Science to support toxics governance: tracking mercury and other pollutants from policy to impacts

by

Amanda Giang

B.A.Sc., University of Toronto (2011)
S.M., Massachusetts Institute of Technology, (2013)

Submitted to the Institute for Data, Systems and Society
in partial fulfillment of the requirements for the degree of

Doctor of Philosophy

at the

MASSACHUSETTS INSTITUTE OF TECHNOLOGY

June 2017

Author . . . .
Institute for Data, Systems and Society
May 10, 2017

Certified by . . . .
Noelle E. Selin
Associate Professor of Data, Systems and Society and Atmospheric Chemistry, MIT
Thesis Supervisor

Certified by . . . .
Lawrence Susskind
Ford Professor of Urban and Environmental Planning, MIT
Committee Member

Certified by . . . .
Elsie M. Sunderland
Associate Professor of Environmental Science and Engineering, Harvard
Committee Member

Accepted by . . . .
John N. Tsitsiklis
Clarence J. Lebel Professor of Electrical Engineering
IDSS Graduate Officer

Science to support toxics governance: tracking mercury and other pollutants from policy to impacts

by

Amanda Giang

Submitted to the Institute for Data, Systems and Society
on May 10, 2017, in partial fulfillment of the
requirements for the degree of
Doctor of Philosophy

Abstract

Persistent and bioaccumulative toxins like mercury pose unique challenges for environmental governance. The complexity of their movement through coupled social, technological, and natural systems can make it difficult to trace their path from emissions to wider impacts, as emissions and impacts can be separated both in time and space. This separation can make it difficult to assess whether different management and policy proposals will effectively reduce negative impacts. Focusing primarily on mercury, this dissertation explores how we can use interdisciplinary tools and approaches—from atmospheric modelling to community engaged research—to better trace this path from policy to human impacts, in support of environmental decision-making at multiple levels of governance. Combining simulation modelling, statistical, and qualitative approaches, it considers three aspects of the path from policy to impacts: how policy translates into emissions changes, how emissions changes translate into changes in environmental concentrations and fluxes, and finally how these environmental concentrations and fluxes impact the well-being of human communities. Taken together, the three studies highlight the need to take into account how social, technical, and natural systems interact, as well as the uncertainty, variability, and pluralism that exist within them, in our efforts to manage these toxic pollutants.

In the first study, I investigate the social and technical factors that affect the domestic implementation of a global environmental treaty (the United Nations Minamata Convention on Mercury) in major emitter countries in Asia, and their potential implications for emissions and global transport using a scenario-based modelling approach. I project that the benefit of avoided emissions and deposition over Asia are large, even when considering a scenario where the Convention allows large flexibility in implementation. These benefits are primarily driven by India, where even modest improvements in mercury capture are projected to result in large emissions decreases given future economic growth. I also find that climate change policies that promote the transitioning away from fossil may be as effective as strict end-of-pipe pollution control approaches for mitigating mercury emissions.

In the second study, driven by interests from community research partners in the Great Lakes region—an area vulnerable to mercury pollution—I use chemical transport modelling experiments to explore the conditions under which regional and global policy change can be statistically detected by wet deposition monitoring networks. I find that, given the magnitude of expected emissions decreases, detecting policy-related decreases in wet deposition in the Great Lakes region on the decadal scale will be challenging as the magnitude of noise—in particular interannual meteorological variability—can exceed this signal. These

results suggest that these variabilities need to be better quantified and taken into account in both the design of policies for effectiveness and evaluation of policy compliance.

In the third study, I investigate the role that university-community partnerships can play in the long-term management of persistent pollutants through an empirical case study of the Superfund Research Program, which has recently required that grantees engage communities impacted by the hazardous substances that they study. I argue that community engagement in practice often supports a community building function—engagement operates as a space where knowledge about pollutants and shared identities of being impacted by these pollutants can be co-produced. Because persistent pollutants can implicate new people across time and space, often in ways that are difficult for those affected to discern, I suggest that supporting the constitution of what I call communities of concern is a critical way that university-based researchers can support the long-term management of persistent pollutants. I propose a conceptual framework to characterize and assess the functions that academic partners can perform in supporting the constitution of communities of concern around persistent pollutants. Further, I call attention to the institutional conditions that can enable this work to continue within academic contexts.

