical mining areas, Hg was reported to be as much as 2 mg kg^{-1} in sediments derived from a floodplain that was downgradient from historical mining areas (Macklin et al., 1997).

It was found that water contamination caused by an influx of sediment from mining sites has the capability of impacting a wider area (Zhuo et al., 2007; Luo et al., 2006; Singhal & Islam, 2008; Rowan et al., 1995). Studies suggest that floodplain sediments that contain particulate-bound trace elements (e.g. Pb, Zn, Cd, and As) were most likely the result of repeated fluvial scouring and re-suspension of sediments (Rowan et al., 1995; Luo et al., 2006). Sediments originating from a mine site can spread contaminants primarily by their transport with water (Luo et al., 2006). Rowan et al. (1995) suggested that (water) metals contamination and subsequent transport is however limited to the inevitable precipitation and sorption of metals onto sediments.

Research shows that trace elements (e.g. Hg, Pb, Cu, Zn, Cr, Ni, As) can be used to illustrate movement of sediments in runoff from mining sites (Sanghoon, 2006; Luo et al., 2006; Singhal & Islam, 2008). Additionally, contamination is not restricted to a small geographic area when transport redistributes sediments enriched with trace elements. Materials that are normally stable in sediments will be mobilized in storm flow and transported great distances from the source (Sanghoon, 2006). In areas with high precipitation, the mobilization of metals begins with leaching of tailings, particularly where the pH is low and SO_4^{2-} content is high (Moncur et al., 2005).

At Lake Coeur D'Alene, Idaho, USA Horowitz et al. (1993) found trace elements (e.g. Ag, As, Pb, Cd) were being deposited into lake sediments from adjacent mining activities. Horowitz et al. (1993) illustrated that trace metal-enriched fine sediments from abandoned mining and milling areas were mobilized from the ground surface to Lake Coeur D'Alene during periods of high precipitation. It was establish that association with Fe oxyhydroxide coatings on grains made the trace metals more environmentally available than if they were associated with other constituents such as sulphide minerals. With relations identified between trace metals and

Fe oxides, Horowitz et al. (1993), were able to determine that the trace metals deposited into Lake Coeur D'Alene can be ascribed to the abandoned mining operations rather than geogenic deposits.

More recently, research has focused on water chemistry and the transport of metals and metalloids in mine drainages (Kwong et al., 2007; Slowey et al., 2007a). For example, As in mine drainage from tailings in Ontario (Canada) was an order of magnitude above the Canada drinking water standard of 0.05 mg L^{-1}. Studies by Riemer and Toth (1970) showed that the chemistry of sediment pore-water will directly influence the transport of As and possibly other trace elements. In the short term, as dissolved trace elements enter a system, they tend to adsorb to suspended matter; however, they could re-dissolve farther down the system when the geochemistry changes due to system variability (Riemer & Toth, 1970).

Ashley and Lottermoser (2003, 2004) and others studied mine waste and the distribution of As in surface waters (Lottermoser & Ashley, 1999). They found AMD with a low pH (4.1) had the highest concentration of dissolved As (up to 13.9 mg L^{-1}) that was entering the Mole River, New South Wales, Australia. Once AMD entered the Mole, surface waters entering the river further diluted the As-rich drainage to background concentration (0.0086 mg L^{-1}) beyond 2.5 km downstream. While their study showed that As was eventually diluted to < background, it was evident that environmental conditions have the ability to alter trace element species over time and distance.

In a semi-arid climate environmental impacts by trace metals are influenced by surface runoff during storm events. Taylor and Kesterton (2002) showed that the distribution of trace metals depended on their solubility and more importantly, their mobility. The movement of trace metals within the sediment profile is also influenced by sedimentation, allowing the distinction between pre-mining, mining, and post mining in semi-arid climates (Meza-Figueroa et al., 2009). However, the above processes are lacking in arid locations where contaminant transport is limited primarily to sediment interactions and subsequent storm-water transport of the affected sediment rather than aqueous mechanisms (Taylor & Kesterton, 2002).

Further research of abandoned mine sites determined whether sites were sources of water contamination caused by influx of sediment containing Pb (Zhuo et al., 2007; Luo et al., 2006; Singhal & Islam, 2008; Rowan et al., 1995). Floodplain sediments were found to contain particulate-bound contaminants such as Pb, Zn, Cd, and As, that were most likely the result of repeated fluvial scouring and re-suspension of sediments, further dispersing trace element contaminated particles (Luo et al., 2006; Singhal & Islam, 2008). According to Luo et al. (2006) movement of sediment will impact the degree of traceability while lateral movement accounts for the age of deposition (Giuliano et al., 2007).

Some contaminants (e.g., Hg, Pb, Cr, As), both soluble and particulate forms, in wetter climates can be used to illustrate movement of contamination in surface waters and sediments adjacent to historical mines (Sanghoon, 2006; Luo et al., 2006; Singhal & Islam, 2008; Da Silva et al., 2005). In areas with high precipitation, the mobilization of metals begins with leaching of tailings (Moncur et al., 2005). More recent research has focused on water chemistry and the transport of metals and metalloids in mine drainage (Kwong et al., 2007; Harris et al., 2003; Leinz et al., 2006). Speciation and mobilization of metalloids and the mechanisms of transport in sediments and water are a direct result of the amount of available moisture (Kwong et al., 2007). As dissolved trace elements enter a system, they tend to adsorb to suspended matter; however, they could re-dissolve farther down the system (Lewis, 1977; Riemer & Toth, 1970).

Research clearly shows that the mobilization of contaminates from processed tailings through surface and ground water flow has been a focus of studies (Macklin et al., 2001; Hughes & Diaz, 2008; Bonzongo et al., 1996). Furthermore, studies of historic mining and the resulting contamination has been the focus in regions that receive generous amounts of precipitation rather than arid or hyperarid areas. In an arid environment water is not an issue. However, once transported to an area or location where water is present such as a desert lake or perched aquifer, Hg, or other trace metals, can transform from a minor environmental issue to a major concern such as methyl-mercury.

5. Chemical Behavior of Trace elements

Speciation and geochemical processes can have a major influence on transport of trace elements at abandoned mine sites. Furthermore, studies have shown that the transformation of trace elements by microbes can influence transport (e.g. Macklin et al., 2001; Chen et al., 2006; Conko et al., 2013). An excellent example of how transformation because of microbes can affect their surrounding environment is the methylation of elemental Hg to methyl-mercury (CH_3Hg^+), a more soluble, bioavailable, and highly toxic form. When CH_3Hg^+ enters a water system it can produce fish kills and large amounts of toxic Hg uptake by multiple species resulting in an impairment in higher species and the food web, thus affecting the surrounding environment more so than elemental Hg (Conaway et al., 2004; Hughes & Diaz, 2008; Bonzongo et al., 1996; Mastrine et al., 1999). In the

Western United States, historically Hg was used in the extraction of precious metals by amalgamation. During this process, excess Hg was routinely dumped into the immediate environment (Greene, 1975). Although portions of Hg will volatilize into the atmosphere, it is possible for Hg to bind with organic matter and remain in soils and sediments for longer periods of time, until transported by erosion.

Methyl-mercury has been a focus of investigations to provide the modes of conversion from elemental to CH_3Hg^+, and the ratios of Hg to CH_3Hg^+ (Hughes & Diaz, 2008; Bonzongo et al., 1996; Mastrine et al., 1999). Macklin et al. (2001) found that methylation occurs in sediments when Hg is < 15.3 mg kg^{-1}, than decreasing above 15.3 mg kg^{-1} because of toxicity (Macklin et al., 2001). Decreased methylation is attributed to reduced activity of microbes as a result of higher levels of total Hg; i.e. more toxic to microbes. A reduced activity of microbes during a winter and spring flooding is also directly related to the presence of Hg at > 15.3 mg kg^{-1} because of increased transport (Macklin et al., 2001).

In an aqueous system, CH_3Hg^+ will increase to toxic levels below a point source by microbial mediated methylation. This process is further enhanced as a direct result of additional Hg entering the system during winter and spring runoff events (Bonzongo et al., 1996). Findings have suggests that CH_3Hg^+ decreases in concentration with distance from a source area, in part from volatilization with turbulence of flowing water as it moves downstream, and in part by retention of Hg and CH_3Hg^+ in bed sediments (Bonzongo et al., 1996).

Macklin et al. (2001) and Mastrine et al. (1999) studied several mining areas with similar climates, sediments, vegetation, and rock where Hg was utilized to amalgamate Au during the milling process. They found Hg and iron (Fe) concentrations were directly correlated in sediments. Mastrine et al. (1999) explained that the relation between Hg and Fe was the result of similar mechanisms that control the aqueous forms of both metals. Their findings showed that sorption of Hg to Fe-oxyhydroxides has a direct relation to the amount of Fe and Hg found in waters originating from contaminated sediments at a mining site. They noted that this relation is caused by the mass of total suspended solids (TSS) containing Fe oxyhydroxides that sorb Hg and Fe to TSS in waters.

Waste from middle 1800s mining activities continues to impact the surrounding environment of the Western United States and beyond (Sims, 2013). For example, Hg has been released from historical mill and mine operations to surface waters of the Truckee River and Steamboat Creek systems in Northern Nevada (Bonzongo et al., 1996; Chen et al., 2006). Total Hg and CH_3Hg^+ in sediments along these river systems have been studied to determine the distribution in relation to precipitation and aeolian transport (Stamenkovic et al., 2004; Kim et al., 2000; Jonasson & Boyle, 1972). If levels of Hg in a river system do not decrease with distance in channel sediments then this suggests that a constant source of Hg is being mobilized (Stamenkovic et al., 2004). Such findings are useful in characterizing movement of sediments and the transformation from elemental Hg to CH_3Hg^+ in similar aquatic systems.

