Impact of El Niño on Environmental Mercury in Madre de Dios, Peru

by

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Executive Summary

Studies show that concentrations of methylmercury in sediment and fish increase in association with flooding. El Niño is a systematic weather phenomenon, occurring every 2-7 years, bringing heavier precipitation and flooding than typical to Peru. Additionally, the southeastern Amazonian region of Peru, Madre de Dios (MDD), is seeing increased development of artisanal and small-scale gold mining (ASGM) and use of mercury to extract the gold. Mercury used in ASGM is released into the environment and methylated by anaerobic bacteria into the more toxic species, methylmercury (MeHg). MeHg biomagnifies in aquatic food chains, and consumption of mercury-contaminated fish can lead to adverse cognitive and kidney effects and cause lasting neurological damage during developmental stages.

Our hypothesis is that due to increased flooding in MDD from El Niño, there will be greater amounts of bacterial activity in sediment that will lead to increased bio-methylation of mercury. This means biomagnification of MeHg in fish downstream of ASGM activity will be higher during and just after El Niño when compared to previous concentrations. This study addresses a public health issue if higher levels of methylmercury are entering aquatic food chains, increasing methylmercury exposure to communities dependent on fish from the river.

Environmental sampling was conducted on four occasions to represent time periods before and during an El Niño event. The first sample sets were collected in March-April and June-July of 2013 by Diringer et al. (2015). The El Niño sample collections were in March-April 2016 and May-June 2016. During each collection, water and fish were collected from a minimum of 11 sites. Environmental sampling sites are spaced out on the MDD river to represent three areas, upriver of ASGM, middle river near ASGM, and downriver of ASGM.

Climate data was obtained for each specific sample collection location. Data comes from the Land Data Assimilation System (LDAS) combining Earth Observations (EO) of meteorological variables, such as Tropical Rainfall Measurement Mission (TRMM) precipitation and time-varying surface properties such as soil moisture, net solar radiation, and temperature. Each sample location had weekly averages for solar radiation, soil moisture, and maximum and minimum temperature, or weekly accumulation in the case of precipitation, for one week lagged periods from the data of collection going back 52 weeks.

Fish mercury concentrations were significantly higher among fish collected during the months of El Nino compared to non-El Nino months. During the non-El Niño year, 47 of 200 (24%) fish samples, and 39 of 123 (32%) carnivorous fish samples surpassed the World Health Organization mercury guideline of

0.5 mg/kg. During the El Niño year, 60 of 64 (94%) of fish samples, and 32 out of 36 (89%) of carnivorous fish samples surpassed the WHO standard. Precipitation, temperature, and soil moisture all had a significant impact on fish tissue mercury concentrations.

In the non-El Niño year, suspended particulate mercury was greater during the March-April wet season than the June-July dry season, similar to the trend between wet and dry season seen in total suspended solids. However, this trend was not seen during the El Niño year, where higher levels were seen in the dry season compared to the wet season. There was no significant difference between the non-El Niño and the El Niño years. Precipitation, temperature, and soil moisture all had a significant impact on particulate mercury concentrations.

This is one of the first studies to investigate the impact of El Niño on environmental mercury concentrations. This study found that El Niño has a large effect in increasing fish tissue mercury concentrations. This could represent a public health threat to communities dependent on fish from waters with methylmercury present, where ASGM and poor mercury regulation is prevalent, and El Niño is an intermittent occurrence. Additionally, this study defined specific meteorological factors that impact environmental mercury concentrations. Both these climate components and El Niño could become more extreme due to climate change (Cai et al., 2014) as predicted by the Intergovernmental Panel on Climate Change Technical Summary (Rolfhus, Hurley, Bodaly, & Perrine, 2015) and the third National Climate Assessment (Rolfhus et al., 2015).

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Introduction

Artisanal and small-scale gold mining (ASGM) is an unregulated, often illegal, operation in many low- and middle-income countries (Basu et al., 2015). ASGM is practiced in over 70 countries (Diringer et al., 2015), mainly Africa, Asia, and South America, and is sometimes the only source of income for families (Nakazawa et al., 2016). At least 10-20 million people are employed in ASGM, and this number continues to grow due to poverty and the increasing price of gold (Basu et al., 2015; Bernaudat & Keane, 2013). Gold from ASGM activity is estimated to contribute 20-30% of the world's gold (Basu et al., 2015). Workers who go into ASGM are usually uneducated and some of the poorest within society (Fraser, 2009; Swenson, Carter, Domec, & Delgado, 2011).

The Madre de Dios (MDD) region of the Peruvian Amazon contains part of one of the largest tropical forest areas remaining (Swenson et al., 2011). It is also considered one of the most biodiverse areas in the world, and contains a high density of floral, faunal, mammal, avian and amphibian species, along with a variety of indigenous cultures (Asner, Llactayo, Tupayachi, & Luna, 2013; Swenson et al., 2011). However, Peru is one of the largest gold producers in the world (International, 2010), with MDD contributing 70% of Peru's gold resource (Brooks, Sandoval, Yepez, & Howard, 2007). As of 2012, it was estimated around 30,000 ASGM miners worked in this area (Gardner, 2012).

ASGM in MDD has grown from less than 10,000 ha in 1999, to more than 50,000 ha by 2012. Up until around 2008, mining in the MDD region was dominated by three large mines, Huepetuhe, Delta-1, and Guacamayo, large enough to be seen on Google Earth images (Asner et al., 2013). Together, these three mines estimated 15,500 ha of mined area as of 2009 (Swenson et al., 2011). Between 1999 and 2012, however, small, diffuse mining operations saw a near 600% increase in area, until they accounted for 51% of total mining in MDD (Asner et al., 2013). Deforestation for ASGM surpassed the amount of forest loss from ranching, agriculture, and logging combined, tripling from 2,166 ha/yr before 2008 to average 6,145 ha/yr by 2012 (Asner et al., 2013).

Swenson et al. correlated this rapid growth of ASGM to rising gold prices (Swenson et al., 2011). Between 2003 and 2011, deforestation rates from ASGM were nonlinearly increasing alongside the increasing rate of gold prices, at 18% per year. Around this same time period, MDD saw a 400% increase in ASGM (Asner et al., 2013). Gold prices have consistently stayed at about $1000/oz since 2010 (Council, 2016). Accordingly, the increase in gold prices was significantly related to increased mercury imports,

which is used as part of the gold processing in ASGM activity. Almost all mercury that is imported into countries like Peru is for illegal ASGM (Buccella, 2014).

ASGM's environmental impact encompasses acid mine drainage, pollution, and deforestation, including areas of high biodiversity such as the Amazon (Swenson et al., 2011) and is also the largest contributor to global emissions of atmospheric mercury. At over 700 tons of emission, ASGM has surpassed fossil fuel burning and makes up 35% of total anthropogenic mercury emissions (Programme, 2013).

In MDD, gold is typically mined through the process of whole ore amalgamation. Prior to mining, an area is deforested and alluvium is extracted for separation. Separation is first carried out on woolen mats or carpets on inclined ladders, and sediment is washed with water pumped from the rivers. At the end of the day, the mats are cleaned out for sediments containing gold. The next step includes adding mercury to the sediment to form amalgams (Valdivia & Ugaya, 2011). Mercury sorbs to the surface of gold particles and due to mercury's surface tension and high density, the formation sinks compared to other, lighter particles. The ratio of mercury used to the amount of gold amalgams formed is about 2 to 1 (Valdivia & Ugaya, 2011). The second separation is carried out by placing sediments into a drum for separation by differing densities. The heaviest gold amalgams are then heated up to 700 °C to vaporize the mercury (Valdivia & Ugaya, 2011).

Unless a retort is used, vaporized mercury is released into the atmosphere, exposing miners through inhalation (Group, 2011). A retort allows the vaporized mercury to be recycled by containing and condensing it, but this is often a neglected practice due to the low price of mercury, time involved, convenience, and lack of knowledge (Veiga, Maxson, & Hylander, 2006). Without the use of a retort, burning off the mercury releases about 300 metric tons per year to the atmosphere. Adding a retort can recycle about 95% of the mercury being released from amalgams (Group, 2011).

Mercury is used because it is inexpensive, easy to use, and readily available (Basu et al., 2015). However, mercury is a neurotoxin to which miners are directly exposed and is also released into water, sediment, and the atmosphere (Ashe, 2012; Diringer et al., 2015). Mixing the mercury into the sediments to form amalgams is sometimes done by miners with their bare feet, which puts miners into direct dermal contact with mercury. Heating the amalgam is done in a metal pan or shovel over an open flame or a small oven with no air filter, leading to elemental mercury inhalation by miners and anyone nearby (Ashe, 2012;

Fraser, 2009). Inhalation of mercury over a long period of time can lead to tremors, kidney dysfunction, and neurological problems (Yard et al., 2012).

However, another important form of exposure, with a wider area of impact, is through ingestion of contaminated aquatic food chains. Fish from nearby rivers accumulate mercury after its disposal into rivers or deposition from the atmosphere (Niane et al., 2015). Whereas only those directly involved in burning the amalgams are exposed to elemental mercury, communities not at all involved in mining operations can be exposed to mercury that has accumulated in aquatic food chains. This form, methylmercury, can cause neurological damage and is especially a concern for the developing fetus (Yard et al., 2012). Mercury from ASGM ends up in aquatic organisms through a chain of environmental and biological processes.