Thesis Supervisor: Noelle E. Selin
Title: Associate Professor of Data, Systems and Society and Atmospheric Chemistry, MIT

Committee Member: Lawrence Susskind
Title: Ford Professor of Urban and Environmental Planning, MIT

Committee Member: Elsie M. Sunderland
Title: Associate Professor of Environmental Science and Engineering, Harvard

Acknowledgments

First and foremost, I would like to thank my advisor Noelle Selin. I've grown so much as a scholar through her mentorship, in ways that I could not have predicted when I started graduate school six years ago. She has been a guiding hand when I've needed it, while also giving me the space to explore and discover my interests. I feel so grateful to have had her as a role model for the sort of professor I hope to be one day.

I have also benefited immensely from my committee members, Larry Susskind and Elsie Sunderland. Their insights have always led me to think in new directions that I hadn't considered, and I think my work is much better for it. And beyond research, I've appreciated their sage advice on academic careers and enjoying life. I would also like to acknowledge my other academic mentors, including Ken Oye and Henrik Selin—their interest in my professional development has enabled me greatly.

Many thanks go to my interdisciplinary collaborators. It has been a delight to work with Leah Stokes, Bess Corbitt, David Streets, Shaojie Song, Marilena Muntean, Abby Harvey, Elizabeth Berg, and the ASEPs team at MTU, DRI, and NWIC. It has also been a great privilege to work with the many community partners in the ASEPs project. I also thank all the interviewees for sharing their time and insights. Without their generosity, this work would not have been possible.

I would like to thank the Selin Group and the Joint Program—members past and present—for creating such an intellectually stimulating and supportive space to do interdisciplinary research. These have been great homes to do exciting work, and to have fun (and pie-sized cookies) with friends and colleagues while doing it.

I'm so grateful to have shared this journey with the TPP/ESD/IDSS community, and in particular, the cohort that I started graduate school with. I'm so inspired by the work of my classmates and friends, and buoyed by everything else we do when we're not working, from book clubs to burgers and beers. I would also like to acknowledge all the support I've received from the ESD/IDSS staff, that has kept me on track over the years.

I'd like to express my gratitude to my other friends and family at and outside of MIT. I'm so privileged to have a support network of real superheroes that make me feel like I can do anything too. Having Sean's family nearby has been a pleasure and a constant source of support. I am forever indebted to my parents, who worked so hard to give me the opportunities to pursue my passions, and to my older brother Wayne, whom I've looked up to my whole life and who inspired my life as an engineer. Finally, I'd like to thank my partner Sean, who has supported me through this process with much grace, laundry, and the very best jokes. He has kept me smiling through all of it.

Contents

List of Figures

List of Tables

Chapter 1

Introduction

1.1 Motivation

Chemical pollution from human activities can perturb the Earth system in ways that ultimately damage human and ecosystem health, and managing this pollution through mitigation of emissions and remediation of existing contamination remains a critical challenge (Rockstrom et al., 2009; Diamond et al., 2015). However, when the environmental impacts of pollution are spatially and temporally separated from their sources—which is often the case for pollutants that are persistent—this challenge may be particularly difficult for both science and governance (Perlinger et al., 2016).

Mercury offers an illustrative example of these complexities. Mercury, especially in the form of methylmercury, is a potent toxin that can cause adverse neurologic and (though the relationship is more uncertain) cardiovascular effects (Karagas et al., 2012; Sundseth et al., 2017). These negative health effects can also have wider implications for individual and community wellbeing (Gagnon, 2016; Ranco et al., 2011), and the economy (Giang & Selin, 2016; Bellanger et al., 2013; Rice et al., 2010). As a result, preventing exposure to mercury has become an important policy goal internationally (UNEP, 2013a; Task Force on Hemispheric Transport of Air Pollution, 2010), within nations like the US (US EPA, 2011b), Canada (Steffen, 2016), and First and Tribal Nations (Gagnon, 2016), and within states and communities (Milford & Pienciak, 2009). However, the complexity of mercury's path through coupled socio-technical-ecological systems can make it difficult to trace mer-

cury's path from emissions to wider impacts, as emissions and impacts can be separated both in time and space, and over a wide range of temporal and spatial scales (Selin, 2011). This separation can make it difficult to assess whether different policy proposals will effectively reduce negative impacts. This dissertation asks how we can use scientific tools and approaches—from atmospheric modelling to community engaged research—to better trace this path from policy to human impacts, in support of environmental decision-making at multiple levels of governance.