Studies have found that anthropogenic Hg at mine sites is one to two orders of magnitude higher than that in pristine streams (Stamenkovic et al., 2004; Macklin et al., 2001). For Hg methylation to occur, conditions favoring an anaerobic environment must be present so that an increase in microbial activity can be available. Stamenkovic et al. (2004) also asserted that minerals and nutrients (i.e. SO_4^{2-}) in sediments might directly correlate high levels of Hg with their reduction. They found that SO_4^{2-} reduction and Hg conversion to CH_3Hg^+ decreases in stream bank sediments containing SO_4^{2-}, which indicated that microbial processes of Hg methylation are inhibited in the presence of significant amounts of dissolved SO_4^{2-}, indicating an aerobic environment.

Speciation is typically used to determine the mobility of Hg in sediments and to identify sources and release potential. Research into Hg speciation in the California Coast Range was undertaken to better understand its mobility in mine waste and how solubility relates to speciation (Kim et al., 2000; Mastrine et al., 1999). However, studies have suggested that Hg and with other anions (i.e. Cl^-) in sediments might have a correlation between solubility and mobility (Kim et al., 2000; Stamenkovic et al., 2004). Determining the correlation between Hg and the anions would provide invaluable information concerning the transport related to Hg in the environment.

Researchers have examined speciation and bioavailability of As, Zn, Cd, Pb, Mn, and Al in mine waste (Wu et al., 2006; Meers et al., 2006; Nair & Robinson, 2000; Chen et al., 2006; Gupta et al., 2007). It was established that the bioavailability of certain metals like Zn, Pb, and Cd are controlled by sediment moisture, pH, and dissolved organic matter. In sediment high in Fe and Mn, it is known that oxidation and adsorption of As (III) and As (V) by pedogenic Fe-Mn rich solid phases at varying sediment pH and organic content controls the speciation of As (Chen et al., 2006; Grafe et al., 2007, 2008; Boyle & Smith, 1994). Slowey et al. (2007b) suggested that As mobilization occurs with the mobility of the particulate phase rather than transport in the

aqueous phase. It has been further shown that the dominant As species in the environment is As (V) and it is adsorbed to amorphous iron-hydroxides that co-precipitated with jarosite (Grafe et al., 2008; Chen et al., 2006; Gupta et al., 2007). Grafe et al. (2008) and others showed that As is released from mine waste as As (V) where a low pH and oxidation convert As (III) to As (V), allowing As to migrate as a particulate verses in the aqueous form (Grafe et al., 2008; Slowey et al., 2007a; Breteler et al., 1981). Suspended materials and electrolytes can influence the transport of metals in water bodies. It has been found that suspended particles can remove soluble contaminants from the water column (Al-Busaidi et al., 2005). Stumm and Morgan (1996) and others have shown that high sediment and salinity suppress suspended particles by their flocculation and subsequent settling out from the water column (Al-Busaidi et al., 2005; Sakata, 1987; Zhang & Selim, 2005).

Another factor influencing the mobility of trace metals is the interaction with the sediment surfaces. Cheng et al. (2009) examined arsenic transport and transformation through columns packed with uncoated sand and natural sand coated with inorganic colloids and/or dissolved organic matter (DOM) using arsenate, As (V) in solution. The occurrence of inorganic colloids and/or DOM evidently enhanced As transport, with a larger fraction of As leaching out of the column compared to the total amount added. Cheng et al. (2008) further explained that when a solution of As (V) was combined with dissolved organic matter the mobility was increased. Therefore, As (V) can leach from the sediments to groundwater due to elevated DOM.

It has been shown that in mining areas with high levels of organic matter in sediments, and significant precipitation, the solubility of As (V) increases (Stumm & Morgan, 1996; Chang et al., 2009). For example, when As (V) was spiked in waters similar to mining site seepage, only As (III) was detected in the effluents for uncoated sand columns while both As (III) and As (V) were detected in the coated sand columns (Chang et al., 2009). It was recognized that when monomethylarsonic acid was injected into sediment columns, all As species were present in the effluents (Cheng et al., 2008; Feng et al., 2005; Hutchinson & Meema, 1987). Thus it appears that areas where organic matter is very low, limited to no moisture reduces the possibility for mobilization and transport of metals and other contaminants are poor due to environmental factors other than overland transport (i.e. arid and hyperarid regions).

6. Siltation Transport

Authors have suggested that siltation possesses several issues to the environment; clogging of the water column and trace element leaching among others. James (1991) suggested that high siltation of waterways can impede aquatic life growth because it can effect oxygen uptake by local biota causing a suffocation. Furthermore, high siltation by mine waste will also provide ideal conditions for enriched trace elements leaching into water systems, further impacting biota (Hudson-Edwards et al., 1997). While siltation clogging the water column is an important issue, the potential release of trace elements contained in silt or sediment that originated from mining areas is significant (e.g. Edwards et al., 1997; Hudson-Edwards et al., 1997; Dennis et al., 2002). Siltation of waterways and associated contamination has the potential to impact a larger radius of a stream or river basin by both sorbed and dissolved metals.

Researchers have used Hg and Si to trace siltation from hydraulic mining areas (e.g Edwards et al., 1997; James, 1991). For example, it has been found that elevated silica concentrations in rivers and streams was observed a distance of 60 km downstream from mining sources in just 100 years in the mining areas of northern California (Edwards et al., 1997; James, 1991). It was found that elevated Si was correlated with Hg, suggesting the source Hg was from the same location as the silica. James (1991) stated this material could have moved even farther than 60 km if it was not for dams and other obstructions put in place since the early 1930s.

Siltation in a river or stream can also affect areas adjacent to water sources such as overbanks and low lying areas (Hudson-Edwards et al., 1997). Studies indicate that sediment contamination has produced metal-contaminated overbank river sediments and increased metal concentrations in waters of local rivers and tributaries (Hudson-Edwards et al., 1997; Dennis et al., 2002). By following siltation patterns, scientists have been able to trace sources of mining-related sediments to their original source. Furthermore, it has been found that in some low lying areas, contamination from mining sites have mixed with agricultural runoff. There have been several incidents reported in the United Kingdom in which a major flood occurred on lands once used for base metal mining that are now used primarily for agricultural (Dennis et al., 2002). In these areas, overbank and channel sediments would already have contained high levels of base metals prior to agricultural development. However, in areas where Pb, Cd, and Zn were mined, there were high levels of contamination corresponding to the source areas of the trace elements (Mlayah et al., 2009; Lecce & Pavlowsky, 2001).

Siltation has the potential to clog water ways as the material moves from an abandoned mine site. However, siltation also can have a greater effect on the environment when tailings dams fail and mine waste enters a water

system. Although this is a somewhat different release mechanism than storm-water transport, it illustrates the effects of a catastrophic tailings transport event on the environment. Thus, it is possible for sites with contaminated sediments, tailings, or mine waste to impact a much larger area, such as a basin or floodplain (Fuente et al., 2007; Miller, 1997; Hernandez et al., 1999). For example, in early 2000, a dam failed in Maramures County, upper Tisa Basin, Northwest Romania releasing a significant amount of silt into the Tisa River (Macklin et al., 2001). During this dam failure over 40,000 metric tons of contaminated sediment was released into major tributaries of the Danube River, with pollutants of cyanide and metals (Pb, Zn, Cu, and Cd). Although metal concentration in surface waters decreased with distance from the source because of a high buffering capacity and dilution, trace metals in sediments still exceeded acceptable levels downstream (Macklin et al., 2001). The Danube River incident illustrates that flood waters originating from a mining area have the potential to pose a significant environmental threat to its surrounding area.

In Seville, Spain, the Aznalcollar tailing dam at the Boliden Apirsa's mine was breached in April of 1998 (Hernandez et al., 1999). The failure impacted 4600 hectares of land located along the Rios Agrio and Guadiamar with 5.5 million cubic meters of acidic waters and 1.3×10^6 cubic meters of mine waste containing metals (Hernandez et al., 1999; Edwards et al., 2003). It was shown that mine waste contained high concentrations of Ag, As, Cd, Cu, Pb, Sb, Tl, and Zn compared to their background levels (Edwards et al., 2003). Studies showed that Zn and Cd had the highest impact on the environment, whereas Pb and As were not mobile unless reducing conditions that could dissolve greater amounts of the reduced species developed in the mine wastes (Edwards et al., 2003). It was also clear that high mineral sulfide content in these sediments can produce AMD, under aerobic conditions, easily releasing trace metals.

When tailings reach surface waters, however, it can transport large quantities of contaminated sediments to downstream areas such as floodplains, basins, or farm lands. Macklin et al. (1997, 2003, 2006) studied transport of localized contaminated sediments by tracking the material downgradient in what they termed "pulses". They explained that pulses consist of material temporarily suspended in flood waters and ultimately deposited in localized spots beyond the source. Finding showed that contaminated deposits decreased downstream in sediments of rivers and streams (Meijer et al., 2002; Coulthard & Macklin, 2003). It is thus important to account for the secondary deposits when evaluating the overall pollution of a mine area. The materials that make up the isolated spots were the result of contaminant laden sediments being transported (secondary deposits) during storm events rather than the legacy of soluble contaminants in the water column.

6.1 Tracking Movement of Contaminated Material

The issue of geogenic contaminants and their mobilization into the environment has been assessed using different techniques (Zhang & Shan, 2008; Chow, 1970; Frignana & Bellucci, 2004). Sutherland and Tolosa (2000) studied the movement of trace element in humid sediments and found that Pb, Zn and Cu were more mobile in the lower level (7.2-10cm) of the sediment than at the surface level. Once trace elements are in dry sediments, their behavior will be similar to terrestrial environments (Sutherland, 2000; Sutherland et al., 2003; Andrews & Sutherland, 2004).

Sutherland (2000) used mass loading and mass per area enrichment ratios (MAER) to study the impact of trace elements in sediments. It was found that input trace elements such as Pb were 4 to 5 times higher in sediments affected by anthropogenic activities. This statistical technique showed that certain forms of metals tend to be more enriched than background metals, hence, presenting a higher anthropogenic signal in sediments (Sutherland, 2000).

