8-2008

Mercury emissions from coal-fired power plants : a local or a global pollutant?

Paul Franklin Tirey 1961-
University of Louisville

MERCURY EMISSIONS FROM COAL-FIRED POWER PLANTS: A LOCAL OR A GLOBAL POLLUTANT?

By

Paul Franklin Tirey
B.S. Mechanical Engineering, University of Kentucky, 1983
Master of Engineering, University of Louisville, 1994

A Dissertation
Submitted to the Faculty of the
Graduate School of the University of Louisville
In Partial Fulfillment of the Requirements
for the Degree of

Doctor of Philosophy

Department of Urban and Public Affairs
University of Louisville
Louisville, Kentucky

August 2008

MERCURY EMISSIONS FROM COAL-FIRED POWER PLANTS: A LOCAL OR A GLOBAL POLLUTANT?

By

Paul Franklin Tirey
B.S., Mechanical Engineering, University of Kentucky, 1983
Master of Engineering, University of Louisville, 1994

A Dissertation Approved on

July 7, 2008

By the following Dissertation Committee:

_____
Dissertation Director

_____

_____

_____

_____

ABSTRACT

MERCURY EMISSIONS FROM COAL-FIRED POWER PLANTS: A LOCAL OR A GLOBAL POLLUTANT?

Paul Franklin Tirey

July 7, 2008

This dissertation tests whether or not mercury emissions from electric power plants are not a significant contributor to mercury measurements in rainfall and argues that the current United States (U.S.) Environmental Protection Agency (EPA) proposed regulatory scheme for controlling mercury from electric power plants, the Clean Air Mercury Rule (CAMR), is an effective regulatory mechanism by using a number of ordinary least square (OLS) and spatial regression models. Two dependent variables are tested, mercury concentration (the average mercury concentration measured in rainfall in nanograms per liter, ng/L) and mercury deposition (the total annual mercury falling at each measurement site in nanograms per square meter, ng/m^2), with mercury concentration determined to be the more valid dependent variable. The source for the mercury concentration and deposition data is the Mercury Deposition Network (MDN), part of the National Atmospheric Deposition Program (NADP), with the data obtained for between 46 and 75 sites operating from 2001 through 2005.

Independent variables include: 1) emissions to the air from power plants, 2) emissions to the air from other industrial sites, 3) emissions to the land from the mining industry, 4) population as a proxy variable for vehicle emissions, 5) burned area from wildfires, 6)

precipitation and 7) dummy variables for year and EPA region. Data for independent variables 1, 2, and 3 were obtained from the EPA's Toxic Release Inventory (TRI) program. Population for each county in the U.S. was obtained from the Census Bureau, and wildfire data was obtained from the U.S. Department of Agriculture satellite based fire mapping system, Moderate Resolution Imaging Spectroradiometer (MODIS). Microsoft Access was utilized to summarize and total the independent variables within a variable radius of the MDN measurement sites, ranging from 25 to 500 miles. The software tool GeoDa 0.95i, made available by the University of Illinois, was used to perform the OLS, spatial lag, and spatial error regressions.

After changing the functional form of the equation to a log-linear model (using the natural log form of the dependent variable and the linear forms of the independent variables) to deal with heteroskedasticity, the results indicate a strong spatial component to the model. Other than precipitation, the most significant predictor of mercury concentration is fire area burned between 50 and 75 miles of the MDN measurement site ($z = 3.08$, $p<0.01$). Other positive and significant predictors in this model include all other industry emissions between 25 and 50 miles ($z = 2.71$, $p<0.01$), fire area burned between 75 and 100 miles ($z = 2.64$, $p<0.01$), population within 25 miles ($z = 1.91$, $p<0.10$), utility emissions between 25 and 50 miles ($z = 1.88$, $p<0.10$), and population between 50 and 75 miles ($z = 1.71$, $p<0.10$). Two of the independent variables are significant and have negative coefficients. These are utility emissions between 50 and 75 miles ($z = -2.49$, $p<0.05$), and fire area burned between 25 and 50 miles ($z = -2.12$, $p<0.05$).

Several conclusions are drawn from this research, including: 1) that utility mercury emissions are marginally significant as a predictor of mercury concentration in rainfall, but only at distances under 50 miles from the measurement point, 2) that there is no known best method for controlling mercury emissions from all utility plants at high levels of collection efficiency (90 percent) although research is ongoing, and 3) that the cap-and-trade provisions of CAMR would be unlikely to result in the creation of new or the exacerbation of existing mercury hotspots. Given that the U.S. District of Columbia Circuit Court of Appeals set aside the CAMR rule in early 2008, two policy prescriptions are provided. One approach makes an economic argument for revising the cap-and-trade provisions of CAMR to include transfer coefficients. The second suggestion involves a less complicated and more politically acceptable change to the trading rules for mercury.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER I

INTRODUCTION

Executive Summary

This study explores whether command-and-control type regulations or market based initiatives are best suited to reduce mercury air emissions from coal-fired electric power plants in the U.S. The debate centers on the question of whether to regulate mercury under a cap-and-trade type regulatory scheme or to require more stringent Maximum Achievable Control Technology (MACT) type regulations (such as those applying to other mercury point sources, for example, municipal incinerators). As the following discussion will make clear, the debate over the most efficient and effective way to regulate mercury emissions centers on the essential research question of this dissertation: Are mercury emissions from coal-fired power plants local or global pollutants?

Utilizing data from the EPA Toxic Chemical Release Inventory (TRI), mercury emissions from coal fired power plants can be included in a regression model to test these emissions as a predictor of mercury present in rainfall. Mercury in rainfall is measured weekly at various monitoring sites in the U.S. as part of the National Atmospheric Deposition Program's Mercury Deposition Network (MDN). Based on a reading of the available literature on this subject, the main hypothesis for this study is that mercury emissions from coal-fired power plants do not correlate with mercury measurements in rainfall. Instead, power plant emissions join the global mercury pool and are not a

significant contributor to the levels of mercury measured in rainfall. This hypothesis will be tested utilizing an OLS regression model and a spatial regression model, where a distance based weight matrix takes into account the distances between the receptors (measurement sites). The dependent variable is mercury concentration in rainfall, and the independent variables include mercury emissions to the air from power plants (identified using the Standard Industrial Classification, SIC, code), mercury emissions to the land from the mining industry, mercury emissions to the air from all other industrial sources, mercury in forest fire smoke, precipitation, and mercury resulting from vehicle emissions (using population as a proxy variable).

In addition to the central research question and main hypothesis, the question of mercury's toxicity is also addressed in this study. Opponents of cap-and-trade for mercury point to the perception that mercury emissions from power plants are so highly toxic to humans and the environment, that it is necessary to regulate mercury with the most stringent means possible. This research project will examine the available information in the literature to determine what is currently known about the extent of mercury contamination in the environment due to power plant emissions, and if the best way to regulate mercury at power plants is in fact, known and can be defined in a MACT type regulation.

Following the executive summary, this introductory chapter includes a section on context, where additional information concerning the current EPA mercury regulatory proposal is provided, as well as a description of how mercury is transported in the environment before becoming a toxicity hazard for humans. Next, the problem addressed by this research is more fully explained and developed, followed by a clear definition of

purpose and the overall significance of the study results (the "so what"). In Chapter II, a literature review discusses eight major areas of mercury research, including the history of mercury emissions, speciation and atmospheric transport, mercury emissions from power plants, emissions trading, hotspots, mercury in vehicle exhaust, mercury toxicity, and mercury in forest fire smoke. Chapter III covers the methodology of the research, discussing the data sources, the regression models used to test the hypothesis, and the limitations of the study. The results of the study are presented in Chapter IV, followed by discussion (Chapter V) and a final summary and conclusion in Chapter VI. Based on the results of the study, two policy prescriptions for the future regulation of mercury emissions are proposed.

Context

The U.S. Environmental Protection Agency (EPA) issued a proposed regulation on May 18, 2005 to regulate mercury emissions from coal-fired power plants. Mercury emissions from these sources have not previously been regulated by the EPA, nor have power plant mercury emissions been the subject of regulation in any other country (U.S. EPA, 2006a). The rule, "Standards of Performance for New and Existing Stationary Sources: Electric Steam Generating Units" is also known as the Clean Air Mercury Rule (CAMR). CAMR creates a cap-and-trade program that will operate in two phases. Phase 1 sets an annual cap of 38 tons of mercury emissions by 2010, and phase 2 reduces the annual cap to 15 tons beginning in 2018. These amounts represent reductions from 1999 emissions (48 tons) of about 20 and 70 percent, respectively. The EPA estimates that phase 1 reductions will result from "co-benefit" reductions achieved under another rule, the Clean Air Interstate Rule (CAIR). CAIR requires reductions in sulfur dioxide and

nitrous oxides emissions (SO_2 and NO_x). The equipment installed to meet CAIR requirements will also collect mercury, with collection efficiency depending on many factors such as coal type and the specific combination of collection equipment installed (U.S. EPA, 2005). The emissions trading program under CAMR allows states to join a nationwide trading program, prohibit interstate or intrastate trading altogether, or develop alternative state rules that will meet caps for each state set up under the rule. As of December 4, 2007, 16 states had notified the EPA that they would not participate in emissions trading, while 34 states indicated they would participate in trading either fully, or with some caveats (NACAA, 2007). CAMR also requires power plants to install mercury emissions monitors on stacks beginning in 2009.

Soon after publishing the final CAMR, the EPA received two Petitions for Reconsideration of the rule requesting more public comment, one from 14 states and one from 5 environmental groups and 4 Native American Tribes. The petitions were granted and the rule reopened for additional public comment with hearings held in October 2005. After considering the results of this additional information, the EPA re-issued the CAMR rule on May 31, 2006 with some slight changes, but with the notion of emissions trading for mercury intact (U.S. EPA, 2006c).

Opponents of CAMR are concerned that mercury emissions from power plants contribute to localized mercury pollution and contamination, and that emissions trading will exacerbate the problem. Those opposed to trading believe that plants wishing to avoid the costs of installing mercury controls can purchase emission allowances and continue to emit mercury. They claim that mercury is a local, not a global pollutant, and cannot be effectively controlled through emissions trading. However, not everyone agrees

that mercury is a local pollutant. In fact, the EPA estimates that 83 percent of the mercury deposited in the U.S. comes from international sources (U.S. EPA, 2006b: 11).

Mercury is a toxic heavy metal that travels through the atmosphere and is deposited back to the earth through dry (soot) and wet (rain) deposition processes, sometimes hundreds or thousands of miles from the source, depending on the form of emission. The sources of mercury include air emissions from industrial and power plants, emissions from incinerators, mercury compounds in industrial and domestic point source water emissions, leachate from landfills, and natural releases from volcanoes and natural erosion processes. The process by which mercury travels through the environment is sometimes referred to as the mercury cycle, as illustrated in Figure 1, below, obtained from the Northeast Waste Managers Association (NEWMOA, n.d.).

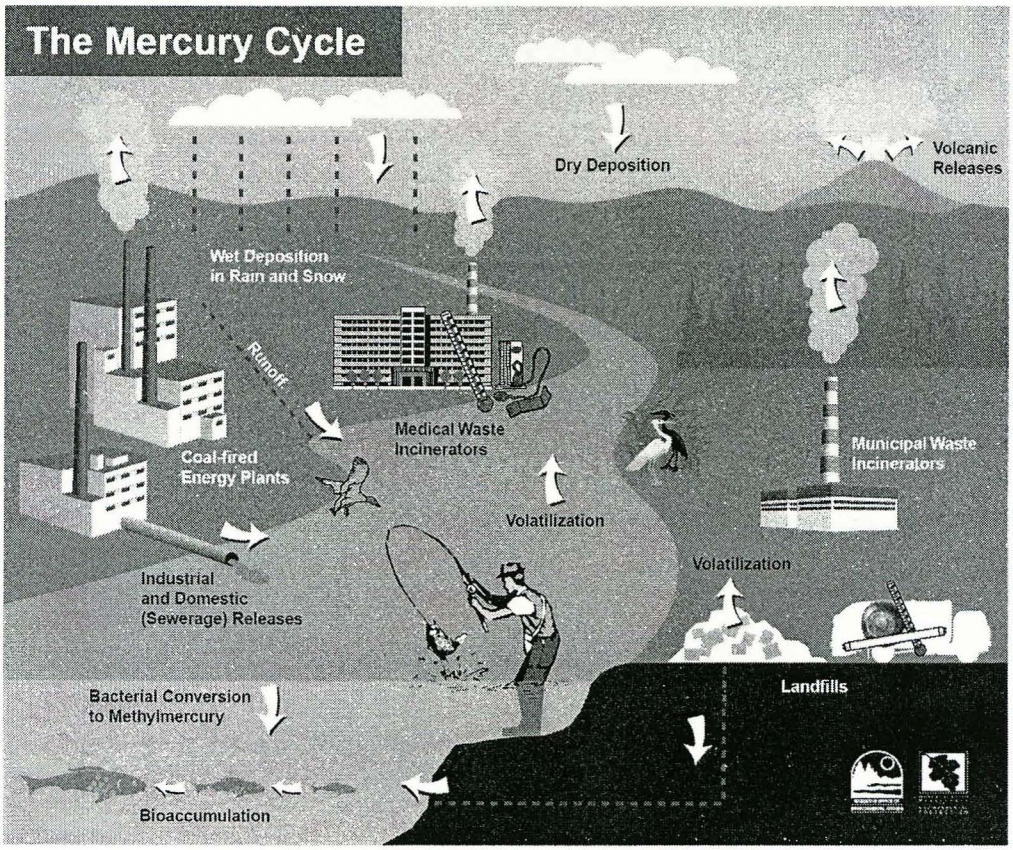

Figure 1. The Mercury Cycle. Source: The Northeast Waste Managers Association, n.d.

As the figure illustrates, the primary exposure to mercury for humans is from eating fish. Mercury emitted from industrial and other natural and anthropogenic sources falls to the earth and ends up in lakes and rivers, where micro organisms convert it into highly toxic methylmercury. Methylmercury bio-accumulates through the aquatic food chain, ending up in the meat of fish and shellfish. At highest risk are women of child bearing age, pregnant and nursing mothers, and young children who consume large amounts of fish. High levels of mercury in young children or fetuses can cause developmental and neurological problems (U.S. EPA, 2006d). However, studies have shown that actual levels of mercury in at risk groups in the U.S. are low, and also that fish consumption can help, rather that inhibit child development. At least one study found that fish consumption by expectant mothers was significantly associated with higher infant cognition. The levels of mercury in the blood and hair of the subjects in this study, which looked at 135 mother-infant pairs in Eastern Massachusetts, were much lower than previous studies of high fish consuming island people used by the EPA to define the toxicity of mercury (Oken, et. al., 2005). However, there have been instances where misuse of mercury resulted in larger scale mercury poisoning.

One of the most famous such episodes of mercury poisoning occurred in the 1950's in the Minamata region of Japan, where a chemical manufacturer dumped 27 tons of mercury in the bay. The locals, mostly fishermen whose daily diet included fish from the bay, were exposed to very high levels of methylmercury. This amount of mercury deposited in a locality was an environmental disaster. It is estimated that over 900 people died and over 2 million suffered health problems from eating the fish from the bay in later years (McCurry, 2006).

It is clear that mercury is a pollutant that represents an exposure hazard to humans through the consumption of fish, and that the electric power industry is a major contributor to the total amount of anthropogenic levels of mercury emitted to the air each year in the U.S. An environmental policy is warranted, but what policy? The debate that is explored in this study is not over whether to regulate mercury at all, but on the choice of the most efficient and effective environmental policy.

Problem and Purpose

The essential problem addressed by this study is whether the mercury emitted by the typical electric power plant acts more like a global, rather than a local pollutant. According to economic theory, this is a crucial question. If mercury is a global pollutant, a cost efficient control solution can be achieved through emissions trading. Cost effective does not mean zero emissions, but rather that the marginal cost of additional control does not exceed the value of the marginal benefit to society, and that the cost is less than the cost of the command-and-control approach. In addition, if the permit market is competitive, emissions trading will also result in incentives for technological innovation (Tietenberg, 2006: 45-46). This means that the optimum level of mercury pollution is not zero. It is a function of how much it will cost society to clean up each ounce of mercury, versus how much the benefit of the cleanup will be worth to society.

In support of CAMR, the EPA concluded that the implementation of mercury emissions trading would not result in mercury contamination hotspots attributable to utilities. In May 2006, the EPA Office of Inspector General published the results of an evaluation report that assesses the basis of this conclusion. The result of that evaluation was that significant uncertainties exist regarding the conclusion that CAMR will not

result in hotspots, including uncertainties associated with: 1) gaps in the data and science associated with mercury emission estimates, 2) limitations with models used to predict deposition, and 3) uncertainty over how mercury reacts in the atmosphere and changes to the more toxic form of methylmercury (U.S. EPA OIG, 2006: 11). Others agree there are significant knowledge gaps.

A 1997 EPA Science Advisory Board reviewed the 1997 EPA Draft Mercury Study Report to Congress and found: 1) that the relative contribution of anthropogenic mercury emissions is highly uncertain, and 2) that the majority of the human population is not exposed to methylmercury at levels that are a concern to human health (U.S. EPA, 1997: 2-3). Another researcher, Carpri (1997), concluded that the speciation (the distinct molecular forms of mercury that are emitted in power plant stack gases) and transport of mercury emissions in the atmosphere are not well understood. He sited the need for additional research, especially associated with the measurement of mercury near emission sources. Indeed, since U.S. power plant mercury emissions are such a small portion of overall worldwide natural and anthropogenic mercury emissions, the cost effectiveness of eliminating power plant mercury has been studied, and the results are mixed. Lipfert, et. al. (2005) explored mercury controls on power plant emissions from a cost-benefit standpoint and found that eliminating coal-fired mercury emissions will have limited public health benefits. Gayer and Hahn (2006) found that while neither a MACT approach nor the cap-and-trade approach of CAMR would result in positive net benefits to society, the gap between costs and benefits is much lower for cap-and-trade. They noted that "...costs are larger than benefits by well over two orders of magnitude for the MACT proposal and well over one order of magnitude for the cap-and-trade proposal

(Gayer and Hahn, 2006: 313). However, Highlander and Goodsite (2006) found that controls on power plant mercury emissions would be cost effective with benefits associated with public health.

Finally, there are conflicting studies regarding whether or not power plant mercury is a local pollutant. The evidence from plant emissions modeling and downwind testing studies is mixed. For example, although one study concluded that coal combustion sources are associated with wet deposition monitoring sites at a single location in Steubenville, Ohio (Keeler, et. al., 2006: 5874), another researcher (Sullivan, 2003 and 2005) documented three studies by the Brookhaven National Laboratory, that found little evidence of local mercury deposition near power plant locations.

The uncertainty associated with these studies involving mercury is very important when considering possible policy approaches. The major downfall associated with the decision of how to regulate mercury from electric power plants is the risk of over-regulation. Since the costs of mercury control will be directly incorporated into the rates that every electricity consumer pays, the over-regulation of mercury will impact everyone, including those who can least afford it. Whenever the marginal benefits to society due to a given level of control of a pollutant are uncertain, it is incumbent on the regulating authority to find the best regulatory solution resulting in the most control at the least cost. In the case of mercury from electric power plants, uncertainties abound. These uncertainties include the following: 1) the form in which mercury is emitted from power plant stacks and how to measure it, 2) the impact of reducing mercury emissions from power plant stacks on the mercury measured in the environment and found in fish, 3) the nature of the risk to humans from the current levels of mercury found in the environment

and what the appropriate control point should be, 4) the most effective technology for controlling mercury in electric generation plants; how much such technology will cost; and how effective cleanup will be, and 5) whether mercury is accumulating in any given area (hotspot) due to the impact of local emission sources. Another reason it is important to consider costs in the face of this uncertainty is because this is not the only area where the impact of regulation is directly affecting the cost of electricity. Carbon legislation, transmission grid security, reliability standards, and financial governance regulations, are all regulatory areas impacting the cost of producing electricity in the U.S. As new regulations are implemented there should be some expectation that benefits will be worth the costs. Chapter II, Literature Review, will explore these uncertainties in more detail.

In a recent development, 16 states and 9 environmental groups filed legal briefs with the District of Columbia Circuit Court of Appeals, claiming that the EPA's CAMR rule will be damaging to the public health (EEI, 2007). On February 8, 2008 the United States Court of Appeals for the District of Columbia ruled in favor of the states and environmental groups, not because CAMR will be damaging to public health, but because the EPA did not follow the strict provisions of Section 112 of the Clean Air Act Amendments of 1990 in setting up the regulatory framework of CAMR. The court set aside CAMR and now the EPA will have to decide whether or not to appeal the decision, go back and try to first delist mercury from section 112 and then re-propose CAMR, or try to implement a MACT style regulation (U.S. Court of Appeals, 2008). If the EPA decides to abandon CAMR and go with MACT type regulations, an electric utilities industry attorney estimates that the most optimistic time frame for a new rule to be finalized would be 2011, with a implementation deadline of 2014 (Cash, 2008).

The purpose of this study is to utilize available data associated with power plant mercury emissions, and mercury measurements taken at wet deposition sites in the U.S., and determine if the level of mercury measured in rainfall is significantly associated with power plant emissions, , and if so, how far from the power plants does this significance holds. If so, then this information would provide additional support for those who oppose CAMR, and the information might be useful in determining how to modify CAMR. If not, then the emissions trading program put in place by the EPA may be the most effective control for mercury emissions, and states that have opted out of emissions trading might wish to re-think this position, in order to minimize the cost and maximize the effectiveness of the overall mercury control policy in the U.S.

Significance

The question of whether atmospheric mercury emissions from power plants are global or local pollutants is important to the arguments for and against cap-and-trade. Those who oppose emissions trading for mercury do not disagree that at least a portion of power plant mercury emissions are global. In a 2003 Study funded by Environmental Defense, Michael Shore (2003) states:

> Atmospheric mercury pollution that has reacted and combined with other pollutants tends to deposit locally or regionally, while unreacted mercury (elemental) tends to enter the global atmospheric pool, enabling it to be deposited virtually anywhere in the world (page 11).

Based on his interpretation of unpublished EPA modeling, Shore claims that most mercury deposition is local, and therefore he and many environmental groups oppose cap-and-trade for mercury. If mercury, however, is more of a global, rather than a local pollutant, then cap-and-trade could be the most cost effective and efficient way to reduce the amount of mercury emitted from power plants.

Emissions trading can reduce mercury emissions and can have an effect on the development of mercury removal technology. Today mercury control technology is highly variant, and is based on the results of limited field testing and not on continued commercial operation over extended periods of time. The EPA CAMR rule is based on the expected efficiency of mercury removal using sorbent (activated carbon) injection in the stack gases upstream of a bag-house filter or electrostatic precipitator. This technology has demonstrated mercury removal rates of between 30 and 95 percent, depending on coal type (GAO, 2005: 16). However, this equipment has not been permanently installed in plants. The tests are promising, but have lasted less than three months (GAO, 2005: 10). A 2005 study by the U.S. Government Accounting Office also found that sorbent technology and cost estimates vary widely and depend on site specific conditions (GAO, 2005:20). More recent studies on mercury removal from power plants support this conclusion, including those from Srivastava, et. al. (2006), Lohman, et. al. (2006), Sigler and Lee (2006), Wedig, et. al. (2008), and Yujin, et. al. (2008).

In the next chapter, a more thorough review of the available literature regarding this problem is presented to discuss these topics and to set up the basis for the hypothesis and research method of this study.

CHAPTER II

LITERATURE REVIEW

Previous research associated with mercury emissions comes from a variety of fields, including Biology, Toxicology, Atmospheric Science, Fuel Processing, Environmental Science, and Economics. This chapter describes some of this research and the conclusions drawn. The research is divided into eight areas including: 1) the history of mercury emissions, 2) speciation and atmospheric transport, 3) research specific to mercury emissions from power plant stacks, 4) research associated with emissions trading versus command-and-control regulations, 5) recent research available as to mercury hotspots in the environment, 6) mercury in vehicle exhaust, 7) mercury toxicity, and 8) mercury in forest fire smoke. The chapter concludes with a summary of the research and an introduction to the research methodology employed for this study.

The History of Mercury Emissions

Pirrone, et. al. (1998) examined the historical record of mercury emissions in North America, mainly due to gold and silver production. North American anthropogenic mercury emissions were as high as 1,700 tons per year in 1879. Changes in gold and silver mining reduced this annual total, but emissions due to manufacturing and fossil fuel combustion peaked in 1947 after the Great Depression (274 tons per year) and again in 1989 at 330 tons per year. Hylander and Meili (2003) examined world-wide mercury

emissions over the last 500 years. They estimated that globally, over one million tons of metallic mercury has been extracted from mercury ores. They also noted that current mining activities may contribute about one third of the total anthropogenic mercury emissions world wide. Hylander and Meili advocate a global ban on mercury mining activities (Hylander and Meili, 2003: 13).

There is some evidence that points to an ongoing decreases of mercury measured in North America. Driscoll, et. al. (2007) found that mercury deposition to sediment in the Northeast has decreased by about 25 percent in recent years, due to the overall reduction in U.S. anthropogenic emissions. They stated that one important factor for this reduction may likely be "controls on particulate matter and sulfur dioxide from electric utilities, and reductions in consumer and industrial Hg [mercury] use..." (20). The article also predicts that the elimination of mercury emissions from U.S. electric utilities will not be enough, in itself, to eliminate the biological hot spots in the Northeast (26). Another study found that mercury levels in fish in the Sacramento River Watershed have neither increased nor decreased over the last 25 years (Sacramento River Watershed Program, SRWP, 2002: 3). In a time-trend study of mercury levels in Wisconsin walleye, Madsen and Stern found a slight annual decrease in the region of about 0.6 percent (Madsen and Stern, 2007). Butler, et. al. examined mercury deposition and concentration measurements for the period 1998 through 2005 in a 2008 article and found statistically significant decreases in the amount of mercury measured in rainfall in the Northeast and Midwest regions of the U.S. (14 percent and 28 percent, respectively), and no significant increase or decrease in the Southeast region (Butler, et. al., 2008: 1591). In this article, they also discuss the relationship between emissions and mercury concentration measurements.

Although the emissions record available to them at the time show decreases in mercury emissions in each region, only the Northeast and Midwest regions had decreases in mercury concentration measurements. They hypothesize, but do not test, that mercury rainfall concentrations in the Southeast are more influenced by global sources of mercury (as opposed to local or regional sources) due to convective and other processes in the upper atmosphere (Butler, et. al., 2008: 1590).

Many authors have modeled and estimated the sources of atmospheric mercury deposition in the U.S. In a 2004 article, Seigneur, et. al. created a global chemical transport model that simulates mercury transport and deposition in North America. They found that 25 to 32 percent of the total atmospheric deposition in North America is due to North American anthropogenic sources, with the remainder coming from Asian anthropogenic sources and natural emissions (Seigneur, et. al., 2004: 566). Another 2005 study based on atmospheric modeling agreed with these results. In this study, Travnikov (2005) found that intercontinental and natural sources contribute from 30 to 70 percent of the mercury deposited on continents in the northern hemisphere. In a more recent study, Gbor, et. al. modeled natural and anthropogenic emissions in North American and calculated a ratio of natural to anthropogenic emissions. They estimated the ratio to vary from 0.7 in January, to 3.2 in July, with a total annual natural mercury deposition to North America (the United States, Canada, and Mexico) of about 250 tons (Gbor, et. al., 2007). Expanding on the study of natural versus man-made emissions, Slemr, et. al. reviewed direct mercury measurements since the 1970s, and found that mercury concentrations in the atmosphere increased during the 1980s, decreased through the 1990s to a minimum in about 1996, and have remained constant since. They studied

measurement sites in the northern and the southern hemisphere, and posited that since mercury modeling and inventory studies point toward increases in man-made mercury, this data suggests that natural mercury emissions are underestimated (Slemr, et. al., 2003). In other words, since there is a world-wide trend of increases in man-made mercury emissions, while mercury measured in the environment is staying the same, the natural sources of mercury emissions must have a more significant impact on the amount of mercury measured in the environment. They concluded,

> The observed worldwide concentration decrease in 1990 – 1996 period is much larger than predicted by the current inventories of anthropogenic and natural (including re-emission) emissions. The discrepancy suggests that either the temporal changes of anthropogenic emissions is substantially larger than estimated or that the approximate ratio of man-made to natural emission of about 1:1 is substantially underestimated (page 23-4).

These types of studies are important to this research. The models constructed by these authors develop useful inventories of mercury emissions and help the reader understand the sources and magnitudes of various natural and man-made sources of atmospheric mercury. These studies also highlight several sources of uncertainty that are present in atmospheric mercury research, including: 1) the relative contribution of natural versus man-made sources of mercury to the mercury measured in the environment, 2) how much the mercury released by power plants impacts the mercury measured locally, 3) whether mercury concentrations in the environment are increasing or decreasing in the world and in the U.S., and 4) how much the releases form Asia (which are believed to be increasing) impact the mercury measured in the U.S.