Though mercury is a naturally occurring element, human activity has greatly increased its mobilization in the environment (Lindberg et al., 2007; Biester et al., 2007; Amos et al., 2013). Because mercury is highly persistent, once human activity extracts it from deep mineral reservoirs, it can continue to cycle through air, water, and land for centuries to millennia (Amos et al., 2013). Major anthropogenic sources of mercury emissions include fossil fuel combustion, metal smelting, and artisanal and small scale gold mining (UNEP, 2013a). The intersection of these activities with economic development mean that mercury emissions occur globally. Once emitted to the atmosphere, mercury can be transported on local to global scales depending on chemical form (Selin, 2009). This mercury can then be deposited into terrestrial and aquatic ecosystems. This deposited mercury enters a legacy pool, which continues to contribute to the global background concentration as it revolatizes and cycles through surface reservoirs (Amos et al., 2013; Selin, 2009). A fraction of mercury deposited in aquatic ecosystems is transformed into methylmercury, a highly toxic and bioaccumulative form. For many in the US and elsewhere in the world, the primary route of exposure to methylmercury is eating contaminated fish and shellfish (Oken et al., 2012), sourced both locally and from a global seafood market (Sunderland, 2007). Chronic, low-level exposure has been linked to cognitive deficits in children who are exposed *in utero* (NRC, 2000; Grandjean et al., 1997; Crump et al., 1998), and, a growing body of evidence suggests, increased risk of cardiovascular disease in adults (Roman et al., 2011).

Some communities are particularly affected by mercury pollution. For instance, the Great Lakes region has been a focus for mercury research because it experiences a large amount of mercury contamination (Evers et al., 2011b), and because it is home to several communities that are particularly vulnerable to mercury exposure through freshwater fish consumption, including recreational anglers, some indigenous peoples, subsistence fishers,

and other individuals for whom eating fish is culturally, spiritually, or socioeconomically important (Moya, 2004; Dellinger et al., 2012). Most mercury in the Great Lakes comes from atmospheric sources that are located not only in other regions in the US, but around the world (Cohen et al., 2016; Evers et al., 2011b; Cain et al., 2011). Mercury is therefore simultaneously a global and local problem—its sources and transport are global in scope, however its impacts are experienced and mediated through local conditions.

Although I have discussed only mercury in detail here, many of the attributes described for mercury's path from emissions to impacts—including diverse anthropogenic (and sometimes natural) emissions sources and the potential for short and long-range multi-media transport—are true of other persistent, bioaccumulative toxics such as persistent organic pollutants (Scheringer et al., 2009; Rodan et al., 1999; Farrington & Takada, 2014; Perlinger et al., 2016), and other metals like arsenic (Punshon et al., 2017; Nachman et al., 2017). How then, might we think about tracing the path that pollutants like this take from the the time and place of their production to their impacts on communities? How can interdisciplinary tools and approaches, from simulation modelling to community engaged research, be used to support this goal in ways that inform real-world environmental decision-making, whether at the local or global scale? In this dissertation, I consider these broader questions by focusing on three aspects of the path that pollutants might take from policy to impacts:

What does policy mean for emissions? Policy—particularly at the global level—seldom prescribes bright-line trajectories for emissions. How does policy translate from words on a page to changes in technologies and processes? How might this translation process differ between countries, and what are the implications of these processes in terms of emissions and global transport?

What do emissions mean for environmental concentrations and fluxes? To what extent are policy-related emissions changes translated into changes in environmental concentrations and fluxes of a pollutant at a specific place? Given the noisiness of real-world technical and natural systems, are policy efforts likely to be statistically detectable through monitoring, and if so, where and on what time-scale?

What do environmental concentrations and fluxes mean for communities? How do communities develop a shared identity of being impacted by a pollutant? Through what mechanisms might communities impacted by these pollutants—once these impacts are recognized—participate in their management? What roles might university-based re-

searchers play in supporting impacted communities?

1.2 Chapter Descriptions

Impacts of the Minamata Convention on mercury emissions and global deposition from coal-fired power generation in Asia

This chapter is adapted from Giang et al. (2015). We explore implications of the United Nations Minamata Convention on Mercury for emissions from Asian coal-fired power generation, and resulting changes to deposition worldwide by 2050. We use engineering analysis, document analysis, and interviews to construct plausible technology scenarios consistent with the Convention. We translate these scenarios into emissions projections for 2050, and use the GEOS-Chem model to calculate global mercury deposition. Where technology requirements in the Convention are flexibly defined, under a global energy and development scenario that relies heavily on coal, we project $\approx$ 90 and 150 Mg·y^{-1} of avoided power sector emissions for China and India respectively in 2050, compared to a scenario in which only current technologies are used. Benefits of this avoided emissions growth are primarily captured regionally, with projected changes in annual average gross deposition over China and India $\approx$ 2 and 13 μg· m^{-2} lower, respectively, than the current technology case. Stricter, but technologically feasible, mercury control requirements in both countries could lead to a combined additional 170 Mg·y^{-1} avoided emissions. Assuming only current technologies but a global transition away from coal avoids 6% and 36% more emissions than this strict technology scenario under heavy coal use for China and India, respectively.