During the process of burning the mercury gold amalgams to release the gold, elemental mercury [Hg(0)] is released into the atmosphere or deposited into waterways through run off. Through chemical and photochemical processes in the atmosphere, Hg(0) is oxidized to Hg(II), an inorganic mercury species with a much shorter atmospheric lifetime. Hg(II) will deposit on land and water surfaces through dry and wet deposition, usually on a regional scale due to its short lifetime. A portion of the deposited Hg(II) will quickly re-volatize to Hg(0) but the rest generally associates with soil and particles (Selin, 2009). Very small amounts of Hg(II) stays in aqueous form but most will sorb to organic matter, sulfide, chloride, or particles (Hsu-Kim, Kucharzyk, Zhang, & Deshusses, 2013). Hg(II) deposited and adsorbed to soils remains there but in water ways, Hg(II) can become methylated to a more toxic form, methylmercury (MeHg). This process is mediated by anaerobic microbial activity, mostly by sulfate- and iron-reducing bacteria, and usually occurs in anoxic sediments. MeHg is more bioavailable to living organisms compared to other mercury species and biomagnifies in aquatic food chains. Predatory fish contain the highest levels as they concentrate MeHg by eating contaminated prey. The high biomagnification factor of MeHg can lead to higher trophic level fish accumulating more than 10^6 times levels in surrounding water (Selin, 2009).

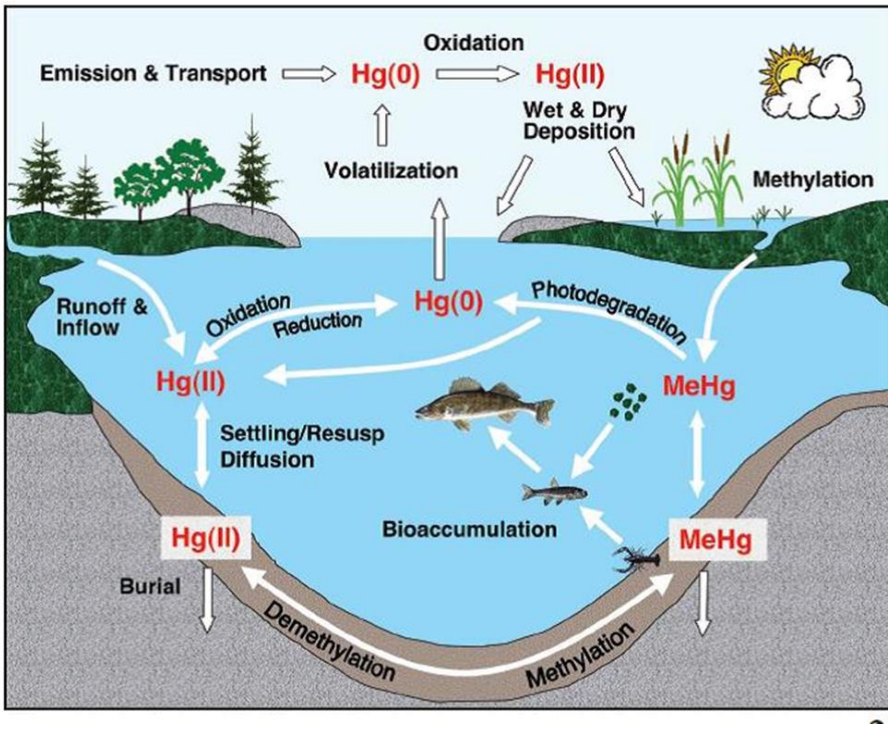

Figure 1. Mercury cycling between the atmosphere and a watershed. Hg(0): elemental mercury, Hg(II): divalent mercury, MeHg: methylmercury (Selin, 2009)

The rate of methylation of inorganic mercury to MeHg is most importantly mediated by the activity of sulfate-reducing bacteria and others, including iron-reducing bacteria and methanogens (Compeau & Bartha, 1985; Gilmour et al., 2013), at oxic/suboxic interface environments such as benthic sediments and inundated soils, like wetlands (Rolfhus et al., 2015). Environmental factors that influence the productivity rates of these methylating bacteria become important in trying to understand when and where observed increases in fish MeHg levels are detected (Kelly et al., 1997; Porvari & Verta, 1995; Rolfhus et al., 2015; St.Louis et al., 2004). Studies at the Experimental Lakes Area (ELA) of northwestern Ontario, Canada revealed a 40-fold increase in MeHg levels after flooding of the reservoir, with high levels persisting over the length of the 9-year study (St.Louis et al., 2004).

During flooding, high amounts of vegetation and organic carbon enter the waterway and decompose. Decomposing organic matter stimulates microbial activity and depletes oxygen levels. The increase in microbial activity and anoxic conditions promotes methylation of inorganic mercury and production of MeHg (Kelly et al., 1997; Porvari & Verta, 1995; Rolfhus et al., 2015; St.Louis et al., 2004). Bioaccumulation in aquatic food chains predictably follows an increase in MeHg levels. The ELA study in

Canada observed that following flooding, zooplankton MeHg levels were 7-10 times higher, correlating to increases in MeHg in the reservoir (St.Louis et al., 2004).

With such a large impact seen after flooding of a reservoir, it is expected that other weather patterns can have an effect on microbial activity. For example, increases in precipitation (e.g., bursts of high precipitation or long-term, sustained precipitation) could begin this chain of events on a small scale that lead to large-scale flooding. Changes in temperature could also influence microbial activity and subsequent mercury methylation by maintaining ideal conditions or becoming too hot or cool for efficient productivity. Additionally, global meteorological oscillations, such as the El Niño Southern Oscillation (ENSO or simply *El Niño*) and the Madden-Julien Oscillation (MJO), cause dramatic changes in regular climate conditions, such as longer rainy seasons, hotter dry seasons, etc., that could have unforeseen impacts on microbial activity and mercury methylation potential.

El Niño is a systematic weather phenomenon in the equatorial Pacific, part of the El Niño Southern Oscillation (ENSO). During normal, non-El Niño years, warmer sea surface water moves toward the western Pacific basin, around Indonesia and the Philippines, and cooler air moves in around the eastern Pacific basin around Ecuador and northern Peru. This contributes to the heavy rainfall during the wet season of Indonesia, northern Australia, and the Philippines and the arid conditions of northern Peru. Occurring every 2-7 years, El Niño events see a reversal in this pattern, where sea surface temperatures warm around the central and eastern Pacific basin instead. This leads to warmer waters and much higher amounts of precipitation in Peru and drought in the western Pacific (Bayer et al., 2014).

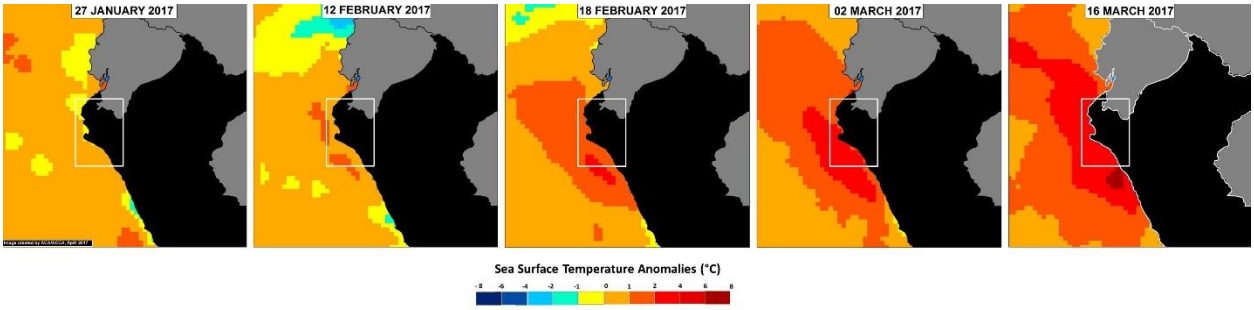

Figure 2. Sea temperature warming off of coast of northern Peru during a "coastal El Niño." January shows primary warming and the following months show intensifying warming. The white inset box shows the primary flooding zone. (Project, 2017)

The unusual sea water warming during an El Niño event leads to extreme weather events such as drought and flooding (Bayer et al., 2014). Peru experiences extensively greater amounts of precipitation and higher temperatures during El Niño(Bayer et al., 2014). This extreme weather can have disastrous impacts on human infrastructure and health and is linked to increased respiratory and enteric diseases (Checkley et al., 2000), malnutrition (Bayer et al., 2014; Danysh et al., 2014), and vector-borne disease (Bouma & Dye, 1997; Gagnon, Smoyer-Tomic, & Bush, 2002). The majority of research on health impacts of El Niño in Peru has been conducted along the Pacific Coast. Very little research has been done on the other side of the Andean Mountain range, where communities experience severe heat and drought, as well as downriver flooding due to heavy rains at headwater streams in the Andean Mountains, the severity of which varies with ENSO (Li, Zhang, Ye, Li, & Baker, 2011).

Additionally, extreme El Niño events are projected to occur more frequently with global warming (Cai et al., 2014). This strong relationship between flooding and high levels of MeHg in sediment and fish may become an even larger concern, as the Intergovernmental Panel on Climate Change Technical Summary (Rolfhus et al., 2015) and the third National Climate Assessment (Rolfhus et al., 2015) are predicting more extreme precipitation and flooding events in the 21st century, including more extreme El Niño events (Cai et al., 2014).

MDD represents a population especially vulnerable to an extreme event such as El Niño. Data from between 2011 and 2012 shows that of children under 10, 46% were anemic, 14% stunted and 19% overweight/obese (Feingold, Pan, Salmon-Mulanovich, & Estela Manrique, 2014). Among adults in 2014, 48% tested positive for recent dengue infection, 23% had reported malaria and 13% were hypertensive (Pan, 2014). Also, the area carries an additional burden of MeHg exposure from fish consumption. High mercury exposure is linked to an increased risk of malaria (Crompton et al., 2002). MeHg seems to be more readily absorbed in malnourished individuals (Hodgson et al., 2014), a possibly significant risk in a population with high mercury exposure and 46% of children are anemic and 14% are stunted, all of which are expected to increase during El Niño.