Speciation and Atmospheric Transport

It is very important to consider the speciation of mercury emissions from coal combustion processes. As the literature reviewed for this section will illustrate, the type of mercury that is emitted from a power plant stack will very much affect whether that

mercury will be more likely to deposit locally, or join the global mercury pool and have a more limited affect on local levels of mercury pollution. Schroeder and Munthe discussed the three main species of mercury (elemental mercury, mercury compounds in the gaseous phase, and particulate mercury) in their 1988 article. Elemental mercury (Hg^0) is insoluble in water and may travel in the atmosphere for thousands of miles before returning to the earth in wet or dry deposition processes. Gaseous phase mercury compounds are soluble in water and are believed to travel anywhere from a few miles to a few hundred miles before returning to the earth in wet deposition processes (rain or other forms of precipitation). Particulate mercury is mercury attached to soot particles that theoretically falls relatively close to the source of emission.

Another important point highlighted by Schroeder and Munthe is the current debate over the range of estimates of the amount of mercury that is emitted from natural sources (1998: 818). According to Rassmussen (1994), recent estimates of natural mercury emissions may be significantly underestimated. The estimate of worldwide mercury emissions is very important. Since the total mercury emitted from coal-fired power plants is about 45 tons per year (Shea, 2004), a significant increase in the estimate of worldwide natural emissions of mercury reduces the overall impact of coal combustion emissions. In some recent research published in the January, 2007 issue of the Journal *Global Biogeochemical Cycles*, Biswas, et. al. found that forest fires release between 19 and 64 millions grams (21 – 70 tons/year) of mercury each year, or between 13 and 42 percent of the estimated annual anthropogenic U.S. emissions.

Gustin (2003) studied geologic emissions in Nevada and found that non-anthropogenic sources of mercury may be underestimated by at least 3 times, and that the

annual natural emissions from the state of Nevada equal 20 percent of the total coal-fired utility boiler mercury emissions in the U.S. In another study, Engle and Gustin (2002) examined naturally occurring mercury emissions in Nevada and California, and found that these emissions may be significantly underestimated. They concluded that "there still is considerable uncertainty in our understanding of the biogeochemical cycle of Hg. Because of this uncertainty, the effectiveness of regulatory controls on anthropogenic point sources of Hg is not known" (2002). Researchers at the Oak Ridge National Laboratory (Betts, 2001) found that elemental mercury in snow and ice in the artic may be released as reactive gaseous mercury. This is the first time that the gaseous form (as opposed to elemental mercury) of mercury was found to be emitted by a non-anthropogenic source (although this is probably at least partly a re-emission).

Another important area of research is the contribution to the global mercury inventory by Asia, especially due to recent increases in energy use by China. Jiang, et. al. (2006) explored the inadequacy of information regarding mercury emissions in China. They estimated the amount of anthropogenic mercury released into the atmosphere in China at about 220 tons, based on estimates of fuel use and mercury content in coal. According to an article in the Journal, *Environmental Science and Technology*, scientists on the island of Okinawa measured mercury in the atmosphere in 2004 and found that actual anthropogenic elemental mercury releases from Asia may be double previous estimates (Renner, 2004). The importance of re-emitted mercury (previous anthropogenic emissions of mercury that are re-emitted in an ongoing global cycle) is the subject of a study by Bergan, et. al. (1999). The study employed a climate model to simulate the global distribution of elemental and reactive gaseous mercury, to test the agreement of

current estimates of natural and manmade mercury emissions with actual air, precipitation, and sediment measurements. They found that man-made emissions are at least as large as 30 percent of the natural emissions, and that the re-emission of previously deposited man-made emissions (from soils and surface waters) tends to increase the world wide deposition rate (from between 50 percent and three times the pre-industrial deposition rate). In areas near industrial centers (China, Europe, North America) the increase in deposition may be as much as 10 times that of pre-industrial times (Bergan, et. al., 1999: 1583).

In August of 2006, the 8th International Conference on Mercury as a Global Pollutant was held in Madison, Wisconsin. This conference brought together 1,150 scientific and technical experts who took a critical look at the available and latest research on mercury and made a number of declarations. Regarding the relative contributions of local, regional, and global sources of mercury to the mercury measured at any given location, one expert panel declared that there remains a large amount of uncertainty, and ascribing such relative contributions is possible, depending on how much uncertainty one is willing to accept. In addition to this conclusion, they also declare that: 1) uncertainty in our understanding of mercury chemistry in the atmosphere significantly impacts our ability to predict source/receptor relationships, 2) while North American and European emissions are decreasing, emissions in Asia and Africa are increasing, and 3) the uncertainty associated with atmospheric modeling needs to be better communicated to policy makers (Lindberg, et. al., 2007). The experts on this panel were Steve Lindberg (Chair), Russell Bullock, Ralf Ebinghous, Daniel Engstrom, Xinbin Fen, William Fitzgerald, Nicola Pirrone, Eric Prestbo, and Christian Seigneur. They were part of the participants in the

conference who represent a "diverse, multinational body of scientific and technical expertise on environmental and mercury pollution" (Madison Declaration, 2007: 62). The panel members are experts in the areas of atmospheric sources of mercury, methylmercury exposure and its effects on humans and wildlife, socioeconomic consequences of mercury pollution, and recovery of mercury-contaminated fisheries.

To summarize, there is a large amount of uncertainty associated with the relative contribution of anthropogenic mercury to overall mercury emissions. In fact, there is evidence that current estimates of natural mercury emissions, especially in the U.S., may be too low. In addition, the type of mercury released (elemental, gaseous, or particulate) or speciation, definitely affects whether the mercury acts as a local or as a global pollutant. Finally, most of the researchers agree that while mercury pollution in North America is either decreasing or staying the same, mercury emissions in the rest of the world are on the increase, perhaps by large amounts, and are also impacted by the re-emission of previously deposited mercury.

Mercury from Power Plants

CAMR and the associated debates regarding mercury have sparked much research into the speciation of mercury emitted from power plant stacks and the capabilities of currently available mercury control technologies. In a comprehensive 77 page article, Pavlish, et. al. (2003) reviewed recent developments in coal-fired power plant mercury control technologies and concluded that "there is no single best technology that can be broadly applied. Combinations of available control methods may be able to provide up to 90 percent control for some plants but not others" (page 94). This variation in control capability is due to the amount and type of mercury in the coal burned, the amount of

chlorine content in the coal, and the type of other existing pollution control devices at the plant. The authors also summarized many of the known issues associated with power plant mercury emissions including mercury levels in coal, the current lack of available stack measurement technology, and the costs and efficiency of currently available control technologies. Their research indicated that "The cost of removing mercury from stack gas using currently available technology is estimated to be very high, ranging from $5,000 to $70,000 per pound of mercury removed and adding up to 5 mil/KWH [one-half cent] to power cost" (page 153). In a more recent article, Srivastava, et. al. (2006) provided an overview of the most recent developments in coal-fired power plant mercury control technologies. They noted that there is a large amount of research and testing ongoing, and that much will be learned about the capability of mercury control technology over the next few years. Their article also summarized what is known about the current state of the capabilities of mercury capture technology as a function of the specific stream of other pollution controls at a given power plant and the type of coal that is burned. Ranges of mercury capture are between 6 percent and 90 percent with existing technology, depending on plant specifics. Srivastava et. al. noted that the EPA predicts that the market created by CAMR will result in low cost mercury emission controls for most boiler types achieving reductions rates from 90 to 95 percent by 2015, but that these reduction rates are not achievable with current technology (that is, there is no current technology now in place achieving such high reduction levels on an ongoing basis). The predicted control estimates of 90 to 95 percent are based on short-term tests only. The authors noted that "full implementation of such a program would take several years to achieve emission reductions, because large numbers of utilities would need time to order,

design, fabricate, and test such units". They also noted that a typical installation of a mercury control technology could take up to 3 years from the signing of the contract with the vendor, and that legal and permitting challenges could lengthen this time frame (Srivastava, et. al., 2006: 1393). In a recent article, Wedig, et. al. reviewed the latest information available regarding ongoing installations of mercury controls at power plants and concluded that the best technology for a given plant is still a function of many factors, including: 1) coal type and concentrations of mercury, chlorine, bromine, fluorine, and sulfur, 2) the level of speciation in the stack, and 3) the amount of mercury oxidation that may be occurring in the other pollution controls in place at the facility. They also summarized information available regarding mercury control efficiencies, ranging from 5 percent to 90 percent, depending on the specific conditions (Wedig, et. al., 2008).

Another important area of research is associated with the speciation of mercury as emitted from power plant stacks. This is important because, as mentioned earlier, gaseous and particulate forms of mercury tend to be deposited closer to the source (local pollutant), while elemental mercury may be transported hundreds or thousands of miles before returning to the earth (global pollutant). Reactive gaseous mercury may be rapidly reduced to elemental mercury in a power plant plume as the gases mix with ambient air. If this is true, then a much greater portion of mercury emissions are in the elemental form, the more global variety of pollutant, than was estimated by the EPA in the models used to develop CAMR. The conclusions of recent research conducted by Edgerton, et. al. (2006) and Lohman, et. al. (2006), are that the amount of elemental mercury in power plant plumes may be underestimated. In each study, the authors measured the ratio of

reactive gaseous mercury to total mercury downwind of several power plant stacks in Georgia (Lohman) and Georgia and Alabama (Edgerton), and each found a much lower percentage of reactive gaseous mercury than predicted by EPA modeling. They propose several explanations, including the reduction of reactive gaseous mercury to elemental mercury in the plant stack, and downwind conversion to elemental mercury through some unknown atmospheric reaction (Lohman, et. al., 2006). In their 2003 inventory of mercury emissions for the Eastern U.S., Walcek et. al. (2003) noted that although the common speciation ratio of 50:30:20 (elemental: reactive gaseous: particulate) is used for most EPA mercury modeling efforts, these speciation factors are highly uncertain and recent measurements show large levels of variation. For example, Walcek et. al. stated that the fraction of the particulate form of mercury is "strongly influenced by the type of coal burned, the level and type of particulate emission control technology implemented at the emission source, and many other factors. Also, recent measurements of individual large utilities show that particulate mercury accounts for less than 1 percent of total mercury emissions" (Walcek, et. al., 2003). Most EPA models do assume that 50 percent of the mercury emissions from coal plants are elemental, with the remaining 50 percent in the gaseous or particulate form. Bullock, et. al. performed a sensitivity analysis on the EPA model and found the model to be highly sensitive to the distribution of mercury emissions speciation (Bullock, 1998: 10). The implications of Bullock's findings are important because if the fraction of elemental mercury is underestimated in a model, then the results of the model based on this lower level of elemental emissions would not be reliable from a local versus global pollutant standpoint. Yujin, et. al. (2008) studied the reactions that convert elemental mercury (Hg^0) to oxidized mercury (Hg^{2+}) in power

plants with Synthetic Catalytic Reduction (SCR) control (used to remove NO_x from the flue gas). They found that the mercury "oxidation across SCR units significantly varies depending on many factors, such as coal type, concentration of other species (HCl, NO_x, and SO_2) in the flue gas, SCR catalyst type, and other operating conditions," and that "the reactions crucial to the transformation of Hg^0 to Hg^{2+} in the SCR reactor are not well understood" (Yujin, et. al., 2008).

Another research team, Sigler and Lee, utilized a combustion tracer methodology to study mercury emissions in the Northeast U.S. They found a disparity in the amount of reactive gaseous mercury measured and what they expected due to emissions from power plants, under the assumption that at least half of the emissions from power plant plumes are reactive gaseous mercury. They propose several explanations, including: 1) the elemental mercury proportion in power plant plumes is higher than thought, 2) emissions from medical and waste incinerators have not been reduced as much as reported, 3) there may be some unreported mercury releases, such as vehicular traffic, or 4) some combination of these explanations results in a smaller contribution to airborne mercury by coal plants (Sigler and Lee, 2006).

The U.S. Department of Energy has an ongoing program in mercury emission control research and development. Their website, last updated on January 18, 2006, summarizes the current state of what is known about mercury control technology:

> Existing pollution control devices such as electrostatic precipitators (which remove solid particles) can be effective in removing elemental and in some cases, oxidized mercury. Typically, removals range between zero and 30 percent, but can be as high as 60 percent for elemental mercury. Wet scrubbers are effective in removing oxidized mercury ranging from 75 to 99 percent, with overall total mercury removals of 55 percent. Dry flue gas desulfurization scrubbers can remove both oxidized and elemental with total mercury removals as high as 90 percent when coupled with a baghouse. Baghouses also remove both forms of mercury, but their effectiveness depends on the type of filter and other power plant specifics (mainly fly ash properties and temperature). In short,

> pollution control systems can be effective in removing as high as 90 percent of the incoming coal's mercury levels in a few cases, but very little mercury is removed in others (U.S. DOE, 2006).

There are several important conclusions that may be drawn from the review of research in this section. First, there is no credible research that concludes there is currently available mercury control technology that can control mercury in all power plant stacks at a high level of efficiency (on the order of 90 percent). In fact, there is no commercially available technology for reliable measurement of all speciations of mercury in power plant stacks on a continuous basis. There are a number of factors that affect the amount of mercury emitted by any given facility, including the type of coal that is burned, the design of the existing pollution control equipment, and the specific operating characteristics of the facility (operating temperatures, number of startups, etc.). The amount of mercury emitted by any utility is a function of a complex number of decisions made by power plant operators, and not simply a matter of whether or not to install a specific type of control technology. These complex decisions include decisions about what kind of fuel to purchase (for example mercury content, sulfur content, ash content, heat content, grindability, costs, and the interaction of these items), decisions about fuel handling, such as washing the coal, (which can be effective at lowing the amount of mercury and sulfur in the coal), decisions about how to control other pollutants (particulate matter, SO_2, and NO_x), decisions about whether to continue operating an older facility versus building an newer cleaner burning unit, and finally, decisions associated with variations in operating conditions in the stream of controls that can affect the amount and speciation of mercury emitted. For a given plant, some of these decisions have already been made in the past, such as the control scheme for particulate matter, SO_2, and NO_x. Some decisions are made on an annual basis, such as what type of coal to

purchase in major fuel procurement contracts. And some are made on a daily or hourly basis, such and operating conditions in the stream of controls. It is difficult to imagine how a control authority can obtain the information necessary to make a MACT technology decision for any given coal-fired utility operator, much less define a MACT standard that will control a high percentage of mercury at all plants. This may become possible in years to come, but it doesn't seem possible today.

In the next section, the issue of emissions trading is explored, beginning with an examination of the economic theories associated with emissions trading.

Emissions Trading

Emissions trading has been the subject of study by economists. An excellent summary of emissions trading theory and practice is available in a 1985 monograph by T. H. Tietenberg, with an updated second edition published in 2006. Tietenberg builds on prior research that he calls classic econometric articles. In 1971, Baumol and Oates made a case for emissions trading for truly global pollutants, where only the amount, and not the location, of the pollution source matters. Then, in 1972, Montgomery made a more general emissions trading argument that includes the case where the location of the emission makes a difference (Tietenberg, 2006: 4-5). Tietenberg examined 14 empirical studies of emissions trading programs that evaluate cost savings and air quality impacts. The studies tended to be either of two types: ex ante studies (computer simulations), and ex post studies (after the fact examinations). He concluded that: 1) for a majority of the ex ante studies, the command-and-control approach was more expensive than the emissions trading approach, 2) free permit distribution rather than permit auctions tend to contribute to the success of emissions trading programs, and 3) at least two emissions

trading cases (acid rain and lead in the U.S.) resulted in substantial air quality improvements. Tietenberg suggested that emissions trading can result in better emissions control technology, but did not declare that finding to be universal. He noted that the specific circumstances where technology is improved more under emissions trading or under traditional approaches is unclear. He also noted that when operators are mandated to install a best available control technology for a certain pollutant (the traditional approach), there will be incentives that result in lowering the cost of installing and operating the mandated units, but that there will be no incentive to improve the pollutant control efficiency or for research and development of alternative technologies (Tietenberg, 2006: 69).

Regarding the initial distribution of emissions trading permits (Tietenberg's second conclusion above), Tietenberg noted that the initial allocation of permits is the most controversial aspect of the design of a trading system. He described four methods of allocation including: 1) random lotteries; 2) first come-first served; 3) administrative rules based on eligibility criteria; and 4) auctions. The first two methods have been used in some wildlife resource management programs (hunting and fishing), but have not been tried in any major air pollution program. The third method is the one most used, usually because it is the only way to institute a program that is politically feasible. Thus, Tietenberg concluded that free distribution contributes to the success of trading programs because it is necessary to gain the support of the stakeholders, and avoid the rent seeking that would accompany permit auctions (Tietenberg, 2006: 195). As noted earlier, CAMR allows each state to determine how to allocate the initial mercury allowances (free or auction), allows each state to govern the administrative rules for trading (even allowing

states to prohibit trading), and allows flexibility regarding set-asides. Most states set-aside around 5 percent of their allowance budget for auctions or as reserve for new generating units (NACAA, 2007).

In some cases Tietenberg noted that air quality is improved under trading programs, but that "the degree to which credit for these reductions can be attributed to emissions trading, as opposed to exogenous factors or complementary policies, is limited" (2006: 72-73). For the U.S. acid rain programs, Tietenberg reviewed ex post data and found that SO_2 emissions dropped by 40 percent and NO_x emissions dropped by 57 percent by 2002. For the lead program, trading not only reduced, but eliminated lead emissions and "was instrumental in achieving that reduction much more quickly than otherwise would have been possible" (Tietenberg, 2006: 63). Tietenberg also considered the spatial dimension of emissions trading programs, and the concern over potential hotspots, particularly in the case of SO_2 trading in the U.S. He found that, although there was concern that hotspots due to SO_2 trading would result in the East and Northeast, "Allowing emissions trading actually resulted in pollutant concentration decreases, rather than increases in the East and Northeast" (page 88). He advocates a zonal approach to controlling hotspots, where a market mechanism controls trades across zonal boundaries.

As Tietenberg and others have pointed out, the best example of an operating emission trading market exists in the U.S. with the regulation of SO_2 and NO_x made possible by the Clean Air Act Amendments of 1990. Amar (2000) studied how technology improved, becoming cheaper and more available after regulatory drivers were put in place for SO_2 and NO_x. He concluded that "a combination of aggressive performance requirements and flexible attainment mechanisms has proven highly successful in the past. An important

benefit of cap-and-trade approaches is that they provide incentives for ongoing technology improvement by creating a market for over compliance at individual facilities" (page xvi).

Burtraw and Mansur (1999) studied the environmental effects of SO_2 trading and concluded that trading has resulted in health related benefits and costs savings in the East and Northeast of the U.S. According to an EPA fact sheet on the results of the Acid Rain program, the SO_2 and NO_x trading program has resulted in the largest quantified human health benefit of any federal regulation in the 10 years prior to 2006, with annual benefits exceeding costs by a factor greater than 40 to 1 (U.S. EPA, 2006f). The EPA has studied the SO_2 and NO_x programs and attribute this success to a number of factors, including: 1) compliance assurance through incentives and automatic penalties, 2) strong quality assurance, 3) a collaborative approach, 4) electronic reporting, 5) flexibility for low-emitting sources, 6) complete emissions data record required, 7) centralized administration, 8) a level playing field, 9) publically available data, 10) a performance based approach, and 11) a reduction in conflicts of interest. The agency noted that all of these provisions have been designed into the CAMR program (Schakenback, et. al. U.S. EPA, 2006).

In a 2000 Monograph, Ellerman, et. al. studied the market success of Phase 1 (1995-1999) of the Acid Rain Program. Phase 1 applied to the 263 dirtiest electric generating units. Phase II applies to the remaining fossil fueled units. They found the program achieved Phase 1 SO_2 reductions with little litigation and at lower costs than expected. In their concluding chapter, they noted "We are unaware of any other U.S. environmental program that has achieved this much, and we find it impossible to believe that any

feasible alternative command-and-control program could have done nearly as well" (Ellerman, et. al., 2000: 314). They outlined the following general lessons from their in-depth study: 1) large scale trading programs can work as the economic theorists, such as Tietenberg, describe in the literature, 2) cost savings over traditional command-and-control regulatory approaches are significant (they suggest 50 percent), 3) trading does result in technology innovation, 4) the political process and rent seeking that accompanies the allowance process did not affect program performance, 5) an efficient trading market developed within two years of the final rules setting up SO_2 trading in 1993, and 6) banking allowances for future use are an important mechanism for controlling over-investment that is common to command-and-control approaches (Ellerman, et. al., 2000: 314-322). They noted one caution, however. An important aspect of the SO_2 and NO_x allowance and trading program is the reliance on accurate emissions reporting and penalty provisions. Attempting to apply such a program at a global level, to control pollutants such as greenhouse gasses, would be very difficult to implement. They noted that there is a difference between the idea of emissions trading, and the implementation of the practical policy details that matter, and these details become more complicated on a global basis. For example, attempting to regulate carbon dioxide with emissions trading would be difficult because there are many more sources (than for example, the electric generating units of the acid rain trading program), and many emit very small amounts that are difficult or too expensive to measure (Ellerman, et. al, 2000: 321-322).

A more general study of the use of economic incentive programs versus traditional command-and-control approaches is the subject of a 2004 monograph by Harrington, et.

al. They examined six case studies to test twelve hypotheses concerning environmental policies. The case studies included:

1. The SO_2 emissions permit market in the U.S. versus SO_2 standards in Germany;
2. Industrial water effluent fees in the Netherlands versus permits and guidelines in the U.S.;
3. The NO_x emissions permit market in the U.S. versus emission taxes in Sweden and France;
4. The Chlorofluorocarbon (CFC) permit market in the U.S. versus mandatory phase outs in other countries;
5. The leaded fuel permits market in the U.S. versus mandatory phase outs and taxes in European countries; and
6. Source regulation of chlorinated solvents in the U.S. versus three other policy approaches in Europe (pages 10-17).

Although Harrington, et. al. acknowledge that their approach was limited by several factors, including: 1) by a small number of observations that are not randomly selected (the common problem of all case studies), 2) by the fact that in addition to policy differences, the case studies are also different in terms of political institutions, history, and pre-existing environmental conditions, and 3) by the fact that the differing approaches cannot be sufficiently analytically separated. However, they tentatively drew the following conclusions:

1. Incentive approaches (trading) are more cost effective than regulatory approaches (command and control);
2. Both incentive approaches and regulatory approaches have large information requirements to be successful;
3. Incentive instruments provide a continuing incentive over time to reduce emissions, provide polluters with more flexibility, and promote new pollution abatement technology;
4. The evidence on the relative effectiveness of incentive and regulatory approaches is mixed;
5. Regulated firms are more likely to oppose incentive approaches than regulatory approaches because they may perceive that in the long run they will have to pay more under an incentive regulatory policy;
6. The evidence on the relative administrative costs of the two approaches is mixed;
7. When spatial or temporal are important, theory says that the performance of incentive based approaches are compromised more that regulatory approaches, but in practice, the situation is less clear, and hybrid approaches mitigate local impacts;
8. There is no strong and consistent evidence that Incentive approaches have more demanding monitoring requirements than regulatory approaches;
9. Taxes on emissions may result in adverse tax interactions;

10. The impact on whether firms choose to act altruistically (voluntarily reduce pollution more than required) does not seem to be affected by the choice of policy instrument;
11. Incentive and regulatory approaches are equally adaptable to new information; and
12. Cost is easier to observe with incentive approaches, where cost is defined by the amount of the emissions fee or market price of the permit (Harrington, 2004: 266-267).

The experience with CAMR in the U.S. contradicts finding number 5 above, with industry groups such as the Edison Electric Institute and the Electric Power Research Institute strongly in support of the cap-and-trade provisions of CAMR (EEI, 2007) (EPRI, n.d.). This might be explained by the comfort and success with the existing cap-and-trade programs for SO_2 and NO_x that the utility industry has achieved over the last 15 years.

Palmer, et. al. performed a 2007 cost benefit analysis of CAMR. They created a model that combines aspects of the electricity generation sector, atmospheric transport, and public health endpoints resulting from the implementation of CAMR. They found that the economic benefits of CAMR are far greater than the cost (Palmer, et. al., 2007).

A number of common themes emerge from this review of the available literature concerning emissions trading. In general, market-based regulatory schemes, such as emissions trading, are proving to be a lower-cost alternative to command-and-control schemes, and in some cases have resulted in substantial air quality improvements. The evidence is less clear regarding the regulatory scheme that results in the least amount of pollution (without regard to cost). In addition to the cost advantage, there is also some evidence that cap-and-trade schemes can contribute to better technology, both in the development of control technology and in better emissions monitoring equipment. Finally, as Tietenberg pointed out, pollutants that result in hotspots can be effectively

managed with tradable permit schemes, utilizing design mechanisms such as zonal based permit management.

In the next section, the available evidence concerning mercury hotspots in the environment is examined. The pertinent questions are: 1) what is a hotspot, 2) where are they found, and 3) what is the evidence that power-plant mercury emissions contribute to the hotspots?

Hotspots

Hotspots may be associated with mercury measured in fish, other wild animals, plants, or soil and water measurements. Evers, et. al. (2007) utilized measured mercury levels in fish and wildlife (7,300 observations) to identify five hotspots and nine areas of concern in the Northeastern U.S. They used the U.S. EPA advisory level of 0.3 micrograms mercury per gram muscle tissue as a threshold for fish (page 30). They also utilized an atmospheric emissions model to examine what sources contribute to mercury deposition in the Northeast and found that emissions from coal-fired power plants in the region account for much of the mercury deposition, and that decreasing these emissions would decrease local deposition. In their model, they assume that the coal fired utilities emit 70 percent of the fraction of emissions as reactive gaseous and particulate mercury (compared to 50 percent in most EPA models). As mentioned earlier, assuming that such a high percentage of the emitted mercury is reactive gaseous instead of elemental may overstate the local polluting effects. They also sited the need for additional monitoring sites to better understand the link between emission sources and mercury levels in the environment. They found that "...large gaps in data and understanding continue to hamper our ability to quantitatively analyze sources, and fully characterize the spatial and

temporal patterns of deposition and biological availability across Canada and the United. States" (Evers, et. al., 2007: 41). Tolme examined the nine biological hotspots (high mercury measurements in wildlife) in the Northeast and Eastern Canada identified by Evers, et. al. and noted that seven of the nine are located far from power plants, "suggesting that airborne mercury is drifting in from elsewhere" (Tolme, 2005: 33).

In the 2007 study mentioned earlier (page 14) by Driscoll, et. al., the authors noted that all of the speciated forms of mercury have the ability to deposit locally or regionally (page 26). However, they also stated that "We expect CAMR will produce important results, but these changes may not be sufficient to protect human and environmental heath....significant additional reductions in Hg emissions will probably be necessary to bring about widespread recovery to Hg levels ...in the Northeastern United States" (Driscoll, et. al, 2007: 26).

Researchers in a 2003 study by the Brookhaven National Laboratory reviewed the emissions from two coal-fired plants and found that "only a few percent (4 to 7 percent) of the mercury emitted from the power plants deposits within 30 km of the plant. The majority of mercury enters the global cycle" (Sullivan, et. al., 2003: 29). The plants included in the Brookhaven study (Mansfield Plant in Shippingport, PA and Monticello in Monticello, TX) were chosen due to their high emission rates for reactive gaseous mercury. Actual emissions from the plants were used to model deposition around the plants. This study also performed a risk assessment to human health associated with fish consumption for people living in the local proximity of these two plants. The results of the risk assessment were that the risks to the general population are small (less that 1 in 10,000). They determined that the risks of mercury exposure are much more a function of

fish consumption rates rather than additional mercury deposition from local power plants (Sullivan, et. al., 2003: 45).

In their defense of CAMR, the Electric Power Research Institute (EPRI), conducted computer modeling based on EPA methodology to determine the likelihood of the development of hotspots after the implementation of CAMR. EPRI found that electric power plants will neither create new hotspots nor contribute to worsening existing hotspots under CAMR (EPRI, n.d.). EPRI also studied the economic aspects of CAMR, and found that of the 253 power plants that emit more that 100 pounds of mercury per year, none would increase their emissions under CAMR and only 6 would stay at their current emission level (EPRI, n.d.).

In a 2004 study, Mark Cohen of the National Oceanic and Atmospheric Administration Air Resources Laboratory developed a mathematical model to estimate what mercury sources in North America and Canada are the greatest contributors to mercury contamination in the Great Lakes. He found that coal combustion sources are the largest contributors to mercury in the great lakes, including sources that are as far away as 2000 km. Cohen admitted that speciation information about the forms of mercury being emitted from coal stacks is scarce (page 249). He used the same assumption about coal combustion speciation as the EPA and other researchers (50% elemental, 45% reactive gaseous (ionic), and 5% particulate) (page 251). Cohen found that both near sources of mercury and distant sources of mercury contribute to the contamination of the Great Lakes (page 260). He concluded, "Long range and regional transport was found to be very significant – at least 50% of the model-estimated deposition was contributed from sources 100-1500 km from each lake", and that coal combustion is the most

significant source category (Cohen, 2004: page 262). In an earlier report submitted to the Commission for Environmental Cooperation, Cohen and Miller studied the possible impacts of mercury deposition in the future under a range of mercury control assumptions. Although their analysis was done too early to consider the impact of CAMR, they again pointed out in this report that "knowledge of the speciation of mercury emissions is very important for predicting the deposition to local, regional, and other receptors. Unfortunately, speciation is not well known for many source categories" (Cohen and Miller, 2003: 10). In 2007 Ryaboshapko, et. al. (including Cohen) conducted a detailed evaluation of the ability of Cohen's model (and other models like it) to predict mercury concentrations by comparing the predictions of the models with mercury measurements taken at monitoring stations in Europe and Ireland. They found that the models are particularly unsuccessful in predicting levels of reactive gaseous mercury. They noted, "These results confirm that our knowledge of reactive gaseous mercury's atmospheric behavior is not sufficient, and there is a high level of uncertainty associated with the reactive gaseous mercury concentrations at any particular time and place" (Ryaboshapko, et. al., 2007: 238). They went on to describe the sources of uncertainty associated with atmospheric mercury modeling, including: 1) the models do not deal with natural emissions or re-emissions of mercury, 2) in many cases the reliability of the anthropogenic emissions data is low, 3) data on the speciation forms of emitted mercury is lacking, 4) knowledge of the atmospheric chemistry of mercury is lacking, 5) large scale models tend to have too coarse of a resolution (50 km) which tends to affect performance, and 6) the models do not deal with temporal variations of mercury emissions (page 238-239).