Study of enrichment ratios (ER) has been applied to agricultural fields globally to evaluate anthropogenic enhancement. While agricultural fields are different from mining sites, they both result in significant alteration to the local sediment environment and therefore are a good comparison. In areas where a long history of intensive agriculture is documented, dating back more than 100 years, change in environmental conditions is easily observed (Zhang & Shan, 2008). Researchers used enrichment factors to evaluate agricultural impacts on sediments in China to assess anthropogenic activities such as fertilizer usage (Zhang & Shan, 2008; Agrawal et al., 1981; Qin & Chen, 1996; Yin et al., 2006). It was found that fertilization over long periods of time will impact the sediment environment with the use of phosphorus fertilizers-containing trace metals such as Cd (Inaba et al., 2006; Zhang & Shan, 2008). As shown by Zhang and Shan (2008) with fertilizers, certain concentrated geogenic trace metals (i.e. Cd) tend to bioaccumulate over time, and may be available for plant uptake.

Environmental models have been widely used to predict mass movement of sediments, as "moving waves" at mine sites (Pickup et al., 1983). Input variables to models have included sediment velocity, dispersion

coefficients, and the mean distributions of loads to simulate the movement of the wave. Pickup et al. (1983) found that modeling the moving wave of sediment with a dispersion model could adequately simulate sediment transport down slope in an area influenced by surface waters.

Pickup et al. (1983) utilized two models for predicting the movement of sediments down stream with one allowing for the distribution of sediment velocity through dispersion co-efficient, the other used a normal mass conservation approach. Their models included datasets collected over a period 43 months, with data from 7 cross-sections of the Kawerong River, Papua-New Guinea, which is polluted with wastes from a copper mine. They were able to predict the distance and concentrations along the river based on precipitation effects.

Over the years, a number of other studies have used modeling techniques to assist in the interpretation of mine waste (Yager & Stanton, 2000; Ge et al., 2005; Mackenzie & Pulford, 2002; Stanton, 2000). Models using topography, precipitation, geophysics, geochemistry, hydrology, and historical records provide the best quantitative results that can be derived from the investigation of historical mine sites. Many of these models (e.g. Coulthard & Macklin, 2003; Stanton, 2000) incorporate numerical and analytical techniques for evaluating the transport and fate of contaminants in surface sediments. Some of the models used today for evaluating contaminant movement are Tracer, Gaussian Distribution Method, and numerical and analytical models to predict contaminant transport and fate in surface sediments.

Coulthard and Macklin (2003) used numerical and analytical approaches for predicting long term contamination issues in rivers and streams in historical mining environments. The approach, Tracer, uses historical mine records, and topographic and hydrologic maps to predict the movement of contaminated sediments in relation to their sources. This approach provides information concerning the possible extent of contamination, and, what Coulthard and Macklin (2003) refer to as "hot spots". Hot spots, as described by Coulthard and Macklin (2003), are secondary sources of contamination that occur some distance from the original source when deposited during periods of high flow. These hot spots can impact other areas that were not historically contaminated (Ge et al., 2005; Coulthard & Macklin, 2003; Mackenzie & Pulford, 2002).

Identifying natural versus anthropogenic input sources of contaminants aids in understanding environmental problems associated with abandoned mine sites. For example, understanding potential sources of Hg will provide information concerning input point source pollution rather then geogenic sources (Coulthard & Macklin, 2003). Studies have evaluated the differences between anthropogenic Hg and geogenic sources in order to gain a better understanding of mine related point- and non-point source pollution. Hg from anthropogenic sources has been found in higher concentration in sediments compared to geogenic sources (Engle et al., 2001; Stanton, 2000; Yager & Stanton, 2000). This higher concentration is due to the use of Hg in large amounts versus geogenic sources, which tend to be much lower and have more time to disperse before moving with transported sediments (Stanton, 2000).

Modeling sediment transport has been used to measure the distance from geogenic and anthropogenic sources in wet climates that impact downstream environments (Engle et al., 2001). Models have been used to illustrate the effects of atmospheric precipitation on Hg sources and the resulting erosion from sediments and alluvial deposits over time in wet climates (Engle et al., 2001; Stanton, 2000; Yager & Stanton, 2000). The Gaussian Distribution Method has been used to calculate the average daily emission of metals from lithologic units with the average metal flux from a given area. It was shown that 89% of Hg released into the environment will come from naturally enriched sources (geogenic) and 11% will most likely originate from anthropogenic sources (Engle et al., 2001). These methods are useful for predicting sources of contamination. However, they are limited in that the variables used will only reflect the data collected and will not account for unknown variables, i.e., most such models are site specific.

Siltation is arid and hyperarid regions are not a major concern because there is little precipitation however, when precipitation occurs, it will transport large amount of sediments due to mass transport rather than a constant siltation affect on nearby surface waters. When transport occurs in arid and hyperarid regions it is usually associated with a short but intense storm event that transports materials into the wider environment, potentially impacting the food web.

7. Biological Environment

7.1 Potential for Flora, Fauna and Human Health Affects

One of the biggest concerns with the transport of contaminants is when it enters the food web. Impacts of mining waste extend beyond the potential effects on water and sediments and ultimately into the biological environment. Biological impacts include the health and well being of plants, animals, and humans exposed to contaminated

waters, sediments and food supplies. The location of mining facilities and their associated contaminated soils, sediments, and wastes, in relation to population and farming, can have an adverse impact on the overall environment through the transport of trace metals (Appanna, 1991; Nicholson et al., 2003).

Mining activity sparked population growth in towns and cities that supplied the labor to mines and milling facilities. Today, many of these towns and cities are larger in population, although the nearby mining and milling sites have been abandoned and left unattended for decades. It should be considered, however, that abandoned mining areas adjacent to towns could still pose potential risks locally to flora, fauna, and ultimately human health (Nicholson et al., 2003; Sierra et al., 2009). The environmental and biological impacts of mining wastes on ecosystems have been studied to determine what, if any, ill effects humans have suffered in areas surrounding abandoned mining sites (Lee et al., 1998; Suner et al., 1999; Appleton et al., 2000).

Thus it appears that siltation and is very low, the mobility and transport of metals and other contaminants are limited due to environmental factors (i.e. arid and hyperarid regions). Although it has been shown that mining can produce adverse effects beyond the mining area itself, the impact of mining activities on local vegetation, fauna, and, eventually, human health in arid environments is not well documented. To understand effects from vegetation, it is important to understand the mechanism in which vegetation becomes a source of contamination. To illustrate, an animal that has ingested contaminated water or food or inhaled contaminated airborne material, can roam long distances before human consumption.

7.2 Uptake and Effects on Flora

Base metal and uranium mining sites have been investigated for contaminants uptake by local vegetation within the continental United States and abroad (Kipp et al., 2009; Kovalchuk et al., 2005). These investigations have studied the uptake of Cu, Ni, Fe, Co, Zn, U, and Pb in vegetation and animal tissues in the vicinity of mining areas. Plants around mining sites have been found to contain higher levels of trace elements (e.g. Pb, As) correlating to concentrations in local sediments. It was found that plants with the highest level of trace elements (e.g. Pb) tended to have broader leaves than small leaf plants. It was found that higher levels of metals is possible in grasses, surrounding mining sites (Nicholson et al., 2003; Vazquez et al., 2008; Wang et al., 2007). Literature has shown that bioaccumulation of As and resulting phytotoxicity to lower plant species were observed in wet mining areas that had several metal-tolerant plant species (e.g., *Angophora floribunda*, *Cassinia laevis*, and *Chrysocephalum apiculatum*) that colonized the periphery of the site (Clemente et al., 2005; Boularbah, et al., 2006; Han et al., 2006). Bermejo et al. (2003) investigated Pb mining in the Sierra Madrona Mountains Spain. Their study also showed higher concentrations of trace elements (Pb, Zn, Cd, Cu, As, and Se) in plants where trace element concentration was high in sediments (Bermejo et al., 2003). However, higher concentrations of trace elements in vegetation were more prevalent in areas like stream sediments near tailings dumps than in areas not impacted by mining (Reglero et al., 2008).

7.3 Uptake and Effects on Fauna

Animals (fauna) are likely to suffer greater affects through the food chain by the ingestion of flora. Researchers identified trends in metal uptake in vegetation and animals from trace element enriched substrata, leachates, and surface water sources (Lee, 2001; Pankakoski et al., 1994; Sánchez-Chardi & Nadal, 2007). It has been shown that metals such as Cd are of great concern because of its bioaccumulation from the consumption of vegetation (Krishnamurti et al., 2005).

Uptake of metals can impact the environment by adversely affecting fauna as a result of grazing on local flora in affected areas (Gammon, 1970; Pagenkopf et al., 1974). Local vegetation near streams and rivers can be impacted when mine waste containing dissolved elements (e.g Cu, Zn, Pb, and As) enters water-bodies (Lewis, 1977; Gammon, 1970; Pagenkopf et al., 1974; Warnick & Bell, 1969). An influx of dissolved trace elements can affect organisms like insect larvae, bivalve mollusks, and microorganisms, which in turn affect larger organisms up the food chain (Wright & Zamuda, 1987; Qiu et al., 2007). As shown in literature (Krishnamurti et al., 2005; Reglero et al., 2008; Moreno-Jimenez et al., 2009), bioaccumulation in local vegetation will directly affect local fauna by the transfer of metals from one organism to another. The level of tolerance for specific metal accumulation is species-specific, resulting in the further transfer of metals from one species to another with the impact being species specific (Moreno-Jimenez et al., 2009).

Studies have shown that Cu, Ni, and Pb are likely to be found in the stomach linings of local fauna (goats), whereas Ni and Fe are more prevalent in skin and fur of some fauna (Cloutier et al., 1985). Studies have shown, for example, that high concentration of Pb found in animals near mining sites can be caused by animals foraging for plant material that are known to uptake Pb into their tissues (Chopin & Alloway, 2007; Reglero et al., 2008). Studies (Reglero et al., 2008) showed that in lead mining areas of the Sierra Madrona Mountains 13 species of

plants contaminated with trace metals resulted in a transfer to local red deer (*Cervus elaphus*) foraging in the area. Reglero et al. (2008) illustrated that in mining impacted soil con tainingh Pb and As at 100 times background concentrations, leafy plants contained Pb and As concentrations 3 times their background levels (Reglero et al., 2008). They further examined the livers of local red deer and found that Pb and As concentrations were 12 times higher than livers from control areas not impacted by mining.