Several studies have aimed to measure the impact of ASGM in MDD on the health of mining communities in the region (Ashe, 2012; Langeland, Hardin, & Neitzel, 2017; Wyatt et al., In-review; Yard et al., 2012). High levels of total mercury have been measured in miners, especially those who are directly involved in burning the gold-mercury amalgams, and methylmercury levels were high in association with high fish consumption (Yard et al., 2012). However little research has investigated exposure to communities downstream of these mining sites. A study by Diringer et al. (2015) was the first peer-

reviewed publication to document long distance transport of Hg from ASGM areas of the MDD watershed to areas that are hundreds of kilometers downstream. This work showed Hg concentrations in sediment, suspended solids, and fish within the MDD River increased in areas downstream of known ASGM activity relative to locations upstream of mining. Fish tissue Hg concentrations in these downstream locations were at levels representing a public health threat, with greater than one-third of carnivorous fish exceeding the international health standard of 0.5 mg kg^{-1}.

The objective of this study is to combine environmental samples obtained in 2016 during El Nino with those previously collected in 2013 (sediment, surface water, and fish) to compare mercury concentrations across time and in response to weather patterns, such as maximum and minimum temperature, precipitation, and soil moisture. Seasonal variation and weather events such as El Niño are evaluated as drivers of change in environmental mercury levels. Mercury concentrations in fish, sediment, and fish are quantified by season and time periods before, during, and after El Niño.

We hypothesize that 1) increased flooding in MDD from El Niño will increase microbial activity in sediment, leading to increased bio-methylation of mercury, and 2) higher biomagnification of MeHg in fish downstream of ASGM activity during and just after El Niño.

Methods

Study Area

Madre de Dios (MDD), a department in the southeastern part of Peru, encompasses around 85,000 km^2 of Amazonian tropical rain forest. The area is globally recognized for its vast biological diversity (Asner et al., 2013; Swenson et al., 2011). The study region includes the MDD watershed, beginning at the headwaters of Rio Madre de Dios at Atalaya downstream to Puerto Pardo, near Peru's border with Bolivia. The river is divided into three sections for a comparison of the impact of ASGM on environmental mercury levels. Section 1 is the area of the river upstream of mining and used as a control. Section 2 is the area of the highest density of mining operations, beginning around Boca Colorado. Two main tributaries, Rio Colorado and Rio Inambari, feed this section of the river where the largest mines, Huepetuhe, Guacamayo, and Delta-1, are located. Section 3, beginning around Laberinto and ending around Puerto Pardo, has less ASGM activity compared to Section 2 but contains the highest population density. This section is where

the capital of MDD, Puerto Maldonado, is located. An oxbow lake, Lago Valencia, was also included in Section 3 as an active area of sediment exchange within the watershed and for its importance in supplying fish in nearby markets.

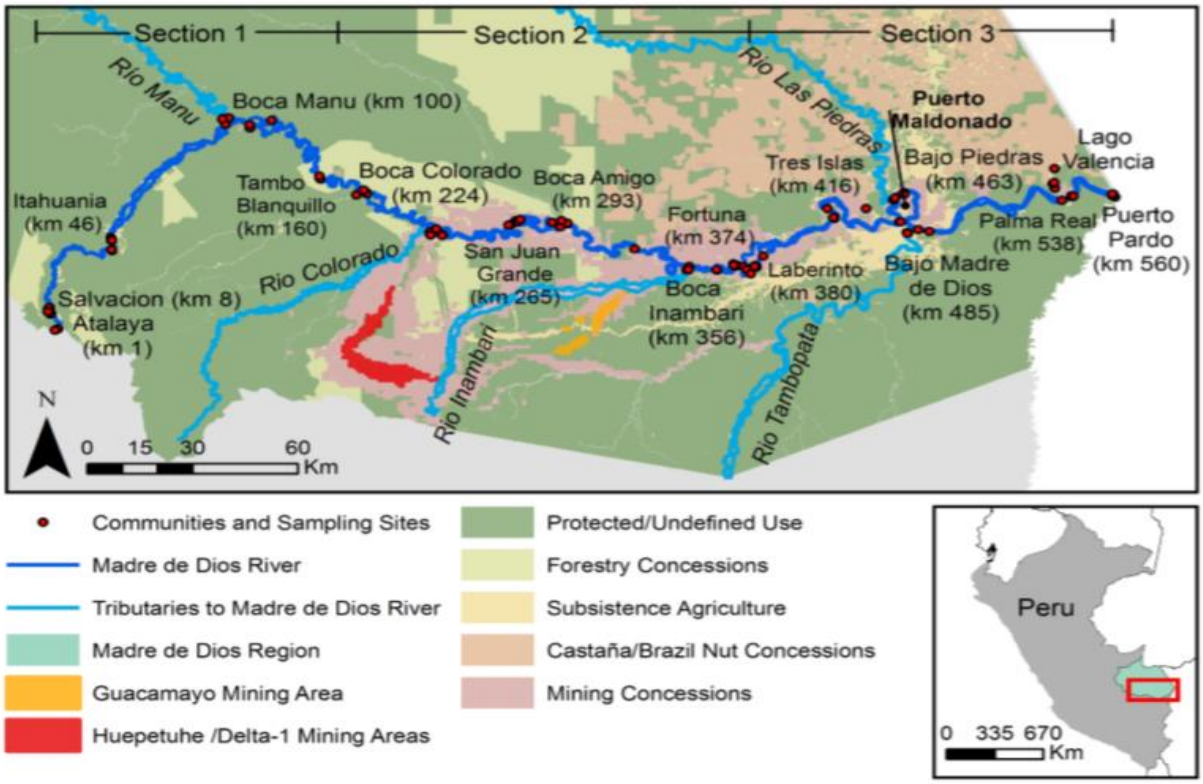

Figure 3. Study location for the 2013 non-El Niño and 2016 El Niño environmental sample collection in Madre de Dios (MDD) Peru. Red dots are locations of communities adjacent to Rio MDD, near which environmental samples were collected. Division of river into three categorical sections is denoted at top. Known mining regions (pink), Huepetuhe/Delta-1 mining area (red), and the Guacamayo mining area (orange) courtesy of the Asociacion para la Conservacion de la Cuenca Amazonica (ACCA) (Diringer et al., 2015).

Environmental Sample Collection

The most recent El Niño occurred in 2015-2016, but the residual, referred to as a coastal El Nino, is ongoing. Environmental sampling was conducted on 4 occasions to represent time periods before, during, and after the El Niño season. The first sample sets were collected in March-April and June-July of 2013 by Diringer et al. (2015). The El Niño sample collections were in March-April 2016 and May-June 2016. During each collection, water, sediment, and fish were collected from a minimum of 11 sites.

Environmental sampling sites are spaced out on the MDD River to represent three areas, upriver of ASGM, middle river near ASGM, and downriver of ASGM. At each site, water samples were collected in triplicate using clean hands technique. Surface water samples were collected from the center of the river using 1 L polyethylene containers. A subset of whole water samples was filtered in the field using disposable syringe filters (0.45 µm nominal pore size) and transported to Duke laboratories for analysis of total suspended particulate Hg (Hg_P). Another subset of whole water samples was filtered in the field using pre-weighted glass fiber filters (0.22 µm) with a polyethylene vacuum filtration apparatus for total suspended solids (TSS) analysis. All samples were immediately frozen in the field with a Sub -18°C Credo Cube (Pelican BioThermal). Water conductivity, temperature, and pH were measured with a YSI Multiparameter Sonde at each site.

At least 10 fish were collected at each site, obtained by traditional fishing techniques similar to previous methods (Diringer et al., 2015), such as cast net, overnight netting, etc. Preference was given to carnivorous fish (i.e., piscivorous, insectivorous, and omnivorous species) and fish with higher reoccurrence from earlier trips. Each fish was identified, categorized by trophic level, weighted, measured, and photographed. Muscle tissue was dissected from the anterior section above the lateral line from each fish, stored, and frozen in the field. Samples were lyophilized after arriving at Duke.

Laboratory Analysis

After the 2016 El Niño sample collection, water samples from the May-June dry season collection were immediately transported and analyzed in Duke laboratories. All other samples were shipped and analyzed in March 2017. All samples were stored and frozen immediately after collection in the field by placing into a Sub -18˚C Credo Cube (Pelican BioThermal). Samples were delivered to Duke laboratories using World Courier and kept frozen using dry ice during shipping.

Total Hg concentrations in whole water and filtered water samples to were quantified with cold vapor fluorescence spectroscopy (EPA, 2001) (BrooksRand T-MERX). Total Hg in filtered water was subtracted from total Hg in unfiltered to give particulate mercury concentration (Hg_p, ng L^{-1}), to compare with previous water mercury measurements. Total Hg concentrations in fish tissue was quantified with direct combustion atomic absorption spectroscopy (EPA, 1998) (Milestone DMA-80). More than 80% of Hg in fish muscle tissue is in the form of MeHg (Huckabee, Elwood, & Hildebrand, 1979) so total Hg concentrations are used to approximate MeHg exposure potential.

In addition to mercury, we will quantify other relevant water quality parameters for surface water samples, such as TSS concentration, via filtration of particulate matter immediately after collection of surface water samples (Diringer et al., 2015), major cation and anion concentrations (ICP-MS and ion chromatography), and total and dissolved organic carbon (Shimadzhu TOC Analyzer). The concentration of Hg per mass of suspended solid (Hgss, ng g^{-1}) was calculated from the Hg_p value divided by the TSS concentration (mg L^{-1}).

Climate

Climate data were obtained for each specific sample collection location. Data come from the Land Data Assimilation System (LDAS) combining Earth Observations (EO) of meteorological variables, such as Tropical Rainfall Measurement Mission (TRMM) precipitation and time-varying surface properties such as soil moisture, net solar radiation, and temperature. Each sample location had weekly averages for solar radiation, soil moisture, and maximum and minimum temperature, or weekly accumulation in the case of precipitation, for one week lagged periods from the date of collection going back 52 weeks.

Statistical analysis

Descriptive statistics for TSS, particulate mercury, and fish mercury were produced using means and 95% confidence intervals. Comparisons of river sections and the non-El Niño year to the El Niño year were performed using ordinary least squares (OLS) regression analysis using averages of triplicate sample collections (for all samples from 2013 and water samples from 2016) or single, measured values (for fish from 2016) for each site location. River sections were used as categorical variables. Multiple regressions for suspended particulate mercury controlled for river section, season, and El Nino. Multiple regressions for mercury in fish controlled for section, season, El Niño, and fish type.