In their more general study of environmental policies from case studies mentioned earlier, Harrington, et. al. examined the hypothesis that incentive-based policy measures such as cap-and-trade would exacerbate pollution hotspots. They found that hotspots can be addressed under either scenario if they develop, and that "incentive-based measures may work to the detriment or to the benefit of any particular area" (Harrington, et. al., 2004: 258). They also found that in cases where there is a limit to the amount of pollution from a given source, the potential for hotspots is reduced (258). This is certainly true in the case of electric utilities, which are already operating near capacity (EIA, 2007).

Will reductions in the atmospheric mercury loading from emission sources result in observable reductions in mercury measured in fish? Munthe, et. al. published a study in 2007 that synthesized the available knowledge associated with methylmercury in fish measurements after reductions in mercury loadings. They found numerous examples where the cessation of mercury from point sources in aquatic systems resulted in clear reductions in the subsequent fish measurements. However, the evidence for similar fish mercury reductions due to reductions in atmospheric mercury emissions is less clear. Their conclusion stated that "For responses to changes in atmospheric loading, evidence is limited to a few cases that currently limit the possibilities to draw firm conclusions" (Munthe, et. al., 2007: 41).

In a more recent study, Manolopoulos et. al. studied mercury measurements at two remote locations in South-Central Wisconsin and found that an upwind power plant does contribute to the level of reactive gaseous mercury, but not to the levels of elemental mercury. They suggested that either the concentration of reactive gaseous mercury in the flue gas of the plant is greater then the EPA estimate of 50 percent, or that there are

atmospheric processes going on that convert the elemental mercury to reactive gaseous mercury (Manolopoulos, et. al., 2007: 500).

The results of this section of the literature review are mixed. Some researchers have found that mercury emissions from power plants do contribute to mercury measured in the environment (Evers, 2007; Driscoll, 2007; Cohen, 2004; Manolopoulos, 2007). This is especially true for those researchers who are utilizing atmospheric models to predict local effects of emissions. Meanwhile, other researches found that there is less evidence of local impact from electric power plants (Sullivan, 2003; EPRI, n.d.). Most everyone agreed, however, that there are many uncertainties associated with the research that cloud the results. The uncertainties include: 1) the correct speciation ratio that should be used to model mercury emissions from power plants, 2) the lack of stack monitoring data and the need for additional monitoring sites, 3) the impact of power plant emissions to the levels of mercury in fish, 4) the ability of atmospheric models to deal with the complexities of mercury speciation, 5) the chemistry of atmospheric mercury and the impact to local and global deposition patterns, and 6) the inability of the atmospheric modeling techniques to deal with temporal variations of mercury emissions. This literature review suggests that mercury may act as both a local, and a global pollutant.

In the next section, a look at another source of mercury emissions to the atmosphere is examined.

Mercury in Vehicle Exhaust

In addition to mercury from industrial sources, mercury has also been identified as a component of vehicle exhaust. A 1996 study in the Journal *Science of the Total Environment* reported that the mercury content of gasoline and diesel fuel is on the order

of 0.22 to 2.97 parts per billion (Liang, et. al, 1996). During the 13th International Emission Inventory Conference, "Working for Clean Air in Clearwater", held in June 2004 in Clearwater, Florida, Hoyer, et. al. presented a paper that reported the results of mercury testing on vehicle exhaust. The research was a joint effort between the U.S. EPA and the University of Michigan Air Quality Laboratory. The authors tested the elemental and particulate phase mercury content of the vehicle exhaust of several light duty vehicles and one heavy duty diesel vehicle. Measuring the amount of mercury in the exhaust, they found mercury levels ranging from 0.3 to 1.4 ng/mile traveled for the light duty gas vehicles, and 6.3 to 11.0 ng/mile for the diesel vehicle. The study was limited in that they could not measure oxidized gaseous mercury in the exhaust, they only tested a few vehicles and fuels, and they did not test the exhaust under cold start conditions, which may make a significant difference (Hoyer, et. al., 2004). These numbers may sound small but when the total miles traveled in a region is considered, the total mercury released to the atmosphere may be significant.

Mercury Toxicity

In the 8th International Conference on Mercury as a Global Pollutant mentioned earlier, another expert panel addressed the topic of mercury toxicity. The members of this panel were Donna Mergler (chair), Anton Scheuhammer (Co-Chair), Henry Anderson, Laurie Chan, Kathryn Mahaffey, Michael Meyer, Michael Murray, Mineshi Sakamoto, Mark Sandheinrich, and Alan Stein. The panelists were experts in human health and toxicology and began meeting one year before the conference to review the existing literature on methylmercury exposure.

The declarations of this panel included: 1) the primary exposure risk for humans to mercury compounds is through the consumption of certain species of fish, 2) although uncertainties remain regarding the developmental neurotoxicity risk of methylmercury to the fetus, there is enough evidence available to advise people (especially expectant mothers and children) to select species of fish lower in mercury content, 3) there is some evidence suggesting that methylmercury exposure could increase the risk of cardiovascular disease, especially in adult men, and 4) since fish contains significant amounts of beneficial omega-3 fatty acids, the selection of specific fish species in the diet can maximize the benefits while minimizing exposure to mercury (Mergler, et. al., 2007: 8). These researchers also declared that mercury in fish and wildlife, especially species at the top of the food chain, are at some risk of toxic effects of methylmercury exposure. The effects on wildlife are most likely associated with reproduction, with an increased level of reproductive success being associated with lower levels of methylmercury exposure (Madison Declaration, 2007: 63-64).

These declarations are well supported by research. In a 2007 study of mercury content in the blood of New York City adults, National Institute of Health researchers found that the blood mercury content of adults who reported eating fish or shellfish 20 or more times in the last 30 days was 3.7 times higher than those reporting no consumption (McKelvey, et. al., 2007). Another recent study examined the mercury levels of 1,024 pregnant women in five Michigan communities, finding mercury levels in the hair ranging from 0.01 to 2.50 micrograms per gram, with a mean of 0.29 micrograms per gram. Total fish consumption and consumption of canned fish, bought fish, and sport-caught fish were all positively correlated with total mercury in the hair (Xue, et. al.,

2007). These studies have resulted in an effort to communicate the health effects and toxicity associated with methylmercury exposure from eating fish. A 2007 brochure published by the New York City Department of Health and Mental Hygiene encourages people to choose fish that are lower in methylmercury content. The brochure lists 31 species as low or very low in mercury including clams, salmon, shrimp, catfish, cod, and light tuna. Another 22 are listed as high or very high, including mackerel, shark, swordfish, and albacore tuna (NYC, 2007).

There have been some who argue that the EPA reference dose for mercury (the amount that an average individual could have in their blood or hair for a lifetime and suffer no ill effects) is set too low and the dietary alternatives to fish for many populations in the world are of poorer nutritional value (Schoen, 2004). However, no one is arguing that mercury emissions should not be controlled at all. The question of course, as it usually is in public policy debates, is one of cost versus benefit.

Mercury in Forest Fire Smoke

As stated in the introductory chapter, mercury from forest fire smoke may contribute up to 70 tons per year to the air in the U.S. This mercury is a combination of mercury that exists naturally in the environment, and mercury that has been deposited on plants and trees over time from both natural and anthropogenic sources. This makes this fraction of mercury in the air very difficult to ascribe to a source. The content of pollutants in forest fire smoke, which in addition to mercury include significant amounts of carbon dioxide, carbon monoxide, oxides of nitrogen, particulate matter, ammonia, sulfur dioxide, and methane, are also very much a function of time of year and fuel loading (Wiedinmyer. C. et. al., 2006). Turetsky, et. al., found that Canadian peat soils contain up to 15 times

more mercury than soils from other forested areas (Turetsky, et. al., 2006). In order to directly measure the mercury content of forest fire smoke, Freidli, et. al. collected vegetation samples from 7 locations around the U.S., then measured the mercury content of the material before and after burning, and the content of the smoke. They found that essentially all of the mercury in vegetation is released during burning and that 95 percent is released as elemental mercury (with the remaining 5 percent as particulate mercury) (Freidli, et. al., 2001). Mercury emissions from forest fires in Europe, North Africa, and Russia have also been the subject of research. Cinnirella and Pirrone estimated emissions utilizing ground based and satellite data, finding the emissions levels to be in the range of 1-9 tons per year for the Mediterranean region, 2-8 tons per year for Europe, and 7-55 tons per year for the Russian Federation (Cinnirella and Pirrone, 2006).

Evidence of the impact of mercury from forest fires has also been linked to increased methylmercury measurements in fish. In a 2006 article, Kelly, et. al. found that methylmercury amounts in rainbow trout from a lake in fire catchment areas were five times higher than those in lakes whose catchment did not experience a fire (Kelly, et. al. 2006). The above research definitely points to the importance of mercury content in wildfire smoke for any air emissions model.

Summary

This review of available literature indicates that there are many unknowns associated with mercury pollution. The amount and form of naturally occurring mercury is not well understood, the speciation of mercury in power plant stacks is unclear, and the results of research tying mercury pollution to local sources are mixed. Although the evidence is mixed regarding whether power plant stacks contribute to biological mercury hotspots,

the creation of such hotspots is the primary reason that several states and environmental groups are opposed to CAMR. No one has opposed CAMR with the argument that cap-and-trade is more costly.

Also, there is some uncertainty over whether mercury pollution, especially in North America, is increasing, decreasing, or staying the same. One fact, however, is clear. The regulation of mercury from power plant stacks under CAMR would be the first time mercury from the electric utility industry has been regulated anywhere in the world. A successful CAMR program could be the first step in the development of a world-wide mercury trading program that would benefit every global citizen.

As Tietenberg and others have demonstrated, market based approaches can work, and are becoming increasingly more acceptable to regulators and environmentalists alike. This research intends to address the main argument against CAMR, that it will result in the creation of new or the exacerbation of existing hotspots of mercury in the U.S. In the next chapter, a methodology is described that will take into account many of the aspects of mercury air emissions identified in this literature review.

CHAPTER III

METHODOLOGY

Introduction

In order to complete this study, data sources were needed for mercury measured in the environment and for mercury emitted from anthropogenic sources. Once the data was obtained, a model was constructed that accounted for levels of emissions near the mercury measurement sites, as well as spatial considerations.

The main argument against cap-and-trade for mercury is that there are "hotspots" of mercury deposition in the United States, and that the cap-and-trade regulatory scheme will exacerbate those hotspots since some polluting plants (supposedly those causing the hotspots in the first place) will buy emissions credits in order to operate more, resulting in higher mercury emissions in these hot spot areas. As the previous chapter illustrates, the available literature paints a different picture. If naturally occurring mercury is underestimated, mercury from Asia is underestimated, and the percentage of gaseous and particulate mercury in power plant emissions is overestimated, then the amount of local mercury deposition will not be significantly affected by the locations of coal-fired power plant stacks. The hypothesis in this study is that mercury measured in rainfall does not correlate with coal-fired power plant emissions. There may be a correlation with some other industrial sources, or no correlation at all, indicating that naturally emitted mercury,

or mercury drifting in from Asia, is the main contributor to mercury measurements in rainfall.

This chapter begins by describing the sources of the data used to model the mercury emitted by industrial sources, mercury released into the air during forest fires, and the population living within a given radius of a mercury measurement site, used as a proxy variable for vehicle mercury emissions. Next, the three regression models used in the study are discussed, followed by an explanation of how the data was manipulated to construct over 60 different models employed to determine whether mercury emitted from coal-fired power plants correlates with mercury measurements in rainfall. The chapter concludes with a discussion of the limitations of the study.

Data Sources

The Mercury Deposition Network (MDN) is part of the National Atmospheric Deposition Program (NADP) network that began measuring airborne mercury in 1995 at 13 measurement sites. The NADP is a cooperative research program sponsored by various state, federal, and non-governmental agencies. The objective of the MDN is "to develop a national database of weekly concentrations of total mercury in precipitation and the seasonal and annual flux of total mercury in wet deposition" (MDN, n.d.). Through the years, additional sites are added and some are removed. Sites are not necessarily selected based on their utility as mercury measurement sites, as a number of air pollutants are measured at each site. The data for this study was taken for the years 2001 through 2005, inclusive, made available in colorful charts on the MDN internet site. Table 1, on the next page, provides the number of measurement sites that were in operation during each of those years.

Table 1

Number of MDN Sites in Operation during Study Period

Year	Number of MDN Sites in Operation
2001	47
2002	46
2003	58
2004	67
2005	75
Total	293

Source: Mercury Deposition Network, 2001-2005.

Utilities were required to report releases of toxic chemicals through the TRI program beginning in 2000, but the data used for forest fire areas was not available until November of that year, so 2001 is the first year of interest for this study. Since the TRI data for calendar year 2006 will not be available until the spring of 2008, 2005 caps the other end of the data used in this analysis. Also, since the data availability for the independent variables is limited to the Continental U.S., the few MDN sites in Canada, Mexico, and Alaska were not considered in the study. For the five years, a total of 293 observations on the dependent variable are included in each model. Figure 2, on the next page, shows the current active and inactive MDN measurement sites.

MDN data has been used by researchers for modeling and investigations. In one study, methylmercury measurements in mosquitoes were shown to be positively related to atmospheric mercury deposition, utilizing MDN data in an OLS model (Hammerschmidt and Fitzgerald, 2005: 3037). However, no journal articles or studies

could be found that have used the MDN data in a spatial regression model, nor in a model where mercury emissions from industrial sources was used to predict mercury wet concentration or deposition measurements. It is unclear why this is the case, but as the literature review for this paper illustrates, most mercury concentration or deposition prediction studies are performed utilizing atmospheric computer models.

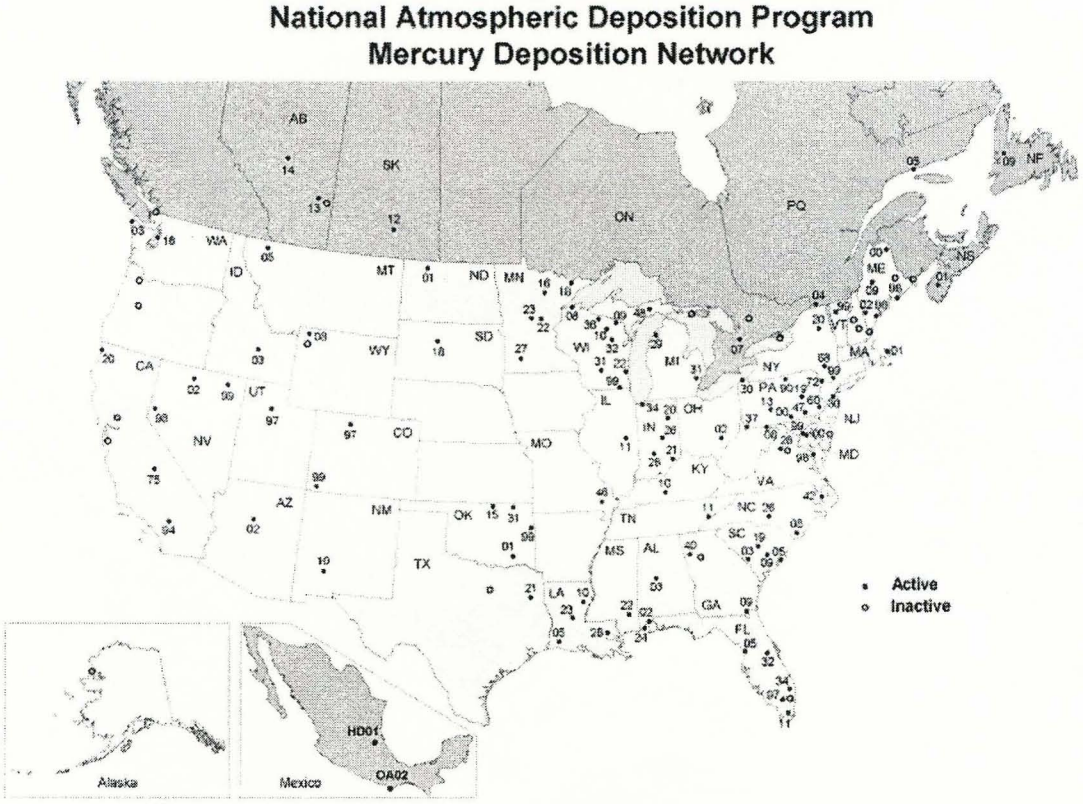

Figure 2. Current Active and Inactive MDN measurement sites. Source: Mercury Deposition Network, n.d.

Each week, precipitation samples are collected at each MDN site and sent to the Mercury Analytical Laboratory at Frontier Geosciences in Seattle, WA. The samples are tested for total mercury using cold vapor atomic fluorescence (MDN, n.d.). Mercury concentration is measured in nanograms per liter (ng/L), which is equivalent to parts per trillion. In addition to the mercury concentration, the precipitation total for each week is

measured in millimeters via a rain gauge. The product of the mercury concentration and the precipitation is recorded for the week and is defined as the mercury deposition. Mercury deposition is measured in nanograms per square meter (ng/m^2). Deposition, then, is the total amount of mercury, in billionths of a gram, falling on a square meter of ground at the measurement site. For each year, the MDN provides maps which illustrate the average weekly mercury concentration, and the total mercury deposition for the year, at each active measurement site. Figures 3 and 4, below and on the next page, show the mercury concentration and total wet deposition for the year 2005, respectively.

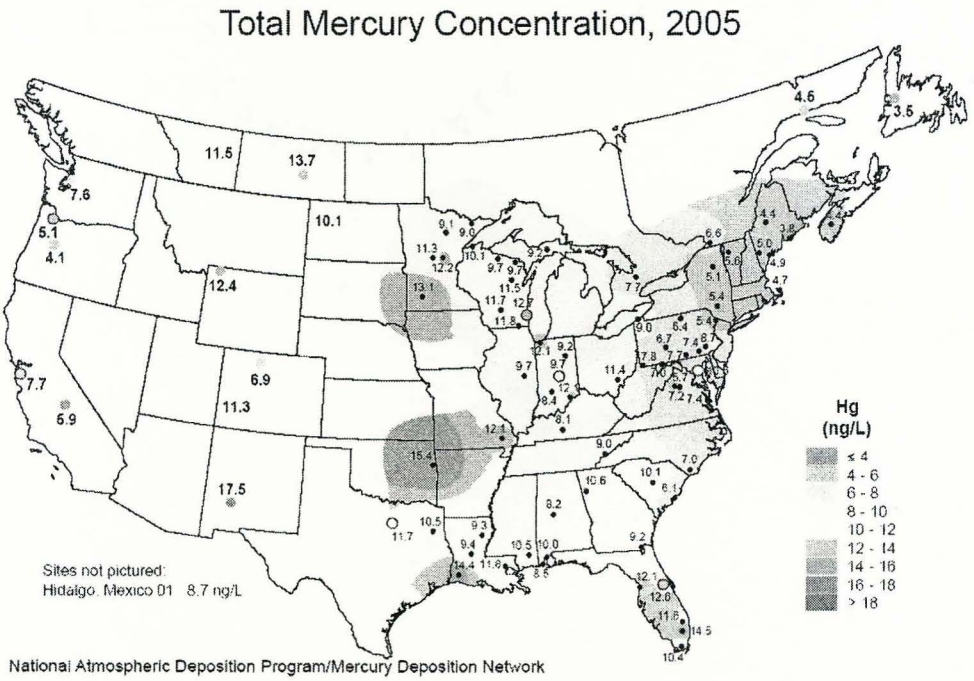

Figure 3. Mercury Concentration, 2005. Source: Mercury Deposition Network, n.d.

For the study, separate models were constructed using both mercury concentration and mercury deposition as the dependent variable. The values for concentration and

deposition were taken directly from the maps published on the MDN web site. The longitude and latitude of all sites is available for download in a text file, which was imported into Microsoft Access. The individual site weekly measurements were also downloaded, and used to calculate total annual precipitation for each measurement site, which was also included as an independent variable.

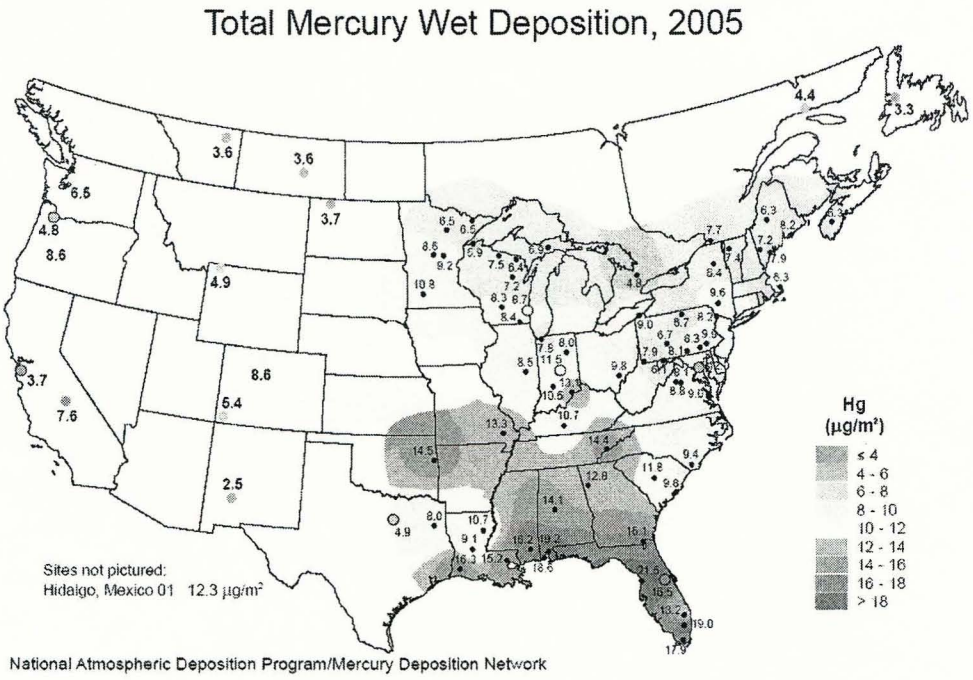

Figure 4. Mercury Deposition, 2005. Source: Mercury Deposition Network, n.d.

Through an analysis of the correlations of the dependent variables versus precipitation, it was determined that precipitation is positively correlated with deposition, and negatively correlated with concentration. This stands to reason since deposition is the product of precipitation and concentration. It is theorized that concentration decreases as precipitation increases because the more rainfall there is; the more the available mercury

in the atmosphere is washed out and diluted. Likewise, mercury deposition is positively correlated with total deposition because the more rainfall you have; the more total mercury will accumulate on any given square meter as long as it can be detected at all in the sample. The correlation between mercury concentration and precipitation is - 0.16, and the correlation between mercury deposition and precipitation is + 0.70, for all data points available from the MDN download, a total of over 25,000 measurements.

Thus, the MDN provided the data for both dependent variables Mercury Concentration (CONC) and mercury deposition (DEP), as well as for the precipitation independent variable. The remainder of this section discusses each of the independent variables, along with their sources of information.

The data source for mercury emissions from industry will be the U.S. EPA's Toxic Chemical Release Inventory (TRI). According to EPA regulations, an industrial facility is required to report emissions of any of about 700 TRI chemicals or chemical categories if the facility falls into certain listed Standard Industrial Classification (SIC) codes, has 10 or more full time employees, and manufactures, processes, or imports more than listed threshold amounts of the material. Reports must be submitted by July 1 for the previous calendar year (for example, reports for 2006 must be submitted by July 1, 2007) on a special EPA form either electronically or in paper format (U.S. EPA, 2001: 2-9). The EPA makes the information submitted in the TRI program available to the public through two websites, *TRI Explorer* at http://www.epa.gov/triexplorer, and *Envirofacts* at http://www.epa.gov /enviro. When the TRI reporting regulations first took effect in 1988 for reporting year 1987, electric utilities were not required to report. Then, in 1998 the EPA increased the number of facilities required to report under TRI, including metal

mining, electric utilities, and hazardous waste treatment facilities. In 2000, the usage threshold that triggers the need to report mercury emissions was reduced from 10,000 lbs to 10 lbs. (U.S. EPA, 2006b: 27).

The EPA conducts a number of data quality and enforcement related activities that address the validity of the TRI data submitted by facilities. These activities include data quality checks, the creation of a facility data profile that gives each reporting facility the chance to check the accuracy of the data submitted, and the performance of technical audits of selected submissions (U.S. EPA, 2007a). In addition, the publicly available nature of the data and the use of the data by numerous environmental groups help make the TRI one of the more transparent and useful government programs.

The EPA also conducts enforcement activities aimed at ensuring compliance with TRI. The statutory authority for the regulations associated with TRI comes from the Emergency Planning and Community Right-to-know Act of 1986 (EPCRA). Section 325(c) of EPCRA authorizes fines of up to $27,500 per day per violation, which can include: failure to report, data quality errors, failure to respond to a notice of non-compliance, repeat violations, failure to supply notification, and failure to maintain records (U.S. EPA, 2007b). Enforcement activities associated with the TRI requirements are available via the Enforcement and Compliance History Online (ECHO) system. This is a searchable online system that allows searches of EPA cases by regulatory citation. A search for all EPA cases associated with TRI between fiscal year 2001 and 2007 year-to-date returned 2,254 cases (out of a total of 36,000 cases total in the system). Of these, 16 were associated with electric power generation facilities. The largest penalty assessed by the EPA in the 16 cases was in a case involving the AES Hawaii Generation Plant in

Kapolei, HI. This facility was assessed a penalty of $45,430 for failure to report TRI data on May 21, 2001 (U.S. EPA, 2007c).

Social researchers have utilized TRI data to investigate various questions and test hypotheses. For example, Decker, et. al. (2005), found that TRI emissions are a significant determinant of residential housing values. The TRI data can be downloaded from the EPA's website into text files that can be imported into Microsoft Excel or Access, or another data analysis program. The files include a large amount of information for each pollutant source including the type of emission (air, water, or land), the latitude and longitude of the source, location information (state, city, county), the amount of emissions in pounds, as well as names of contacts associated with the source and information about how the emissions were estimated.

For this study, the entire TRI reporting files for reporting years 2001 through 2005 were downloaded from the EPA web pages and subsequently imported into Microsoft Access. Table 2, on the next page, lists the mercury air emission totals from the TRI data from all sources, from only electric utilities (SIC code 49), and also emissions to the land from the mining industry. The table illustrates that the electric power generation industry, SIC code 49, is the source of between 45 and 48 tons of emissions to the air, out of total releases to the air of between 70 and 75 tons, for the study period 2001-2005. However, it is clear that a very large amount of mercury is deposited to the land from industries with SIC codes starting with 10 through 19. These SIC codes represent the mining industry. Although these emissions are not directly to the air, emissions from the mining industry were included as an independent variable due the large values associated with these

emissions. Mining emissions to land are from 38 to 74 times the electric utility emissions to the air, on an annual basis.

Table 2

Mercury Releases from TRI data, 2001-2005

	2005	2004	2003	2002	2001
Hg Air Releases, all sources, lbs	140,285	139,344	141,572	142,518	149,520
Hg Air Releases, SIC Code 49, lbs	96,091	93,889	89,913	89,706	90,640
Hg Land Releases, Mining, lbs	3,630,838	3,960,379	6,633,961	4,806,800	4,295,491

Source: U.S. EPA TRI Data, 2001-2005.

Although there were mercury air emissions from 28 different two-digit SIC coded industries, most of these emissions were very small amounts, and no other industry had aggregate air emissions large enough to be considered as a separate independent variable. Therefore, all other industrial emissions to the air except for SIC codes 49 were lumped together and considered as a separate independent variable. One minor exception was the case of SIC Code 4953, Electric Services – Refuse Systems. Since this SIC code represents facilities that incinerate garbage and do not burn coal, these facilities were not included in the independent variable for SIC 49, and were instead included in the variable for all other industries.

Thus, three independent variables were created from the TRI Data: 1) mercury emissions to the air from the electric power industry (SIC49AIR), 2) mercury emissions to the land from the mining industry (SIC10LND), and 3) mercury emissions to the air from all other industries (SICOTAIR). Figures 5, 6, and 7, on the next two pages show the emission sites for these three variables for emission year 2005 on a map of the U.S.

These maps were created from GeoDa, utilizing the longitude and latitude made available in the EPA TRI data.