7.4 Uptake and Effects on Human Health

Contamination of the environment and resulting impact to human health is well known and has been noted as far back as 2000 years in the Roman Empire. In ancient Rome, Romans used lead, a documented neurotoxin, for making water pipes, cooking utensils, water tanks, and storage vessels (Bishop, 1989). Lead water pipes were used in most major cities in the empire. In ancient Rome, wine was contaminated with Pb from as many as 14 sources during its preparation. Romans also used Pb as a preservative and a flavor enhancer (Bishop, 1989; Nriagu, 1990). Rome was not the only culture that was contaminating itself with toxic metals, as even the Christian sacramental cups of wine were made of Pb or leaded bronze (Nriagu, 1990). Other trace elements (e.g. Hg, Cd, As, Se, Cr) are well-documented as toxic to ecosystems and humans, and modern health agencies such as the World Health Organization and USEPA have published guidelines for limiting ingestions of these elements in food and drinking water.

Throughout most of the 20th century, the mining in the United States has resulted in impacts on the local and wider environment, such as the USEPA Superfund site from Coeur d'Alene, Idaho that dates back to the 1870s. The Coeur d'Alene area had more than 100 mining and mill operations producing Ag, Pb, Zn, and other metals (USEPA, 2005). According to the USEPA one of the most notorious sites was the Bunker Hill Mine and Smelting Complex located in Kellogg, Idaho. Many of the mine tailings in the region were discharged directly to the Coeur d'Alene River and its tributaries until 1968 when the practice was banned by the US Government. Smelting operations at Bunker Hill discharged large quantities of sulfur dioxide waste containing Pb and other metals that affected local communities and the environment (USEPA, 2005).

The Bunker Hill site was investigated to evaluate the impact to water, ecological and human health downstream from the source of pollution to the river. The USEPA studied the human health risk of Pb intake by local population to determine future population impacts to children from the groundwater and vegetation (USEPA, 2005). It was found that downstream and adjacent farms to the Coeur d'Alene River contained Pb 5 to 10 times higher than regulatory levels (0.05 mg kg^{-1}) as a result of contaminated soils and sediments. The pollution was found a far as 80 km downstream, impacting more than 40 square km (USEPA, 2005).

The effects that mining sites have on surrounding environments also have been studied in great detail in Japan (Vahter et al., 2007). Inaba et al. (2006) and others reported that the disposal of mine waste from a nearby mining operation impacted the local chain (Yamagami et al., 2006). They found that Cd-contaminated waste water from mining operations was used for growing rice (Vahter et al., 2007; Inaba et al., 2006; Yamagami et al., 2006; Takagi et al., 2004). It was found that locals that consumed 3.1 g to 3.8 g of Cd in rice over a lifetime, developed mild to severe mitochondrial dysfunction called Itai–itai disease (Inaba et al., 2005). Studies have shown that over time bioaccumulation of trace metals (e.g. Cd) in tissues of local indigenous peoples affected their tubular epithelial cells after 80-weeks of exposure by ingestion (Takagi et al., 2004; National Academy of Science, 1973). This disease was found to be the direct result of Cd-enriched mine waste water that discharged to local rivers, and ultimately used in the rice fields.

Impacts of mining activities on local flora have shown that vegetation uptake will transfer contaminants to fauna and ultimately up the food chain. The uptake of contaminants into local vegetation will eventually impact humans due to biomagnifications and that the true effect is based on the specific species rather than the actual dose or level of contaminants found in the food chain.

8. Summary and Conclusions

Although it is well documented that mining contamination in wetter climates has the possibility of migrating long distances by water transport, there has been limited research in arid and hyperarid regions. Studies have shown that mining activities can influence the mobilization of geogenic metals into the surrounding environment. Furthermore, trace metals (Cu, Zn, Pb, Ni, Cd, and Cr) have been the focus of interest because of their long residence times and significant toxicities to biota in wet climates. Evaluating sediments contaminated with geogenic and anthropogenic inputs in dry regions will provide valuable data on the extent, origin, and distribution of contaminants from source areas in hyperarid regions. Although there is limited research in hyperarid climates, studies have shown that mining is of great concern because of possible transport. It has become quite clear that trace metals and contaminates are mobile in both the particulate and soluble phase and

are transported long distances from source areas contained in sediments. Thus it is imperative that research be conducted in these arid and hyperarid regions where significant amounts of mining and milling have occurred to further understanding in the mechanism of transport in similar environments.

References

Agrawal, Y. K., Patel, M. P., & Merh, S. S. (1981). Lead in soil and plants: Its relationship to traffic volume and proximity to highway (lalbag, Baroda City). *International J. of Environ. Studies, 16*, 222-224. http://dx.doi.org/10.1080/00207238108709873

Al-Busaidi, A., Cookson, P., & Yamamoto, T. (2005). Methods of pH determination in calcareous soils: use of electrolytes and suspension effect. Aust. *J. Soil Res., 43*, 541-545. http://dx.doi.org/10.1071/SR04102

Andrews, S., & Sutherland, R. A. (2004). Cu, Pb, and Zn contamination in Nuuanu watershed, Oahu, Hawaii. *Science of the Total Environment, 324*, 173-182. http://dx.doi.org/10.1016/j.scitotenv.2003.10.032

Appanna, V. (1991). Impact of heavy metals on an arctic rhizobium. *Bull Environ. Contam. Toxical, 46*, 450-455. http://dx.doi.org/10.1007/BF01688946

Appleton, J., Lee, K. M., Kapusta, K. S., Damek, M., & Cooke, M. (2000). The heavy metal content of the teeth of the bank vole (Clethrionomys glareolus) as an exposure marker of environmental pollution in Poland. *Environmental Pollution, 110*, 441-449. http://dx.doi.org/10.1016/S0269-7491(99)00318-8

Ashley, P. M., Lottermoser, B. G., & Chubb, A. J. (2003). Environmental geochemistry of the Mt. Perry copper mines area, SE Queensland, Australia. *Geochemistry: Exploration, Environment, Analysis, 3*, 345-357. http://dx.doi.org/10.1144/1467-7873/03-014

Ashley, P. M., Lottermoser, B. G., Collins, A. J., & Grant, C. D. (2004). Environmental geochemistry of the derelict Webbs Consols mine, New South Wales, Australia. *Environmental Geology, 46*, 591-604. http://dx.doi.org/10.1007/s00254-004-1063-7

Berger, A. C., Bethke, C. M., & Krumhansl, J. L. (2000). A process model of natural attenuation in drainage from a historic mining district. *Applied Geochemistry, 15*(5), 655-666. http://dx.doi.org/10.1016/S0883-2927(99)00074-8

Bermejo, J. C. S., Beltrán, R., & Ariza, J. L. G. (2003). Spatial variations of heavy metals contamination in sediments from Odiel River (Southwest Spain). *Environmental International, 29*, 69-77. http://dx.doi.org/10.1016/S0160-4120(02)00147-2

Besser, J. M., & Leib, K. J. (2000). *Toxicity of Metals in Water and Sediment to Aquatic Biota*. Chapter E19: Integrated Investigations of Environmental Effects if Historical Mining in the Animas River Watershed, San Juan County, Colorado.

Besser, J. M., Finger, S. E., & Church, S. E. (2008). Impacts of Historical Mining on Aquatic Ecosystems–An Ecological Risk Assessment. *Integrated investigations of Environmental Effects of Historical Ming in the Animas River Watershed, San Juan County, Colorado, 1651*, 87-106.

Blowes, D. W., Robertson, W. D., Ptacek, C. J., & Merkley, L. C. (1994). Removal of agricultural nitrate from tile-drainage effluent using in-line bioreactors. *J. Contam. Hydrol. 15*, 207-221. http://dx.doi.org/10.1016/0169-7722(94)90025-6

Bonzongo, J. C., Heim, K. J., Warwick, J. J., & Lyons, W. B. (1996). Mercury levels in surface waters of the Carson River-Lahontan reservoir system, Nevada: Influence of historic mining activities. *Environmental Pollution, 92*(2), 193-201. http://dx.doi.org/10.1016/0269-7491(95)00102-6

Boularbah, A., Schwartz, C., Bitton, G., Aboudrar, W., Ouhammou, A., & Morel, J. L. (2006). Heavy Metal contamination from mining sites in South Morocco: 2. Assessment of metal accumulation and toxicity in plants. *Chemosphere, 63*, 811-817. http://dx.doi.org/10.1016/j.chemosphere.2005.07.076

Boyle, D. R., & Smith, C. N. (1994). Mobilization of mercury from a Gossan tailings pile, Murry Brook Precious metal vat leaching operation, New Brunswick, Canada. *International Land Reclamation and mine Drainage Conference and the Third International Conference on the Abatement of acidic Drainage*, Pittsburgh, PA, April 24-29, 1994.

Breteler, R. J., Valiela, I., & Teal, J. M. (1981). Bioavailability of mercury in several north-eastern U.S. *Spartina* ecosystems. *Estuarine Coastal and Shelf Science, 12*(2), 155-166. http://dx.doi.org/10.1016/S0302-3524(81)80093-X

Camm, G. S, Glass, H. J., Bryce D. W., & Butcher, A. R. (2004). Characterization of a mining-related arsenic-contaminated site, Cornwall, UK. *Journal of Geochemical Exploration, 82*(1-3), 1-15. http://dx.doi.org/10.1016/j.gexplo.2004.01.004

Camm, G. S., Butcher, A. R., Pirrie, D., Hughes, P. K., & Glass, H. J. (2003). Secondary mineral phase associated with a historic arsenic calciner identified using automated scanning electron microscopy; a pilot study from Cornwall, UK. *Minerals Engineering, 16*(11), 1269-1277. http://dx.doi.org/10.1016/j.mineng.2003.07.005

Chang, J.-S., Yoon, I.-H., & Kim, K.-W. (2009). Heavy metal and arsenic accumulating fern species as potential ecological indicators in As-contaminated abandoned mines. *Ecological Indicators, 9*, 1275-1279. http://dx.doi.org/10.1016/j.ecolind.2009.03.011

Chen, Y., Bonzongo, J. C., & Miller, G. C. (2006). Levels of Methylmercury and controlling factors in surface sediments of the Carson River system, Nevada. *Environmental Pollution, 92*(3), 281-287. http://dx.doi.org/10.1016/0269-7491(95)00112-3

Chow, T. J. (1970). Lead Accumulation in roadside soil and grass. *Nature, 225*, 295-296. http://dx.doi.org/10.1038/225295a0

Churchill, R. (1999). Insights into California mercury production and mercury availability for the gold mining industry from the historical record. *Geol. Soc. Am. Abs., 21*, 45.