OLS regression analyses were performed for each lagged week of precipitation, maximum and minimum temperature, and soil moisture estimates in regard to particulate mercury and fish mercury. Significant lagged weeks were identified as possible influential factors on environmental mercury concentrations. Pearson correlations examined multicollinearity between lagged weeks for the same climate variable. Non-collinear lags were included in multiple regressions to identify the most significant lags on particulate or fish mercury. Significant lagged weeks were included in an overall multiple regression including all important lags of the different climate variables.

Results

47 locations were visited in 2013 and 23 locations in 2016. At the time of this Master's Project, fish samples obtained in section 2 had not yet been shipped from Peru, nor had 25 samples of sediment been analyzed in the laboratory.

Fish mercury concentrations were significantly higher among fish collected during the months of El Nino compared to non-El Niño months (Table 1). Fish mercury concentrations were 2-7 times higher during El Nino for sections 1 and 3 compared to non-El Niño months (Table 1), with significant increases in mercury levels in both carnivorous and non-carnivorous fish (Table 1).

In contrast to fish, suspended particulate mercury appear to have declined during the El Nino months compared to non-El Niño months even when comparing concentrations by section and by predefined wet/dry seasons (Table 1). The dry season was the only time a significant increase in mercury in El Nino months compared to non-El Niño months was observed. However, consistent with prior findings, mercury levels were highest in section 2, the section with the highest density of ASGM activity, relative to the other river sections.

Table 1: t-test of Means for Fish Mercury and Suspended Particulate Mercury

	Non-El Nino Year (2013)	El Nino Year (2016)	t-test p-value
Fish Mercury (mg/kg)			
River Section 1	0.16 (0.1, 0.2)	1.16 (0.7, 1.6)	< .001
River Section 2	0.45 (0.3, 0.6)	*	
River Section 3	0.4 (0.3, 0.5)	0.94 (0.6, 1.3)	0.01
Carnivorous	0.43 (0.3, 0.5)	1.45 (0.9, 2.0)	< .001
Non-Carnivorous	0.20 (0.2, 0.2)	0.61 (0.4, 0.8)	< .001
Total	0.35 (0.3, 0.4)	1.07 (0.8, 1.4)	< .001
Suspended Particulate Hg (ng/L)			
River Section 1	4.2 (1.5, 6.9)	3.1 (-4.0, 10.3)	0.67
River Section 2	19.2 (10.4, 28.0)	15.9 (8.6, 23.1)	0.57
River Section 3	16.8 (10.7, 22.8)	3.5 (-3.7, 10.7)	0.01
Wet Season	18.1 (12.3, 24.0)	9.8 (0.04, 19.6)	0.14
Dry Season	4.8 (0.5, 9.0)	12.8 (6.8, 18.8)	0.03
Total	14.9 (10.1, 19.7)	11.3 (6.0, 16.6)	0.35

Model Results

OLS regression analysis was used to explain the impact of El Niño on fish mercury controlling for river section, dry or wet season, and trophic level of the fish, or Hg = River Section + Season + Fish Type + El Niño year. Fish muscle tissue mercury concentrations were not significantly different between the March-April wet season and the June-July dry season (Table 2). Fish mercury was significantly greater in river section 2 by 1.02 mg/kg (on a log scale) (95% CI 0.6, 1.5; $p < 0.001$) compared to section 1 (Table 2). Section 3 fish mercury increased by 0.65 mg/kg (on a log scale) (95% CI 0.2, 1.1; $p = 0.004$) relative to section 1 (Table 2). This increase in mercury levels cannot be fully explained by differences in weight, which could be related to higher contaminant burdens. Carnivorous fish had greater levels of muscle

tissue mercury, 0.64 mg/kg (on a log scale) (95% CI 0.3, 1.0; $p < 0.001$), than non-carnivorous fish (Table 2).

OLS regression analysis was used to explain the impact of El Niño on TSS controlling for river section and dry or wet season, or TSS = River Section + Season + El Niño Year. Total suspended solids (TSS) significantly decreased from the March-April wet season to the June-July dry season by 324.9 mg/kg (95% CI -436, -214; $p < 0.001$) (Table 3). TSS in section 2 is significantly greater than section 1 by 133.5 mg/kg (95% CI 4.1, 262.8; $p = 0.04$) but there is no significant difference between section 1 and section 3 or between the non-El Niño data collection and the El Niño year (Table 3).

OLS regression analysis was used to explain the impact of El Niño on suspended particulate mercury controlling for river section and dry or wet season, or Hg_P = River Section + Season + El Niño Year. Suspended particulate mercury did not significantly change from the March-April wet season to the June-July dry season (Table 4). However, particulate mercury increased in river section 2 relative to section 1 (Table 4). Particulate mercury concentrations in section 2 were 2.26 ng/L (on a square root scale) (95% CI 1.2, 3.4; $p < 0.001$) greater than section 1 (Table 4). Relative to section 1, section 3 particulate mercury concentrations increased 1.27 ng/L (on a square root scale) (95% CI 0.03, 2.5; $p = .05$) (Table 4). There was no significant difference in particulate mercury concentrations between the non-El Niño data collection and the El Niño year (Table 4).

Following the OLS models demonstrating the importance of El Nino in predicting fish and suspended particulate mercury levels, we wanted to test whether specific meteorological variables measured in the same location as where the sample was collected at the time they were collected, could be associated with mercury levels in fish or particulates. We did this by exploring averaged weekly time lags for minimum/maximum temperature, cumulative precipitation and soil moisture of the top 10 cm of soil. However, including multiple lags into the same regression equation can lead to significant problems associated with collinearity due to the correlation of climate measures week to week. Since we are not aware of prior research looking at meteorological lags on mercury production, we began by identifying the most important lags that were related to fish or particulate mercury by fitting univariate OLS models for each meteorological parameter for every weekly time lag going back 24 weeks (~6 months) for suspended particulate mercury and 52 weeks (~1 year) for fish mercury. The resulting beta estimates for the fish mercury univariate models are shown for precipitation (Figure 4a), soil moisture (Figure 4b), and minimum and maximum temperatures (Figures 4c, 4d). The resulting beta estimates for the particulate

mercury univariate models are shown for precipitation (Figure 5a), soil moisture (Figure 5b), and minimum and maximum temperatures (Figures 5c, 5d).

Results demonstrate that the most important lags for fish mercury were lagged weeks 9, 16, 19, and 25 for cumulative precipitation, lagged weeks 2 and 10 for maximum temperature, lagged weeks 10 and 27 for minimum temperature and lagged week 48 for soil moisture in the top 10 cm. Next we determined, among these most important lags, which lags were least correlated based on a Pearson correlation. This will allow us to preliminarily identify which lags to include in a multivariate model. We identified lagged weeks 16 and 19 for cumulative precipitation, lagged week 27 for minimum temperature, and lagged week 48 for soil moisture significantly influenced fish mercury. Cumulative precipitation 16 weeks previous decreased mercury levels by 0.011 mg/kg (on a log scale) (95% CI -0.02, -0.004; $p = 0.001$) (Table 2). Cumulative precipitation 19 weeks previous increased mercury levels by 0.023 mg/kg (on a log scale) (95% CI 0.01, 0.03; $p < 0.001$) (Table 2). Minimum temperature 27 weeks previous increased mercury levels by 0.24 mg/kg (on a log scale) (95% CI 0.04, 0.4; $p = 0.02$) (Table 2). Soil moisture 48 weeks previous increased mercury levels by 0.060 mg/kg (on a log scale) (95% CI 0.03, 0.09; $p < 0.001$) (Table 2).

Results demonstrate that the most important lags for suspended particulate mercury were lagged weeks 15 and 20 for precipitation, lagged week 3 for maximum temperature, lagged week 4 for minimum temperature, and lagged week 3 for soil. Next we determined, among these most important lags, which lags were least correlated based on a Pearson correlation. This will allow us to preliminarily identify which lags to include in a multivariate model. We identified lagged week 15 for precipitation, lagged week 3 for maximum temperature, and lagged week 3 for soil moisture significantly influenced particulate mercury concentrations. Cumulative precipitation 15 weeks previous increased mercury levels by 0.013 ng/L (on a square root scale) (95% CI 0.003, 0.03; $p = 0.01$) (Table 4). Maximum temperature 3 weeks previous decreased mercury levels by 0.74 ng/L (on a square root scale) (95% CI -1.4, -0.1; $p = 0.02$) (Table 4). Soil moisture 3 weeks previous increased mercury levels by 0.095 ng/L (on a square root scale) (95% CI 0.02, 0.2; $p = 0.01$) (Table 4).