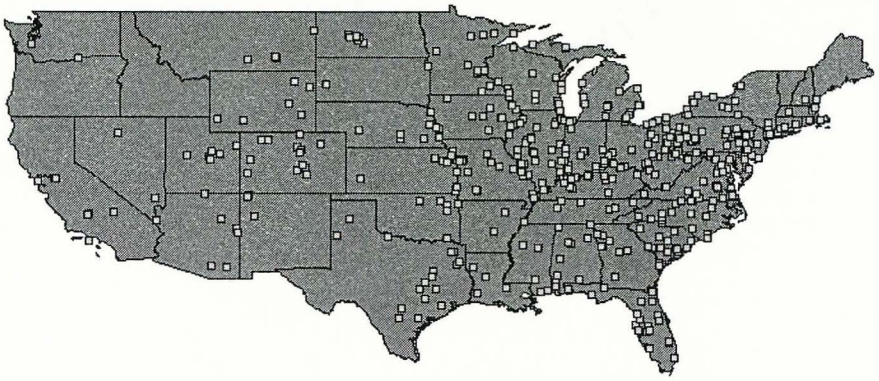

Figure 5. Mercury Emission Sites (to air) from Electric Power Plants, 2005. Source: Tirey, 2008.

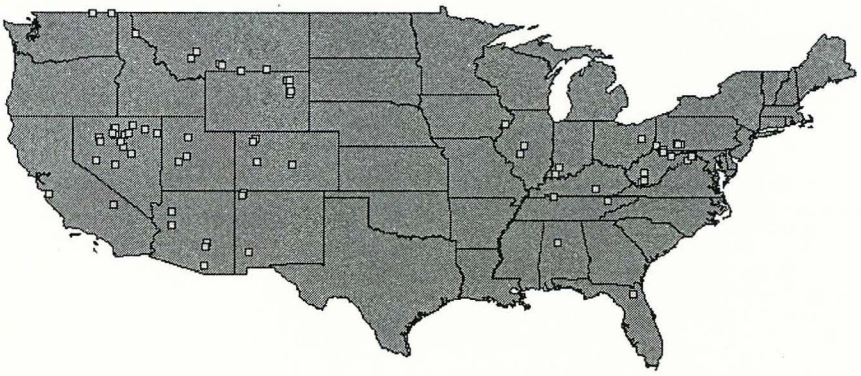

Figure 6. Mercury Emission Sites (to land) from the Mining Industry, 2005. Source: Tirey, 2008.

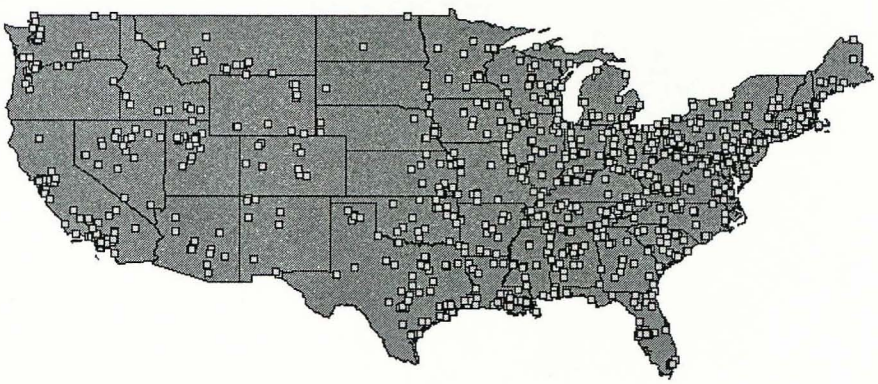

Figure 7. Mercury Emission Sites (to air) from all other Industry, 2005. Source: Tirey, 2008.

Mercury emissions from forest fires are difficult to tie down to a specific longitude and latitude, but fortunately, technology has helped deal with this problem. The U.S. Department of Agriculture operates a satellite based fire mapping program called MODIS (Moderate Resolution Imaging Spectroradiometer). MODIS is an imaging instrument mounted on two NASA satellites, Aqua (launched in 2002), and Terra (launched in 1999). These satellites pass over the U.S. twice a day and detect fire events. Data from the program is downloadable and contains information on each fire detection including the date, longitude, latitude, length and width of the fire, and other information. The data is available beginning in November 2000 (USDA, 2007). The data from the MODIS system has been validated by researchers utilizing ground based techniques, and the system is being used for estimating greenhouse gas emissions from natural wildfires (Kaufman, et. al., 2003), (Roy, et. al., 2007). For this study, all fire detections from the MODIS system were downloaded for each year of the analysis, 2001 through 2005. Figure 8, below illustrates the locations of each of the 73,464 fires detected by MODIS during calendar year 2005. Figure 8 was created using GeoDa.

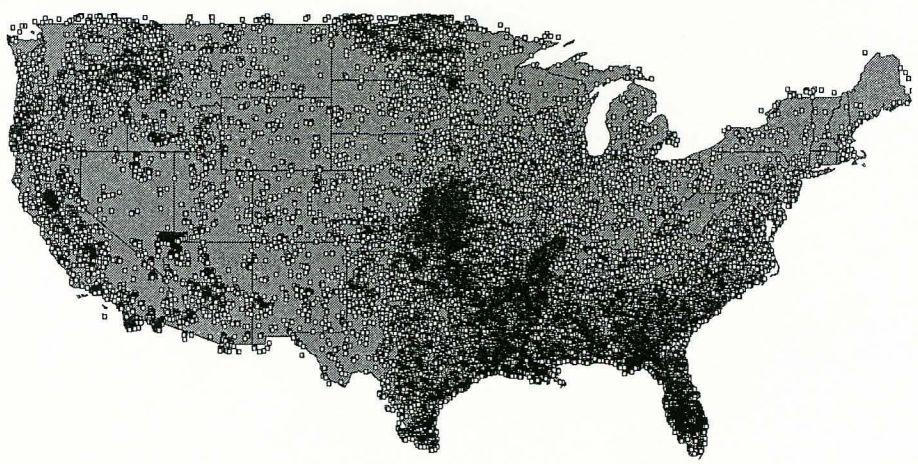

Figure 8. MODIS Fire Detections, 2005. Source: Tirey, 2008.

After importing the MODIS data into Access, the area of each fire was calculated, the distance from each fire site to each MDN site was calculated, and the total fire area within a given radius of an MDN measurement site was summarized and entered into the models as an independent variable (FIREAREA).

The final independent variable included in each model is the population living within 75 miles of the MDN mercury measurement sites. A table containing the population of 3,219 counties was downloaded from the U.S. Census Bureau (2007). This table also included the longitude and latitude of the geometric centroid of each county. The population within 75 miles was chosen as the radius to model around each MDN site under the assumption that this would be a good indicator of the amount of vehicle emissions (vehicles operating anywhere within an hour of an MDN site). This radius was held constant for population (under the assumption that vehicle emissions are area sources as opposed to point sources), even as the radius for industrial emissions and wildfire events was varied between 25 and 500 miles. The variable name for population within 75 miles was termed POP75 in the models.

In summary, two dependent variables and five independent variables are included in the OLS and spatial regression models that are described in the next section. Since the spatial regression models are impacted by the distance between the dependent variables, the longitude and latitude for each mercury measurement site becomes a key input to the models, and is used to calculate these distances. Table 3, on the next page, contains a summary of the variables in the initial 60 models, the code used in Access and GeoDa for

each variable, and the source of the data. The unit of measurement for each variable is also listed in the variable description in Table 3.

Table 3

Summary of Dependent and Independent Variables, Initial Models

Variable Description	Type	Variable Name	Data Source
Mercury concentration at each MDN site, ng/L	Dependent	CONC (mercury concentration)	MDN
Mercury deposition at each MDN site, ng/m^2	Dependent	DEP (mercury deposition)	MDN
Total precipitation (or 1 over the precipitation) at each MDN site, mm	Independent	PRECIP or INVPRECIP	MDN
Mercury releases to the air from coal fired power plants within model radius, lbs	Independent	SIC49AIR	EPA TRI
Mercury releases to land from mining Operations within model radius, lbs	Independent	SIC10LND	EPA TRI
Mercury releases to the air from all other industrial sources within model radius, lbs	Independent	SICOTAIR	EPA TRI
Population living within 75 miles of each MDN measurement site, millions	Independent	POP75	U.S. Census
Total wildfire burned area with model radius, km^2	Independent	FIREAREA	USDA

Source: Tirey, 2008.

Regression Models

Two types of statistical models were constructed to explore the correlation of mercury concentration and deposition with coal-fired power plant emissions. The first model is an OLS regression model with the MDN measurement sites as the unit of analysis, and the second is a spatial regression model, utilizing a distance based weight matrix. In both cases, two dependent variables are modeled, mercury concentration and mercury deposition. Computing resources for construction and analysis of the OLS and spatial models was a geographical data analysis tool called GeoDa.

The OLS model does not take into account the variation that might be present in the data due to the spatial nature of the mercury measurement sites. Since the geographical locations of the measurement sites can be defined, a spatial regression model can be constructed to explore the spatial relationships. If these relationships are not considered and there is a spatial component to the relationships, it has been shown that the results of the OLS regressions will be inefficient (the standard errors, t-statistics, and measures of fit will be biased) and the results unreliable (Anselin, 2007).

Since the tools available in typical regression analysis programs, such as SPSS (Statistical Package for the Social Sciences), do not include the methodology for exploring spatial relationships, some other analysis tool must be used. Dr. Luc Anselin, Director of the Spatial Analysis Laboratory at the University of Illinois Department of Geography, made available for free download a software tool just for this purpose. The tool is called GeoDa 0.95i, and it provides a number of useful spatial data analysis methods, including cartograms, maps, conditional plots, and spatial regression (University of Illinois, 2007). The GeoDa web page provides a number of user manuals, example papers, and sample data downloads that enable a researcher to learn how to utilize this important software tool to explore spatial relationships in data.

Anselin outlines two basic spatial regression model types that can be explored utilizing the GeoDa software tool, the spatial lag model and the spatial error model (Anselin, 1988). In the spatial lag model (Anselin also calls this the mixed regressive, spatial autoregressive model) the spatial autocorrelation pertains to the dependent variable. A spatially lagged dependent variable term, ρWy, is included on the right hand side of the common OLS regression equation:

$$y = \rho Wy + X\beta + e,$$

where y is the vector of observations on the dependent variable, ρ is the spatial autoregressive parameter, Wy is a spatially lagged dependent variable for weight matrix W, X is the matrix of observations on the independent variables, β is the vector of independent variable coefficients, and e is a vector of random error terms. The spatial weights matrix contains non-zero values in the row-column combinations where the dependent variables are expected to interact. (Anselin, 2007).

In the spatial error model the autocorrelation is limited to the error term and the model is expressed as:

$$y = X\beta + u, \text{ with } u = \lambda Wu + e,$$

where the error term u is the sum of the error vector e and a spatially lagged error term λWu where λ is the autoregressive coefficient (Anselin, 2007). Although the spatial lag and error models are similar, Anselin noted that, in the spatial lag model, in addition to being affected by the values of the independent variables, each dependent variable is also affected by the spatially weighted values of the dependent variables in neighboring nodes. While in the spatial error model, spatial autocorrelation comes from omitted variables that follow a spatial pattern (Anselin, 2001).

In order to construct a regression model that relates the TRI releases (and each of the other independent variables) to the MDN measurement sites, the geographical locations of the independent variables had to be tied in some way to the mercury measurement sites. One possible model included using the U.S. states as the unit of analysis for the regression models (and averaging the mercury measurements from all MDN sites in each state), however, not all states have MDN sites and this model would not allow an

examination of the local affects of mercury emission sources that are close to the measurement sites. Instead, since the longitude and latitude of each industrial site emitting mercury is known from the TRI data, the distance between each emission source and each measurement site can be calculated, and all the mercury emissions for all sources within a given radius of a measurement site can be added together. The latter is the model constructed for this study. Models were built that included all TRI mercury emissions within a given radius of the mercury measurement sites, with the radius set equal to 25, 50, 75, 100, and 500 miles.

After importing the TRI data into Microsoft Access, a query was written that selected the records of all facilities reporting emissions of mercury or mercury compounds. Table 4, below, lists how many facilities reported mercury emissions for each of the five study years.

Table 4

Number of Facilities Reporting Mercury Emissions, 2001-2005

	2001	2002	2003	2004	2005
Number of facilities reporting mercury emissions	1,633	1,641	1,706	1,740	1,745

Source: U.S. EPA TRI Data, 2001-2005.

Then, for each year, the emission tables were joined with the table containing the measurement site information from the MDN. The resulting table for 2005 contained 130,875 records (1,745 TRI emission sources times 75 MDN measurement sites), with the longitude and latitude of both the site and source contained in the record. In the same query that produced this table, distance between each site and source was calculated using the great circle formulas from trigonometry. This distance formula is a function of

the inverse cosine. Although Microsoft Access does not offer a built in function for the inverse cosine, it does offer the built-in function for the inverse tangent. Therefore, the distance formula was created in a series of three steps: 1) convert the longitude and latitude of both the TRI source and the MDN site to radians, 2) calculate the argument for the inverse cosine function for the great circle distance formula, and 3) calculate the inverse cosine as a function of the inverse tangent. Table 5, below, shows the actual formulas coded into the queries in Microsoft Access.

Table 5

Microsoft Access Formulas for the Great Circle Distance

sitelatrad	Site_Data!latitude*3.14159/180
sitelatrad	Site_Data!longitude*3.14159/180
ARCCOS_ Arg	Cos([sitelatrad])*Cos([sitelongrad])*Cos([sourcelatrad])*Cos([sourcelongrad])+Cos([sitelatrad])*Sin([sitelongrad])*Cos([sourcelatrad])*Sin([sourcelongrad])+Sin([sitelatrad])*Sin([sourcelatrad])
Distance	3963.1*(Atn(-[ARCCOS_Arg]/(Sqr(-[ARCCOS_Arg]*[ARCCOS_Arg]+1)))+3.14159/2)

Source: Tirey, 2008.

This formula was tested by utilizing a web based longitude and latitude distance calculator each time a query was written that contained the great circle distance formula.

Another factor built into the models in this study was wind direction. Prevailing wind direction for most areas of the U.S. are known and available from various sources. For this study, a wind direction map was obtained from the National Oceanic and Atmospheric Administration (NOAA, 2005) that could be compared to the MDN mercury measurement site map (see Figure 9 on the next page).

Using this map, a prevailing wind direction was defined for each of the 124 current or past MDN measurement sites. For example, the prevailing wind direction for the KY10 MDN measurement site is primarily from the South. In the same query that the distance

of the measurement source to the MDN site was calculated, the direction was also determined through a series of simple formulas that compared the longitude and latitude of the MDN site to the TRI emission source. In the above example, only those emission sources located to the South of the KY10 MDN site were considered in the query that added up the total amount of mercury released within the modeled distances.

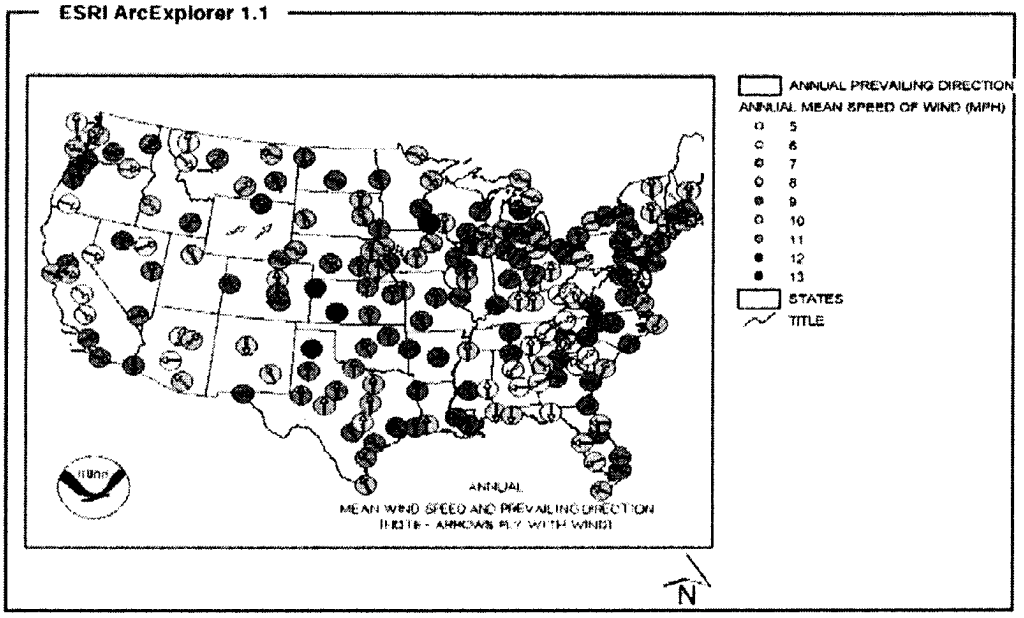

Figure 9. NOAA Prevailing Wind Direction Map. Source National Oceanic and Atmospheric Administration, 2005.

Once the site-source distance tables (including wind direction) were complete, queries were written that found all facilities in SIC code 49 (electric utilities), SIC code 10 through 19 (mining), or all other SIC codes within each of the modeled distances (25, 50, 75, 100, or 500 miles) for each MDN site for each year of the study. The emissions from all facilities within the radius were then summed and placed in a table containing the MDN site records. A similar process was followed for the wildfire area and for the population within 75 miles. A bivariate correlation table created using SPSS is listed below that shows the correlations between the independent variables (see Table 6 below).

This table illustrates the expected significant correlations between mercury concentration and precipitation (negative), and between mercury deposition and precipitation (positive). The correlations between the independent variables include a positive relationship between precipitation and SIC10LND and FIREAREA, and a positive relationship between SICOTAIR and population. As the multicollinarity diagnostics will later illustrate, none of these bivariate correlations presented a concern in the analysis of the results from GeoDa.

Table 6

Correlations for the Model with Radius equal to 50 Miles

Correlations

		CONC	DEP	SIC10LND	SIC49AIR	SICOTAIR	LNPRECIP	PRECIP	INVPRECP	POP50	POP75	FIREAREA
CONC	Pearson Correlation	1	.235**	.495**	.048	.009	-.457**	-.340**	.551**	.079	-.016	.145*
	Sig. (2-tailed)	.	.000	.001	.511	.883	.000	.000	.000	.176	.785	.014
	N	293	293	40	189	260	293	293	293	293	293	288
DEP	Pearson Correlation	.235**	1	-.311	.009	.020	.693**	.740**	-.529**	.123*	.083	.371**
	Sig. (2-tailed)	.000	.	.051	.898	.751	.000	.000	.000	.036	.156	.000
	N	293	293	40	189	260	293	293	293	293	293	288
SIC10LND	Pearson Correlation	.495**	-.311	1	-.167	.015	-.453**	-.352*	.542**	-.252	-.259	-.115
	Sig. (2-tailed)	.001	.051	.	.331	.925	.003	.026	.000	.116	.106	.478
	N	40	40	40	36	40	40	40	40	40	40	40
SIC49AIR	Pearson Correlation	.048	.009	-.167	1	.060	.021	-.011	-.055	.077	.038	-.133
	Sig. (2-tailed)	.511	.898	.331	.	.425	.770	.886	.456	.291	.602	.069
	N	189	189	36	189	180	189	189	189	189	189	188
SICOTAIR	Pearson Correlation	.009	.020	.015	.060	1	-.002	.004	.031	.340**	.241**	.070
	Sig. (2-tailed)	.883	.751	.925	.425	.	.979	.950	.623	.000	.000	.261
	N	260	260	40	180	260	260	260	260	260	260	259
LNPRECIP	Pearson Correlation	-.457**	.693**	-.453**	.021	-.002	1	.938**	-.910**	.006	.077	.226**
	Sig. (2-tailed)	.000	.000	.003	.770	.979	.	.000	.000	.921	.187	.000
	N	293	293	40	189	260	293	293	293	293	293	288
PRECIP	Pearson Correlation	-.340**	.740**	-.352*	-.011	.004	.938**	1	-.724**	-.014	.038	.303**
	Sig. (2-tailed)	.000	.000	.026	.886	.950	.000	.	.000	.815	.522	.000
	N	293	293	40	189	260	293	293	293	293	293	288
INVPRECP	Pearson Correlation	.551**	-.529**	.542**	-.055	.031	-.910**	-.724**	1	-.026	-.094	-.115
	Sig. (2-tailed)	.000	.000	.000	.456	.623	.000	.000	.	.659	.107	.052
	N	293	293	40	189	260	293	293	293	293	293	288
POP50	Pearson Correlation	.079	.123*	-.252	.077	.340**	.006	-.014	-.026	1	.806**	.050
	Sig. (2-tailed)	.176	.036	.116	.291	.000	.921	.815	.659	.	.000	.395
	N	293	293	40	189	260	293	293	293	293	293	288
POP75	Pearson Correlation	-.016	.083	-.259	.038	.241**	.077	.038	-.094	.806**	1	-.067
	Sig. (2-tailed)	.785	.156	.106	.602	.000	.187	.522	.107	.000	.	.260
	N	293	293	40	189	260	293	293	293	293	293	288
FIREAREA	Pearson Correlation	.145*	.371**	-.115	-.133	.070	.226**	.303**	-.115	.050	-.067	1
	Sig. (2-tailed)	.014	.000	.478	.069	.261	.000	.000	.052	.395	.260	.
	N	288	288	40	188	259	288	288	288	288	288	288

**. Correlation is significant at the 0.01 level (2-tailed).

*. Correlation is significant at the 0.05 level (2-tailed).

Source: Tirey, 2008.

Since the number of cases was 293 for each model, there were enough degrees of freedom available to include a number of dummy variables in the model to explore the impact of time and region. One set of dummy variables was included for year (D2001,

D2002, D2003, and D2004), and one set was included for EPA region (DREG1, ..., DREG9). One year (2005) and one region (DREG10) were omitted to avoid perfect collinarity. Table 7 on the next page provides the list of states that make up each EPA region and Figure 10, below, shows the regions on a map of the U.S. Figure 10 was obtained from the EPA Internet site (U.S. EPA, 2007d).

Table 7

States in Each EPA Region

EPA Region	States
DREG1	Maine, Vermont, New Hampshire, Massachusetts, Rhode Island, Connecticut
DREG2	New York, New Jersey
DREG3	Pennsylvania, West Virginia, Virginia, Delaware, Maryland, Washington DC
DREG4	Kentucky, Tennessee, Mississippi, Alabama, Georgia, South Carolina, North Carolina, Florida
DREG5	Minnesota, Wisconsin, Illinois, Michigan, Indiana, Ohio
DREG6	New Mexico, Texas, Oklahoma, Arkansas, Louisiana
DREG7	Nebraska, Iowa, Kansas, Missouri
DREG8	Montana, North Dakota, South Dakota, Wyoming, Colorado, Utah
DREG9	California, Arizona, Nevada, Hawaii
DREG10	Washington, Oregon, Idaho, Alaska (not included as a dummy variable)

Source: U.S. EPA, 2007d.

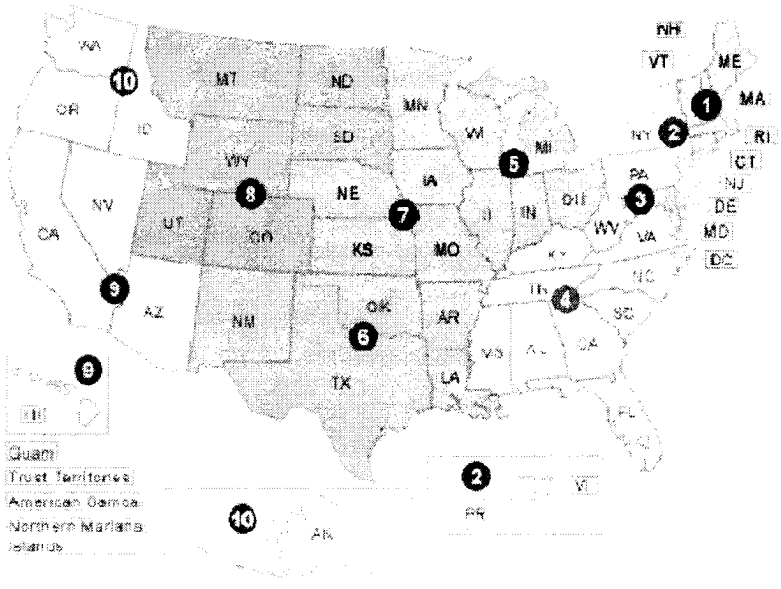

Figure 10. U.S. EPA Regions. Source: U.S. EPA, 2007d.

Limitations

As expected in many social science studies, there are a number of limitations associated with this analysis. The final section of this methodology chapter explores these limitations, including: 1) the MDN data does not, nor does any other available database, contain measurements associated with dry deposition, 2) even though there are up to 75 MDN measurement sites operating in any given year of the study, the measurement sites are still limited and this may bias the results, especially for the lower radius models (25 and 50 miles), 3) mercury from wildfire emissions may be dependent on the type of flora and land type and little data is available regarding mercury emissions that result from various land types, 4) there is error associated with TRI data which is difficult to quantify, and finally, 5) even if there is a correlation between air emissions and mercury measured in rainfall, this correlation may not extend to mercury levels in fish, which is the primary environmental vector of concern regarding impact on humans.

There is no dry deposition measuring network. In their estimation and mapping of mercury deposition across the Northeastern U.S., Miller et. al. noted that wet deposition represents somewhere between one-half to two-thirds of total deposition (Miller, et. al., 2005). Their analysis is based on the EPA estimate that 20 percent of utility emissions are in the form of particulate mercury (which may be overestimated as noted earlier), but much of the particulate mercury from all sources does fall to the earth through dry deposition, as opposed to wet deposition processes. This knowledge gap is understandable. Particulate mercury attached to solid particles that cannot be seen with the naked eye is difficult to capture, and analytical methods to measure mercury attached to these small particles if they can be captured have yet to be developed. An article in the

January 31, 2007 edition of Environmental Science and Technology describes the current state of the art associated with measurement of mercury dry deposition. The authors describe efforts at two MDN measurement sites in Nevada to measure dry mercury deposition. They found a large seasonal and locational variation and highlighted the need to develop a network of reliable dry deposition measurements to supplement the reliable wet deposition measurement network (Lyman, et. al., 2007).

In addition to the inclusion of dry deposition measurement data, this study would be strengthened by the addition of more wet deposition measurement sites. Models were constructed by taking into account the number of industrial emission sources, wildfire area, and population within a given radius of the MDN measurement sites (25, 50, 75, 100, and 500 miles). With 75 or fewer MDN sites spaced about the U.S., there is a fairly large distance between measurement sites, and at the smaller radii, there are larger numbers of empty cells in the model, particularly for the TRI related independent variables. In table 8, below, the number of non-zero cells for each of the independent variables impacted by distance from the MDN measurement sites is provided to illustrate this point.

Table 8

Number of Non-Zero Cells for each Independent Variable

n = 293	SIC49AIR	SIC10LND	SICOTAIR	FIREAREA	POP75
25 mile radius	93	7	126	233	293
50 mile radius	171	39	234	278	293
75 mile radius	243	58	278	291	293
100 mile radius	263	73	291	293	293
500 mile radius	281	250	293	293	293

Source: Tirey, 2008.

As the table shows, there are non-zero values for FIREAREA and POP75 for most of the cells in each model, but the number of non-zero cells falls off greatly for SIC10LND below 500 miles, for SICOTAIR below 50 miles, and for SIC49AIR below 75 miles. The larger number of cells containing values of zero increases the standard error for the 25 mile and 50 mile radius models. The good news is that between 2001 and 2005, the number of active MDN sites increased from 47 to 75, or about 60 percent. It would be beneficial to conduct more studies of this type in the future as more MDN sites become active.

The third limitation discussed here is that it is unlikely that the mercury contained in wildfire emissions contains a constant level of mercury depending on location, time of year, or proximity to other emission sources. This model contains no provision to account for any of this variation. In fact, the mercury component of wildfire smoke is a fairly recent subject of study. As additional information becomes available about this phenomenon, perhaps this model could be modified to account for that variation.

Although the TRI program is considered by many to be very successful, there is a source of error associated with this data that is difficult to quantify. However, because the total mercury emitted from coal-fired power plants has been studied by the EPA and is a function of the mercury contained in coal, there most likely is less error in the numbers reported by utilities than there is in the numbers reported by other industrial emitters. The data sources section of the chapter provided some details about the programs the EPA has in place to validate TRI reporting. It would, however, be safe to assume that mercury emissions are under reported, rather than over reported. Thus, if the error in the TRI data is associated with a tendency for the mercury emissions of non-utility sources to be under

reported, this would tend to mask the significance of the SICOTAIR coefficient, and not materially impact the significance of the SIC49AIR coefficient.

Finally, in order to make the logical leap from controlling mercury from power plants to reducing the human health threat from mercury exposure, there is a primary assumption that there is a direct and significant impact between the mercury measured in rainfall and the methylmercury that ends up in fish. Although this seems very logical, mercury that enters the water table from sources unrelated to air emissions may also be a significant contributor to mercury levels in fish. The millions of pounds of mercury emitted from mining operations that theoretically never enter the air are many times greater than the mercury emitted from all other anthropogenic sources. If mercury releases from power plants significantly impact mercury measured in rainfall, the environment provides another level of dilution. That is, power plant mercury is not a direct threat to humans until it is converted to methylmercury, shows up in the meat of fish, and is consumed by humans in high enough amounts to become a health threat.