Clemente, R., Walker, D. J., & Bernal, M. P. (2005). Uptake of heavy metals and As by Brassica Juncea grown in a contaminated soil in Aznalcóllar (Spain): The effect of soil Amendments. *Environmental Pollution, 138*(1), 46-58. http://dx.doi.org/10.1016/j.envpol.2005.02.019

Cloutier, N, Clulow, F. V., Lim, T. P., & Dave, N. K. (1985). Metal (Cu, Ni, Fe, Co, Zn, Pb) and Ra-226 levels in meadow voles Microtus Pennsylvanicus living on nickel and uranium mine tailings in Ontario, Canada: environmental and tissue levels. *Environmental Pollution Series B, Chemical and Physical, 10*(1), 19-46. http://dx.doi.org/10.1016/0143-148X(85)90014-X

Conaway, H. H., Watson, E. B., Flanders, J. R., & Flegal, A. R. (2004). Mercury deposition in a tidal marsh of south San Fransisco Bay downstream of the historic New ALdaden mining district, California. *Marine Chemistry, 90*(1-4), 175-184. http://dx.doi.org/10.1016/j.marchem.2004.02.023

Conko, K. M., Landa, E. R., Kolker, A., Kozlov, K., Gibb, H. J., Centeno, J. A. . . . Panov, Y. B. (2013). Arsenic and mercury in the soils of an industrial city in the Donets Basin, Ukraine. *Soil and Sediment Contamination: An Int. J., 22*(5), 574-593. http://dx.doi.org/10.1080/15320383.2013.750270

Coulthard, T. J., & Macklin, M. G. (2003). Modeling long-term contamination in river systems from historical metal mining. *Geology, 31*(5), 451-454. http://dx.doi.org/10.1130/0091-7613(2003)031<0451:MLCIRS>2.0.CO;2

Craw, D. (2005). Potential anthropogenic mobilization of mercury and arsenic from soils on mineralized rocks, Northland, New Zealand. *Journal of Environmental Management, 74*, 283-292. http://dx.doi.org/10.1016/j.jenvman.2004.10.005

Da Silva, E. F., Fonseca, E. C., Matos, J. X., Patinha, C., Reis, P., & Oliveira, J. M. S. (2005). The effect of unconfined mine tailings on the geochemistry of soils, sediments and surface waters of the Lousal area (Iberian Pyrite Belt, Southern Portugal). *Land Degrad. Develop., 16*, 213-228. http://dx.doi.org/10.1002/ldr.659

Davis, B. E. (1976). Mercury Content of Soils in Western Britain with Special Reference to Contamination from Base Metal Mining. *Geoderma, 16*(3), 183-192. http://dx.doi.org/10.1016/0016-7061(76)90020-3

Davis, B. E., & White, H. M. (1981). Environmental Pollution by Wind Blown Lead Mine Waste: A Case Study in Wales, U.K. *The Science of the Total Environment, 20*(1), 57-74. http://dx.doi.org/10.1016/0048-9697(81)90036-X

Dennis, I. A., Macklin, M. G., Coulthard T. J., & Brewer, P. A. (2002). The impact of the October-November 2000 floods on contaminated metals dispersal in the River Swale catchment, North Yorkshire, U.K. *Hydrological Processes, 17*(8), 1641-1657. http://dx.doi.org/10.1002/hyp.1206

Dolenec, T., Serefimvski, T., Dobnnikar, M., Tasev, G., & Dolenec, M. (2005). Mineralolgical and heavy metal signature of acid mine drainage impacted paddy soil from the western part of Kocani field (Macedonia). *Materials and Environment, 52*, 397-402.

Earman, S. (1996). *The Impact of Non-point Source Pollution from Mining Waste on Water Quality, Elko County, Nevada* (Unpublished Master thesis in the Geology Department, University of Nevada, Las Vegas).

Edwards, D. P., Lamarque, J., Attié, J., Emmons, L., Richter, A., Cammas, J., . . . Burrows, J. (2003). Tropospheric ozone over the tropical Atlantic: A satellite perspective. *J. Geophys. Res., 108*(D8), 4237. http://dx.doi.org/10.1029/2002JD002927

Edwards, K., Macklin, M. & Taylor, M. (1997). Historic metal mining inputs to Tees River basin. *Science of the Total Environment, 194-195*, 437-445. http://dx.doi.org/10.1016/S0048-9697(96)05381-8

Engle, M. A., Gustin. M. S., & Zhang, H. (2001). Quantifying natural source mercury emissions from the Ivanhoe Mining District, north-central Nevada, U.S.A. *Atmospheric Environment, 35*(23), 3987-3997. http://dx.doi.org/10.1016/S1352-2310(01)00184-4

Feng, M., Schrlau, J. E., Snyder, R., Snyder, G. H., Chen, M., Cisar, J. L., & Cai, Y. (2005). Arsenic transport and transformation associated with MSMA application on a golf course green. *J. Agric Food Chem, 53*, 3556-3562. http://dx.doi.org/10.1021/jf047908j

Figueroa, J. A. L., Wrobel, K., Afton, S., Caruso, J. A., Corona, J. F. G., & Wrobel, K. (2008). Effect of some heavy metals and soil humic substances on the phytochelatin production in wild plants from silver mine areas of Guanajuato, Mexico. *Chemosphere, 70*, 2084-2091. http://dx.doi.org/10.1016/j.chemosphere.2007.08.066

Frignana, M., & Bellucci, L. G. (2004). Heavy metals in marine coastal sediments: Assessing source, flux, history and trends. *Ann. Chim., 94*, 479-486. http://dx.doi.org/10.1002/adic.200490061

Fuente, C. de la, Clemente, R., & Bernal, M. P. (2007). *Changes in metal speciation and pH in olive processing waste and sulphur-treated contaminated soil*. Ecotoxicology and Environmental Safety, Corrected Proof.

Gammon, J. R. (1970). *Effect of Inorganic Sediment on Stream Biota*. USEPA Rep. U.S. Government Printing Office, Washington D.C.

Ge, Y., MacDonald, D., Sauvé, S., & Hendershot, W. (2005). Modeling of Cd and Pb speciation in soil solutions by WinHumicV and NICA-Donnan model. *Environmental Modeling & Software, 20*(3), 353-359. http://dx.doi.org/10.1016/j.envsoft.2003.12.014

Gemici, Ü., & Oyman, T. (2003). The influence of the abandoned Kalecik, Hg mine on water and stream sediments (Karaburun, Izmir, Turkey). *The Science of the Total Environment, 312*, 155-166. http://dx.doi.org/10.1016/S0048-9697(03)00008-1

Giuliano, V., Pagnanelli, F., Bornoroni, L., Toro, L., & Abbruzzese, C. (2007). Toxic elements at a disused mine district: Particle size distribution and total concentration in steam sediments and mine tailings. *Journal of Hazardous Materials, 148*, 409-418. http://dx.doi.org/10.1016/j.jhazmat.2007.02.063

Gräfe, M., Singh, B., & Balasubramanian, M. (2007). Surface speciation of Cd(II) and Pb(II) on kaolinite by XAFS spectroscopy. *Journal of Colloid and Interface Science, 315*(1), 21-32. http://dx.doi.org/10.1016/j.jcis.2007.05.022

Gräfe, M., Tappero, R. V., Marcus, M. A., & Sparks, D. L. (2008). Arsenic speciation in multiple metal environments II. Micro-spectroscopic investigation of a CCA contaminated soil. *Journal of Colloid and Interface Science, 321*(1), 1-20. http://dx.doi.org/10.1016/j.jcis.2008.01.033

Greene, J. M. (1975). *Life in Nelson Township, Eldorado Canyon, and Boulder City* (Unpublished Master Thesis, History Department, University of Nevada, Las Vegas).

Gupta, P. K., Jha, A. K., Koul, S., Sharma, P., Pradhan, V., Gupta, V., . . . Singh, N. (2007). Methane and Nitrous Oxide Emission from Bovine Manure Management ractices in India. *Journal of Environmental Pollution 146*(1), 219-224. http://dx.doi.org/10.1016/j.envpol.2006.04.039

Han, F. X., Su, Y., Monts, D. L., Waggoner, C. A., & Plodinec, M. J. (2006). Binding, distribution, and plant uptake of mercury in a soil from Oak Ridge, Tennessee, USA. *Science of The Total Environment, 368*(2-3), 753-768. http://dx.doi.org/10.1016/j.scitotenv.2006.02.026

Harikumar, P. S., & Jisha, T. S. (2010). Distribution pattern of trace metal pollutants in the sediments of an urban wetland in the southwest coast of India. *International Journal of Engineering Science and Technology, 2*(5), 840-850.