Table 2. OLS Regression Models for Fish Mercury Concentrations (in mg/kg) (on a log scale) by El Niño Season and Meteorological Data

Variable	MODEL 1 (El Niño Season)			MODEL 2 (Meteorological Data)		
	Beta	95% CI	p-value	Beta	96% CI	p-value
Intercept	-2.65	(-3.1, -2.2)	< .001	-74.04	(-131, -17.3)	0.01
Section (section 1 the reference)						
Section 2	1.02	(0.6, 1.5)	< .001	0.98	(0.5, 1.4)	< .001
Section 3	0.65	(0.2, 1.1)	0.004	0.40	(0.007, 0.8)	0.05
Dry Season (vs. Wet)	0.28	(-0.1, 0.7)	0.16	N/A		
El Niño (vs. non-El Niño year)	1.31	(0.9, 1.7)	< .001	N/A		
Carnivorous Fish (vs. non-carnivorous)	0.64	(0.3, 1.0)	< .001	N/A		
Meteorological Lags						
Precipitation Lag 16	N/A			-0.011	(-0.017, -0.004)	0.001
Precipitation Lag 19	N/A			0.023	(1.3, 0.03)	< .001
Minimum Temperature Lag 27	N/A			0.237	(0.04, 0.4)	0.02
Soil Moisture Top 10 cm Lag 48	N/A			0.060	(0.03, 0.09)	< .001

Table 3. OLS Regression Model for Total Suspended Solids (in mg/kg)

Variable	MODEL 1 (El Niño Season)		
	Beta	95% CI	p-value
Intercept	406.59	(291.6, 521.6)	< .001
Section (section 1 the reference)			
Section 2	133.46	(4.1, 262.8)	0.04
Section 3	11.8	(-134.4, 158.0)	0.87
Dry Season (vs. Wet)	-324.87	(-436.1, -213.7)	< .001
El Niño (vs. non-El Niño year)	-44.73	(-156.6, 67.2)	0.43

Table 2. OLS Regression Models for Suspended Particulate Mercury Concentrations (in ng/L) (on a square root scale) by El Niño Season and Meteorological Data

Variable	MODEL 1 (El Niño Season)			MODEL 2 (Meteorological Data)		
	Beta	95% CI	p-value	Beta	96% CI	p-value
Intercept	2.11	(1.1, 3.1)	< .001	221.57	(35.3, 407.8)	0.02
Section (section 1 the reference)						
Section 2	2.26	(1.2, 3.5)	< .001	1.38	(0.4, 2.4)	0.009
Section 3	1.27	(0.03, 2.5)	0.05	-0.41	(-1.8, 0.9)	0.54
Dry Season (vs. Wet)	-0.57	(-1.5, 0.3)	0.22	N/A		
El Niño (vs. non-El Niño year)	-0.80	(-1.7, 0.1)	0.09	N/A		
Meteorological Lags						
Precipitation Lag 15	N/A			0.01	(0.003, 0.02)	0.01
Maximum Temperature Lag 3	N/A			-0.74	(-1.4, -0.1)	0.02
Soil Moisture Top 10 cm Lag 3	N/A			0.10	(0.02, 0.2)	0.01

Figure 4a-4d. Beta estimates for fish mercury (mg/kg) from a univariate OLS testing each meteorological (weekly) lag up to a maximum of 52 lags (1 year).

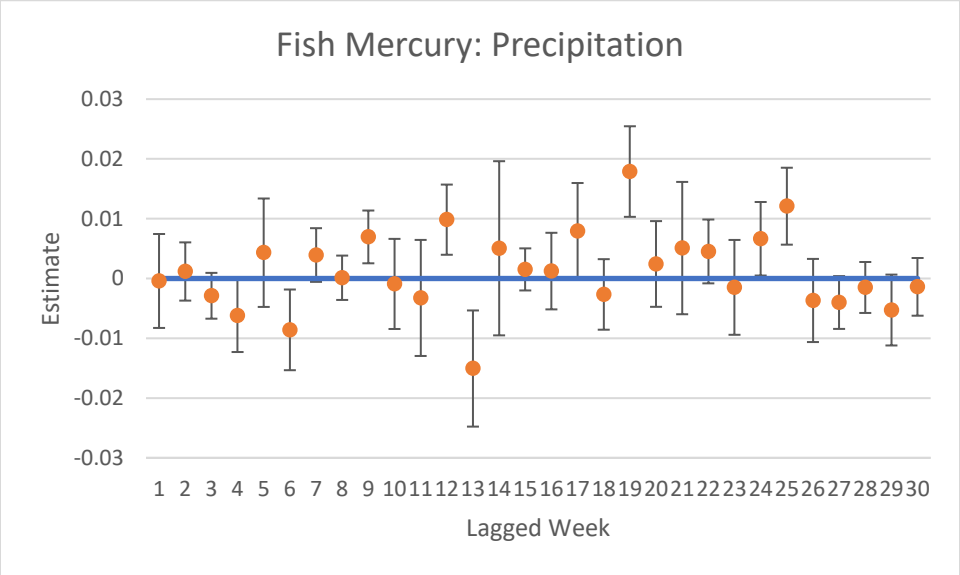

Figure 4a. Fish Mercury Estimate and Cumulative Weekly Precipitation (mm)

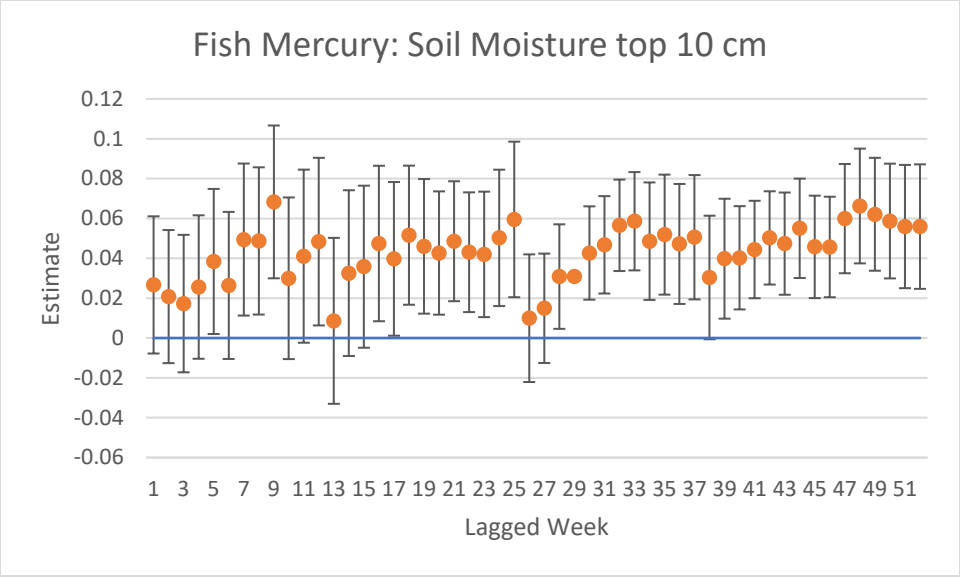

Figure 5b. Fish Mercury Estimate and Average Weekly Soil Moisture top 10 cm (mm)

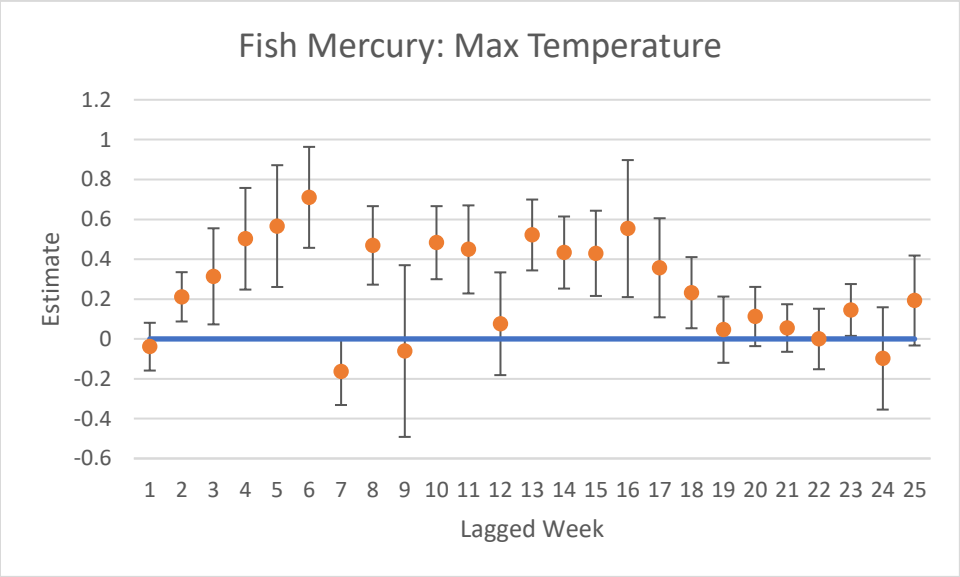

Figure 6c. Fish Mercury Estimate and Average Weekly Maximum Temperature (°C)

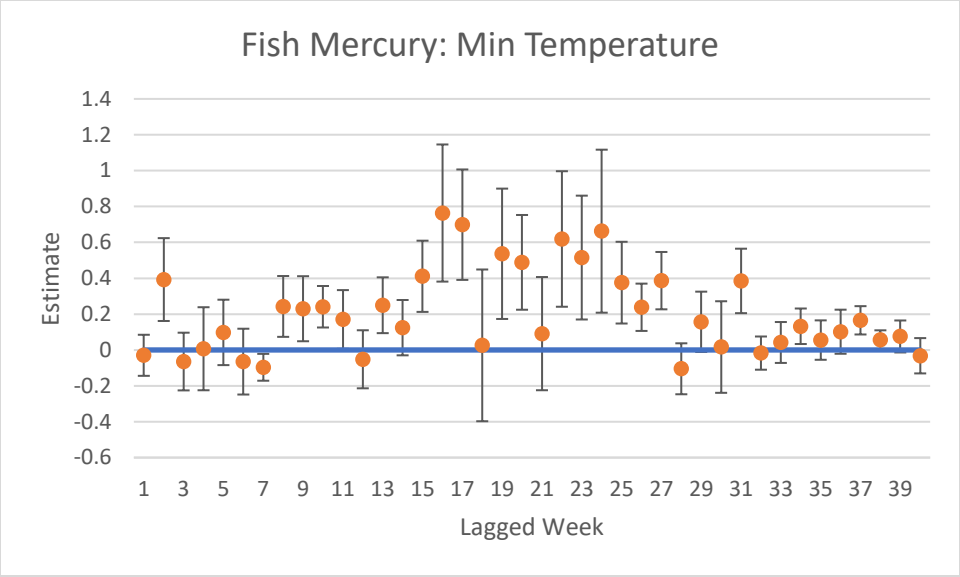

Figure 7d. Fish Mercury Estimate and Average Weekly Minimum Temperature (°C)

Figure 5a-5d. Beta estimates for suspended particulate mercury (ng/L) from a univariate OLS testing each meteorological (weekly) lag up to a maximum of 24 lags (6 months).