Now that the methodology, data sources, model descriptions, and limitations have been discussed, the next chapter presents the results of the regression models.

CHAPTER IV

RESULTS

Initial Results

A total of 60 initial models were processed in GeoDa for this study. All independent variables were regressed against each of the two dependent variables (mercury concentration and mercury deposition) for each of the three types of models (OLS, spatial lag, and spatial error), and five increasingly higher radii of concern (25, 50, 75, 100, and 500 miles). This equals a total of 30 initial models (two dependent variables times five radii times three types of models) that were all repeated both considering and not considering the wind direction. Three tables in Appendix 1 contain a summary of the output from GeoDa for these 60 models. Appendix 2 contains a typical printout from GeoDa for the no wind condition for the OLS, spatial lag, and spatial error 25 mile radius models with concentration as the dependent variable. The results shown in Appendix 2, labeled case numbers 1A, 2A, and 3A, are included in the lower section of the first column of the table in Appendix 1.

As Appendix 1 illustrates, the F-statistic (on the null hypothesis that all regression coefficients are jointly zero) for the OLS models was significant in every case at the 99 percent probability or better, ranging from a low of 27.9 (concentration, 500 miles, wind case) to a high of 45.8 (deposition, 25 miles, no wind case). Adjusted R^2 ranged from a low of 0.637 to a high of 0.745 for these same cases. The OLS models explain about 65

percent of the variation in the concentration dependent variable and about 74 percent of the variation in the deposition independent variable.

More interestingly, the GeoDa printout for the OLS models provides three statistics to compare the fit of the OLS model to the spatial models. These are the log likelihood, the Akaike information criterion, and the Schwarz criterion. According to Anselin (2005: 175), a comparison of these statistics is an indication of whether or not the spatial model is a better fit than the OLS model. For the log likelihood, the more positive the measure, the better the fit. For the Akaike information criterion and the Schwarz criterion, the opposite is true (the lower the number the better the fit). By examining the log likelihood of cases 1A, 2A, and 3A in the GeoDa printouts in Appendix 2, it is clear that the statistic increases (gets more positive) from the OLS to the spatial models, moving from negative 588 to negative 559 for the spatial lag model and negative 561 for the spatial error model. Likewise, the Akaike information criterion and the Schwarz criterion both get smaller, moving from 1216 to 1159 or 1162 in the former case and from 1289 to 1236 or 1236 in the latter case. Appendix 1 provides the log likelihood statistic for all 20 OLS models. In every case this statistic gets more positive, indicating the spatial models are the better fit. Although not listed in Appendix 1, the same is true for the Akaike information criterion and the Schwarz criterion; the statistics indicate the spatial models are a better fit that the OLS model.

In the center of the GeoDa printout (Appendix 2) the variable name, the value of the coefficient, the standard error, the t-statistic, and the probability are provided. Appendix 1 lists the value of the t-statistic for each of the initial 60 models if the coefficient is significant at $p < 0.01$ (in bold text) or $p < 0.05$ (in normal text). The table below, Table

9, summarizes the significance of the independent variables in the initial models, where the probability the t-statistic is zero is less than 5 percent (bolded and normal test t- or z-statistics in Appendix 1). Since there is very little difference between the results of the first 30 models that considered the wind direction and the second 30 models that did not, only the no wind results are shown in the table.

Table 9

Summary of the Significance of the Independent Variables in the Initial Models

Independent Variable	Significant for Concentration ($p < 0.05$)	Significant for Deposition ($p < 0.05$)
SIC10LND – Hg emissions to Land from the mining industry	OLS - 500 Lag - None Error - None	OLS - 500 Lag - 100 Error - None
SIC49AIR – Hg emissions to air from electric generation	OLS – 25, 50 Lag – 25, 50 Error - 25, 50	OLS – 25, 50 Lag - None Error - 50
SICOTAIR – Hg emissions to air from all other industry	OLS - None Lag - None Error - None	OLS - None Lag - None Error - None
POP75 – population within 75 miles of MDN site	OLS - All Lag - All Error - All	OLS - All Lag - All Error - All
FIREAREA – MODIS wildfire area within radius of MDN site	OLS - 25, 50, 75, 100 Lag - 25, 50, 75, 100 Error - 100	OLS – 25, 50, 75, 100 Lag - None Error - 50
PRECIP/INVPRECIP – total annual precipitation (or inverse) at MDN site	OLS - All Lag - All Error - All	OLS - All Lag - All Error - All
Dummy variable for year	2001 and 2003 show positive coefficients in most models	2001 and 2003 show positive coefficients in most models
Dummy variable for EPA region	Regions 4, 5, and 6 have positive coefficients and region 9 has a negative coefficient	Regions 4, 5, and 6 have positive coefficients

Source: Tirey, 2008.

The most significant independent variable, other than precipitation, in the initial models is POP75. This independent variable is the most significant contributor to the variance in the mercury concentration dependent variable at all radii and in all three types of regressions, OLS, spatial lag, and spatial error. For the deposition dependent variable, POP75 is a significant contributor in the OLS model at every radius, in the spatial lag model at 25, 50, and 100 miles, and is not significant in the spatial error model. The value of the t-statistic is positive in every case, and exceeds the values of all other independent variables in every spatial lag model except one.

The next most significant independent variable is FIREAREA. This variable is significant for concentration in all OLS and spatial lag models up through 100 miles. It becomes insignificant in these models at 500 miles and in all the spatial error models except 100 miles for concentration and 50 miles for deposition. FIREAREA is also positive when it is significant, has the highest t-statistic value in the OLS models and in the 100 mile concentration model.

SIC49AIR is significant in the OLS model for concentration at 25 and 50 miles, and in both spatial models at 25 and 50 miles. It is not significant in any model above 50 miles. It is significant for the deposition model in the OLS model at 25 and 50 miles, and in the spatial error model at 50 miles

SIC10LND is significant at 500 miles in the OLS models for both dependent variables with a negative coefficient and for deposition at 100 miles in the spatial lag model. SICOTAIR is not significant in any model, and precipitation (for deposition) or inverse precipitation (for concentration) is always significant. Dummy variables for year

are significant in many cases for 2001 and 2003, while the dummy variables for EPA region are significant in most models for regions 4, 5, 6, and 9.

The lower sections of the GeoDa printouts, illustrated for three cases in Appendix 2, provide additional helpful diagnostic statistics. On the printout for the OLS model, diagnostic statistics are provided for: 1) multicollinarity condition number and Jarque-Bera test on normality of errors, 2) three statistics for heteroskedasticity (Breusch-Pagan test, Koenker-Bassett test, and the White specification robust test), and 3) six statistics for spatial dependence. The spatial dependence statistics include the Moran's I (a significant statistic means the model contains spatial autocorrelation), the Lagrange Multiplier for spatial lag, the Lagrange Multiplier for spatial error, the Robust LM for lag, the Robust LM for error, and the LM SARMA. Anselin noted that these statistics should be considered in a given sequence. If the Moran's I statistic is significant, then the model is likely to be affected by spatial autocorrelation. The Lagrange Multiplier statistics test whether the model is of the spatial lag or error variety. If both Lagrange Multiplier statistics are significant, then only the Robust LM statistics should be considered (2005: 198). For the spatial models, diagnostics are also provided for heteroskedasticity (Breusch-Pagan) and for remaining spatial dependence (Likelihood Ratio Test). A significant Likelihood Ratio Test is a confirmation of the strength and significance of the spatial autoregressive coefficients (ρ or λ) in the spatial models (2005: 209)

A multicollinarity condition number greater than 30 suggests the independent variables are correlated. For the 20 OLS models, the multicollinarity condition number starts out at about 12 in the 25 mile models and increases to about 20 in the 500 mile

model. Based on these numbers and the results of the correlation table (see Table 6), multicollinarity does not appear to be a significant issue.

The Breusch-Pagan and Koenker-Bassett tests for heteroskedasticity are significant in all the OLS models. GeoDa is not able to calculate the value of the White test, a more general test for heteroskedasticity, so the printout returns (N/A). A summary of the outputs of the spatial dependence statistics for the OLS models is contained in Table 10.

Table 10

Summary of Spatial Dependence Statistics for the OLS Models

Model Number	**1A**	**4A**	**1**	**4**	**7**	**10**	**13**	**16**	**19**	**22**
Dependent Variable	Conc	Dep	Conc	Dep	Conc	Dep	Conc	Dep	Conc	Dep
Radius (miles)	25	25	50	50	75	75	100	100	500	500
Model Type	OLS	OLS	OLS	OLS	OLS	OLS	OLS	OLS	OLS	OLS
Adjusted R^2	0.661	0.740	0.652	0.731	0.646	0.729	0.643	0.731	0.637	0.729
Moran's I	X	X	X	X	X	X	X	X	X	X
LM lag	**X**	X	X	X	X	X	X	X	X	X
Robust LM lag	•	X	•	X	•	•	•	•	•	• **(NW)**
LM error	X	**X**	X	X	X	X	X	X	X	X
Robust LM error	X	•	X	•	X	X	X	X	X	• **(W)**
SARMA	X	X	X	X	X	X	X	X	X	X

Source: Tirey, 2008.

The top row of Table 10 lists the model number. An X in a cell indicates the statistic is highly significant (every statistic is significant in all models). A dot (•) indicates which of the spatial models (error or lag) has the higher statistic value, and is therefore the most significant. Anselin noted that in the case where both Robust LM statistics are highly significant, the model with the highest value for the test statistic should be specified (Anselin, 2005: 200). The spatial lag model has the highest Robust LM statistic in every

case for the mercury concentration dependent variable (cases 1A, 1, 7, 13, and 19) and for the mercury deposition independent variable in cases 10, 16, and 22 (no wind condition). The spatial error model is specified only for the deposition dependent variable in cases 4A, 4, and 22 (wind condition).

The heteroskedasticity (Breusch-Pagan) test statistic remains highly significant for the mercury concentration dependent variable models in both the spatial lag and error models for both the wind and no wind conditions. For the mercury deposition dependent variable, heteroskedasticity is less of a problem. It is either not significant (for the lag model at 25 miles for both wind and no wind conditions and at 50 miles for the wind condition) or GeoDa returns N/A. In all 40 spatial models, the Likelihood Ratio Test is highly significant, attesting to the strength and significance of the spatial autoregressive coefficient (ρ) in the spatial lag model and the spatial weight coefficient (λ) in the spatial error models.

Summarizing these initial results, all indications point to the specification of a spatial model for mercury measurements in rainfall. The log likelihood, the Akaike Information Criterion, and the Schwarz Criterion all indicate that the spatial models are superior to the OLS models. In order to take into account a large number of model variations (two dependent variables, wind conditions, radius of concern, and type of spatial dependence) a large number of initial models (60) were processed in GeoDa. The spatial lag model was the more robust type of spatial dependence, especially for the mercury concentration dependent variable, even though the spatial error model statistics remained highly significant. Heteroskedasticity was a problem in all the OLS models, and remains a problem in the mercury concentration spatial models. Discounting precipitation, the most

significant contributor to mercury concentration was POP75 at every radius, followed by FIREAREA at radii up to 100 miles, and SIC49AIR at 25 and 50 miles.

In the next section of this chapter, the models will be respecified, narrowed, and defined to focus on only one dependent variable and wind assumption. The functional form of the regression equation will be adjusted to deal with the heteroskedasticity problem. Heteroskedasticity does not bias the estimators, but may cause over estimation of the size of the t-statistics or z-statistics.

Model Refinements

With very little difference between the wind and no wind results, the wind direction condition was dropped from consideration in the subsequent analysis. Had the inclusion of wind conditions resulted in more significant differences in the results, it would be prudent to continue including wind direction as a model variation. However, even though there is a prevailing wind direction for every point in the U.S., there are also times throughout the year where the wind direction is highly variable. Wind direction will therefore not be included in the respecification of the model.

A more significant and difficult question is: Which dependent variable is the best measure of mercury in rainfall (concentration or deposition)? It would be tempting to simply conclude that mercury deposition is the more valid dependent variable. Since heteroskedasticity was less of a problem in the mercury deposition models, and SIC49AIR was not significant in any of the spatial lag models for mercury deposition, a conclusion that mercury releases from coal-fired plants do not impact mercury measurements in rainfall could be drawn and reported as the main finding of this research. However, there is no theoretical basis for making mercury deposition the more

valid measurement. Quite the contrary, mercury concentration is a more valid measurement of mercury in rainfall for the following reasons. Concentration is an instantaneous measurement of the amount of mercury in the precipitation at a point in time, whereas deposition is a totalization of all the concentration measurements over a year's time. Summing mercury deposition by adding up the mercury concentration amount in rainfall is not as valid due to the impact of dry deposition. Unfortunately, there is no dry deposition measuring network to supplement the wet deposition data. As stated earlier, some research suggests that dry deposition represents somewhere between one-half to two-thirds of total deposition (Miller, et. al., 2005). A wet mercury concentration measurement provides an indication of how much mercury is in the air at the time of a rainfall event, including any soluble dry deposition that might be present at the time. However, the mercury falling in-between rain events as a result of dry deposition processes is not captured or measured using wet deposition measurement techniques. Thus, totalizing mercury falling on a given spot from mercury measurements in rain on that same spot is not valid without taking into account dry deposition. Therefore concentration is the more valid measurement, and thus the remaining models consider only mercury concentration as the dependent variable.

A third consideration for model respecification is the selection of the spatial lag or the spatial error model. Therefore, according to Anselin's guidance for the use of the spatial dependence statistics, the spatial lag model is the model to specify (see table 10). The value of the Robust LM statistic for lag slightly exceeded that of the Robust LM statistic for error in every case.

Fourth, separate independent variables for population were constructed for each radius of concern, rather than being fixed at 75 miles. Independent variables were developed for POP25, POP50, POP75, and POP100 (the population within 25, 50, 75, and 100 miles of each mercury measurement point, respectively). Tying the population variable to the radius of concern was done to make the construction of all the independent variable more consistent.

With the above changes, an additional series of models were constructed to include the independent variables for each of the radii of concern (25, 50, 75, and 100 miles) in the model simultaneously, including the dummy variables for year and region. This change was made in order to see which radius of concern was the most significant and was possible due to the large degrees of freedom in the model. At first, an attempt was made to include the overlapping independent variables in the model. For example, utility emissions from 0-25 miles, 0-50 miles, 0-75 miles, and 0-100 miles were each included in the model. This however, resulted in a very high level of multicollinarity as well as continued high levels of heteroskedasticity. Heteroskedasticity results when the error terms are not equally distributed (have a constant variance). Heteroskedasticity may result in an overestimation of t-statistics or z-statistics. Thus, independent variables were calculated that included only the incremental amounts of emissions. For example, utility emissions from 0-25 miles, 25-50 miles, 50-75 miles, and 75-100 miles were included as separate variables. Similar incremental variables were constructed for mining emissions to land (SIC code 10), air emissions from all other industry, area burned by land fires, and population. In addition, the natural log of the dependent variable (concentration) and the natural log of the inverse precipitation independent variable were also calculated and

Table 11.

Revised Model Results

	Model 27	Model 28	Model 29	Model 30
Model Type	**OLS**	**Spatial Lag**	**OLS**	**Spatial Lag**
Dependent Variable/ Independent Variable	**Natural Log of Concentration**	**Natural Log of Concentration**	**Natural Log of Concentration**	**Natural Log of Concentration**
W_"Conc" – the spatial autoregressive coefficient (ρ)		0.54096*** (0.05087)		0.48888*** (0.05541)
Constant	4.27168*** (0.23349)	2.4230*** (0.25918)	1.57888*** (0.06784)	0.64287*** (0.01282)
10_25DIV - Mining emissions to land within 25 miles	-0.00707 (0.04653)	-0.00701 (0.03702)	-0.00004 (0.04525)	0.00022 (0.03761)
10_50-25 – Mining emissions to land between 25 & 50 miles	-0.01969 (0.02951)	-0.02135 (0.02348)	-0.03173 (0.02874)	-0.02938 (0.02389
10_75-50 – Mining emissions to land between 50 & 75 miles	0.00010 (0.00008)	0.00000 (0.00006)	0.00007 (0.00008)	0.00000 (0.00007)
10100-75 – Mining emissions to land between 75 & 100 miles	0.02825 (0.04183)	0.03067 (0.03328)	0.04517 (0.04075)	0.04194 (0.03386)
49_25 – Utility emissions to air within 25 miles	0.00006* (0.00004)	0.00004 (0.00003)	0.00008** (0.00003)	0.00005* (0.00003)
49_50-25 - Utility emissions to air between 25 & 50 miles	0.00004 (0.00003)	0.00005* (0.00002)	0.00002 (0.00003)	0.00003 (0.00002)
49_75-50 - Utility emissions to air between 50 & 75 miles	-0.00005** (0.00002)	-0.00005** (0.00002)	-0.00005 (0.00002)	-0.00005** (0.00002)
49_100-75 - Utility emissions to air between 75 & 100 miles	0.00002 (0.00002)	0.00000 (0.00002)	0.00002 (0.00002)	-0.00000 (0.00002)
OT_25 - All other emissions to air within 25 miles	0.00005 (0.00005)	0.00006 (0.00004)	0.00011** (0.00005)	0.00009** (0.00004)
OT_50-25 – All other emissions to air between 25 & 50 miles	0.00006 (0.00005)	0.00010*** (0.00004)	0.00008* (0.00005)	0.00011*** (0.00004)
OT_75-50 – All other emissions to air between 50 & 75 miles	0.00001 (0.00003)	0.00001 (0.00002)	0.00001 (0.00003)	0.00001 (0.00003)
OT_100-75 – All other missions to air between 75 & 100 miles	-0.00003 (0.00003)	-0.00002 (0.00003)	-0.00000 (0.00003)	-0.00000 (0.00003)
FA_25 – Area burned within 25 miles	0.00021* 0.00011	0.00009 (0.00009)	0.00032*** (0.00011)	0.00017* (0.00009)
FA_50-25 – Area burned between 25 and 50 miles	-0.00012** (0.00005)	-0.00009** (0.00004)	-0.00014*** (0.00005)	-0.00011** (0.00004)
FA_75-50 – Area burned between 50 and 75 miles	0.00028*** (0.00007)	0.00017*** (0.00005)	0.00018*** (0.00007)	0.00012** (0.00006)
FA_100-75 – Area burned between 75 and 100 miles	0.00004 (0.00004)	0.00009*** (0.00003)	0.00010** (0.00004)	0.00012*** (0.00004)
POP25 – Population within 25 miles	-0.00193 (0.02421)	0.03705* (0.01937)	0.01901 (0.02331)	0.05023*** (0.01949)
POP50-25 – Population between 25 and 50 miles	0.02382* (0.01296)	0.00854 (0.01036)	0.02203* (0.01261)	0.00845 (0.01055)
POP75-50 – Population between 50 and 75 miles	0.00507 (0.00817)	0.01118* (0.00653)	0.00465 (0.00796)	0.01060 (0.00665)

Table 11, Continued.

Revised Model Results

	Model 27	Model 28	Model 29	Model 30
Model Type	**OLS**	**Spatial Lag**	**OLS**	**Spatial Lag**
Dependent Variable/ Independent Variable	**Natural Log of Concentration**	**Natural Log of Concentration**	**Natural Log of Concentration**	**Natural Log of Concentration**
POP00-75 – Population between 75 and 100 miles	0.00558 (0.00834)	-0.00153 (0.00667)	0.00094 (0.00805)	-0.00486 (0.00673)
INVPRECP – The inverse of precipitation			187.22820*** (16.46526)	124.75690*** (15.41638)
LNINVPCP – The natural log of the inverse of precipitation	0.35133*** (0.03362)	0.24870*** (0.02840)		
D2001 – Dummy variable for year, 2001	0.17912*** (0.03837)	0.16465*** (0.03053)	0.19339*** (0.03732)	0.17687*** (0.03103)
D2002 – Dummy variable for year, 2002	0.11579*** (0.03893)	0.10167*** (0.03098)	0.11011*** (0.03792)	0.10047*** (0.03152)
D2003 – Dummy variable for year, 2003	0.15989*** (0.03439)	0.15552*** (0.02736)	0.15201*** (0.03350)	0.15108*** (0.02784)
D2004 – Dummy variable for year, 2004	0.07072** (0.03337)	0.06592** (0.02655)	0.06857** (0.03248)	0.06343** (0.02699)
DREG1 – Dummy variable for EPA region, region 1	-0.15545** (0.06973)	-0.15090*** (0.05650)	-0.09577 (0.06872)	-0.11865** (0.05781)
DREG2 – Dummy variable for EPA region, region 2	-0.10763 (0.10051)	-0.03537 (0.08181)	-0.04364 (0.09842)	-0.00558 (0.08334)
DREG3 – Dummy variable for EPA region, region 3	0.03196 (0.07212)	0.06156 (0.05839)	0.09602 (0.07077)	0.09801 (0.05966)
DREG4 – Dummy variable for EPA region, region 4	0.31724*** (0.05228)	0.18373*** (0.04188)	0.30947*** (0.05057)	0.18186*** (0.04239)
DREG5 – Dummy variable for EPA region, region 5	0.29499*** (0.05906)	0.23714*** (0.04699)	0.40110*** (0.05762)	0.31707*** (0.04792)
DREG6 – Dummy variable for EPA region, region 6	0.13327*** (0.03361)	0.06263** (0.02693)	0.12076*** (0.03279)	0.06231** 0.02742
DREG7 – Dummy variable for EPA region, region 7	0.33834** (0.14725)	0.29006** (0.11716)	0.40197*** (0.14344)	0.33783*** (0.11922)
DREG8 – Dummy variable for EPA region, region 8	0.10978 (0.08277)	0.07593 (0.06594)	0.19704** (0.07887)	0.15226** (0.06562)
DREG9 – Dummy variable for EPA region, region 9	-0.33533*** (0.10647)	-0.32259*** (0.08539)	-0.37264*** (0.10411)	-0.33294*** (0.08774)
R Squared	0.668	0.761	0.685	0.753
Log-Likelihood	85.2	130	93.0	126
Breusch-Pagan Test for heteroskedasticity		29.2		35.6
Likelihood Ratio Test		89.5***		65.5***

Notation for Table 11: *** = $p < 0.01$; ** = $p < 0.05$; * = $p < 0.10$.
Values in table for independent variables are coefficients, with standard errors in parentheses.
Source: Tirey, 2008.

included in the revised models. Thus, the revised model is a log-linear model in all the explanatory variables except inverse precipitation, in which case the model is a double-

log model. Table 11, on pages 79 and 80, illustrates the results of the OLS and spatial regressions in the revised models. The additional models are numbered 27 through 30. Models 27 and 28 are the OLS and spatial lag regressions for the all inclusive model with the natural log of concentration as the dependent variable, and the inverse precipitation independent variable. Models 29 and 30 are the OLS and spatial lag regressions for the all inclusive model with the natural log of concentration as the dependent variable, and the natural log of inverse precipitation independent variable. Although the OLS models have high multicollinarity and heteroskedasticity, the statistics provided by GeoDa indicate the spatial lag models are superior to the OLS model. The log likelihood increases in both sets of models, while the Akaike info criterion and the Schwarz criterion decrease. Also, both the spatial lag and spatial error statistics are significant in models 27 and 29, with the statistic for the spatial lag model holding the higher value. The Likelihood Ratio Test is highly significant in models 28 and 30, indicating a high level of significance for the spatial autoregressive coefficient (ρ).

With slightly higher log likelihood, R squared value, and Likelihood Ratio Test, the best model is number 28, which includes the natural log form of the inverse precipitation variable. This model also has a lower Breusch-Pagan test statistic for heteroskedasticity, meaning a higher probability that the model is not heteroskedastic. Other than inverse precipitation, the most significant independent variable in model 28, judging by the highest z-score, is fire area burned between 50 and 75 miles ($z = 3.08$, $p<0.01$). Other positive and significant predictors in this model include all other industry emissions between 25 and 50 miles ($z = 2.71$, $p<0.01$), fire area burned between 75 and 100 miles ($z = 2.64$, $p<0.01$), population within 25 miles ($z = 1.91$, $p<0.10$), utility emissions between

25 and 50 miles ($z = 1.88$, $p<0.10$), and population between 50 and 75 miles ($z = 1.71$, $p<0.10$). Two of the independent variables are significant and have negative coefficients. These are utility emissions between 50 and 75 miles ($z = -2.49$, $p<0.05$), and fire area burned between 25 and 50 miles ($z = -2.12$, $p<0.05$).

The dummy variables for year are positive and significant or highly significant in every year, while the dummy variables for EPA region are positive and significant for regions 4, 5, 6, and 7, but are negative and significant for regions 1 and 9.

The coefficients for the significant variables in each of these best models are shown in Table 12, below. Only coefficients for independent variables that are significant at 90 percent confidence ($p<0.10$) or greater are listed in Table 12.

Table 12

Coefficient Values for the Significant Independent Variables in Model 28

Independent Variable	**Coefficient Value, β**
49_50-25 - Utility emissions to air between 25 & 50 miles, lbs	0.0000454
49_75-50 - Utility emissions to air between 50 & 75 miles, lbs	-0.0000470
OT_50-25 – All other emissions to air between 25 & 50 miles, lbs	0.000102
FA_50-25 – Area burned between 25 and 50 miles, km^2	-0.0000919
FA_75-50 – Area burned between 50 and 75 miles, km^2	0.000169
FA_100-75 – Area burned between 75 and 100 miles, km^2	0.0000898
POP25 – Population within 25 miles, millions	0.03705
POP75-50 – Population between 50 and 75 miles, millions	0.01117

Source: Tirey, 2008

Since the functional form of model 28 is a so called log-linear model of the form

$$\ln Y = \alpha + \beta X + e,$$

the interpretation of the coefficients of the independent variables is that when there is a absolute change in X, there will be a proportionate change in Y of β. So,

$$\Delta Y/Y = \beta \Delta X.$$

When ΔX is one unit, then the proportionate change in Y will be β units, or a percentage change of 100 β percent (Feinstein and Thomas, 2002: 348). This is also known as the marginal effect of the independent variable. Thus, based on this interpretation of the coefficients in the log-linear model, an increase in one pound of mercury emissions from utilities between 25 and 50 miles, will increase mercury concentration measured in precipitation by 0.005 percent, while an increase of one pound of mercury from all other industries between 25 and 50 miles will increase mercury concentration by 0.01 percent. Increasing area burned between 50 and 75 miles by one square kilometer increases concentration by 0.02 percent. Increasing population by one million people within 25 miles increases mercury concentration by 4 percent, and the same increase in population between 50 and 75 miles increases mercury concentration by 1 percent.

Elasticities can also be calculated from the coefficients in a log-linear model. Elasticity is the ratio of the proportional change in the dependent variable to the proportional change in the independent variable. For a log-linear model, elasticities can be calculated from the formula: $E = \beta$ (Xbar), where β is the unstandardized coefficient and Xbar is the mean of the independent variable (Feinstein and Thomas, 2002: 350). Table 13, on the next page, lists the elasticity for each independent variable in model 28. In this context, elasticity means the percentage change in the natural log of the dependent variable resulting from a one percent change in the independent variable. As Table 13 illustrates, a one percent increase in mercury emissions from electric plants between 25 and 50 miles would result in a three percent increase in mercury concentration, with the opposite effect (a three percent decrease) resulting from a one percent increase in utility plant emissions between 50 and 75 miles. Area burned between 50 and 75 miles has the

highest elasticity, with a one percent increase in area burned resulting in a 5 percent mercury concentration increase. A one percent increase in population, either within 25 miles or between 50 and 75 miles, increases mercury concentration by about 1.5 percent.

Table 13

Elasticities for the Significant Independent Variables

Significant Independent Variable	IV Mean (Xbar)	IV Coefficient (β)	Elasticity
49_50-25 - Utility emissions to air between 25 & 50 miles, lbs mercury	624	0.0000454	0.0283
49_75-50 - Utility emissions to air between 50 & 75 miles, lbs mercury	618	-0.0000470	-0.0291
OT_50-25 – All other emissions to air between 25 & 50 miles, lbs mercury	196	0.000102	0.0200
FA_50-25 – Area burned between 25 and 50 miles, square kilometers	236	-0.0000919	-0.0217
FA_75-50 – Area burned between 50 and 75 miles, square kilometers	276	0.000169	0.0466
FA_100-75 – Area burned between 75 and 100 miles, square kilometers	281	0.0000898	0.0252
POP25 – Population within 25 miles, millions	0.4	0.03705	0.0148
POP75-50 – Population between 50 and 75 miles	1.5	0.01117	0.0168

Source: Tirey, 2008.

The next chapter discusses the results presented for this study in more detail, addresses the main research question, and explores the policy implications that result.

CHAPTER V

DISCUSSION

Initial Models

The initial set of 60 models confirms the existence of a spatial component to the relationship between the dependent variables and the estimators. In each case, the diagnostic statistics provided by GeoDa indicate that the spatial models are superior to the OLS models. While both the spatial error and spatial lag statistics were significant in every case, the higher value of the Robust LM lag statistic results in the specification of the spatial lag model. Recalling that the spatial error specification means there is correlation across space in the error term and is usually caused by missing variables, while the special lag specification means the dependent variables in each space are affected by the independent variables in that space as well as the independent variables in other spaces. The results presented here indicate that while there are elements of spatial lag and spatial error present, the spatial lag model is slightly more significant. In other words, while there is evidence the model is affected by missing variables, there is stronger evidence that the dependent variable (mercury concentration) at each measurement point is affected by the independent variables in adjacent spaces. Missing variables could be other anthropogenic emissions of mercury, underestimated mercury, natural emissions, or mercury coming in from outside the U.S. (from China for example).