Harris, D. L., Lottermoser, B. G., & Duchesne, J. (2003). Ephemeral acid mine drainage at the Montalbion silver

mine, north Queensland. *Australian Journal of Earth Sciences, 50*, 797-809. http://dx.doi.org/10.1111/j.1440-0952.2003.01029.x

Harrison, R. M., & Laxen, D. P. H. (1981). *Lead Pollution–Causes and Control*. UK: Cambridge University Press. http://dx.doi.org/10.1007/978-94-009-5830-2

Hernández, L. M., Gómara, B., Fernández, M., Jiménez, B., González, M. J., Baos, R., . . . Montoro, R. (1999). Accumulation of heavy metals and As in wetland birds in the area around Donñana National Park affected by the Aznalcollar toxic spill. *The Science of The Total Environment, 242*(1-3), 293-308. http://dx.doi.org/10.1016/S0048-9697(99)00397-6

Horowitz, A. J. (2008). Contaminated Sediments: Inorganic Constituents. In M. G. Anderson (Ed.), *Encyclopedia of Hydrological Sciences*. Wiley Inter-science. http://dx.doi.org/10.1002/0470848944.hsa317

Horowitz, A. J., & Stephens, V. C. (2008). The effects of land use on fluvial sediment chemistry for the conterminous U.S.—Results from the first cycle of the NAWQA Program: Trace and major elements, phosphorus, carbon, and sulfur. *Science of The Total Environment, 400*, 290-314. http://dx.doi.org/10.1016/j.scitotenv.2008.04.027

Horowitz, A. J., Elrick, K. A., & Callender, E. (1988). The Effect of Mining on The Sediment-Trace Element Geochemistry of Cores from The Cheyenne River Arm of Lake Oahe, South Dakota (USA). *Chemical Geology, 67*, 17-33. http://dx.doi.org/10.1016/0009-2541(88)90003-4

Horowitz, A. J., Elrick, K. A., & Cook, R. B. (1993). Effect of Mining and Related Activities on The Sediment Trace Element Geochemistry of Lake Coeur D'Alene, Idaho, USA. Part 1: Surface Sediments. *Hydrological Processes, 7*, 403-423. http://dx.doi.org/10.1002/hyp.3360070406

Hsu, S.-C., Liu, S. C., Jeng, W.-L., Lin, F.-J., Huang, Y.-T., Lung, S.-C. C., . . . Tu, J.-Y. (2005). Variations of Cd/Pb and Zn/Pb ratios in Taopei aerosols reflecting long-range transport or local pollution emissions. *Science of the Total Environment, 347*(1-3), 111-121. http://dx.doi.org/10.1016/j.scitotenv.2004.12.021

Hudson-Edwards, K., Macklin, M., & Taylor, M. (1997). Historic metal mining inputs to Tees river sediment. *Science of The Total Environment, 194-195*, 437-445. http://dx.doi.org/10.1016/S0048-9697(96)05381-8

Hughes, M. K., & Diaz, H. F. (2008). Climate variability and change in the drylands of Western North America. *Global and Planetary Change, 64*, 111-118. http://dx.doi.org/10.1016/j.gloplacha.2008.07.005

Hutchinson, I., & Ellison, R. (1992). *Mine Waste Management*. London: Lewis Publishers.

Hutchinson, T. C., & Meema, K. M. (1987). Lead Mercury, Cadmium and Arsenic in the Environment. *Scientific Committee on Problems of the Environment (SCOPE), (Series): 31*.

Inaba K., Murakami, S., Suzuki, M., Nakagawa, A., Yamashita, E., Okada, K., & Ito, K. (2006). Crystal structure of the DsbB=DsbA complex reveals a mechanism of disulfide bond generation. *Cell, 127*, 789-801. http://dx.doi.org/10.1016/j.cell.2006.10.034

Ismail, I., Abdel-Monem, N., Fateen, S.-E., & Abdelazeem, W. (2009). Treatment of a synthetic solution of galvanization effluent via the conversion of sodium cyanide into an insoluble safe complex. *Journal of Hazardous Materials, 166*, 978-983. http://dx.doi.org/10.1016/j.jhazmat.2008.12.005

Jambor, J. L., Martin, A. J., & Gerits, J. (2009). The post-depositional accumulation of metal-rich cyanide phases in submerged tailings deposits. *Applied Geochemistry, 24*(12), 2256-2265 http://dx.doi.org/10.1016/j.apgeochem.2009.09.011

James, L. A. (1991). Quartz concentration as an index of sediment mixing: hydraulic mine-tailings in the Sierra Nevada, California. *Geomorphology, 4*(2). 125-144. http://dx.doi.org/10.1016/0169-555X(91)90024-5

Jonasson, I. R., & Boyle, R. W. (1972). Geochemistry of Mercury and origins of natural contamination of the environment. *CIM Bulletin, 65*(717), 30-49.

Khalid, A. M., El-Tawil, A. H., Ashy, M., & Elbeih, F. K. (1981). Constituents of local plants. Part-8: Distribution of some coumarins in plants of different plant families grown in Saudi Arabia. *Pharmazia, 36*(8), 569-571.

Kim, C. S., Brown Jr., G. E., & Rytuba, J. J. (2000). Characterization and speciation of mercury-bearing mine waste using X-ray absorption spectroscopy. *The Science of the Total Environment, 261*(1-3), 157-168. http://dx.doi.org/10.1016/S0048-9697(00)00640-9

Kim, K. W., Lee, H. K., & Yoo, B. C. (1998). The environmental imact of gold mines in the Yugu-Kwangcheon

Au-Ag metallogenic province, Republic of Korea. *Environmental Technology, 19*, 291-298. http://dx.doi.org/10.1080/09593331908616683

Kim. C. S., Rytuba, J. J. & Brown Jr., G. E. (2004). Geological and Anthropogenic factors influencing mercury speciation in mine waste: an EXAFS spectroscopy study. *Applied Geochemistry, 19*(3), 379-393. http://dx.doi.org/10.1016/S0883-2927(03)00147-1

Kipp, G. G., Stone, J. J., & Stetler, L. D. (2009). Arsenic and uranium transport in sediments near abandoned uranium mines in Harding County, South Dakota. *Applied Geochemistry, 24*, 2246-2255. http://dx.doi.org/10.1016/j.apgeochem.2009.09.017

Kovalchuk, I., Titov, V., Hohn, B., & Kovalchuk, O. (2005). Transcriptome profiling reveals similarities and differences in plant responses to cadmium and lead. *Mutation Research, 570*, 149-161. http://dx.doi.org/10.1016/j.mrfmmm.2004.10.004

Krishnamurti, G. S. R., McArthur, D. F. E., Wang, M. K., Kozak, L. M., & Huang, P. M. (2005). Chapter 7–Biogeochemistry of soil cadmium and the impact of terrestrial food chain contamination. *Biogeochemistry of Trace Elements in the Rhizosphere*, 197-257. http://dx.doi.org/10.1016/B978-044451997-9/50009-X

Kwong, Y. T. J., Beauchemin, S., Hossain, M. F., & Gould, W. D. (2007). Transformation and mobilization of arsenic in the historic Cobalt mining camp, Ontario, Canada. *Journal of Geochemical Exploration, 92*(2-3), 133-150. http://dx.doi.org/10.1016/j.gexplo.2006.08.002

Lecce, S., & Pavlowsky, R. T. (2001). Use of mining-contaminated sediment tracers to investigate the timing and rates of historical flood plain sedimentation. *Geomorphology, 38*, 85-108. http://dx.doi.org/10.1016/S0169-555X(00)00071-4

Lechler, P. J., Miller, J. R., Hsu, L.-C., & Desilets, M. O. (1997). Mercury mobility at the Carson River Superfund Site, west-central Nevada, USA: Interpretation of mercury speciation data in mill tailings, soils, and sediments. *Journal of Geochemical Exploration, 58*(2-3), 259-267. http://dx.doi.org/10.1016/S0375-6742(96)00071-4

Lee, J. S., Chon, H. T., Kim, J. S., Kim, K. W., & Moon, H. S. (1998). Enrichment of potentially toxic elements in areas underlain by black shales and slates in Korea. *Environmental Geochemistry and Health, 20*, 135-147. http://dx.doi.org/10.1023/A:1006571223295

Leinz, R. W., Sutley, S. J., Desborough, G. A., & Briggs, P. H. (2006). *An Investigation of the Partitioning of Metals in Mine Wastes Using Sequential Extractions*. U.S. Geological Survey.

Lewis, M. (1977). *Aquatic Inhabitants of a Mine Waste Stream in Arizona*. United States Forest Service; U.S. Department of Agriculture, Issue 349.

López, J. M., Callén, M. S., Murillo, R.. García, T., Navarro, M. V., de la Cruz. M. T.. & Mastral, A. M. (2005). Levels of selected metals in ambient air PM_{10} in an urban site of Zaragoza (Spain). *Environmental Research, 99*, 58-67. http://dx.doi.org/10.1016/j.envres.2005.01.007

Lottermoser, B. G., & Ashley, P. M. (2005). Tailings dam seepage at the rehabilitated Mary Kathleen Uranium mine, Australia. *Journal of Geochemical Exploration, 85*, 119-137. http://dx.doi.org/10.1016/j.gexplo.2005.01.001

Lottermoser, B. G., Ashley, P. M., & Lawie, D. C. (1999). Environmental geochemistry of the Gulf Creek copper mine area, north-eastern New South Wales, Australia. *Environmental Geology, 39*(1). http://dx.doi.org/10.1007/s002540050437

Lu, W., Ma Y., & Lin, C. (2012). Spatial variation and fractionation of bed sediment-borne copper, zinc, lead and cadmium in a stream system affected by acid mine grainage. *Soil and Sediment Contamination; An Int. J. 21*(5), 831-849. http://dx.doi.org/10.1080/15320383.2012.697933

Luo, X.-S., Zhou, D.-M., Liu, X.-H., & Wang, Y.-J. (2006). Solids/solution partitioning and speciation of heavy metals in the contaminated agricultural soils around a copper mine in eastern Nanjing city, China. *Journal of Hazardous Materials, 131*(1-3), 19-27. http://dx.doi.org/10.1016/j.jhazmat.2005.09.033

Mackenzie, A. B., & Pulford, I. D. (2002). Investigation of contaminant metal dispersal from a disused mine site at Tyndrum, Scotland, using concentration gradients and stable Pb isotope ratios. *Applied Geochemistry, 17*(8), 1093-1103. http://dx.doi.org/10.1016/S0883-2927(02)00007-0

Macklin M. G., Brewer, P. A.. Belteanu, D., Coulthard, T. J., Driga, B., Howard, A. J., & Zaharia, S. (2001). The

long term fate and environmental significance of contaminated metals released by the January and Mack 2000 mining tailings dam failures in Maramures County, Upper Tisa Basin, Romania. *Applied Geochemistry, 18*, 241-257. http://dx.doi.org/10.1016/S0883-2927(02)00123-3

Macklin, M. G., Brewer, P. A., Hudson-Edwards, K. A., Bird, G., Coulthard, T. J., Dennis, I. A., . . . Turner, J. N. (2006). A Geomorphological approach to the management of rivers contaminated by metal mining. *Geomorphology, 79*, 423-447. http://dx.doi.org/10.1016/j.geomorph.2006.06.024

Macklin, M. G., Brewer, P. A., Hudson-Edwards, K. A., Bird, G., Coulthard, T. J., Dennis, I. A. . . . Turner, J. N. (2003). A geomorphological approach to the management of rivers contaminated by metal mining. *Geomorphology, 79*(2006), 423-447.