Understanding factors influencing the detection of mercury policies in modelled Laurentian Great Lakes wet deposition

We use chemical transport modelling to better understand the extent to which policy-related anthropogenic mercury emissions changes (a policy signal) can be statistically detected in wet deposition measurements in the Great Lakes region on the decadal scale, given sources of noise. In our modelling experiment, we consider hypothetical regional (North American) and global (rest of the world) step policy changes, consistent with existing policy efforts, that divide an eight-year period (Δ_{global} = -13%; $\Delta_{regional}$ = -30%). The magnitude

of statistically significant (p<0.1) pre- and post-policy period wet deposition differences, holding all else constant except for the policy change, ranges from -0.4 to -2.4% for the regional policy and -0.9 to -2.5% for the global policy. We then introduce sources of noise—trends and variability in factors that are exogenous to the policy action—and evaluate the extent to which the policy signal can still be detected. We find that global trends in emissions of realistic magnitudes (for instance, due to economic activity) reduce the areas where the policy signal can be detected to the immediate vicinity of targeted emissions sources, as these global trends dominate elsewhere. We find that interannual variability in emissions magnitude and speciation can reduce the magnitude of wet deposition differences between periods (but not their significance) by up to 35% for the regional policy, and up to 80% for the global policy. Interannual variability in meteorology has the largest effect, driving wet deposition differences between periods ±20%, far exceeding the magnitude of the policy signal. These results highlight the potential challenges of detecting statistically significant policy-related changes in Great Lakes wet deposition within the short-term.

Creating and sustaining communities of concern for the long-term management of persistent pollutants

This chapter explores the role that university-community partnerships can play in the long-term management of persistent pollutants through an empirical case study of the Superfund Research Program (SRP), a research program with a mandate to support the management of (often persistent) hazardous substances with a new emphasis on working with and for communities affected by these substances. It argues that because persistent pollutants implicate new places and people across time and space, their management requires the continued constitution of new *communities of concern*. It observes that community engagement in practice often supports this community building role, helping to trace pollutants across space and time and bringing together those potentially affected along those dimensions as a community of shared identity and interest. Community engagement operates as a space where knowledge about pollutants and shared identities of being impacted by these pollutants can be co-produced. Further, the case study illustrates the active work that university-based researchers and community-organizers engage in to help build communities of concern, particularly when the reach of pollutants is large in time and space. I propose a conceptual framework for categorizing and assessing the roles that academic

partners in particular can play to support the constitution of these communities. Through the application of this framework to the SRP, the chapter identifies potential challenges that university-based researchers may face in fulfilling these functions and suggests that a better understanding of the institutional conditions that enable these researchers to participate in this work of creating and sustaining communities of concern is necessary for the improved management of persistent pollutants.

Chapter 2

Impacts of the Minamata Convention on mercury emissions and global deposition from coal-fired power generation in Asia

Abstract

We explore implications of the United Nations Minamata Convention on Mercury for emissions from Asian coal-fired power generation, and resulting changes to deposition worldwide by 2050. We use engineering analysis, document analysis, and interviews to construct plausible technology scenarios consistent with the Convention. We translate these scenarios into emissions projections for 2050, and use the GEOS-Chem model to calculate global mercury deposition. Where technology requirements in the Convention are flexibly defined, under a global energy and development scenario that relies heavily on coal, we project $\approx$ 90 and 150 Mg·y^{-1} of avoided power sector emissions for China and India respectively in 2050, compared to a scenario in which only current technologies are used. Benefits of this avoided emissions growth are primarily captured regionally, with projected changes in annual average gross deposition over China and India $\approx$ 2 and 13 μg· m^{-2} lower, respectively, than the current technology case. Stricter, but technologically feasible, mercury control requirements in both countries could lead to a combined additional 170 Mg·y^{-1} avoided emissions.

Assuming only current technologies but a global transition away from coal avoids 6% and 36% more emissions than this strict technology scenario under heavy coal use for China and India, respectively.