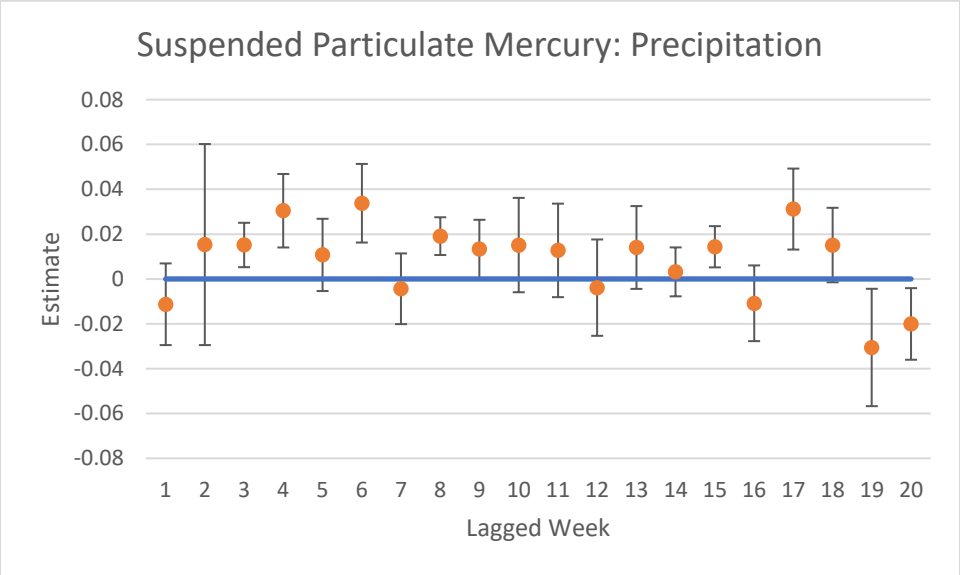

Figure 5a. Suspended Particulate Mercury Estimate and Cumulative Weekly Precipitation (mm)

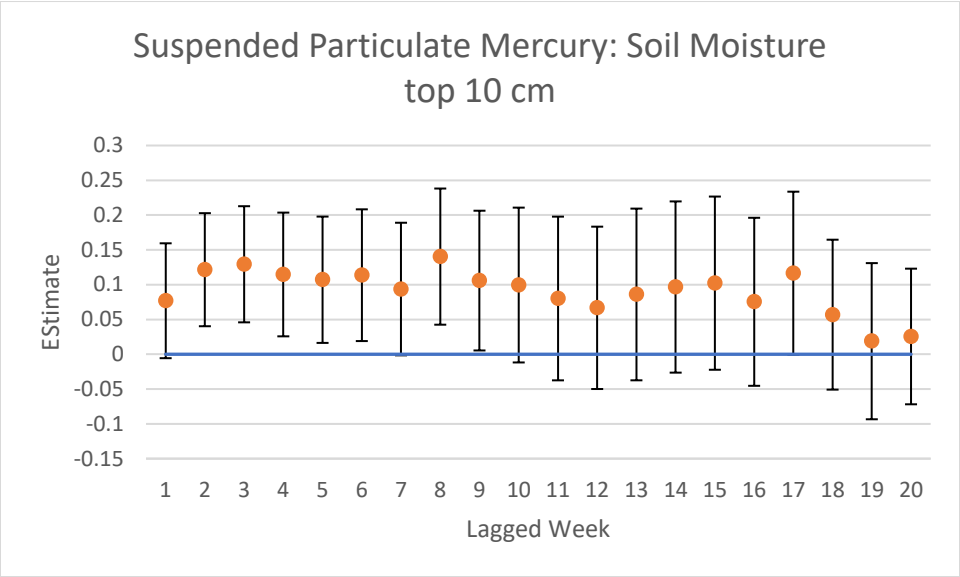

Figure 5b. Suspended Particulate Mercury Estimate and Average Weekly Soil Moisture top 10 cm (mm)

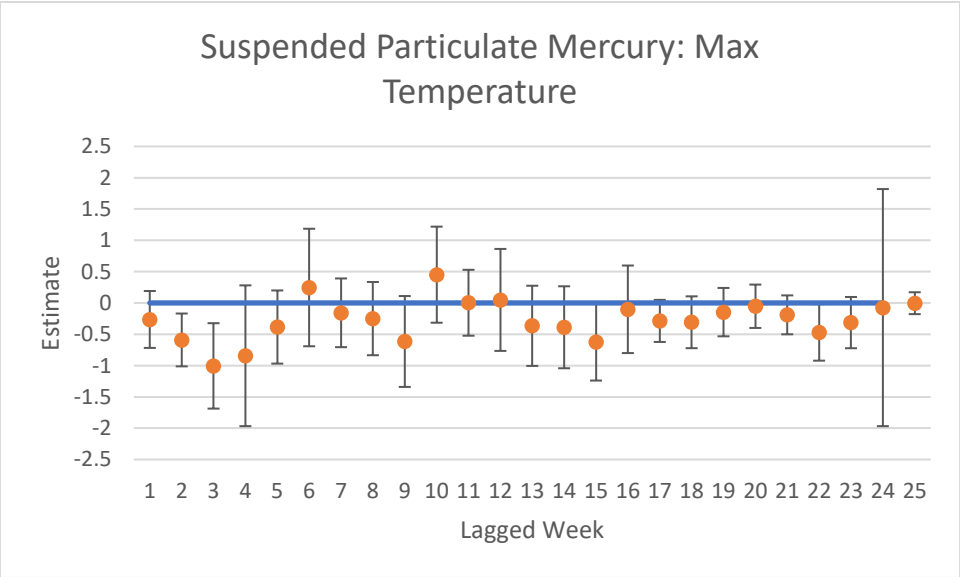

Figure 5c. Suspended Particulate Mercury Estimate and Average Weekly Maximum Temperature (°C)

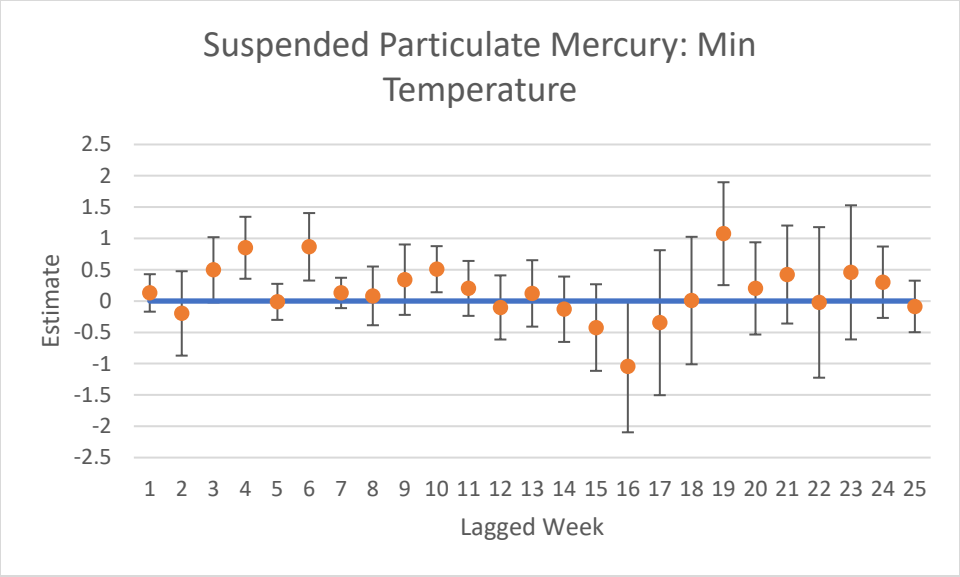

Figure 5d. Suspended Particulate Mercury Estimate and Average Weekly Minimum Temperature (°C)

Discussion

El Niño was found to significantly increase fish muscle tissue mercury compared to the non-El Niño year. Both sections 1 and 3 were significantly increased from the non-El Niño year, with section 1 seven times as high and section 3 two times as high as the non-El Niño year. Fish from section 2 in the El Niño year collection were not analyzed in time to be included. Interestingly, fish mercury in the non-El Niño year followed a similar trend to particulate mercury in the same year, with the highest levels in section 2 and lowest levels in section 1, but the El Niño collection showed greater concentrations in section 1 than in section 3. This was true for both carnivorous and non-carnivorous fish and could not be explained by the weight of the fish.

Carnivorous fish concentrations were higher than non-carnivorous fish. Carnivorous and non-carnivorous fish were also significantly greater in the El Niño year compared to the non-El Niño year. During the non-El Niño year, 47 of 200 (24%) fish samples, and 39 of 123 (32%) carnivorous fish samples surpassed the World Health Organization mercury guideline of 0.5 mg/kg. During the El Niño year, 60 of 64 (94%) of fish samples, and 32 out of 36 (89%) of carnivorous fish samples surpassed the WHO standard of 0.5 mg/kg. These high levels represent a public health risk especially considering the frequency of

occurrence, i.e. 94% of fish samples exceeding the WHO guideline in the El Niño year. Considering the significant difference for both carnivorous and non-carnivorous fish between the non-El Niño year and the El Niño year, El Niño events can greatly impact methylmercury bioaccumulation in aquatic food chains and lead to a public health risk.

There was no significant difference in TSS or suspended particulate mercury between El Niño and non-El Niño years. Suspended solids are an important indicator of mercury transport because mercury is generally adsorbed to particles rather than a free species in the water column (Diringer et al., 2015). TSS concentrations were consistently higher during the March-April wet seasons compared to the May-June dry seasons. Higher precipitation during the wet season likely contributes to greater sediment loading in the river. Section 2 had the highest levels of TSS of the river for each season and between the El Niño and non-El Niño years. The high TSS concentrations in section 2 coincide with the greater density of ASGM operations. ASGM contributes to high rates of deforestation which would impact the amount of sediment loading in that section of the river. Deforestation contributes to enhanced sediment loading in rivers due to increased runoff (Diringer et al., 2015).