Macklin, M. G., Hudson-Edwards K. A., & Dawson, E. J. (1997). The significance of pollution from historic metal mining in the Pennine Orefields on river sediment contaminant fluxes to the North Sea. *Science of the total Environment, 194-195*, 391-397. http://dx.doi.org/10.1016/S0048-9697(96)05378-8

Mastrine, J. A., Bonzongo, J. J., & Lyons, W. B. (1999). Mercury concentrations in surface waters from fluvial systems draining historical precious metals mining areas in Southeastern U.S.A. *Applied Geochemistry, 14*(2), 147-158. http://dx.doi.org/10.1016/S0883-2927(98)00043-2

Meers, E., Unamuno, V. R., Du Laing G., Vangronsveld, J., Vanbroekhoven, K., Samson, R., . . . Tack, F. M. G. (2006). Zn in the soil solution of unpolluted and polluted soils as affected by soil characteristics. *Geoderma, 136*(1-2), 107-119. http://dx.doi.org/10.1016/j.geoderma.2006.03.031

Meijer, R. J., Bosboom, J., Cloin, B., Katopodi, I., Kitou, N., Koomans, R. L., & Manso, F. (2002). Gradation effects in sediment transport. *Coastal Engineering*, 47, 179-210. http://dx.doi.org/10.1016/S0378-3839(02)00125-4

Meza-Figueroa, D., Maier, R. M., de la O-Villanueva, M., Gónez-Alvarez, A., Moreno-Zazueta, A., Rivera, J., . . . Palafox-Reyes, J. (2009). The Impact of unconfined mine tailings in residential areas from a mining town in a semi-arid environment: Nacozari, Sonora, Mexico. *Chemosphere, 77*, 140-147. http://dx.doi.org/10.1016/j.chemosphere.2009.04.068

Miller, J. R. (1997). The role of fluvial geomorphic processes in the dispersal of heavy metals from mine sites. *J. Geochem Explor, 58*, 101-118. http://dx.doi.org/10.1016/S0375-6742(96)00073-8

Miranda-Aviles, R., Puy-Alquiza, M. J., & Arvizu, O. P. (2012). Anthropogenic metal content and natural background of overbank sediments from the Mining District of Guanajuato, Mexico. *Soil and Sediment Contamination; An Int. J., 21*(5), 604-624. http://dx.doi.org/10.1080/15320383.2012.672488

Mlayah, A., da Silva, E. F., Rocha, F., Hamza, Ch. B., Charef, A., & Noronha, F. (2009). The Oued Mellègue: Mining activity, stream sediments and dispersion of base metals in natural environments, North-western Tunisia. *Journal of Geochemical Exploration, 102*, 27-36. http://dx.doi.org/10.1016/j.gexplo.2008.11.016

Moncur, M. C., Ptacek, C. J., Blowes, D. W., & Birks, S. J. (2006). Groundwater and surface water discharge from an abandoned tailings impoundment: Implications for watershed water quality. *American Geophysical Union Fall meeting, San Francisco, CA*, 11-15 December.

Moncur, M. C., Ptacek, C. J., Blowes, D. W., & Jambor, J. L. (2005). Release, transport and attenuation of metals from an old tailings impoundment. *Applied Geochemistry, 20*(3), 639-659. http://dx.doi.org/10.1016/j.apgeochem.2004.09.019

Moreno-Jiménez, E., Peňalosa, J. M., Manzano, E., & Carpena-Ruiz, R. O. (2009). Heavy metals distribution in soils surrounding an abandoned mine in NW Madrid (Spain) and their transference to wild flora. *Journal of Hazardous Materials, 162*, 854-859. http://dx.doi.org/10.1016/j.jhazmat.2008.05.109

Nair, P. S., & Robinson, W. E. (2000). Cadmium speciation and transport in the blood of the bivalve Mytilus edulis. *Marine Environmental Research, 50*(1-5), 99-102. http://dx.doi.org/10.1016/S0141-1136(00)00097-0

Navarro, M. C., Pérez-Sirvent, C., Martínez-Sánchez, M. J., Vidal, J., Tovar, P. J., & Bech, J. (2008). Abandoned mine sites as a source of contamination by heavy metals: A case study in a semi-arid zone. *Journal of Geochemical Exploration, 96*, 183-193. http://dx.doi.org/10.1016/j.gexplo.2007.04.011

Nicholson, F. A., Smith, S. R., Alloway, B .J., Carlton-Smith, C., & Chambers, B. J. (2003). An inventory of heavy metals inputs to agricultural soils in England and Wales. *The Science of the Total Environment, 311*, 205-219. http://dx.doi.org/10.1016/S0048-9697(03)00139-6

Nishida, H., Miyai, M., Tada, F., & Suzuk, I. S. (1982). Computation of the index of pollution caused by heavy metals in river sediment. *Environmental Pollution Series B, Chemical and Physical, 4*(4), 241-248. http://dx.doi.org/10.1016/0143-148X(82)90010-6

Pagenkopf, G. K., Russo, R. C., & Thurston, R. V. (1974). Effects of complexation on toxicity of copper to fishes. *Journal of Fish. Res. Bd. Canada, 31*, 462-465. http://dx.doi.org/10.1139/f74-077

Pankakoski, E., Koivisto, I., Hyvarinen, H., & Terhivuo, J. (1994). Shrews as indicators of heavy metal pollution. (pp. 137-149). In J. F. Merritt et al. (Eds.), *Advances in the biology of shrews.* Pittsburgh, PA.: Spec. Publ. 18. Carnegie Museum of Nat. Hist.

Patterson, J. W., & Passino, R. (1987). *Metals Speciation, Separation, and Recovery*. MI.: Lewis Publishers, Inc.

Petaloti, C., Triantafyllou, A., Kouimtzis, T., Samara, C. (2006). Trace elements in atmospheric particulate matter over a coal burning power production area of western Macedonia, Greece. *Chemosphere, 65*(11), 2233-2243. http://dx.doi.org/10.1016/j.chemosphere.2006.05.053

Pickup, G, Higgins, R. J., & Grant, I. (1983). Modeling Sediment Transport as a Moving Wave–The Transfer and Deposition of Mining Waste. *Journal of Hydrology, 60*(1-4), 281-301. http://dx.doi.org/10.1016/0022-1694(83)90027-6

Prusty, B. G., Sahu, K. C., & Godgul, G. (1994). Metal contamination due to mining and milling activities at the Zawar zinc mine, Rajasthan, India 1. Contamination of Stream sediments. *Chemical Geology, 112*, 275-291. http://dx.doi.org/10.1016/0009-2541(94)90029-9

Purves, D. (1985). *Trace Element Contamination of the Environment*. NY: Elsevier.

Qin, D. M., & Chen, X. H. (1996). The agriculture development of Yantze-Haihe region in six dynasties period. *Agric. Hist. China, 15*, 9-12.

Qiu, J.-W., Tang, X., Zheng, C., Li, Y., & Huang, Y. (2007). Copper complexation by fulvic acid affects copper toxicity to the larvae of the polychaete Hydroides elegans. *Marin Environmental Research, 64*(5), 563-573. http://dx.doi.org/10.1016/j.marenvres.2007.06.001

Rampe, J. J., & Runnells, D. D. (1989). Contamination of water and sediment in a desert stream by metals from an abandoned gold mine and mill, Eureka District, Arizona, U.S.A. *Applied Geochemistry, 4*(5), 445-454. http://dx.doi.org/10.1016/0883-2927(89)90002-4

Reglero, M. M., Monsalve-González, L., Taggart, M. A., & Mateo, R. (2008). Transfer of metals to plants and red deer in an old lead mining area in Spain. *Science of The Total Environment, 406*, 287-297. http://dx.doi.org/10.1016/j.scitotenv.2008.06.001

Reheis, M. C. (2006). A 16-year record of eolian dust in Southern Nevada and California, USA: Controls on dust generation and accumulation. *J. of Arid Environments, 67*(3), 487-520. http://dx.doi.org/10.1016/j.jaridenv.2006.03.006

Rennert, T., & Mansfeldt, T. (2002). Sorption of Iron-Cyanide Complexes in Soils. *Soil Science Society of America Journal, 66*, 437-444. http://dx.doi.org/10.2136/sssaj2002.0437

Riemer, D. N., & Toth, S. J. (1970). Adsorption of copper by clay minerals, humic acids and bottom muds. *J. of American Water Works Association, 62*, 195-197.

Rieuwerta J. S., & Farago, M. (1996). Mercury concentrations in historic lead mining and smelting town in the Czech Republic: A pilot study. *Science of the Total Environment, 188*(2-3), 167-171. http://dx.doi.org/10.1016/0048-9697(96)05167-4

Ross, C. (2008). Preserving the Culture While Closing the Holes-Abandoned Mine Reclamation in Nevada. *Presented at the 30th annual National Association of Abandoned Mine Land Program Conference*, October 26-29, Durango, CO.