2.1   Introduction

Atmospheric mercury emissions can travel short or long distances depending on chemical form, leading to both local and global mercury contamination (Selin, 2009). In aquatic ecosystems, mercury poses risks to human and wildlife health as the potent neurotoxin methylmercury (Mergler et al., 2007; Karagas et al., 2012; Depew et al., 2012). Human populations are typically exposed to methylmercury through fish and shellfish consumption, but also through rice and other food sources (McKelvey & Oken, 2012; Jiang & Shi, 2006).

Concern about mercury's global transport and human health impacts led to the 2013 adoption of the United Nations (UN) Minamata Convention on mercury. The Minamata Convention takes a life-cycle approach to regulating mercury and its compounds, with obligations for mining, use, emissions and releases, and disposal (Selin, 2014a). More than 50% of mercury emissions are estimated to be by-product: mercury is a trace impurity in raw coal, oil, and ores, released upon combustion or smelting (UNEP, 2013a). Thus, efforts to control these emissions sources can interact with energy and development interests, as well as with traditional air quality priorities. Coal combustion is estimated to be the second largest global source of anthropogenic mercury emissions to air (24% of emissions in 2010) (UNEP, 2013a, 2008, 2002). For coal combustion, Asia is the largest regional contributor to global emissions, and these emissions are projected to increase with continued economic growth (UNEP, 2013a; Sloss, 2012b; UNEP, 2011; Streets et al., 2011; Rafaj et al., 2013). Historically, however, Europe and North America were major contributors, and this previously emitted mercury can continue to be re-emitted from soils and the ocean (Streets et al., 2011). Existing projections suggest that emissions growth in Asia will drive the global trajectory until 2050 (Streets et al., 2009; Rafaj et al., 2013).

The 2013 UNEP Global Mercury Assessment estimated that China alone contributes approximately one-third of the global anthropogenic emissions total (UNEP, 2013a). India is estimated to be the second largest national contributor, though the gap between India and China is large—in 2010, India contributed an estimated 7% of global emissions, com-

pared to 29% from China (Arctic Monitoring and Assessment Programme & United Nations Environment Programme, 2013). For both countries, rapid economic development drives emissions: primarily, coal combustion for electricity generation and industry, but also other industrial and mining activity, such as cement production and non-ferrous metal smelting (Sloss, 2012b; UNEP, 2013a).

Coal consumption in both countries is expected to grow substantially. The World Resources Institute estimates that China was responsible for 46% of world coal consumption in 2010—more than three times the next two largest consumers, the US at 13% and India at 9% respectively—and increases in coal electricity generating capacity are still planned (Yang & Cui, 2012). Growth in India may be even steeper, as electrification is a major near-term government priority (India Central Electricity Authority, 2012), and close to 25% of the population may lack electricity access (Remme et al., 2011; Ahn & Graczyk, 2012). The Indian government plans to add $\approx$160 GW of additional capacity by 2022, and coal-fired power plants (CFPPs) will be a large part of this expansion (India Central Electricity Authority, 2012).

While the Minamata Convention contains obligations for regulating mercury emissions from coal, it does not specify quantitative emissions limits or require specific technologies. Instead, parties must control, and where feasible, reduce mercury emissions in new CFPPs by applying Best Available Techniques (BAT) and Best Environmental Practices (BEP) within five years of the treatys entry into force. For existing plants, parties can choose between implementing BAT and BEP, quantified goals, emission limit values, multi-pollutant control strategies, or alternative reduction measures, within 10 years of the treaty's entry into force (UNEP, 2013c). Given the flexibility of these requirements, what the Convention will mean in practice will differ by country. Specific guidance on BAT and BEP will be developed by the Convention's Conference of Parties, and will likely include a suite of approaches, with country representatives taking into account country-specific economic and technological considerations, as well as other pollutant controls (UNEP, 2013c). Here, we explore how different technological approaches might affect the implications of the Minamata Convention for large, CFPP boilers in two major mercury-emitting countries in Asia: India and China. We develop technology scenarios consistent with the Convention, taking into account local political, technological, and geological factors. We examine the impacts of these technology scenarios on atmospheric mercury emissions from coal, and on mercury

deposition worldwide. We also compare the magnitude of emissions and deposition benefits from increased use of control technology to that from a less carbon-intensive global energy trajectory.