There was no significant difference between the non-El Niño and the El Niño years for suspended particulate mercury, nor did the El Niño year follow the patterns of particulate mercury or TSS in the non-El Niño year. In the non-El Niño year, suspended particulate mercury was greater during the March-April wet season than the June-July dry season, similar to the trend between wet and dry season seen in TSS. However, this trend was not seen during the El Niño year, where higher levels were seen in the dry season compared to the wet season. This could be impacted by limitations noted below. Section 2 had the highest levels of particulate mercury across seasons and years except for the June-July dry season of the non-El Niño, which was surpassed by levels in section 3. Interesting to note is that, where particulate mercury levels were expected to increase during the El Niño event, most sections decreased in concentrations. However, between the dry seasons particulate mercury was higher in sections 1 and 2 in the El Niño year.

Despite the low concentrations seen in the wet season during the El Niño year and the possibility of mercury evaporation, section 3 during the dry season also saw a decrease in mercury concentrations compared to both sample collections of the non-El Niño year. This is an observation opposed to the hypothesis on El Niño levels. However, particulate mercury is more an indication of mercury transport in the river system (Diringer et al., 2015) rather than a spatial representation of mercury like sediment measurements would be. There may be more ecological factors involved in the lower mercury concentrations during the El Niño year when such high levels of mercury in fish is observed. Additionally,

it is difficult to target locations where major introduction of mercury into the river or production of methylmercury may be located. Where sampling an entire transect of the river may be the only way to truly capture a picture of particulate mercury concentrations in the river, only one location for an area of the river is sampled.

Precipitation, temperature, and soil moisture were all related to fish tissue mercury concentrations. Precipitation significantly impacted mercury concentrations from 16 and 19 weeks before the date of sample collection. Lagged week 16 decreased fish mercury levels by 0.011 mg/kg (on a log scale), but lagged week 19 increased mercury by 0.023 mg/kg (on a log scale) for every 1 mm increase in precipitation. This time line was not much longer than the timeline of precipitation affecting particulate mercury despite the idea that bioaccumulation in fish should take some time. Minimum temperature from 27 weeks before increased mercury levels. This is on a much longer time scale than the effect of maximum temperature on particulate mercury and had a positive impact on mercury rather than a negative impact. With the long-term time scale, at the 27 week previous mark, minimum temperature could foster ideal conditions for microbial activity and an increase rate in mercury methylation. This could explain an increase of 0.24 mg/kg (on a log scale) per degree Celsius in fish 27 weeks later. Soil moisture 48 weeks previous increased mercury levels. This large time scale indicates climate factors up to a year back can impact aquatic food chains.

Precipitation, temperature, and soil moisture were all related to particulate mercury concentrations. Temperature and soil moisture were found to significantly impact mercury concentrations from 3 weeks before the date of sample collection. Maximum temperature from 3 weeks previous decreased mercury, and soil moisture from 3 weeks previous increased mercury. A short time scale of a few weeks makes sense particulate mercury measured in the water samples was likely recently loaded into the waterway or methylated by microbial activity and before transport or uptake into organisms. Maximum temperature could decrease mercury levels if those temperatures are surpassing ideal temperature conditions for efficient microbial methylation activity. Soil moisture may be an indicator of earlier precipitation and flood-like conditions that input organic matter and developed conditions favorable to higher methylation rates.

Interestingly, precipitation 15 weeks previous significantly impacted particulate mercury. In other words, high precipitation will increase particulate mercury levels by 0.013 ng/L (on a square root scale) for every 1 mm precipitation 15 weeks later. This time scale is longer than temperature or soil moisture influence but may involve other ecological factors between the meteorological impact and the result in

mercury concentrations. These ecological factors could include the timeline between input of organic matter and mercury and an increase in microbial activity, the development of conditions favorable to microbial activity, such as anoxia, or the actual rate of methylation by bacteria.

There are several findings that reaffirm conclusions of the Diringer et al. study (2015). Section 2 had the highest levels of particulate mercury in both the wet and dry seasons of the El Niño year which reaffirms the past study's conclusion that ASGM contributed mercury to the river due to higher mercury concentrations near the highest density of ASGM activity. TSS levels were higher during the wet season compared to the dry season, demonstrating an increased input of sediment during high precipitation. Carnivorous fish also had considerably higher levels of mercury than non-carnivorous fish. Diringer et al. demonstrated that communities not directly involved in ASGM are exposed to an environmental contaminant, methylmercury, imposing a significant health impact. This is exemplified by the fact that in the El Niño year, methylmercury in fish was several times higher than the non-El Niño year.

The biggest limitation to this study was an incomplete account of samples. Some fish samples were not included in the El Niño collection for sections 1 and 3 and no samples were included from section 2. Fish samples are still in Peru and unable to be analyzed in laboratories. The low concentrations during the March-April wet season during the El Niño year may have been affected by a delayed analysis. Whereas water samples from all other seasons were analyzed between 1-3 weeks after field collection, filtered and unfiltered water samples took up to several months for mercury concentrations to be measured. Mercury in the water could have evaporated during this period and compromised the results for that season.

Also, limited ability in time and expense to collect more water samples across different areas of the width of the river may limit obtaining a full picture of mercury concentrations in the water column. Limited number of sample collections could mean important areas of the river or floodplain where methylation is occurring are missed. Sediment was also collected to obtain a spatial understand of total and methylmercury in the river, but those were also not analyzed in time. Measuring mercury in sediment will allow for a clearer understanding of the process occurring between mercury in the water, methylation in soils and sediments, and eventual bioaccumulation in fish.

Conclusion

This is one of the first studies to investigate the impact of El Niño on environmental mercury concentrations. This study found that El Niño has a large effect in increasing fish tissue mercury concentrations. This could represent a public health threat to communities dependent on fish from waters with methylmercury present, where ASGM and poor mercury regulation is prevalent, and El Niño is an intermittent occurrence. Additionally, this study defined specific meteorological factors that impact environmental mercury concentrations. Both these climate components and El Niño could become more extreme due to climate change (Cai et al., 2014) as predicted by the Intergovernmental Panel on Climate Change Technical Summary (Rolfhus et al., 2015) and the third National Climate Assessment (Rolfhus et al., 2015).

Many studies have investigated the implication that El Niño increases health risk such as respiratory and enteric diseases (Checkley et al., 2000), malnutrition (Bayer et al., 2014; Danysh et al., 2014), and vector-borne disease (Bouma & Dye, 1997; Gagnon et al., 2002). However, this study provides evidence that El Niño also modifies environmental toxin exposures. The potential consequences of high precipitation events for methylmercury exposure are also not limited to populations in Peru. The implication of El Niño's effect on environmental contaminants means that in areas around the world that experience El Niño or extreme climate conditions could be vulnerable to a multiplicative effect in exposure to an environmental contaminant such as methylmercury. This phenomenon could also apply to other regions, such as the upper Midwest and Northeast United States, where climate change is expected to increase the frequency of wet weather events (Walsh, Wuebbles, & Hayhoe, 2014). Methylmercury accumulation in fish is also a concern for many lakes, streams, and reservoirs in these regions of the U.S.

Considering the large impact El Niño had on fish mercury levels, further study should investigate more extensively how these more specific meteorological factors are influencing mercury levels. The period between a specific day of flooding, precipitation, temperature, soil moisture, or solar radiation and its eventual impact on particulate, sediment, or fish mercury is not well or easily characterized. To better predict how extreme weather, such as El Niño, will affect environmental mercury or other contaminants for hazard preparation, these influences should be better studied.

Rio Madre de Dios needs to become better characterized in order to understand key locations of methylmercury production. With the disconnect between low particulate mercury levels and high fish mercury, it is likely that areas of high methylmercury production are not being represented in sample

collection. With the health risk ASGM imposes from its high input of mercury into this river, any remediation effort would depend on understand those locations within the MDD watershed.

Based on conclusions from the previous study (Diringer et al., 2015), and considering the higher frequency of fish with mercury concentrations exceeding WHO standards compared to the 2013 collection, children and pregnant women living near the central part of the river could not safely consume carnivorous fish. During El Niño events, this could extend to other parts of the river considering the high levels found in sections 1 and 2 of the river. The risks of high mercury levels in fish and the impact of extreme weather conditions should be communicated to these communities. The most accessible solution is to put forth policies that could regulate ASGM and the mercury released from operations. With high prices of gold, ASGM is not likely to go away, but containing the mercury burned off is a solution to decreasing atmospheric mercury from the amalgam burning.

Works Cited

Ashe, K. (2012). Elevated mercury concentrations in humans of madre de dios, peru. *PLoS ONE, 7*(3), e33305. doi:10.1371/journal.pone.0033305

Asner, G. P., Llactayo, W., Tupayachi, R., & Luna, E. R. (2013). Elevated rates of gold mining in the amazon revealed through high-resolution monitoring. *Proceedings of the National Academy of Sciences of the United States of America, 110*(46), 18454-18459. doi:10.1073/pnas.1318271110

Basu, N., Clarke, E., Green, A., Calys-Tagoe, B., Chan, L., Dzodzomenyo, M., . . . Wilson, M. L. (2015). Integrated assessment of artisanal and small-scale gold mining in ghana--part 1: Human health review. *Int J Environ Res Public Health, 12*(5), 5143-5176. doi:10.3390/ijerph120505143

Bayer, A. M., Danysh, H. E., Garvich, M., Gonzálvez, G., Checkley, W., Alvarez, M., & Gilman, R. H. (2014). The 1997–1998 el niño as an unforgettable phenomenon in northern peru: A qualitative study. *Disasters, 38*(2), 351-374. doi:10.1111/disa.12046

Bernaudat, L., & Keane, S. (2013). Partnership on reducing mercury in artisanal and small-scale gold mining (asgm). *E3S Web of Conferences, 1*, 36003.