Rowan, J. S., Barnes, S. J. A., Hetherington, S. L., & Lambers, B. (1995). Geomorphology and pollution: the environmental impacts of lead mining, Leadhills, Scotland. *Journal of geochemical Exploration, 52*(1-2), 57-65. http://dx.doi.org/10.1016/0375-6742(94)00053-E

Sakata, M. (1987). Relationship between adsorption of arsenic(III) and boron by soil properties. *Environ. Sci. Technol., 21*, 1126-1130. http://dx.doi.org/10.1021/es00164a016

Sanchez-Chardi, A., & Nadal, J. (2007). Bioaccumulation of metals and effects of landfill pollution in small mammals, Part I. The greate white-toothed shrew, Crocidura russula. *Chemosphere, 68*, 703-711.

http://dx.doi.org/10.1016/j.chemosphere.2007.01.042

Sanghoon, L. (2006). Geochemistry and partitioning of trace metals in paddy soils affected by metal mine tailings in Korea. *Geoderma, 135*, 26-37. http://dx.doi.org/10.1016/j.geoderma.2005.11.004

Sierra, M. J., Millán, R., & Esteban, E. (2009). Mercury uptake and distribution in *Lavandula Stoechas* plants grown in soil from Almadén mining district (Spain). *Food and Chemical Toxicology, 47*(11), 2761-2767. http://dx.doi.org/10.1016/j.fct.2009.08.008

Sims, D. B. (1997). The Migration of Arsenic and Lead in Surface Sediments at Three Kids Mine: Henderson, Nevada (Unpublished Master thesis in the Geology Department, University of Nevada, Las Vegas).

Sims, D. B. (2010). Contamination mobilization from anthropogenic influences in the Techatticup wash, Nelson, NV (USA). *Soil and Sediment Contamination; An Int. J., 19*(5), 515-530.

Sims, D. B. (2011). *Fate of Contaminants at an Abandoned Mining site in an Arid Environment* (Unpublished dissertation in environmental science. Kingston University, London).

Sims, D. B., & Bottenberg, B. C. (2007). Environmental Hazards Associated with Historic Mining on Water Quality in Nelson, Nevada: A Test Case for Environmental Issues Associated with Historic Mining in Nevada. *Journal of the Nevada Water Resources Association, 4*(2), 21-27.

Sims, D. B., Hooda, P. S., & Gillmore, G. K. (2013). Sediment Contamination along Desert Wash Systems from Historic Mining Sites in a Hyperarid Region of Southern Nevada, USA. *Soil and Sediment Contamination: An Int. J., 22*(7), 737-752. http://dx.doi.org/10.1080/15320383.2013.768200

Sims, D., B., & Bottenberg, B. C. (2008). Arsenic and Lead Contaminator in Wash Sediments at Historic Three Kids Mine-Henderson, Nevada: The environmental hazards Associated with Historic Mining Sites and Their Possible Impact on Water Quality. *Journal of the Arizona-Nevada Academy of Science*, pending final publication. http://dx.doi.org/10.2181/1533-6085(2008)40[16:AALCIW]2.0.CO;2

Singhal, N., & Islam, J. (2008). One-dimensional model for biogeochemical interactions and permeability reduction in soils during leachate permeation. *Journal of Contaminant Hydrology, 96*(1-4), 32-47. http://dx.doi.org/10.1016/j.jconhyd.2007.09.007

Slowey A. J., Rytube, J. J., Hothem, R. L., & May, J. T. (2007b). Mercury at the Oat Hill Extension Mine and James Creek, Napa County, California; tailings, sediment, water, and biota, 2003-2004. *USGS report: OF 2007-1132, Vol. 60*.

Slowey, A. J., Johnson, S. B., Newville, M., & Brown, G. E. (2007a). Speciation and colloid transport of arsenic from mine tailings. *Applied Geochemistry, 22*(9), 1884-1898. http://dx.doi.org/10.1016/j.apgeochem.2007.03.053

Stamenkovic, J., Gustin, M. S., Marvin-DiPasquale, M. C., Thomas, B. A., & Agee, J. L. (2004). Distribution of total methyl mercury in sediments along Steamboat Creek (Nevada, USA). S*cience of The Total Environment, 322*(1-3), 167-177. http://dx.doi.org/10.1016/j.scitotenv.2003.10.029

Stanton, M. R. (2000). *The Role of Weathering in Trace Metal Distributions in subsurface samples from the Mayday Mine Dump near Silverton, Colorado*.

Stover, Bruce; Colorado Division of Reclamation, Mining & Safety. (1996). *Mine-Drainage 319 Project, Bulkhead Feasibility Investigation, 2008 Reconnaissance Report*, Underground Source Control Sites (1, 2, 3, 4, CD).

Stumm, W., & Morgan, J. J. (1996). *Aquatic chemistry, chemical equilibrium and rates in natural waters*. New York: John Wiley & Sons.

Suner, M. A., Devesa, V., Muñoz, O., López, F., Montoro, R., Arias, A. M., & Blasco, J. (1999). Total and inorganic arsenic in the fauna of the Guadalquivir estuary: environmental and human health implications. *The Science of The Total Environment, 242*(1-3), 261-270. http://dx.doi.org/10.1016/S0048-9697(99)00399-X

Sutherland, R. A. (2000). Bed sediment-associated trace metals in an urban stream, Oahu, Hawaii. *Environmental Geology, 39*(6), 611-627http://dx.doi.org/10.1007/s002540050473

Sutherland, R. A., & Tolosa, C. A. (2000). Multi-element analysis of road-deposited sediment in an urban drainage basin, Honolulu, Hawaii. *Environmental Pollution, 110*, 483-495. http://dx.doi.org/10.1016/S0269-7491(99)00311-5

Sutherland, R. A., Day, J. P., & Bussen, J. O. (2003). Lead Concentration, Isotope Ratios, and Source Apportionment in Road Deposited Sediments, Honolulu, Oahu, Hawaii. *Water, Air, and Soil Pollution, 142*, 165-186. http://dx.doi.org/10.1023/A:1022026612922

Takagi, Y., Nomizu, M., Ui-Tei, K., Tokushige, N., & Hirohashi, S. (2004). Active sites in the carboxyl-terminal region of the laminin a chain in Drosophila neuronal cell spreading. *Arch. Insect Biochem. Physiol., 56*(4), 162-169. http://dx.doi.org/10.1002/arch.20006

Tasdemir, Y., Kural, C., Cindoruk, S. S., & Vardar, N. (2006). Assessment of trace element concentrations and their estimated dry deposition fluxes in an urban atmosphere. *Atmospheric Research, 81*(1), 17-35. http://dx.doi.org/10.1016/j.atmosres.2005.10.003

Taylor, M. P., & Kesterton, R. G. H. (2002). Heavy metal contamination of an arid river environment: Gruben River, Namibia. *Geomorphology, 42*, 311-327. http://dx.doi.org/10.1016/S0169-555X(01)00093-9

USEPA. (1995). *Historic hard rock mining: The west's toxic legacy*. USEPA Region 8; EPA 908-F-95-002

USEPA. (2005). *Five-Year Review Report: Second Five-Year Review for the Bunker Hill Mining and Metallurgical Complex Superfund Site, Operable Units 1, 2, and 3, Idaho and Washington*.

Vázquez, S., Carpena, R. O., & Bernal, M. P. (2008). Contribution of heavy metals and As-loaded lupin root mineralization to the availability of the pollutants in multi-contaminated soils. *Environmental Pollution, 152*(2), 373-379. http://dx.doi.org/10.1016/j.envpol.2007.06.018

Wang, Y.-P., Shi, J.-Y., Lin, Q., Chen, X.-C., & Chen, Y.-X. (2007). Heavy metal availability and impact on activity of soil microorganisms along a Cu/Zn contamination gradient. *Journal of Environmental Sciences, 19*(7), 848-853. http://dx.doi.org/10.1016/S1001-0742(07)60141-7

Warnick, S. L., & Bell, H. L. (1969). The acute toxicity of some heavy metals to different species of aquatic insects. *Journal of Water Pollution Cont. Fed., 4*, 280-285.

Wright, D. A., & Zamuda, C. D. (1987). Copper accumulation by two bivalve molluscs: Salinity effect is independent of cupric ion activity. *Marine Environmental Research, 23*(1), 1-14. http://dx.doi.org/10.1016/0141-1136(87)90013-4

Wu, S. C., Luo, Y. M., Cheung, K. C., & Wong, M. H. (2006). Influence of bacteria on Pb and Zn speciation mobility and bioavailability in soil: A laboratory study. *Environmental Pollution, 144*(3), 765-773. http://dx.doi.org/10.1016/j.envpol.2006.02.022

Yadav, S. K., & Rajamani, V. (2006). Air Quality and trace metal chemistry of different size fractions of aerosols in N-NW India—implications for source diversity. *Atmospheric Environment, 40*(4), 698-712. http://dx.doi.org/10.1016/j.atmosenv.2005.10.005

Yager, B. D., & Stanton, M. R. (2000). Multidimensional Spatial Modeling of the May Day Mine Waste Pile, Silverton, Colorado. *Society for Mining, Metalling & Exploration, 1*, 297-303.

Yeddou, A. R., Nadjemi, B., Halet, F., Ould-Dris, A. A., & Capart, R. (2010). Removal of cyanide in aqueous solution by oxidation with hydrogen peroxide in presence of activated carbon prepared from olive stones. *Minerals Engineering, 23*, 32-39. http://dx.doi.org/10.1016/j.mineng.2009.09.009

Yin, C. Q., Shan, B. Q., & Mao, Z. P. (2006). Sustainable water management by using wetlands in catchments with intensive land use. *Wetlands and natural resource management, 190*, 53-65. http://dx.doi.org/10.1007/978-3-540-33187-2_4

Yngard, R., Damrongsiri, S., Osathaphan, K., & Sharma, V. K. (2007). Ferrate(VI) oxidation of zinc-cyanide complex. *Chemosphere, 69*(5), 729-735. http://dx.doi.org/10.1016/j.chemosphere.2007.05.017

Zhang, H., & Selim, H. M. (2005). Kinetics of arsenate adsorption-desorption in soils. *Environ. Sci. Technol., 39*, 6201-6208. http://dx.doi.org/10.1021/es050334u

Zhang, H., & Shan, B. (2008). Historical records of heavy metal accumulation in sediments and the relationship with agricultural intensification in the Yangtze-Huaihe region, China. *Science of the Total Environment, 399*, 113-120. http://dx.doi.org/10.1016/j.scitotenv.2008.03.036

Zhou, J. M., Dang, Z., Cai, M. F., & Liu, C. Q. (2007). Soil heavy metal pollution around the Dabaoshan mine, Guangdong Province, China. *Pedosfhere, 17*(5), 588-594. http://dx.doi.org/10.1016/S1002-0160(07)60069-1

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