2.2 Methods

2.2.1 Technology scenario development

To better understand mercury-related policy, energy, and technology trends in India and China, we conduct an extensive literature review, including both peer-reviewed sources and technical reports from governments, and international agencies such as UNEP and the International Energy Agency (IEA). We supplement this literature review with semi-structured interviews with Convention negotiators, air pollution regulators, and coal and air pollution control experts. More information on these interviews and how interview data was used to support analysis is provided in Appendix A. We combine insights from these sources to develop technology scenarios consistent with the Convention, focusing on how BAT and BEP are likely to be applied in China and India.

We consider three representative technology scenarios: a no additional control (NAC) scenario, a Minamata-Flexible (MF) scenario, and a Minamata-Strict scenario (MS). Under NAC, we specify technologies and techniques currently widely in use. We define the MF scenario as technologies and techniques consistent with existing domestic (not necessarily mercury specific) policy plans, and which could fall under a flexible definition of BAT. We define the MS scenario as technologies and techniques that represent a progression in stringency of mercury control beyond those specified in MF. For each scenario, we specify a representative, most likely suite of technologies and techniques for each country based on our review of the literature and interviews, and assume countrywide implementation. We use a representative scenario approach (in contrast to a plant-by-plant approach that would focus on variability in pollution control strategies) to better isolate the impacts of specific technological choices on emissions and transport.

For each technology scenario and country, we estimate most likely values for the mercury removal efficiency fraction, f$_{capture}$, and the resulting speciation of emissions among three chemical forms of mercury: gaseous elemental mercury ($Hg(0)$), gaseous oxidized mercury

26

$(Hg(II)_g)$, and particulate-bound mercury $(Hg(II)_p)$. Additional analysis considering the reported range of f$_{capture}$ and emissions speciation for control technologies under each scenario is given in Appendix A Section A.6. Different pollution control technologies can alter emissions speciation, which has implications for atmospheric transport. $Hg(0)$ has an atmospheric lifetime of six months to a year. In contrast, $Hg(II)$ is readily deposited, leading to atmospheric lifetimes of days to weeks (Selin, 2009). Thus, $Hg(II)$ acts as a more regional pollutant, while $Hg(0)$ is capable of global transport—though atmospheric chemical reactions can transform one species to another (Hynes et al., 2009; Landis et al., 2014). Mercury deposited from the atmosphere is not lost to a stable surface reservoir however, but may be re-emitted and continue to cycle in surface atmosphere, terrestrial, and water reservoirs for decades to centuries (Selin, 2009).

Where available, we use values for mercury capture efficiency and emissions speciation from measurements at Chinese and Indian plants. Where measurements are unavailable, we use the Process Optimization Guidance for Reducing Mercury Emissions from Coal Combustion in Power Plants (UNEP, 2010) and the Interactive Process Optimization Guidance (iPOG) tool (Krishnakumar et al., 2012), to estimate removal efficiencies and speciation, based on average reported coal characteristics for each country. While we use deterministic estimates for these values in our baseline analysis, any given technology is likely to lead to a distribution of capture efficiencies and speciation fractions, due to variability in coal characteristics and implementation. As a result, we apply sensitivity analysis (discussed further below) to explore how variability in our estimates of f$_{capture}$ and emissions speciation affects our transport estimates.

2.2.2 Emissions estimation

To estimate the impact of technology choices on emissions, we combine assumptions about the mercury removal efficiency and speciation effects of technologies with projections for future coal use. Following Streets et al. (2009, 2005) and Wu et al. (2010), we calculate total mercury emissions from CFPPs in a given region as:

$$E_{totHg} = Hg_c \cdot C \cdot f_{release}(1 - f_{capture} \cdot f_{uptake})$$

$$E_{Hg,i} = E_{totHg} \cdot f_{Hg,i}$$

where E_{totHg} represents total mercury emissions, Hg_c is coal mercury content, C is the amount of coal burned, $f_{release}$ is the fraction of mercury released to the gas phase when burning coal, $f_{capture}$ is the removal efficiency of the technology scenario, and f_{uptake} is the uptake rate of that technology suite. To calculate emissions of specific mercury species, $E_{Hg,i}$, where $i \in 0$ (gaseous elemental), 2 (gaseous oxidized), P (particulate-bound), we then apply the speciation fraction $f_{Hg,i}$, to E_{totHg}. We apply values for Hg_c, C, and $f_{release}$ from Streets et al. (2009), who use forecasts based on the IPCC SRES scenarios from the IMAGE group (RIVM, 2001). We assume a constant f_{uptake} across scenarios of 0.95, the maximum control technology penetration rate under IMAGE assumptions (RIVM, 2001; Streets et al., 2009). Our baseline analysis uses the A1B SRES scenario, which is broadly consistent with Representative Concentration Pathway 6.0 and Shared Socio-economic Pathway 2, under the new scenario framework developed by the climate research community for future-looking global change modeling studies (van Vuuren & Carter, 2013). This scenario has been characterized as "Business as Usual," with continued growth in coal and limited global environmental cooperation (van Vuuren & Carter, 2013; Streets et al., 2009). To assess the relative effects of technology on mercury emissions compared to broader socio-economic development trends, we also consider energy use under the SRES B1 scenario (RIVM, 2001; Streets et al., 2009). This scenario is characterized by global cooperation on sustainable development, leading to transitions away from coal (van Vuuren & Carter, 2013; Streets et al., 2009).

2.2.3 Chemical transport modeling

To estimate worldwide mercury deposition, we use the GEOS-Chem global mercury model, version 9-02 (http://acmg.seas.harvard.edu/geos). The model includes a 3-D atmosphere from Holmes et al. (2010) and Amos et al. (2012), and 2-D ocean and terrestrial slabs from Soerensen et al. (2010) and Selin et al. (2008) respectively. GEOS-Chem is driven by assimilated meteorology from the NASA Goddard Earth Observing System (GEOS-5).

The model has been extensively compared in previous work to observed concentrations and wet deposition (Amos et al., 2012; Zhang et al., 2012; Holmes et al., 2010; Soerensen et al., 2010; Selin & Jacob, 2008; Chen et al., 2014; Muntean et al., 2014). There are numerous uncertainties in modeling global mercury. Corbitt et al. (2011) discuss uncertainties in quantifying source-receptor relationships, and identify mercury speciation and atmospheric reduction processes as particularly relevant. We quantitatively evaluate these uncertainties here through sensitivity analysis and an alternative chemistry scenario, described below.

The model includes gaseous elemental mercury, $Hg(0)$, and gaseous and particulate-bound divalent mercury, $Hg(II)_g$ and $Hg(II)_p$. In GEOS-Chem, $Hg(0)$ is emitted by natural and anthropogenic sources, while $Hg(II)$ is emitted predominantly by anthropogenic sources. Divalent mercury can deposit via wet and dry deposition, and $Hg(0)$ can undergo dry deposition (Selin, 2009). Re-emissions of mercury from terrestrial and aquatic reservoirs occur only as $Hg(0)$ (Selin, 2009). Divalent mercury follows an empirical gas-particle partitioning relationship based on air temperature and aerosol concentration, following Amos et al. (2012). $Hg(0)$ oxidizes to $Hg(II)$ by reaction with Br, and $Hg(II)$ is photoreduced to $Hg(0)$ in cloud droplets (Holmes et al., 2010).

Reduction of $Hg(II)$ to $Hg(0)$ in power plant plumes has been hypothesized to occur, based on comparisons of measured speciation fractions from the stack and downwind, though its mechanism has not been identified (Edgerton et al., 2006; Weiss-Penzias et al., 2011; Lohman et al., 2006; Landis et al., 2014). Recent evidence suggests that the occurrence and extent of in-plume reduction (IPR) may depend on coal composition characteristics (Landis et al., 2014). Implementing IPR in mercury transport models has improved correlations between modeled and observed wet deposition and concentrations within North America (Vijayaraghavan et al., 2008; Amos et al., 2012; Zhang et al., 2012). We therefore run model simulations both with and without IPR, where we use the without IPR scenario as our baseline. Based on measurement and experimental studies (Tong et al., 2014; Edgerton et al., 2006; Lohman et al., 2006; Landis et al., 2014), we implement IPR as a conversion of 70% of power sector $Hg(II)$ emissions to $Hg(0)$.

Emissions gridded at 1°x1° from Corbitt et al. (2011), and scaled to 2050 projections from Streets et al. (2009), are scaled on a national basis for China and India given technology scenario totals constructed as described above. As the IMAGE 2.2 energy (RIVM, 2001) and Streets et al. (2009) emissions projections underlying the analysis are for East Asia and

South Asia as regions (see Figure A-1), we apply these regional scaling factors to China and India respectively (Corbitt et al., 2011).