The initial models also provide information regarding which independent variables might be significant at various radii. As illustrated by the results presented in Table 9, emissions to land from the mining industry (SIC10LND) was significant at $p<0.05$ at a radius of 500 miles in the OLS models for both dependent variables and at 100 miles in the spatial lag model for deposition. In addition, the sign of the coefficient is negative in the 500 mile models. One possible explanation for this may be that the emission sites are remotely located, and one significant finding of this study is the positive correlation between mercury measurements and population, which will be discussed in more detail below. As Figure 6 on page 54 illustrates, the raw number of emission points from the mining industry is very small, even though the total mercury released is large, 3.6 million lbs in 2005 (see Table 2). The results of this study indicate that the mercury released to land does not end up in the air, and does not appreciably affect the measurement of mercury at the MDN sites. Whether this mercury makes its way into the water table, and eventually into lakes and streams and the tissue of fish, is a separate question not addressed by these results.

Based on the results of the initial 60 models, mercury emissions to the air from all other industries (SICOTAIR) is another independent variable that does not appear to affect mercury measurements at MDN sites. These emission sources are more numerous than the utility emission sources (compare Figures 5 and 7 on page 54), but collectively make up a smaller proportion of air emissions (see Table 2 on page 53). This proportion also decreases over the period of the study, from 39 percent in 2001 to 32 percent in 2005. The SICOTAIR variable was not significant ($p<0.05$) at any radius for either the concentration or deposition dependent variable, regardless of whether or not wind

direction was taken into account. However, this variable was significant in the model respecification discussed in the next section of his chapter.

Mercury emissions to the air from electric utilities (SIC49AIR) was significant at $p<0.05$ in the initial OLS models at radii of 25 and 50 miles, and in both spatial models at 25 and 50 miles. At 75 miles or greater, this variable was not significant in any model. The implications of this result and the results of the model respecifications to the central research questions of this study will be discussed below.

The area of land burned during wildfire events (FIREAREA) was highly significant in the initial models at 25, 50, 75, and 100 miles. It should be kept in mind that mercury emissions from wildfires are really re-emissions of mercury that have been deposited onto forests and grasses through ongoing wet and dry deposition, from all sources, anthropogenic and natural.

Population within 75 miles of the MDN sites (POP75) was significant in every OLS model and in all of the spatial models for concentration. This independent variable also had the highest t-statistic (OLS models) or z-statistic (spatial models) for every concentration model except the spatial error model at 50 miles. The significance of the population variable, included in the model as a proxy variable for automobile emissions, was not fully expected. Thus, this led to changing the population variable to include the population at the model radius, rather than using the population at 75 miles for every model.

Although multicollinarity was not an issue in the initial models, heteroskedasticity was an issue for all of the models where concentration was the dependent variable. As discussed in the results chapter, concentration is a more valid dependent variable than

deposition, so only concentration was modeled as the dependent variable in the respecification.

Model Respecification

The results chapter describes how the model was respecified, to deal with the above issues. The respecification includes all of the changes listed below:

- Wind direction, since it did not appreciably change the results of the initial models was not included in the respecification.
- Concentration, a more valid measurement than deposition, was selected as the dependent variable.
- Since the robust LM statistic for lag was slightly more significant than the robust LM statistic for error, the model was specified as a spatial lag model.
- The respecified model focused on the radii where the SIC49AIR independent variable showed the higher probability of being significant (25, 50, and 75, and 100 miles).
- The radius of the population variable was changed to correspond to the radius of the other independent variables. Population variables were created for 25, 50, and 100 miles.
- The model was respecified as a log-linear model in all the explanatory variables except inverse precipitation, and this variable was treated as both the double log model (models 27 and 28) and the log-linear model (models 29 and 30).

Once the above changes were made, the GeoDa regression results (illustrated in Tables 11, 12, and 13) show that model 28, a log-linear model in the explanatory

variables except for inverse precipitation, is the best model. Model 28 has a higher log likelihood, a higher R squared value, a higher Likelihood Ratio Test, and a higher probability that the model is not heteroskedastic. Based on the values of the z-statistics in model 28, area burned within 50 and 75 miles was the most significant predictor with a positive coefficient, followed by all other industry emissions between 25 and 50 miles, area burned between 75 and 100 miles, population within 25 miles, utility emissions between 25 and 50 miles, and population within 50 and 75 miles. Two predictors have negative and significant coefficients: utility emissions between 50 and 75 miles, and area burned between 25 and 50 miles.

Central Research Question

Returning to the central research question for this study, "is mercury from electric power plants a local or a global pollutant?". The answer is both yes and no. When all utility emissions within a radius of 25 miles or more of a mercury rainfall concentration measurement site are considered, there is no correlation. However, there is a marginal positive correlation between utility emissions between 25 and 50 miles ($z = 1.88$, $p<0.10$). At distances between 50 and 75 miles, the correlation is more significant, but the relationship is reversed ($z = -2.49$, $p<0.05$). At distances greater than 75 miles, there is no correlation. The explanation for why the correlation reverses in not readily apparent. There may be an atmospheric explanation for this result. As the literature review in this study suggested, there is a portion of mercury from power plant stacks that is soluble in water and is washed out of the atmosphere through precipitation events. It could be that this portion of mercury is efficiently returned through precipitation events within 50 miles of the stacks, and very little soluble mercury is available after that distance. The

non-soluble mercury (the elemental mercury), does not return to the earth as a local pollutant, and is independent of the spatial effects of its release. Thus, the answer to the central research question is that there is evidence that power plant mercury can act as a local pollutant within 50 miles of a stack, but acts more as a global pollutant at distances greater than 50 miles. The significance of the regional dummy variables might be associated with the prevalence of coal fired power plants in the EPA regions that had a positive significant correlation (regions 4, 5, 6, and 7), and the lack of coal plants in the regions that have a negative correlation (regions 1 and 9). The significance of the positive correlation for the year dummy variables is less clear. These results are important, especially in light of the reasons that various states and environmental groups have opposed CAMR. How these results might be understood from a policy standpoint, and two proposals for going forward with federal mercury regulation are the subject of the next section of this chapter.

In order to have concluded that mercury emissions from power plants were a highly significant local contributor to mercury measured in rainfall, and that a MACT type regulatory standard was necessary to control mercury emissions, two results would have been necessary from this research. First, a higher level of significance would be necessary for the power plant mercury emissions independent variables. Indeed, the z-statistic for mercury emissions from power plants between 25 and 50 miles was exceeded by that of four other independent variables in the best model, including: 1) fire area burned between 50 and 75 miles, 2) non-utility industry emissions between 25 and 50 miles, 3) fire area burned between 75 and 100 miles, and 4) population within 25 miles. Power plant mercury emissions were not significant and positively correlated at any other

distance in the model. The term marginally significant is used here since the probability that the coefficient for power plant mercury emissions is zero is greater that 5 percent but less than 10 percent. In addition, the coefficient for power plant mercury emissions between 50 and 75 miles has a negative sign, and a higher level of significance ($p<0.05$). For whatever reason or reasons, mercury measured in precipitation is lower with higher levels of power plant emissions between 50 and 75 miles, a totally unexpected correlation.

Secondly, the literature available on the technological capability of power plant mercury removal equipment and options indicates that consistent high levels (90 percent removal rates or higher) of mercury removal is not currently possible at all facilities. In addition, measurement equipment capable of continuous mercury emissions monitoring in the power plant stack is not available today. CAMR is a better option than MACT due to the incentive for improved technology that would have resulted. It is not possible to implement MACT unless there is equipment that is available to reach the removal rates mandated by such a standard, and some way to measure whether or not the equipment is working. CAMR addressed both of these problems.

Policy Implications

The main hypothesis in this study, that mercury emissions from electric utilities do not affect mercury measurements in rainfall, is rejected. However the evidence of a correlation is not overwhelming, and the answer to the central research question is mixed. Electric utility emissions can act as a local pollutant when power plant stacks are between 25 and 50 miles of the measurement location. At closer distances (25 miles) or longer distances (greater than 50 miles), these emissions are better classified as global

pollutants. This is a key finding from the perspective of those who are in opposition to CAMR, which sought to set up a cap-and-trade strategy for mercury emissions from utility plants. As described previously, the main arguments against CAMR is that plants would be able to purchase allowances to continue operating without installing mercury controls, thus exacerbating already existing hotspots. Table 14 lists the states that have joined the suit against the EPA seeking to strike down CAMR. Ten of the 16 states in Table 14 have less that 1000 lbs/year of mercury emissions, and are very far away from the concentration of utility plants in the Midwest. This research suggests that

Table 14

Mercury Emissions from States Suing the EPA over CAMR (New York State, 2006)

State	2005 Mercury Electric Utility Emissions
California	52
Connecticut	108
Delaware	323
Illinois	4,164
Maine	0
Massachusetts	211
Michigan	2,933
Minnesota	1,714
New Hampshire	141
New Jersey	395
New Mexico	1,318
New York	708
Pennsylvania	6,287
Rhode Island	0
Vermont	0
Wisconsin	2,574

Source: New York State Attorney General, 2006.

the mercury emitted from utility plants greater than 50 miles away does not affect the mercury measured in rainfall. The remaining six states, four of which are in the top 15 states for mercury emissions from utility plants, are free under CAMR to prohibit trading,

or to implement other mercury regulations at the state level to control mercury emitted within their borders.

The victory of these states in the February 2008 District of Columbia Court of Appeals opinion has been touted as a "rebuke" of the George W. Bush administration (CBS News, 2008). With this judicial result, the EPA may decide to appeal the case to the U.S. Court, or to try to re-write CAMR under Section 112 of the Clean Air Act to incorporate trading, or to mandate command-and-control type provisions. Since there is no currently available technology that can be applied in all coal burning plants to achieve a specified control percentage, the utility companies (and state public service commissions) will oppose attempts to require expensive controls that have not been commercially demonstrated. The ultimate result of this court case may be a delay in any effective mercury control regulations for many years. It is interesting to note that all of the states joining in the suit except for New Mexico, voted for John Kerry in 2004 and are considered to be so-called "blue" states. It would certainly be ironic if these states' so-called victory over the Bush administration occurs at the cost of the implementation of meaningful market-based mercury regulations that, over time, would have resulted in efficient and effective mercury control in coal-fired power plants.

Perhaps it will not be possible for the EPA under the current administration to resubmit a mercury control regulation that is acceptable to the states and environmental groups who opposed CAMR. However, two proposals are suggested below that take into account the results of this research. The first proposal is to modify CAMR to include the provision of transfer coefficients, and the second is to modify CAMR to include a 50 miles trading rule.

Policy Proposal 1 – Modify CAMR to Include Transfer Coefficients

When the location of the source of pollutants is important, economic theory calls such pollutants non-uniformly mixed pollutants, and deals with such pollutants by building into a transferable emissions permit system the concept of transfer coefficients. The relationship between sources (emission points) and receptors (measurement points) for a non-uniformly mixed pollutant is described by the following formula:

$C_R = a_1E_1 + a_2E_2 + a_iE_i + B,$

where C_R is the concentration at receptor R, a_i is the transfer coefficient for source i, E_i is the emissions from source i, and B is the background level from natural sources or sources outside the control area. The transfer coefficient is intended to capture the amount the concentration will rise at the receptor for one additional unit of pollution from the source. Using this approach, the emissions allowance for each source is calculated from the formula $t_i = a_iF$, where t_i is the per unit charge paid by source i, and F is the marginal cost of a unit of concentration reduction (Tietenberg, 2003: 351).

The problem here is that we do not know the values of the transfer coefficients, a_i, or the marginal cost of reducing mercury at the wet deposition measurement points, F. Although mercury from power plants is an air pollutant, we do not know the true impact of one additional unit of power plant emissions to the concentration of mercury in the air, nor do we know how to relate that to the ultimate endpoint of human risk, the amount of methylmercury contained in fish tissue. One way around this dilemma would be to calculate transfer coefficients based on mercury as a water pollutant, rather than as an air pollutant. If we assume that some portion of the mercury emitted from power plants will end up returning to the earth in the watershed within which the utility plant is located or

is close to, we can estimate transfer coefficients for each watershed. The coefficients can be estimated based on the levels of mercury contained in the fish in each watershed. Watersheds containing fish with high levels of methylmercury content would have higher transfer coefficients, and thus higher cost mercury allowances, than water sheds containing fish with lower levels of mercury. In this way, plants located in or within 50 miles of a watershed would be required to pay more for allowances to emit mercury, and those plants located in the same areas would have a greater economic incentive to install mercury reduction technology.

The drawback to this proposal of course, is its complexity. The estimation of transfer coefficients would require analysis of complex information regarding mercury levels in fish, and the agreement of many stakeholders in the process. However, in theory, this would work. Plants located in watersheds feeding bodies of water with high levels of fish tissue methylmercury content would have higher cost mercury allowances. This higher cost sends an economic signal to power plants to locate in a lower cost area, or sends a signal to existing power plants to install mercury control technology, or shut down/curtail operations. For trading, the price of the allowance would be based on the transfer coefficient of the watershed for which the allowance will be used. This would shift allowances to facilities in lower cost, and thus lower risk, mercury regions.

Policy Proposal 2 – Modify CAMR with a 50 Mile Trading Rule

The alternative takes advantage of the knowledge gained from this research that mercury emissions are a local pollutant only at distances up to 50 miles from the power plant stack, and a global pollutant otherwise. The proposal is that the CAMR rule be modified such that sources located within 50 miles of another state must comply with the

trading rules of both that state(s) as well as the state in which the plant is located. This would allow those states having little mercury emissions within their own borders, to have a say regarding emissions within 50 miles of their border. For example, Maine, Vermont, and Rhode Island, with zero mercury power plant emissions, could prohibit the facilities within 50 miles of their borders from purchasing mercury allowances on the open market. It is in the best interests of these states to allow trading between facilities that are further away. Since, over time, the number of mercury allowances will decrease, the amount of mercury emitted will decrease, and those states (states concerned about hot spots) will benefit from trading in distant states (a lower overall cost of electricity generation), while not being exposed to a possible increase in emissions from nearby plants that could affect local mercury measurements. This is a simple adjustment to CAMR, which addresses the local pollutant concern, while maintaining many of the benefits associated with a market based mechanism.

Both of these proposals would avoid the major problem associated with creating a MACT type standard for mercury. That is, that the best mercury control technology would not need to be defined by the control authority. In addition, power plant operators would be free to reduce mercury by the most economical means possible, including installing new technology, configuring existing pollutions controls to maximize mercury removal rates, switching or washing fuel, reducing the operating hours of high mercury releasing plants, and even shutting down facilities. Both proposals also address the concerns of the states and environmental groups that oppose CAMR, the creation or exacerbation of mercury hot spots.

The next and final chapter summarizes this research, draws a number of conclusions, and defines a number of areas which present opportunities for further research into mercury emissions from power plants.

CHAPTER VI

SUMMARY AND CONCLUSIONS

Research Summary

This study utilized available data to examine the significance of mercury air releases from coal-fired power plants in the U.S. in a geospatial model. Although examples of the use of EPA TRI data for social science explorations are plentiful, no study found has combined TRI data and MDN data before. In addition, the exploration of the spatial effects and implications for mercury emissions has not been published before. Due to the spatial nature of how we measure mercury in rainfall, and how we report mercury emissions from industry, a spatial model is especially well suited to the problem. The application of a spatial model was achievable since geographic details for all the data identified for the independent variables were available. Even with the availability of the data, however, this research would not have been possible without the easy accessible availability of the spatial regression software, GeoDa 9.5i, and the associated manuals and case studies made available by Luc Anselin at the University of Illinois.

Another important aspect of this research was the use of Microsoft Access to analyze and develop the independent variables to relate them to the dependent variables. The use of the great circle formula from spherical geometry to calculate the distance from each of the MDN sites to hundreds of thousands of TRI emission sites, county centroids, and wildfire locations was efficient and may have application in other research endeavors.

Wind direction was incorporated into the analysis through the comparison of longitude and latitude of the emission sources and the MDN sites. Grouping and summarizing the data for a variable radius around the MDN sites required the use of more than 400 tables and 1,000 queries in Microsoft Access. Although time consuming, this methodology allowed total control over the use of each of the fields made available by the various data sources, including longitude, latitude, emissions, and SIC code. Some of the procedures certainly taxed the capabilities of a typical home computer. For example, the process of calculating the distance from each of the 75 MDN measurement sites to each of the about 75,000 fires detected by the MODIS system in 2005 required the calculation and creation of a table in Microsoft Access that contained over 5.5 million records. The query that created this table took about 2 hours to process on a 3 gigahertz Pentium Gateway computer, with 512 MB ram. The research required the processing of 10 similar queries just for the creation of the fire area data tables.

The results of this study are both significant and timely. Sixteen states filed suit against CAMR in 2005, and the result of that lawsuit (setting aside CAMR) came down from the United States District of Columbia Circuit Court just a couple of months before the defense of this dissertation was conducted. The stakeholders have not yet announced whether or not they will appeal the decision to the U. S. Supreme Court. The policy prescriptions outlined in the previous chapter provide two suggestions for revising CAMR that take into account the results of this research. The conclusions detailed below may be useful as policy makers either implement a new and revised CAMR, or appeal the court decision.

Conclusions

Several conclusions can be drawn from the results of this research. Each is listed below in bullet format, followed by supporting discussion. The final section of this chapter discusses opportunities for future research.

The six main conclusions resulting from this study are as follows:

- The available literature documents a number of uncertainties associated with mercury research that affect any cost benefit conclusions that might be reached regarding the best way to regulate this pollutant.
- When considered at a distance between 25 and 50 miles, mercury emissions from electric utility plants are marginally significant as a predictor of mercury concentration measured in rainfall. Between 50 and 75 miles, emissions from utility plants are negatively correlated with rainfall mercury concentration. At 75 miles or greater, mercury emissions from coal plants do not significantly affect mercury measurements in rainfall. At 50 miles or less, mercury from power plants is at least partially a local pollutant. For sources of mercury that are over 50 miles away, the evidence is that mercury is more of a global pollutant.
- The most significant predictor of mercury concentration in rainfall, other than precipitation, is the fire area burned. This variable is highly significant and positive at distances between 50 and 75 miles, and is marginally significant and positive above 75 miles. Between 25 and 50 miles fire area was significant and negatively correlated. Population was marginally significant and positive at distances up to 25 miles, and between 50 and 75 miles.

Although population was included in this research model as a proxy variable for vehicle emissions, the true nature of the correlation between population and mercury measured in rainfall remains an open question.

- There are components of both spatial lag and spatial error in a mercury air emission model.
- It is unlikely that a cap-and-trade approach to the regulation of mercury emissions would result in an exacerbation of hot spots, even if CAMR were to be implemented in its original form, since coal-fired power plants are already operating at capacity (EIA, 2007), and since the location of coal fired units to mercury measurement sites is significant only within a narrow distance range (25 to 50 miles).
- The debate over CAMR may boil down to a battle between two approaches to regulation: the Precautionary Principle versus market-based regulation.

Uncertainties – The first conclusion of this research is a summary of the uncertainties resulting from a review of the available literature on mercury air emissions. These uncertainties include: 1) The fraction of elemental mercury released from coal burning power plants may be underestimated, 2) it is not clear whether the amount of mercury measured in the environment is increasing or decreasing in the U.S., 3) the amount of mercury released from natural processes may be underestimated, and 4) the best available mercury control technology for power plants is currently not clear. The next few paragraphs support these uncertainties with references previously cited.

Although the EPA assumes in its modeling programs that about half the mercury in a typical power plant stack is elemental, with the remaining half either reactive gaseous

compounds of mercury or mercury attached to particulate matter, there is a great deal of research that concludes the elemental mercury may be underestimated (Sigler and Lee, 2006; Walcek, et. al., 2003; Lohman, et. al., 2006; Edgerton, 2006). Recalling that the elemental mercury is insoluble in water and does not fall out locally in wet and dry deposition processes, an underestimation of elemental mercury in power plant stacks would significantly change the results of deposition modeling on which the EPA and others draw conclusions about mercury as a local or global pollutant. This is a very important and significant uncertainty associated with mercury research, but there are others as well. For example, the risks of mercury to human populations is also a matter of uncertainty regarding whether mercury levels are increasing or decreasing in the environment (Driscoll, 2007; Madsen and Stern, 2007; Slemr, et. al., 2003; Sacramento River Watershed Program, 2002), whether the developmental benefits of fish consumption are exceeded by the risk of mercury exposure (Mergler, et. al., 2007), and whether the mercury released by anthropogenic sources are only a small percentage of mercury released by natural processes (Engle and Gustin, 2002; Rassmussen, 1994; Gustin; 2003).

The greatest uncertainty evident in the research is that the best way to control mercury is not yet known. There is no single mercury control technology that can be applied in every coal-fired power plant that will control mercury at a high level of efficiency (GAO, 2005; Pavlish, et. al., 2002). In fact, we cannot now reliably measure the fractions of elemental, reactive gaseous, and particulate forms of mercury on a continuous basis in the stack. The technology to measure mercury on a continuous basis in all its speciated forms in the harsh environment of the power plant stack is still being

developed (Pavlish, 2003). If the EPA abandons its attempts to implement a cap-and-trade mechanism and is forced to implement a command-and-control standard, the standard will require controls equal to the best 12 percent of all currently operating units. Since the best control technology is not yet known, both the implementation of the regulation and the legal battles over individual sites could extend for years, and there will be no incentive for improvements in mercury collection or mercury measurement technology. In fact the incentive will be just the opposite. Utility strategists will take a wait and see attitude, while these legal battles play out. Unfortunately, the resources that would have gone into the development of new technology will instead go into legal costs. Hopefully, the EPA will re-implement cap-and-trade for mercury under section 112 of the Clean Air Act, possibly with the incorporation of changes outlined in the policy prescriptions in the last chapter.

Local versus Global Pollutant - The main hypothesis for this research, that mercury from power plants does not correlate with mercury measurements in rainfall, is rejected. Utility mercury is at least partially a local pollutant within 50 miles of the stack. After dealing with heteroskedasticity, mercury emissions from electric utilities between 25 and 50 miles from the MDN measurement sites was marginally significant ($z = 1.88$, $p<0.10$) in a spatial lag model that utilized mercury concentration in rainfall as the dependent variable. Thus, the emissions in this distance band could contribute to a local hotspot. In the last chapter two policy changes to CAMR (incorporating the concept of transfer coefficients into a redesigned CAMR rule or allowing states to prohibit mercury trades that increase emissions within 50 miles of the state's border) are intended to deal with this concern.

Emissions at distances between 50 and 75 miles were even more significant ($z = -2.49$, $p<0.05$), but the sign of the coefficient was negative. Emissions from sources up to 25 miles or greater than 75 miles are not a significant predictor of mercury measurements in rainfall. Further research is necessary to explore the reason for the negative correlation. These results are highly significant to those who are concerned that mercury from power plants is drifting from the Midwest and South to the Northeast and is a major contributor there to mercury pollution. The results of this research suggest that at up to 25 miles, or more than 75 miles, the amount of mercury released from power plants was not a significant predictor of mercury measurements in rainfall. Only between 25 and 50 miles was a marginal level of significance indicated in the spatial model. For example, collective mercury emissions from power plants in Illinois, Kentucky, and Georgia are not a significant predictor of mercury measured in the rainfall in New York, New Hampshire, or Maine in any of the spatial models.

CAMR, if implemented, would reduce mercury emissions from most power plants. Very few will increase emissions through the purchase of emission credits. There may be some older facilities that continue to operate at current capacity by purchasing allowances, but coal-fired base load units already operate at or near capacity (EIA, 2007).

Fire Area and Population – Fire area burned is a significant predictor of mercury measured in rainfall at MDN sites greater than 50 miles in distance. Forest fires seem to act as a diluting force in the distribution of mercury on the land. Mercury from all sources (man-made and natural) falls to the earth in wet and dry deposition processes and is distributed again through fire processes depending on the direction of the wind.

In the initial spatial lag models, the most significant predictor of mercury concentration is the total population living within the radius of the model. In the revised models, population was significant up to 25 miles, and between 50 and 75 miles. These results were not expected and perhaps are a good opportunity for future research. Population was chosen as a proxy variable for vehicle emissions in the absence of available geospatial data for number of vehicles. This does not necessarily mean that mercury from vehicle exhaust completely explains the contribution of the population independent variable. There may be other explanations. Some of these possibilities include: 1) heating oil combustion in home heating systems (the fuel oil contains trace amounts of mercury), 2) landfills (although usually landfills are located in rural areas and do not correlate spatially with population centers), and 3) industrial emissions that are not being reported in TRI data. In the best model, mercury emissions from industries other than utilities were significant at distances between 25 and 50 miles.

Spatial Nature of the Model - The spatial nature of this model was confirmed in every case through the use of the spatial statistics available in the GeoDa software. In all 60 of the initial models, the log likelihood statistic for the spatial lag and error models was greater, and the Akaike info criterion and Schwarz criterion were lower, than the corresponding statistics in the OLS models. In addition, the diagnostic for spatial dependence (the likelihood ratio test), was highly significant in every spatial model. Through the utilization of procedures defined by Anselin, it was determined that the spatial error model, though significant, was slightly less indicated than the spatial lag model. The existence of the spatial error component in the model means that there may be missing variables. This could be mercury coming from outside the U.S., or a missing

anthropogenic or natural source of mercury within the U.S. The fire area burned independent variable would account somewhat for any missing variable, since the mercury emissions from wildfires are a re-admission of whatever mercury has fallen onto forests and plants, from all sources. Likewise, the population variable may also account for a missing variable. The spatial lag component takes into account the impact of mercury measurements at the other spatially weighted sites at each measurement site in the model. The results of the spatial regressions shed more light on the relationships between the dependent variable and the independent variables than would a simple linear regression model, and allow greater reliance to be placed on the resulting coefficients, especially after changing the form of the model to eliminate high levels of heteroskedasticity.

Hotspots – Most likely, the implementation of CAMR in its original form would not result in the creation of new, or the significant exacerbation of existing, hotspots. The average capacity factor for coal-fired electric utility plants in the U.S. is about 73 percent, and from a power source standpoint is exceeded only by the capacity factor for nuclear generated electricity (about 90 percent) (EIA: 2007). There is a practical limit to the capacity factor due to the need to conduct annual maintenance and the occasional unexpected event which causes downtime. These two categories limit the practical capacity factor of coal plants to between 85 and 90 percent. Capacity factor is a measurement of how much time the unit runs compared to the total time available. The fact is that there is not much room to ramp up the production of coal-fired units, perhaps a maximum 15 percent for any plant that is not fully utilized today. In fact, larger coal-fired units, currently the least cost to operate, are already considered to be the base load

units for most utilities, and are operated at their highest capacity already. The addition of the cost of allowances for mercury due to the implementation of CAMR will result in a higher cost to operate these units, not a lower cost. Thus, the capacity factor for coal units will most likely decrease, rather than increase under CAMR. In addition, as new facilities are built, CAMR would result in technological competition to install more and more efficient means of controlling mercury. Over time, this technological competition would result in lower cost, more efficient mercury control options. These options would in turn be available, not only to utilities building new facilities in the U.S., but also to companies building new coal units in China, India, and all over the world. The new technology that results from incentive approaches to environmental regulation is well documented in the literature. Ellerman, et. al. (2000), Amar (2000), and Harrington, et. al. (2004) all found technological improvement resulting from cap-and-trade approaches as detailed in Chapter II. On the other hand, as Tietenberg (2006) points out, MACT standards produce no incentive for regulated facilities to research or introduce new and better technology. Why would utilities take the risk? With CAMR not set aside by the courts, research on new mercury control technology will most likely slow or come to a halt, as the regulated community waits to see what action the EPA takes.

In addition, as discussed above in the section on the local versus the global nature of power plant mercury emissions, the pollutant is a significant predictor of mercury measured in rainfall only within a narrow distance band, 25 to 50 miles. At less than 25 or more than 50 miles, power plant mercury does not significantly affect mercury in rainfall. As pointed out in the literature review, many of the mercury hotspots that have been identified in the Northeast are far away from power plants. The policy suggestions

in the last chapter would be a way to deal with hotspots as well, by allowing states to prohibit trades that would allow a plant within 50 miles of the state's border (assuming the state had identified the area as a hotspot), or by implementing transfer coefficients that would increase the price of mercury allowances to the point that trades would be uneconomical.

Furthermore, cap-and-trade is the better policy alternative for power plant mercury since MACT standards for mercury are not currently feasible because there is no best available mercury control technology for the wide variety of equipment and coal types now in use. As CAMR is implemented, a market for new technology should develop, and better mercury controls can be expected to emerge in the long run. The market-based regulatory approach for SO_2 and NO_x resulted in advances in scrubbing technology such that today, no utility would consider building or be able to obtain permits for a new coal plant without the latest SO_2 and NO_x controls available. With the development of a viable mercury allowance market, the same result for mercury controls is inevitable.