Bouma, M. J., & Dye, C. (1997). Cycles of malaria associated with el niño in venezuela. *JAMA, 278*. doi:10.1001/jama.1997.03550210070041

Brooks, W. E., Sandoval, E., Yepez, M. A., & Howard, H. (2007). Peru mercury inventory 2006: U.S. Geological survey open-file report 2007-1252.

Buccella, A. (2014). Can the minamata convention on mercury solve peru's illegal artisanal gold mining problem? *Yearbook of International Environmental Law, 24*(1), 166-187. doi:10.1093/yiel/yvu063

Cai, W., Borlace, S., Lengaigne, M., van Rensch, P., Collins, M., Vecchi, G., . . . Jin, F.-F. (2014). Increasing frequency of extreme el nino events due to greenhouse warming. *Nature Clim. Change, 4*(2), 111-116. doi:10.1038/nclimate2100

http://www.nature.com/nclimate/journal/v4/n2/abs/nclimate2100.html#supplementary-information

Checkley, W., Epstein, L. D., Gilman, R. H., Figueroa, D., Cama, R. I., Patz, J. A., & Black, R. E. (2000). Effect of el niño and ambient temperature on hospital admissions for diarrhoeal diseases in peruvian children. *Lancet, 355*.

Compeau, G. C., & Bartha, R. (1985). Sulfate-reducing bacteria: Principal methylators of mercury in anoxic estuarine sediment. *Applied and Environmental Microbiology, 50*(2), 498-502.

Council, W. G. (2016). Retrieved from http://www.gold.org/

Crompton, P., Ventura, A. M., de Souza, J. M., Santos, E., Strickland, G. T., & Silbergeld, E. (2002). Assessment of mercury exposure and malaria in a brazilian amazon riverine community. *Environ Res, 90*(2), 69-75.

Danysh, H. E., Gilman, R. H., Wells, J. C., Pan, W. K., Zaitchik, B., Gonzálvez, G., . . . Checkley, W. (2014). El niño adversely affected childhood stature and lean mass in northern peru. *Climate Change Responses, 1*(1), 7. doi:10.1186/s40665-014-0007-z

Diringer, S. E., Feingold, B. J., Ortiz, E. J., Gallis, J. A., Araujo-Flores, J. M., Berky, A., . . . Hsu-Kim, H. (2015). River transport of mercury from artisanal and small-scale gold mining and risks for dietary mercury exposure in madre de dios, peru. *Environ Sci Process Impacts, 17*(2), 478-487. doi:10.1039/c4em00567h

EPA, U. (1998). Method 7473 mercury in solids and solutions by thermal decomposition, amalgamation, and atomic absorption spectrophotometry. *Washington, DC*.

EPA, U. (2001). Method 1630 methyl mercury in water by distillation, aqueous ethylation, purge and trap, and cold vapor atomic fluorescence spectrometry

Washington, DC.
Feingold, B., Pan, W., Salmon-Mulanovich, G., & Estela Manrique, A. (2014). *Land use, household vulnerability and dual disease burden among children living in the peruvian amazon*. Paper presented at the 26th Conference of the International Society for Environmental Epidemiology, Seattle, WA.
Fraser, B. (2009). Peruvian gold rush threatens health and the environment. *Environ Sci Technol, 43*(19), 7162-7164. doi:10.1021/es902347z
Gagnon, A. S., Smoyer-Tomic, K. E., & Bush, A. B. (2002). The el nino southern oscillation and malaria epidemics in south america. *Int J Biometeorol, 46*(2), 81-89.
Gardner, E. (2012). Peru battles the golden curse of madre de dios. *Nature, 486*(7403), 306-307. doi:10.1038/486306a
Gilmour, C. C., Podar, M., Bullock, A. L., Graham, A. M., Brown, S. D., Somenahally, A. C., . . . Elias, D. A. (2013). Mercury methylation by novel microorganisms from new environments. *Environ Sci Technol, 47*(20), 11810-11820. doi:10.1021/es403075t
Group, Z. M. W. (2011). Artisanal and small scale gold mining. *EBB Zero Merucy Campaign.* Retrieved from http://www.zeromercury.org/index.php?option=com_content&view=article&id=207&Itemid=95
Hodgson, N. W., Waly, M. I., Al-Farsi, Y. M., Al-Sharbati, M. M., Al-Farsi, O., Ali, A., . . . Deth, R. C. (2014). Decreased glutathione and elevated hair mercury levels are associated with nutritional deficiency-based autism in oman. *Exp Biol Med (Maywood), 239*(6), 697-706. doi:10.1177/1535370214527900
Hsu-Kim, H., Kucharzyk, K. H., Zhang, T., & Deshusses, M. A. (2013). Mechanisms regulating mercury bioavailability for methylating microorganisms in the aquatic environment: A critical review. *Environ Sci Technol, 47*(6), 2441-2456. doi:10.1021/es304370g
Huckabee, J., Elwood, J., & Hildebrand, S. (1979). *Accumulation of mercury in freshwater biota huckabee*. Amsterdam: Elsevier/North Holland Biomedical Press.
International, U. P. (2010). Peru poised to become 5th largest gold producer. Retrieved from http://www.upi.com/Peru-poised-to-become-5th-largest-gold-producer/83651274470647/
Kelly, C. A., Rudd, J. W. M., Bodaly, R. A., Roulet, N. P., St.Louis, V. L., Heyes, A., . . . Edwards, G. (1997). Increases in fluxes of greenhouse gases and methyl mercury following flooding of an experimental reservoir. *Environ Sci Technol, 31*(5), 1334-1344. doi:10.1021/es9604931
Langeland, A. L., Hardin, R. D., & Neitzel, R. L. (2017). Mercury levels in human hair and farmed fish near artisanal and small-scale gold mining communities in the madre de dios river basin, peru. *Int J Environ Res Public Health, 14*(3). doi:10.3390/ijerph14030302
Li, W., Zhang, P., Ye, J., Li, L., & Baker, P. A. (2011). Impact of two different types of el niño events on the amazon climate and ecosystem productivity. *Journal of Plant Ecology, 4*(1-2), 91-99. doi:10.1093/jpe/rtq039
Nakazawa, K., Nagafuchi, O., Kawakami, T., Inoue, T., Yokota, K., Serikawa, Y., . . . Elvince, R. (2016). Human health risk assessment of mercury vapor around artisanal small-scale gold mining area, palu city, central sulawesi, indonesia. *Ecotoxicology and Environmental Safety, 124*, 155-162. doi:http://dx.doi.org/10.1016/j.ecoenv.2015.09.042
Niane, B., Guedron, S., Moritz, R., Cosio, C., Ngom, P. M., Deverajan, N., . . . Pote, J. (2015). Human exposure to mercury in artisanal small-scale gold mining areas of kedougou region, senegal, as a function of occupational activity and fish consumption. *Environ Sci Pollut Res Int, 22*(9), 7101-7111. doi:10.1007/s11356-014-3913-5

Pan, S. (2014). Exploring the effect of short term mobility on dengue in the peruvian amazon [masters thesis]: University of north carolina at chapel hill.

Porvari, P., & Verta, M. (1995). Methylmercury production in flooded soils: A laboratory study. In D. B. Porcella, J. W. Huckabee, & B. Wheatley (Eds.), *Mercury as a global pollutant: Proceedings of the third international conference held in whistler, british columbia, july 10–14, 1994* (pp. 765-773). Dordrecht: Springer Netherlands.

Programme, U. N. E. (2013). Global mercury assessment 2013; sources, emissions, releases, and environmental transport. *Geneva, Switzerland: UNEP Chemicals Branch*.

Project, M. o. t. A. A. (2017). Maap #58: Link between peru's flooding and warm coastal waters. Retrieved from http://maaproject.org/2017/climate-flooding/

Rolfhus, K. R., Hurley, J. P., Bodaly, R. A., & Perrine, G. (2015). Production and retention of methylmercury in inundated boreal forest soils. *Environ Sci Technol, 49*(6), 3482-3489. doi:10.1021/es505398z

Selin, N. E. (2009). Global biogeochemical cycling of mercury: A review. *Annual Review of Environment and Resources, 34*(1), 43-63. doi:10.1146/annurev.environ.051308.084314

St.Louis, V. L., Rudd, J. W. M., Kelly, C. A., Bodaly, R. A., Paterson, M. J., Beaty, K. G., . . . Majewski, A. R. (2004). The rise and fall of mercury methylation in an experimental reservoir. *Environ Sci Technol, 38*(5), 1348-1358. doi:10.1021/es034424f

Swenson, J. J., Carter, C. E., Domec, J.-C., & Delgado, C. I. (2011). Gold mining in the peruvian amazon: Global prices, deforestation, and mercury imports. *PLoS ONE, 6*(4), e18875. doi:10.1371/journal.pone.0018875

Valdivia, S. M., & Ugaya, C. M. L. (2011). Life cycle inventories of gold artisanal and small-scale mining activities in peru. *Journal of Industrial Ecology, 15*(6), 922-936. doi:10.1111/j.1530-9290.2011.00379.x

Veiga, M. M., Maxson, P. A., & Hylander, L. D. (2006). Origin and consumption of mercury in small-scale gold mining. *Journal of Cleaner Production, 14*(3–4), 436-447. doi:http://dx.doi.org/10.1016/j.jclepro.2004.08.010

Walsh, J., Wuebbles, D., & Hayhoe, K. (2014). Ch. 2: Our changing climate. Climate change impacts in the united states. *The Third National Climate Assessment*.

Wyatt, L., Ortiz, E., Feingold, B., Berky, A., Diringer, S., Ana Maria Morales, A. M., . . . Pan, W. (In-review). Mercury exposure in peruvian riverine communities near artisanal and small-scale gold mining: The importance of household diet. *PLoS ONE*.

Yard, E. E., Horton, J., Schier, J. G., Caldwell, K., Sanchez, C., Lewis, L., & Gastanaga, C. (2012). Mercury exposure among artisanal gold miners in madre de dios, peru: A cross-sectional study. *J Med Toxicol, 8*(4), 441-448. doi:10.1007/s13181-012-0252-0