As mercury emissions are reduced in the U.S. due to CAMR, emissions from Asia (especially China), will become more and more important and significant to air quality in the U.S. The development of new technology resulting from the implementation of CAMR is more likely to result in the technology being used elsewhere in the world. The market will drive the cost of the technology down, making the inclusion of the technology in any new coal plant design, anywhere in the world, much more likely. A command-and-control standard that implements a higher cost of technology that is not proven to work will make the implementation of such controls in other parts of the world less likely.

Precautionary Principle - Finally, it is interesting to discuss the debate over the implementation of CAMR within the context of the Precautionary Principle (PP). Simply stated, the PP states that "[w]hen an activity raises threats of harm to human health or the environment, precautionary measures should be taken even if some cause and effect relationships are not fully established scientifically" (Ahteensuu, 2007:366). The PP has been criticized for being vague and incoherent, and for causing unintended consequences (Morris, 2007). Examples would be malaria deaths resulting from a worldwide ban on DDT or starvation resulting from banning genetically engineered food. However, the PP has been used to justify policy-making in various countries, especially in Europe (Ahteensuu, 2007). In the case of mercury pollution, the logical extension of the PP would be that, since mercury pollution poses a threat to human health and the environment, burning coal should be banned altogether. Indeed, this seems to be the position of some environmental organizations (Sierra Club, n.d.; Terra Nature, 2007). Short of a complete ban, the PP might also be employed to justify a command-and-control standard instead of the market-based policy of CAMR.

There are a number of problems with this argument. Lipfert, et. al., listed a number of unintended consequences of an expensive command-and-control style approach to mercury control including: 1) higher costs of electricity, 2) expending societal resources on a mercury MACT policy diminishes expenditures on other mercury pollution initiatives, 3) creating a false sense of security concerning methylmercury in fish, and 4) creating a concentrated mercury disposal problem (Lipfert, et. al., 2005). Lipfert's first and second points are very salient. Some would argue that the cost of coal-fired electricity should be higher in order to build in all of the life cycle costs of coal. This is a

valid argument, but only where there is a clear link between the use of coal and the subsequent costs to society. The amount of mercury in the coal may be easily identifiable and it may be easy to estimate mercury emissions, but there has been no clear link between the emissions from coal plants and the costs of subsequent mercury exposure. Imposing a command-and-control regulatory requirement that raises costs, without a subsequent reduction in downstream costs to society, hurts everyone who has to pay for electricity. If all the mercury emissions from coal plants were eliminated, and there was no subsequent reduction in mercury measurement in fish or in the environment, then the resources used up in eliminating the mercury would be wasted.

The second point is also very important, and is a matter of opportunity costs. Every dollar spent on a command-and-control type mercury regulation that exceeds what would be spent under a more efficient cap-and-trade policy diminishes the resources that are available to be spent on other initiatives in the electric power industry. This is especially true in an industry where most of the capital expenditures are controlled and approved by state public service commissions through regulatory mechanisms. The other opportunities are significant, including demand side policies (such as peak load control mechanisms and smart metering), new power plants that are much more efficient than older units, carbon capture and sequestration initiatives and research, and transmission grid and infrastructure investments to improve reliability. Indeed, this is the great problem with the precautionary principle. Every expenditure decision we make, whether we like it or not, results in reducing risk in one area, while simultaneously ignoring risk in every other area. Once all the resources are expended, there will inevitably remain some risk that is not addressed. The requirement of the PP to always take action if there is any risk to

human health or the environment ignores the need to make decisions based on a ranking of those risks.

Finally, one other unintended consequence of the abandonment of CAMR is the inevitable legal challenges expected to result from a command-and-control approach. Since there is no single easily-identifiable technology that would result in maximum mercury control at all utility plants, there would be endless legal wrangling over which controls to install, not only in new plants, but also in existing plants. As previously noted, the electric industry is already on record in support of CAMR, and is comfortable with the market-based approach as evidenced by the success of the SO_2 and NO_x trading programs. CAMR is the right policy at the right time for the right pollutant. The implementation of the PP in this case, might result in no control at all, or at least in the delay of controls for years, as litigants argue over the best way to control mercury emissions. Cap-and-trade would work to regulate mercury emissions from utility plants for each of the following reasons:

- The main argument against CAMR, that trading will worsen or create hotspots, is weak at best. Base load coal plants are already operating near capacity, and this research indicates that power plant mercury significantly affects mercury measured in rainfall only within a narrow distance band of 25 to 50 miles. Adjustments to CAMR (trading rules or transfer coefficients) can address this concern.
- The best way to control mercury is unknown. Cap-and-trade would have created incentives for facilities to reduce mercury emissions due to a wide variety of methods, including fuel switching, fuel washing, limiting

operations, and the implementation of a wide variety of technological control schemes.

- The amount of mercury and its speciation in the stack is unknown. CAMR included a provision to require continuous stack monitoring, which is currently not available. This could be reason alone for leaving CAMR in place. Until facilities fully understand how much mercury they are emitting in what form, maximizing mercury collection efficiency will not be possible.
- The regulated community supported CAMR, due to positive and successful experience with the acid rain programs for SO_2 and NO_X. The focus under CAMR would have been on the search for new technology at minimum cost, rather than on the legal wrangling that may very well result under a MACT standard.
- CAMR is a least cost solution. This is backed up by both general research into the differences between the costs of incentive based approaches versus command-and-control type approaches (Tietenberg, 2006) as well as by specific research in the costs and benefits of CAMR and MACT for mercury (Gayer and Hahn, 2006).

The above reasons summarize and clarify why CAMR is a better option than MACT for reducing mercury emissions from power plants. This is an important question with many stakeholders, not the least of which is everybody who has to pay for electricity (all of us). Because there are limited resources available in society, policy makers have a duty to consider risk in the policy-making process. The harm that might result from mercury emissions from power plants must be weighed in light of the harm that would result from

a MACT standard that 1) would result in higher electricity costs for everyone, and 2) might not result in lower mercury from power plant or lower mercury measurements in the environment. This research is an important contribution to this public policy debate because of the light shed on the impact of mercury emissions from power plant to the mercury measured in rainfall.

Opportunities for Further Research

The opportunities for additional research in this area are plentiful and include: 1) utilizing the latest available data to construct updated versions of the models built in this research, 2) further exploring the population variable by including a variable that accounts more directly for vehicle emissions or by accounting for the impact of the population variable in some other way, 3) creating a dependent variable related to the mercury content in fish, 4) conducing a similar analysis on data in other countries, 5) building a model that takes more short term mercury rainfall content measurements and wind speed and direction into account (such as weekly), and 6) determining what impact the mercury in a given air shed has on the mercury content in fish in each watershed affected by that air shed (in order to contribute to the determination of appropriate transfer coefficients), and 7) exploring the impact of CAMR (or its replacement) after the regulation has been in place for some time (policy evaluation) .

First, as utilities and other industries report their mercury emissions each year, the same regression model constructed in this research can be built with the new data as it becomes available. For example, 2006 TRI emissions will be available in mid 2008, and the corresponding MDN data is available now. There were 75 active MDN sites for the full year of 2006, and since the beginning of 2006 an additional 20 sites have been added.

Eleven of the new sites are in the Midwest and West where the MDN sites are sparsely spaced. The data for the remaining independent variables is also available for 2006. The addition of the new MDN sites addresses one of the limitations of this study, the limited number of mercury wet deposition measurement sites.

A second area for further research would be an exploration of the population independent variable. The addition of a spatially defined independent variable for number of vehicles, vehicle emissions, or vehicle-miles traveled would help identify whether or not the significance of the population independent variable is really due to vehicle emissions, or some other cause. The addition of a vehicle related variable would need to be at a sufficiently granular level that covers the entire county, such as the county or precinct level, and would need to include the longitude and latitude of the geographical unit. Once identified, the data could be analyzed the same way the population data was analyzed in this research. The variable could be summed for all geographical centroids within a given radius of the mercury measurement sites. The use of city or metropolitan area data would not be as useful, because the there would be many areas not included in city level data.

As stated previously, perhaps another explanation of the significance of this variable is that there is some unidentified or unreported source of mercury air emissions from population centers. One way to research the question of whether any industries are underreporting mercury emissions would be to map each individual industry currently reporting TRI data by SIC code on a map of the U.S. and see if there are any industries clustered around the highest concentration MDN sites (see figure 7 on page 54 for a map

of all 1,745 industries reporting mercury emissions to the EPA in 2005). Research into the question of unreported mercury would be more difficult.

A third area of research would be to operationalize a dependent variable that would take into account the current data that is available for mercury content of fish. The problem here of course is that fish tend to move around, and it would be difficult to assign mercury fish content levels to a specific geographic location (longitude and latitude). Perhaps the centroid of a lake, or the mid point of a river in which the fish is found could be used. Another problem with this approach is that mercury content in fish varies greatly by species. It would be difficult to use a fish species for the analysis that is found in all areas of the U.S. Analyzing a dependent variable based on fish mercury content may need to be limited to a smaller geographical area in which the fish are commonly found, such as a state or a region. Such a model would still make use of the methodology introduced in this research, and the independent variables (population, forest fire area burned, and industrial emissions) would still be applicable. The confounding nature of the precipitation variable would not be a problem in a fish content model, and the researcher could correctly assume that whatever mercury ending up in the fish would get there through both dry and wet deposition processes.

Fourth, if mercury emissions and mercury deposition data could be located in other countries, a similar spatial model could be constructed for that area. According to the U.S. EPA, a number of other countries have emission reporting programs similar to the Toxic Release Inventory, including Canada, the United Kingdom, Mexico, Japan, and Australia (U.S. EPA 2007). Some of these programs are voluntary (Mexico) and would therefore be less useful in a similar research effort. However, if the geographic locations

of the emission sources and a database of mercury rainfall measurements are available, it would be a straightforward process to duplicate this research for that country.

Fifth, the model constructed in this research considered the annual average mercury content in rainfall, and the average wind direction at each measurement site. Since the data for the mercury measurements and the wind speed and direction are available on a weekly basis, a model could be constructed around this weekly data. Perhaps a correlation could be found between spikes in wind speed and higher levels of mercury in rainfall. Then a more detailed investigation of what was going on with regard to emission sources during that specific time could be conducted.

Sixth, as described in the literature review and in the second proposal in the Discussion chapter, the primary risk to humans from mercury emissions is through the accumulation of methylmercury in the tissue of fish that is subsequently consumed by people. In order to determine valid watershed transfer coefficients, more research is necessary to try to determine what impact the mercury in a given air shed has on the mercury content in fish in each watershed affected by that air shed. This relationship, if it does exist, must be estimated in order to properly price the emissions permits for each watershed.

A final area of research would be applicable after CAMR (or command-and-control regulations, whichever prevail) has been in place. The impact of the regulation could be reviewed from a policy analysis standpoint, to determine: 1) whether mercury emissions are lower, 2) whether technology has developed (both collection and measurement technology) and/or is less expensive, 3) whether lower mercury emissions have impacted mercury levels in fish or other wildlife, and 4) whether the development of a strong

market for emissions in the U.S. can be implemented at a larger scale or can otherwise affect mercury control in other countries. More importantly an analysis of the impact of the regulations could be performed to determine whether mercury measurements in rainfall form the MDN data are decreasing, whether fish content levels are improving, and whether human measurements (blood and hair) are improving.

REFERENCES

Ahteensuu, M. (2007). "Defending the Precautionary Principle Against Three Criticisms", *TRANES: A Journal of the Humanities and Social Sciences*, 11, 4, 366-381.

Amar, P. K. (2000). "Environmental Regulation and Technology Regulation: Controlling Mercury Emissions from Coal-Fired Boilers", Northeast States for Coordinated Air Use Management (NESCAUM), September 2000.

Anselin, L. (2007). "A Spatial Econometric Approach to the Economies of Site-Specific Nitrogen Management in Corn Production", Luc Anselin current working papers web page, not dated, obtained from the internet on November 25, 2007 from http://sal. uiuc.edu/users/anselin/papers.html.

Anselin, L. (2005). "*Exploring Spatial Data with GeoDa: A Workbook*", Center for Spatially Integrated Social Science, Urbana, IL.

Anselin, L. (2001). " Measuring the Benefits of Air Quality Improvement: A Spatial Hedonic Approach", Forthcoming in *Journal of Environmental Economics and Management*, Luc Anselin current working papers web page, not dated, obtained from the internet on November 25, 2007 from http://sal.uiuc.edu/users/anselin /papers.html#seoul.

Anselin, L. (1988). *Spatial Econometrics: Methods and Models*, Kluwer Academic Publishers, Dordrecht, Netherlands.

Baumol, W. J. and W. E. Oates (1971). "The Use of Standards and Prices for Protection of the Environment", *Swedish Journal of Economics*, 65, 2, 421-445.

Bergan, T. et. al. (1999). "Mercury in the Global Troposphere: A Three Dimensional Model Study ", *Atmospheric Environment*, 33, 1575-1585.

Betts, K. S. (2001). "Arctic May be Naturally Generating Reactive Gaseous Mercury", *Environmental Science and Technology*, 35, 21, 434A-435A.

Biswas, A. et. al. (2007). " Release of Mercury from Rocky Mountain Forest Fires", *Global Biogeochemical Cycles*, 21 ,1, GB1002.

Bullock, O. R. Jr. et. al. (1998). "Lagrangian Model of Mercury Air Emission, Transport, and Deposition: An Analysis of Model Sensitivity to Emissions Uncertainty", *Science of the Total Environment*, 213, 1-3, 10.

Burtraw, D. and E. Mansur (1999). "Environmental Effects of SO2 Trading and Banking", *Environmental Science and Technology*, 33, 20, 3489-3494.

Butler, T. J., et. al. (2008). "Regional Precipitation Mercury Trends in the Eastern USA, 1998-2005: Declines in the Northeast and Midwest, No Trend in the Southeast", *Atmospheric Environment*, 42, 1582-1592.

Carpri, A. (1997). "Mercury From Combustion Sources: A review of the chemical species emitted and their transport in the atmosphere", *Air, Water and Soil Pollution*, 98, 241-254.

Cash, C. (2008). "UARG Awaits Decision to Rehear Mercury Opinion", Platts web page dated April 14, 2008 obtained from the internet on April 14, 2008 from www.platts.com.

CBS News. (2008). "Court Strikes Down EPA's Plan on Mercury", CBS News web page dated February 10, 2008 obtained from the internet on February 17, 2008 from http://www.cbsnews.com/stories/2008/02/10/ap/government/main3814350.shtml.

Cinnirella, S. and N. Pirrone. (2006). "Spatial and Temporal Distributions of Mercury Emissions from Forest Fires in Mediterranean Region and Russian Federation", *Atmospheric Environment*, 40, 38, 7346-7361).

Cohen, Mark. (2004). "Modeling the Atmospheric Transport and Deposition of Mercury to the Great Lakes", *Environmental Research*, 95, 247-265.

Cohen, M. and P. Miller. (2003). "Atmospheric Mercury Deposition Impacts of Future Electric Power Generation", Report submitted to the Commission for Environmental Cooperation, December 8, 2003.

Decker, C. S. et. al. (2005). "Residential Property Values and Community-Right-to-Know Laws: Has the Toxic Release Inventory had an Impact?", *Growth and Change*, 36, 1, 113-133.

Driscoll, C. T. (2007). "Mercury Contamination in Forest and Freshwater Ecosystems in the Northeastern United States", *BioScience*, 57, 1, 17-28.

Edgerton, E. S. et. al. (2006). "Mercury Speciation in Coal-Fired Power Plant Plumes Observed at Three Surface Sites in the Southeastern U.S.", *Environmental Science and Technology*, 40, 4563-4570.

Edison Electric Institute (EEI) (2007). "16 States, 9 Groups File Legal Briefs Opposing EPA Mercury Rule", *EEI Daily Energy News*, Tuesday, January 16, 2007.

Electric Power Research Institute (EPRI) (n.d.), "Mercury Hot Spots – Emissions and Deposition Patterns, Electric Power Research Institute web page, not dated, obtained from the Internet on December 26, 2007 from http://www.epriweb.com/public/000000000001014428.pdf.

Ellerman, A. D., et. al. (2000). *Markets for Clean Air*, Cambridge University Press, Cambridge, UK.

Energy Information Administration (EIA) (2007). "Figure ES3. Average Capacity Factor by Energy Source", web page dated October 22, 2007, obtained from the internet on December 23, 2007 from http://www.eia.doe.gov/cneaf/electricity/epa/figes3.html.

Engle, M. A. and M. S. Gustin, (2002). "Scaling of Atmospheric Mercury Emissions from Three Naturally Enriched Areas: Flowery Peak, Nevada; Peavine Peak, Nevada; and Long Valley Caldera, California", *Science of the Total Environment*, 290, 1-3, 91-104.

Evers, D. C. (2007). "Biological Mercury Hotspots in the Northeastern United States and Southeastern Canada", *BioScience*, 57, 1, 29-43.

Feinstein, C. H. and M. Thomas. (2002), *Making History Count: A Primer in Quantitative Methods for Historians*, Cambridge University Press, Cambridge, UK.

Freidli, H. R., et. al. (2001). "Mercury in Smoke from Biomass Fires", *Geophysical Research Letters*, 28, 17, 3223-3226.

Gayer, T., and R. W. Hahn. (2006). "Designing Environmental Policy: Lessons from the Regulation of Mercury Emissions, *Journal of Regulatory Economics*, 30, 291-315.

Gbor, P. K., et. al. (2007). "Modeling of Mercury Emission, Transport, and Deposition in North America", *Atmospheric Environment*, 41, 6, 1135-1149.

Gustin, M. S. (2003). "Are Mercury Emissions from Geologic Sources Significant? A Status Report", *Science of the Total Environment*, 304, 1-3, 153-167.

Hammerschmidt, C. R. and W. F. Fitzgerald. (2005). "Methylmercury in Mosquitoes Related to Atmospheric Mercury Deposition and Contamination", *Environmental Science and Technology*, 39, 9, 3034-3039.

Harrington, W. et. al. (2004). *Choosing Environmental Policy: Comparing Instruments and Outcomes in the United States and Europe*, Resources for the Future, Washington, D.C.

Hoyer, M., et. al. (2004). "Mercury Emissions from Motor Vehicles", paper presented at the *13th International Emission Inventory Conference: Working for Clean Air in Clearwater*, Clearwater, Fl., June 8-10, 2004.

Hylander, L. D. and M. Meili, (2003). "500 Years of Mercury Production: Global Annual Inventory by Region Until 2000 and Associated Emissions", *Science of the Total Environment*, 304, 13-27.

Hylander, L. D. and M. E. Goodsite (2006). "Environmental Costs of Mercury Pollution", *Science of the Total Environment*, 368, 352-370.

Jiang, G. B., et. al. (2006). "Mercury Pollution in China", *Environmental Science and Technology*, June 15, 2006, 3673-3678.

Kaufman, Y. J., et. al. (2003). "Fire and Smoke Observed from the Earth Observing System MODIS Instrument – Products, Validation, and Operational Use", *International Journal of Remote Sensing*, 24, 8, 1765-1781.

Keeler, G. J. et. al. (2006). "Sources of Mercury Wet Deposition in Eastern Ohio, USA", *Environmental Science and Technology*, 40, 19, 5874-5881).

Kelly, E. N., et. al. (2006). "Forest Fire Increases Mercury Accumulation by Fishes Via Food Web Restructuring and Increased Mercury Inputs", *Proceedings of the National Academy of Sciences*, 103, 51, 19380-19385.

Liang, L. et. al. (1996). "A Novel Analytical Method for the Determination of Picogram Levels of Total Mercury in Gasoline and Other Petroleum Based Products", *Science of the Total Environment*, 187, 57-64.

Lindberg, S. et. al. (2007). "A Synthesis of Progress and Uncertainties in Attributing the Sources of Mercury in Deposition", *Ambio*, 36, 1, 19-32.

Lipfert, F. et. al. (2005). "Methylmercury, Fish Consumption, and the Precautionary Principle", *Journal of the Air and Waste Management Association*, 55, 388-398.

Lohman, et. al. (2006). "Modeling Mercury in Power Plant Plumes", *Atmospheric Science and Technology*, 40, 3848-3854.

Lyman, S. N. et. al. (2007). "Estimation of Dry Deposition of Atmospheric Mercury in Nevada by Direct and Indirect Methods", *Environmental Science and Technology*, 41, 6, 1970-1976.

Madison Declaration. (2007). "The Madison Declaration on Mercury Pollution", *Ambio*, 36, 1, 62-65.

Madsen, E. and Stern, H. (2007). "Time Trends of Methylmercury in Walleye in Northern Wisconsin: A Hierarchical Bayesian Analysis", *Environmental Science and Technology*, 41, 13, 4568-4573.

Manolopoulos, H., et. al. (2006), "Local and Regional Factors Affecting Atmospheric Mercury Speciation at a Remote Location", *Journal of Environmental Engineering and Science*, 6, 5, 491-501.

McCurry, J. (2006). "Japan Remembers Minamata", *The Lancet*, 397, 99-100.

McKelvey, W., et. al. (2007). "A Biomonitoring Study of Lead, Cadmium, and Mercury in the Blood of New York City Adults", Environmental Health Perspectives, July 23, 2007, 10056, 1-38.

MDN (n.d.). "Mercury Deposition Network: a NADP Network", NADP web page, not dated, obtained from the internet on January 21, 2007 from http://nadp.sws.uiuc.edu/mdn/.

Mergler, D., et. al. (2007). "Methylmercury Exposure and Health Effects in Humans: A Worldwide Concern", *Ambio*, 36, 1, 3-11.

Miller, E. K., et. al. (2005). "Estimation and Mapping of Wet and Dry Mercury Deposition Across Northeastern North America", *Ecotoxicology*, 14, 53-70.

Montgomery, W. D. (1972). "Markets in Licenses and Efficient Pollution Control Programs", *Journal of Economic Theory*, 5, 3, 395-418.

Morris, J. (2002). "The Relationship Between Risk and the Precautionary Principle", *Toxicology*, 181-182, 127-130.

Munthe, J. (2007). "Recovery of Mercury-Contaminated Fisheries", *Ambio*, 31, 1, 33-44.

NACAA, (2007). "State Mercury Control Programs", National Association of Clean Air Agencies web page, dated December 4, 2007, obtained from the internet on April 13, 2008 from http://www.4cleanair.org/Documents/StateTable.pdf.

New York City Department of Health and Mental Hygiene. (2007). "Eat Fish, Choose Wisely, Protect Against Mercury", Brochure dated June 2007, available at http:// nyc.gov.health.

New York State Attorney General Andrew M. Cuomo (2006). "Coalition of 16 States File Lawsuit Challenging Final EPA Rules That Will Perpetuate Dangerous Mercury Hotspots", web page dated June 19, 2006, obtained from the internet on December 23, 2007 from http://www.oag.state.ny.us/press/2006/jun/jun19a_06.html.

NOAA Satellite and Information Service (2005), "Climate Maps of the United States", web page dated June 24, 2005, obtained on the internet on September 20, 2007 from http://cdo.ncdc.noaa.gov/climaps/wnd60a13.pdf.

Northeast Waste Managers Association (NEWMOA). (n.d.). "The Mercury Cycle", web page, not dated, obtained from the Internet on December 27, 2007 from http://www.newmoa.org/prevention/topichub/22/mercury_cycle.pdf.

Oken, E. et. al. (2005). "Maternal Fish Consumption, Hair Mercury, and Infant Cognition in a U. S. Cohort", *Environmental Health Perspectives*, 113, 10, 1376-1380.

Palmer, K., et. al. (2007). "The Benefits and Costs of Reducing Emissions from the Electricity Sector", *Journal of Environmental Management*, 83, 1, 115-130.

Pavlish, J. H. et. al. (2003). "Status Review of Mercury Control Options for Coal-fired Power Plants", *Fuel Processing Technology*, 82, 2-3, 89-165.

Pennsylvania Department of Environmental Protection (2005), "Data Collected Over Eight Years Show Mercury Levels 47% Higher in Areas Near Power Plants", web page dated May 31, 2005 obtained from the Internet on March 17, 2008 from http://www.depweb.state.pa.us/news/cwp/ view.asp?a=3&q=507034.

Pirrone, N. et. al. (1998). "Historical Atmospheric Mercury Emissions and Depositions in North America Compared to Mercury Accumulations in Sedimentary Records", *Atmospheric Environment*, 32, 5, 928-940.

Rassmussen, P. (1994). "Current Methods of Estimating Atmospheric Mercury Fluxes in Remote Areas", *Atmospheric Science and Technology*, 28, 13, 2233-2241.

Renner, R. (2004). "Asia Pumps Out More Mercury than Previously Thought", *Environmental Science and Technology*, March 1, 2005.

Roy, B., et. al. (2007). "Refining Fire Emissions for Air Quality Modeling with Remotely Sensed Fire Counts: A Wildfire Case Study", *Atmospheric Environment*, 41, 3, 655-665.

Ryaboshapko, A. et. al. (2007). Intercomparison Study of Atmospheric Mercury Models: 1. Comparison of Models with Short-Term Measurements", *Science of the Total Environment*, 376, 228-240)

Sacramento River Watershed Program, (2002), "Strategic Plan for the Reduction of Mercury-Related Risk in the Sacramento River Watershed", obtained from the internet on January 7, 2007 from http://www.sacriver.org/subcommittees/dtmc/documents/DTMCMercuryStrategyPlan.pdf.

Schakenback, J. et. al., U.S. EPA. (2006). "Fundamentals of Successful Monitoring, Reporting, and Verification under a Cap-and-Trade Program", *Journal of the Air and Waste Management Association*, 56, 1576-1583.

Schoen, D., (2004). "Exceeding the Methylmercury Reference Dose: How Dangerous is it?", *Environmental Health Perspectives* – (correspondence), 112, 6, A337.

Schroeder, W. H., and J. Munthe. (1998). "Atmospheric Mercury – An Overview", *Atmospheric Environment*, 32, 5, 809-822.

Seigneur, C., et. al. (2004). "Global Source Attribution for Mercury Deposition in the United States", *Environmental Science and Technology,* 38, 555-569.

Shea, Q. (2004). "Mercury Control Requires Flexibility", *Power*, 148, 8, 30.

Shore, M. (2003). "Out of Control and Close to Home: Mercury Pollution from Power Plants", Environmental Defense, obtained on the internet on January 7, 2007 from http://www.environmentaldefense.org/documents/3370_MercuryPowerPlants.pdf.

Sierra Club, (n.d.). "The Dirty Truth About Coal", Sierra Club web page not dated obtained from the internet on January 13, 2008 from http://sierraclub.org/coal/dirtytruth/.

Sigler, J. M. and X. Lee, (2006). "Recent Trends in Anthropogenic Mercury Emissions in the Northeast United States", *Journal of Geophysical Research*, 110, D14316, 1-11.

Slemr, F., et. al. (2003). "Woldwide trend of atmospheric mercury since 1977", *Geophysical Research Letters*, 30, 10, 23-1 - 23-4.

Srivastava, R. K. et. al. (2006). "An Overview of the Status of Mercury Control Technologies", *Environmental Science and Technology*, March 1, 2006, 1385-1393.

Sullivan, T. M. et. al. (2003). "The Local Impacts of Mercury Emissions from Coal Fired Power Plants on Human Health Risk- Progress Report for the Period of March 2002 – March 2003", Brookhaven National Laboratory under contract for the U.S. Department of Energy, May2003.

Sullivan, T. M. et. al. (2005). "Mercury Emissions From Coal-fired Power Plants: Local Impacts on Human Health Risk", Brookhaven National Laboratory, December 2005.

TerraNature, (2007). "Global Warming: Climate Change Becomes a Campaign Issue in the U.S. Democrat Race for the White House", TerraNature web page dated March 26, 2007 obtained from the internet on January 13, 2008 from http://www.terranature.org/coalPlants_Edwards.htm.

Tietenberg, T. H. (2006). *Emissions Trading: Principles and Practice*, Second Edition, Resources for the Future: Washington, D.C.

Tietenberg, T.H. (2003). *Environmental and Natural Resource Economics*, Sixth Edition, Addison Wesley, Boston.

Tolme, P. (2005). "Mercury Rising", *National Wildlife*, 43, 5, 33-37.

Travnikov, O. (2005). "Contribution of the Intercontinental Atmospheric Transport to Mercury Pollution in the Northern Hemisphere", *Atmospheric Environment*, 39, 7541-7548.

Turetsky, M. R. et. al. (2006). "Wildfires Threaten Mercury Stocks in Northern Soils", *Geophysical Research Letters*, 33, L16403, 1-6.

University of Illinois. (2007). GeoDa: An Introduction to Spatial Data Analysis", University of Illinois Spatial Analysis Lab web page obtained from the internet on November 25, 2007 from https://www.geoda.uiuc.edu/.

USDA Forest Service (2007), "Continental United States Fire Detection GIS Data", web page not dated but updated daily, obtained on the internet on October 1, 2007 from http://activefiremaps.fs.fed.us/gisdata.php.

U.S. Census Bureau (2007), "United States Census 2000", web page dated October 27, 2007 obtained on the internet from http://www.census.gov/main/www/cen2000.html.

U.S. Court of Appeals (2007). "State of New Jersey, et. al. versus the Environmental Protection Agency",U.S.Court of Appeals for the District of Columbia Circuit opinion dated February 8, 2008. No 05-1097.

U.S. Department of Energy (U.S. DOE) (2006). "Mercury Emission Control R & D", U.S. DOE web page dated January 18, 2006, obtained on the internet on April 6, 2008 from C:\Documents and Settings\Owner\My Documents\Dissertation\Mar 2008\DOE - Fossil Energy Mercury Emission Control R&D.htm.

U.S. EPA (1997). "An SAB (Science Advisory Board) Report: Review of the EPA Draft Mercury Study Report to Congress", Report Number EPA-SAB-EC-98-001, October 1997.

U.S. EPA (2001). "The Emergency Planning and Community Right-To-Know Act: Section 313 Release and Other Waste Management Reporting Requirements", Brochure number EPA 260/K-01-001, Office of Environmental Information, February 2001.

U.S. EPA (2005). "Fact Sheet – EPA's Clean Air Mercury Rule", U.S. EPA web page dated March 15, 2005, obtained on the internet on January 7, 2007 from http://www.epa.gov/air/mercuryrule/factsheetfin.htm.

U.S. EPA (2006a). "Controlling Power Plant Emissions: Overview", U.S. EPA web page dated November 27, 2006, obtained on the internet on January 14, 2007 from http://www.epa.gov/mercury/control_emissions/index.htm.

U.S. EPA (2006b). "EPA's Roadmap for Mercury", Report Number EPA-HQ-OPPT-2005-00013, obtained on the internet on January 7, 2007 from http://www.epa.gov/mercury/pdfs/FINAL-Mercury-Roadmap-6-29.pdf.

U.S. EPA (2006c). "Controlling Power Plant Emissions: Decision Process and Chronology", U.S. EPA web page dated November 27, 2006, obtained on the internet on January 14, 2007 from http://www.epa.gov/mercury/control_emissions/decision.htm.

U.S. EPA (2006d). "Basic Information", U.S. EPA web page dated November 27, 2006, obtained on the internet on January 14, 2007 from http://www.epa.gov/mercury/about.htm.

U.S. EPA (2007a), "TRI Data Quality Program Information", U.S. EPA web page dated September 27, 2007, obtained on the internet on November 25, 2007 from http://www.epa.gov/tri/tridata/data_quality_reports/index2.htm.

U.S. EPA (2007b), "Toxics in Your Community", U.S. EPA web page, not dated, obtained from the internet on November 25, 2007 from http://yosemite.epa.gov/R10/OWCM.NSF/webpage/Toxics+in+Your+Community+(TRI).

U.S. EPA (2007c), "Enforcement and Compliance History Online", U.S. EPA web page dated November 21, 2007, obtained from the internet on November 25, 2007 from http://www.epa-echo.gov/echo/index.html.

U.S. EPA (2007d). "Contact Us", web page dated December 18, 2007 obtained from the Internet on December 27, 2007 from http://www.epa.gov/epahome/comments2.htm.

U.S. EPA (2007e). "International TRI", web page dated September 21, 2007 obtained from the Internet on February 19, 2008 from http://www.epa.gov/tri/programs/prtrs.htm.

U.S. EPA (2007f). "Clean Air Markets: Cap and TradeI", web page dated August 14, 2007 obtained from the Internet on April 9, 2008 from http://www.epa.gov/AIRMARKET/cap-trade/.

U.S. EPA Office of Inspector General (2006). "Evaluation Report: Monitoring Needed to Address Impact of EPA's Clean Air Mercury Rule on Potential Hotspots", Report Number 2006-P-00025, May 15, 2006.

U.S. Government Accountability Office (GAO) (2005). "Clean Air Act: Emerging Mercury Control Technologies Have Shown Promising Results, but Data on Long-Term Results are Limited", Report Number GAO-05-612, May 2005.

Walcek, C. et. al. (2003). "Preparation of Mercury Emissions Inventory for Eastern North America", *Environmental Pollution*, 123, 3, 375-381.

Wedig, C., et. al. (2008). "Future of National Mercury Rule Now Uncertain", *Power*, 152, 5, 26-30.

Wiedinmyer, C. et. al. (2006). "Estimating Emissions from Fires in North America for Air Quality Modeling", *Atmospheric Environment*, 40, 19, 3419-3432.

Xue, F. et. al. (2007). "Maternal Fish Consumption, Mercury Levels, and Risk of Preterm Delivery", *Environmental Health Perspectives*, 115, 42-47.

Yujin, E. et. al. (2008). "Heterogeneous Mercury Reaction on a Selective Catalytic Reduction (SCR) Catalyst", *Catalyst Letters*, 121, 3/4, 219-225.

APPENDICIES

Appendix 1. Summary of Geoda Output for 60 Initial Models, 25 and 50 Mile Radius

	MODEL #	1A	2A	3A	4A	5A	6A	1	2	3	4	5	6
	DEP VAR	CONC	CONC	CONC	DEP	DEP	DEP	CONC	CONC	CONC	DEP	DEP	DEP
	RADIUS	25	25	25	25	25	25	50	50	50	50	50	50
	MODEL	OLS	LAG	ERROR	OLS	LAG	ERROR	OLS	LAG	ERROR	OLS	LAG	ERROR
Panel	F-Stat	**30.9**			**44.8**			**29.7**			**42.8**		
Data	Adj. R^2				0.740			0.652			0.731		
with	log-like	-590	-560	-562	-666	-619	-619	-594	-559	-558	-671	-618	-615
Wind	SIC10LND											*1.85*	
	SIC49AIR	**2.75**	2.54	2.02	**2.86**		*1.84*	**3.01**	**3.51**	**3.49**	**2.60**		2.51
	SICOTAIR												
	POP75	**2.95**	**3.48**	2.57	**3.05**	**2.87**	2.41	**3.59**	**4.04**	**2.76**	**3.54**	**2.85**	2.32
	FIREAREA	**4.45**	**3.21**	*1.86*	**3.35**			**3.03**	2.34				-2.33
	INVP/PRECIP	**17.2**	**8.64**	**9.60**	**16.2**	**13.7**	**15.4**	**16.3**	**8.26**	**9.14**	**15.5**	**13.7**	**15.8**
	D2001	**4.02**	**4.45**	**3.93**	**3.24**	**2.66**	2.53	**3.86**	**4.45**	**3.96**	**3.01**	**2.62**	2.42
	D2002	*1.83*	2.23	*1.74*				*1.80*		*1.81*			
	D2003	**4.52**	**5.55**	**5.61**	**3.65**	**4.52**	**4.55**	**4.22**	**5.40**	**5.60**	**3.49**	**4.71**	**4.83**
	D2004				*1.89*	**2.68**	2.28					2.55	2.21
	DREG1	**-3.22**		-2.41	**-2.99**			**-3.54**		**-2.61**	**-3.36**		
	DREG2	*-1.67*		*-1.91*	*-1.73*			*-1.93*		-2.20	-2.01		
	DREG3									*-1.49*			
	DREG4	**5.25**	**3.32**		**5.94**	**2.94**	2.01	**5.07**	**2.96**		**5.85**	**2.92**	
	DREG5	**4.87**	**3.45**	*1.75*	**3.91**	**4.62**	2.14	**4.18**	**2.83**		**3.12**	**4.29**	2.00
	DREG6	**5.24**	**3.99**	**3.40**	**4.67**	**3.64**	**3.26**	**5.02**	**3.76**	**3.16**	**4.51**	**3.68**	**3.48**
	DREG7	*1.96*	*1.78*				2.05	*1.73*					2.11
	DREG8					**2.97**						**2.82**	
	DREG9	**-6.58**	**-4.39**	-2.05		*1.67*		**-5.77**	**-3.73**	*-1.76*		2.04	
Panel	F-Stat	**31.8**			**45.8**			**30.5**			**43.4**		
Data	Adj. R^2	0.667			0.745			0.658			0.734		
No	log-like	-588	-559	-561	-663	-621	-619	-592	-558	-558	-669	-620	-616
Wind	SIC10LND											*1.79*	
	SIC49AIR	**2.64**	2.33	1.97	**2.61**		*1.80*	**2.66**	**3.16**	**3.32**	2.20		2.52
	SICOTAIR	*1.83*	*1.65*										
	POP75	**3.04**	**3.59**	**2.70**	**2.98**	**2.62**	2.29	**3.43**	**3.90**	**2.68**	**3.23**	**2.60**	2.02
	FIREAREA	**4.64**	**3.36**	*1.71*	**3.82**			**4.06**	**3.18**		2.58		-2.15
	INVP/PRECIP	**17.5**	**8.97**	**9.60**	**16.4**	**13.7**	**15.4**	**16.6**	**8.70**	**9.25**	**15.3**	**13.8**	**15.8**
	D2001	**4.02**	**4.43**	**3.87**	**3.28**	**2.65**	2.43	**3.98**	**4.51**	**3.90**	**3.09**	2.54	2.22
	D2002	*1.81*	2.19	*1.71*				*1.84*	2.27	*1.76*			
	D2003	**4.57**	**5.57**	**5.64**	**3.72**	**4.53**	**4.58**	**4.07**	**5.24**	**5.49**	**3.29**	**4.53**	**4.53**
	D2004				2.03	**2.72**	2.28				*1.73*	2.50	2.16
	DREG1	**-2.81**		-2.30	-2.54			**-2.67**		-2.41	-2.50		
	DREG2			*-1.83*						-2.01			
	DREG3									*-1.68*			
	DREG4	**5.59**	**3.69**		**6.19**	**3.08**	2.25	**5.59**	**3.48**		**6.28**	**2.88**	
	DREG5	**5.26**	**3.80**	*1.92*	**4.36**	**4.74**	2.39	**4.85**	**3.44**		**3.80**	**4.28**	2.24
	DREG6	**5.18**	**3.94**	**3.59**	**4.62**	**3.66**	**3.64**	**4.81**	**3.53**	**3.20**	**4.34**	**3.62**	**3.58**
	DREG7	2.19	**1.98**	*1.88*	*1.79*		2.20	*1.94*	*1.72*	*1.69*			2.39
	DREG8					**3.07**						**3.04**	
	DREG9	**-6.61**	**-4.48**	-2.01				**-6.17**	**-4.08**	*-1.80*		*1.77*	

Note: 1. The convention used here for t statistics (OLS regressions) and z statistics (spatial regressions), is italic text for p<0.10, normal text for p<0.05, bold text for p<0.01.

Appendix 1. Summary of Geoda Output for 60 Initial Models, 75 and 100 Mile Radius

	MODEL #	7	8	9	10	11	12	13	14	15	16	17	18
	DEP VAR	CONC	CONC	CONC	DEP	DEP	DEP	CONC	CONC	CONC	DEP	DEP	DEP
	RADIUS	75	75	75	75	75	75	100	100	100	100	100	100
	MODEL	OLS	LAG	ERROR	OLS	LAG	ERROR	OLS	LAG	ERROR	OLS	LAG	ERROR
Panel	F-Stat	**30.0**			**42.3**			**28.9**			**42.8**		
Data	Adj. R^2	0.646			0.729			0.643			0.731		
with	log-like	-597	-563	-564	-672	-619	-620	-598	-562	-563	-671	-618	-620
Wind	SIC10LND					2.38						2.56	
	SIC49AIR										*1.68*		
	SICOTAIR												
	POP75	**3.65**	**4.05**	**2.81**	**3.68**	**2.82**	2.38	**3.70**	**4.10**	**2.86**	**3.70**	**2.89**	2.37
	FIREAREA	**3.44**	**2.77**		2.09			**2.96**	**2.89**		2.32		
	INVP/PRECIP	**15.6**	**8.05**	**9.03**	**15.3**	**13.9**	**15.1**	**15.7**	**8.18**	**9.05**	**14.9**	**14.1**	**15.2**
	D2001	**3.99**	**4.48**	**3.89**	**3.14**	**2.63**	2.20	**3.97**	**4.57**	**3.89**	**3.26**	2.52	2.11
	D2002	2.00	2.38	*1.80*				2.08	2.49	*1.85*			
	D2003	**4.17**	**5.30**	**5.42**	**3.46**	**4.56**	**4.58**	**4.28**	**5.37**	**5.44**	**3.59**	**4.63**	**4.53**
	D2004					2.42	2.18				*1.68*	2.37	2.19
	DREG1	**-2.93**		-2.33	**-2.71**			**-2.80**		-2.27	-2.43		
	DREG2			*-1.90*	*-1.65*					*-1.86*			
	DREG3												
	DREG4	**5.25**	**3.14**		**6.15**	**3.29**	1.99	**5.15**	**3.07**		**6.20**	**3.16**	2.06
	DREG5	**4.68**	**3.27**		**3.74**	**4.91**	2.34	**4.35**	**3.30**		**3.62**	**4.57**	2.30
	DREG6	**4.90**	**3.65**	**2.99**	**4.50**	**3.84**	**3.34**	**4.65**	**3.40**	**2.86**	**4.32**	**3.85**	**3.35**
	DREG7	*1.75*				*1.66*	2.27	*1.75*				*1.82*	2.24
	DREG8					**3.51**						**3.36**	
	DREG9	**-5.62**		*-1.83*		*1.95*		**-5.69**	**-3.68**	*-1.83*		1.98	
Panel	F-Stat	**28.4**			**42.4**			**30.5**			**45.4**		
Data	Adj. R^2	0.641			0.729			0.658					
No	log-like	-599	-567	-568	-672	-621	-621	-592	-557	-561	-664	-619	-621
Wind	SIC10LND					*1.89*						2.22	
	SIC49AIR												
	SICOTAIR												
	POP75	**3.56**	**3.96**	**2.72**	**3.46**	2.54	2.06	**3.59**	**3.99**	**2.72**	**3.56**	**2.72**	2.07
	FIREAREA	**4.14**	**3.19**		**3.53**			**4.67**	**4.43**	2.32	**4.56**		
	INVP/PRECIP	**16.0**	**8.45**	**9.09**	**14.6**	**13.7**	**14.9**	**16.4**	**9.01**	**9.17**	**15.1**	**14.0**	**15.2**
	D2001	1.98	2.18	*1.94*				**4.48**	**5.04**	**4.19**	**3.87**	**2.61**	2.32
	D2002							2.43	**2.85**	2.11	*1.88*		
	D2003	**2.91**	**3.92**	**4.40**	2.40	**3.86**	**4.03**	**4.24**	**5.34**	**5.41**	**3.54**	**4.53**	**4.38**
	D2004					*1.74*	*1.67*		*1.66*		2.04	2.43	2.28
	DREG1	-2.15		-2.30	*-1.80*					-2.07			
	DREG2			*-1.90*						*-1.68*			
	DREG3												
	DREG4	**5.63**	**3.49**		**6.53**	**3.14**	*1.95*	**5.98**	**3.95**		**6.87**	**3.10**	2.19
	DREG5	**4.97**	**3.44**		**4.21**	**4.54**	2.35	**5.53**	**4.45**	2.02	**5.01**	**4.49**	2.38
	DREG6	**4.58**	**3.33**	**2.96**	**4.27**	**3.80**	**3.30**	**4.45**	**3.21**	**2.81**	**4.09**	**3.86**	**3.20**
	DREG7	*1.76*					2.19	2.00	*1.72*		*1.67*	*1.73*	2.03
	DREG8					**3.36**					*1.69*	**3.42**	
	DREG9	**-5.89**	**-3.99**	*-1.83*		*1.76*			**-3.75**	*-1.87*		*1.90*	

Note: 1. The convention used here for t statistics (OLS regressions) and z statistics (spatial regressions), is italic text for $p<0.10$, normal text for $p<0.05$, bold text for $p<0.01$.

Appendix 1. Summary of Geoda Output for 60 Initial Models, 500 Mile Radius

	MODEL #	19	20	21	22	23	24
	DEP VAR	CONC	CONC	CONC	DEP	DEP	DEP
	RADIUS	500	500	500	500	500	500
	MODEL	OLS	LAG	ERROR	OLS	LAG	ERROR
Panel	F-Stat	**27.9**			**42.4**		
Data	Adj. R^2	0.637			0.729		
with	log-like	-601	-566	-563	-672	-618	-621
Wind	SIC10LND	*-1.78*			*-1.75*		
	SIC49AIR			*1.69*			
	SICOTAIR						
	POP75	**3.46**	**4.03**	**2.93**	**3.56**	**2.85**	2.54
	FIREAREA						
	INVP/PRECIP	**15.4**	**7.40**	**8.52**	**15.7**	**14.0**	**15.1**
	D2001	2.36	**3.54**	**3.48**	2.27	2.46	2.55
	D2002		1.97	*1.83*			
	D2003	**2.99**	**4.73**	**5.08**	*1.95*	**2.85**	**3.08**
	D2004					**2.63**	2.39
	DREG1	**-4.95**	-2.08	-2.14	**-4.07**		
	DREG2	**-2.90**	*-1.67*	*-1.81*	-2.28		
	DREG3	-2.17					
	DREG4	**4.28**	2.37		**5.38**	**3.27**	*1.94*
	DREG5	2.19			2.56	**4.67**	2.37
	DREG6	4.47	**3.26**	**2.69**	**4.17**	**3.91**	**3.04**
	DREG7	2.02			*1.72*	*1.83*	*1.71*
	DREG8					**3.12**	
	DREG9	**-6.09**	**-3.74**				
Panel	F-Stat	**28.7**			**40.7**		
Data	Adj. R^2	0.643			0.721		
No	log-like	-598	-564	-563	-676	-618	-621
Wind	SIC10LND	**-4.02**	*-1.96*		**-4.02**		
	SIC49AIR						
	SICOTAIR						
	POP75	**3.61**	**4.12**	**2.88**	**3.55**	2.47	2.46
	FIREAREA						
	INVP/PRECIP	**15.4**	**7.64**	**8.66**	**15.2**	**14.0**	**14.7**
	D2001	**2.81**	**3.88**	**3.32**	**2.97**	*1.82*	2.17
	D2002		2.33	*1.83*			
	D2003	2.50	**4.20**	**4.44**		2.54	**2.68**
	D2004				*1.90*	2.37	2.23
	DREG1	**-4.41**	*-1.90*	-2.15	**-3.03**	-2.14	
	DREG2	-2.59		*-1.84*			
	DREG3						
	DREG4	**4.10**	2.47		**5.50**	**2.67**	2.06
	DREG5	2.24	*1.74*		**3.08**	**3.74**	2.32
	DREG6	**4.07**	**2.86**	**2.77**	**3.73**	**3.96**	**3.13**
	DREG7	*1.77*				2.11	*1.89*
	DREG8					**2.93**	
	DREG9		**-2.63**		*1.82*		

Note: 1. The convention used here for t statistics (OLS regressions) and z statistics (spatial regressions), is italic text for $p<0.10$, normal text for $p<0.05$, bold text for $p<0.01$.

Appendix 2. OLS Model, 25 mile radius, Concentration as DV, No Wind

Model 1A. REGRESSION
SUMMARY OF OUTPUT: ORDINARY LEAST SQUARES ESTIMATION

Data set	:	**Panel_Data_WI25Miles_NoWind**			
Dependent Variable	:	**"CONC"**	Number of Observations:		293
Mean dependent var	:	10.0584	Number of Variables	:	20
S.D. dependent var	:	3.22296	Degrees of Freedom	:	273
R-squared	:	0.688458	F-statistic	:	31.752
Adjusted R-squared	:	0.666776	Prob(F-statistic)	:	0
Sum squared residual:		948.187	Log likelihood	:	-587.795
Sigma-square	:	3.47321	Akaike info criterion	:	1215.59
S.E. of regression	:	1.86366	Schwarz criterion	:	1289.19
Sigma-square ML	:	3.23613			
S.E of regression ML:		1.79893			

Variable	Coefficient	Std.Error	t-Statistic	Probability
CONSTANT	4.361644	0.5151547	8.46667	0.0000000
"SIC10LND"	3.415137e-005	0.0003377665	0.1011094	0.9195530
"SIC49AIR"	0.0007923332	0.0002997825	2.643027	0.0086916
"SICOTAIR"	0.0008402122	0.0004583269	1.833216	0.0678596
"POP75"	0.1153247	0.03788612	3.043982	0.0025625
"FIREAREA"	0.004138451	0.0008926286	4.636252	0.0000055
"INVPRECP"	2539.655	145.2487	17.48486	0.0000000
"D2001"	1.419602	0.3530606	4.020844	0.0000751
"D2002"	0.6484636	0.3574159	1.814311	0.0707273
"D2003"	1.500331	0.328327	4.569625	0.0000074
"D2004"	0.4820099	0.3170248	1.520417	0.1295632
"DREG1"	-1.589534	0.5651849	-2.812415	0.0052739
"DREG2"	-1.226171	0.8640099	-1.419164	0.1569912
"DREG3"	-0.05924487	0.5204133	-0.113842	0.9094704
"DREG4"	2.498675	0.4466253	5.594567	0.0000001
"DREG5"	2.233042	0.4245233	5.260117	0.0000003
"DREG6"	1.209393	0.2332794	5.184312	0.0000004
"DREG7"	3.031458	1.381725	2.193966	0.0290807
"DREG8"	-0.2779877	0.7024442	-0.3957435	0.6926053
"DREG9"	-5.859828	0.886064	-6.613324	0.0000000

REGRESSION DIAGNOSTICS
MULTICOLLINEARITY CONDITION NUMBER 11.98041
TEST ON NORMALITY OF ERRORS

TEST	DF	VALUE	PROB
Jarque-Bera	2	21.81091	0.0000184

DIAGNOSTICS FOR HETEROSKEDASTICITY
RANDOM COEFFICIENTS

TEST	DF	VALUE	PROB
Breusch-Pagan test	19	92.64639	0.0000000
Koenker-Bassett test	19	56.41672	0.0000141

SPECIFICATION ROBUST TEST

TEST	DF	VALUE	PROB
White	209	N/A	N/A

DIAGNOSTICS FOR SPATIAL DEPENDENCE
FOR WEIGHT MATRIX : **Weight25NW.GWT** (row-standardized weights)

TEST	**MI/DF**	**VALUE**	**PROB**
Moran's I (error)	0.152201	11.6271344	0.0000000
Lagrange Multiplier (lag)	1	87.6034738	0.0000000
Robust LM (lag)	1	41.9131239	0.0000000
Lagrange Multiplier (error)	1	54.5963441	0.0000000
Robust LM (error)	1	8.9059942	0.0028424
Lagrange Multiplier (SARMA)	2	96.5094680	0.0000000

========== END OF REPORT ==========

Appendix 2. Spatial Lag Model, 25 mile radius, Concentration as DV, No Wind

Model 2A. REGRESSION
SUMMARY OF OUTPUT: SPATIAL LAG MODEL - MAXIMUM LIKELIHOOD ESTIMATION

Data set	:	**Panel_Data_WI25Miles_NoWind**			
Spatial Weight	:	**Weight25NW.GWT**			
Dependent Variable	:	**"CONC"**	Number of Observations:		293
Mean dependent var	:	10.0584	Number of Variables	:	21
S.D. dependent var	:	3.22296	Degrees of Freedom	:	272
Lag coeff. (Rho)	:	0.477904			
R-squared	:	0.749520	Log likelihood	:	-558.516
Sq. Correlation	:	-	Akaike info criterion	:	1159.03
Sigma-square	:	2.60186	Schwarz criterion	:	1236.31
S.E of regression	:	1.61303			

Variable	Coefficient	Std.Error	z-value	Probability
W_"CONC"	0.4779039	0.06372229	7.499792	0.0000000
CONSTANT	1.191937	0.6513124	1.830054	0.0672417
"SIC10LND"	8.05422e-006	0.0002923978	0.02754542	0.9780246
"SIC49AIR"	0.0006059908	0.0002596883	2.333531	0.0196202
"SICOTAIR"	0.0006545622	0.0003968308	1.649474	0.0990505
"POP75"	0.1179232	0.03280814	3.594328	0.0003253
"FIREAREA"	0.002615989	0.0007778617	3.363052	0.0007710
"INVPRECP"	1597.847	178.0634	8.973471	0.0000000
"D2001"	1.355235	0.3056002	4.434666	0.0000092
"D2002"	0.6776114	0.3095511	2.189013	0.0285958
"D2003"	1.583069	0.2844061	5.566226	0.0000000
"D2004"	0.4101412	0.2744413	1.494459	0.1350558
"DREG1"	-0.5466774	0.5211247	-1.049034	0.2941626
"DREG2"	-0.6103094	0.7570016	-0.8062194	0.4201162
"DREG3"	0.05758601	0.4535672	0.1269625	0.8989700
"DREG4"	1.459441	0.3953619	3.691405	0.0002231
"DREG5"	1.425105	0.3751584	3.798675	0.0001455
"DREG6"	0.814099	0.2065776	3.940888	0.0000812
"DREG7"	2.370504	1.19867	1.977612	0.0479724
"DREG8"	0.03272814	0.6137358	0.05332611	0.9574719
"DREG9"	-3.700904	0.8266325	-4.477085	0.0000076

REGRESSION DIAGNOSTICS
DIAGNOSTICS FOR HETEROSKEDASTICITY
RANDOM COEFFICIENTS

TEST	**DF**	**VALUE**	**PROB**
Breusch-Pagan test	19	116.4674	0.0000000

DIAGNOSTICS FOR SPATIAL DEPENDENCE
SPATIAL LAG DEPENDENCE FOR WEIGHT MATRIX : **Weight25NW.GWT**

TEST	**DF**	**VALUE**	**PROB**
Likelihood Ratio Test	1	58.55987	0.0000000

========== END OF REPORT ==========

Appendix 2. Spatial Error Model, 25 mile radius, Concentration as DV, No Wind

Model 3A. REGRESSION
SUMMARY OF OUTPUT: SPATIAL ERROR MODEL - MAXIMUM LIKELIHOOD ESTIMATION

Data set	: **Panel_Data_WI25Miles_NoWind**		
Spatial Weight	: **Weight25NW.GWT**		
Dependent Variable	: **"CONC"**	Number of Observations:	293
Mean dependent var	: 10.058362	Number of Variables	: 20
S.D. dependent var	: 3.222962	Degree of Freedom	: 273
Lag coeff. (Lambda)	: 0.740895		
R-squared	: 0.755584	R-squared (BUSE)	: -
Sq. Correlation	: -	Log likelihood	: -560.877881
Sigma-square	: 2.538866	Akaike info criterion	: 1161.76
S.E of regression	: 1.59338	Schwarz criterion	: 1235.359214

Variable	Coefficient	Std.Error	z-value	Probability
CONSTANT	6.508149	0.7116238	9.145491	0.0000000
"SIC10LND"	0.0001007641	0.0002648229	0.3804961	0.7035773
"SIC49AIR"	0.0005142701	0.0002607257	1.972457	0.0485574
"SICOTAIR"	0.0006087454	0.0004024905	1.512447	0.1304203
"POP75"	0.09353595	0.03464511	2.699831	0.0069376
"FIREAREA"	0.00135723	0.0007926938	1.712174	0.0868646
"INVPRECP"	1903.881	198.3199	9.60005	0.0000000
"D2001"	1.138066	0.2937542	3.874213	0.0001070
"D2002"	0.5058678	0.2959331	1.709399	0.0873769
"D2003"	1.532156	0.2716529	5.640122	0.0000000
"D2004"	0.3841541	0.2579255	1.4894	0.1363823
"DREG1"	-2.125622	0.9245913	-2.298985	0.0215057
"DREG2"	-1.841954	1.006307	-1.83041	0.0671885
"DREG3"	-0.9891232	0.8277129	-1.195008	0.2320842
"DREG4"	0.3380479	0.5499969	0.6146359	0.5387951
"DREG5"	1.49645	0.7778557	1.923814	0.0543778
"DREG6"	0.8627223	0.2403491	3.589455	0.0003314
"DREG7"	2.570647	1.367497	1.879819	0.0601326
"DREG8"	-0.6564279	2.055377	-0.3193711	0.7494453
"DREG9"	-5.26469	2.613924	-2.014095	0.0439995
LAMBDA	0.7408954	0.05503827	13.46146	0.0000000

REGRESSION DIAGNOSTICS
DIAGNOSTICS FOR HETEROSKEDASTICITY
RANDOM COEFFICIENTS

TEST	DF	VALUE	PROB
Breusch-Pagan test	19	172.3074	0.0000000

DIAGNOSTICS FOR SPATIAL DEPENDENCE
SPATIAL ERROR DEPENDENCE FOR WEIGHT MATRIX : **Weight25NW.GWT**

TEST	DF	VALUE	PROB
Likelihood Ratio Test	1	53.8352	0.0000000

========== END OF REPORT ==========

CIRICULUM VITAE

NAME:	Paul F. Tirey
ADDRESS:	5312 Barkwood Drive LaGrange, KY 40031
DOB:	Charleston, SC – September 8, 1961
EDUCATON & CERTIFICATION:	B. S. Mechanical Engineering University of Kentucky Lexington, KY 1983
	Master of Engineering, Engineering Management University of Louisville Louisville, KY 1994
	Ph. D. Candidate, Urban and Public Affairs University of Louisville Louisville, KY Expected Graduation – August 2008
	Licensed Professional Engineer, Kentucky - 1994 Certified Hazardous Materials Manager – 1994 Certified Internal Auditor - 2002 Certified Information Systems Auditor – 2004
PROFESSIONAL SOCIETIES:	Institute of Hazardous Materials Management Institute of Internal Auditors Information Systems and Control Association