Our GEOS-Chem simulations have a horizontal resolution of $4°\text{x}5°$ latitude-longitude, with 47 vertical layers. We simulate meteorological years 2007-2012, with 2050 emissions, using the first three years as initialization. We present 2010-2012 averages to account for inter-annual variability. We archive simulated total gross deposition as the sum of wet and dry deposition of all mercury species. We do not consider the effects of future meteorology in the present study to better isolate the effect of technology choices and emissions. Our GEOS-Chem simulations track the effect of primary anthropogenic emissions changes on mercury in surface reservoirs and resulting deposition. Additional deposition effects due to mercury in longer-lived soil and ocean pools (legacy pools) are addressed in Appendix A and Discussion.

To better understand the effects of the technology scenarios on mercury deposition patterns, we perform sensitivity analysis on the removal efficiency, and speciation of emissions for each country ($f_{capture}$ and $f_{Hg(0)}$). We perturb each of these variables $\pm20\%$ from the MF scenario baseline, and evaluate the corresponding impacts on average total gross deposition flux over India, China, the US, and ocean basins. Because deposition responses to these perturbations are approximately linear in this range, we then calculate response ratios, α, for average deposition fluxes over India, China, and the US (to illustrate long-range impacts). The response ratio represents the average percent change in average deposition flux due to a 1% change in the perturbed variable.

2.3 Results

2.3.1 Synthesis of available mercury control technologies and techniques

Mercury control measures for large CFPP boilers have been extensively reviewed elsewhere (UNEP, 2010; Krishnakumar et al., 2012; Pacyna et al., 2010b; Srivastava et al., 2006; UNEP, 2011). We assemble this information to apply to our technology scenarios, focusing on the performance of control measures in China and India specifically. Table 2.1 summarizes reported mercury reduction ranges for different control measures for India, China, and the US (where measurements are the most abundant). A more in-depth review of this

information is provided in Appendix A.

Table 2.1 summarizes three major categories of mercury reduction approaches for CF-PPs: improvements to plant efficiency, pre-combustion controls, and post-combustion controls (UNEP, 2010; Pacyna et al., 2010b).

Efficiency improvements can include upgrading equipment, optimizing combustion, minimizing short cycling and air leakages, and changing operations and maintenance. As shown in Table 2.1, these upgrades have been estimated to reduce emissions by up to 7%. Efficiency improvements can also have economic and climate benefits.

Pre-combustion controls include processing coal to improve efficiency and reduce unwanted trace elements, such as mercury and sulfur. Conventional coal cleaning targeting ash (which lowers the energy value of coal and combustion efficiency of boilers) and sulfur can also reduce mercury content by $\approx 30\%$ (Toole-O'Neil et al., 1999; US EPA, 1997; Wu et al., 2010; UNEP, 2010). However, a wide range of removal efficiencies has been reported (0-78%), as efficacy depends on coal source and characteristics (Toole-O'Neil et al., 1999; US EPA, 1997; Wu et al., 2010; UNEP, 2010). Coal blending, additives, and beneficiation are discussed in Appendix A.

Post-combustion controls can be either mercury-specific, or aimed at other pollutants but with co-benefits for mercury. Substantial divalent, but not elemental, mercury reduction co-benefits can be achieved by controlling for particulate matter (PM), SO_2, and NO_x. Standard PM controls like electrostastic precipitators (ESP) and fabric filters (FF) are effective at capturing $Hg(II)_p$, while controls for sulfur, such as wet flue gas desulfurization (wFGD) and spray dry absorber (SDA) systems, are effective at capturing mercury as $Hg(II)_g$. Selective catalytic reduction (SCR), used for NO_x control, increases the fraction of mercury in the stack gas as $Hg(II)_g$, facilitating removal by desulfurization measures. As with coal cleaning, the efficacy of these measures depends on coal characteristics such as moisture, ash, sulfur, and halogen, and mercury content (Krishnakumar et al., 2012). Table 2.1 presents reported capture ranges for selected combinations of cold-side ESP (CS-ESP), FF, wFGD, SDA, and SCR. The majority of these values are based on US measurements, though measurements for CS-ESP are available for both India and China, and measurements for FF and wFGD are also available for China. The ranges among countries are consistent, though reported efficacy of PM and SO_2 control tends to be lower in China than in the US.

Post-combustion control technologies that specifically target mercury (particularly $Hg(0)$)

Table 2.1: Reported mercury reduction (%) of control strategies in China (UNEP, 2010; Zhu et al., 2002; Wang et al., 2010; Wang & Shen, 2000; Zhang et al., 2008; Wang et al., 2009; Zhou et al., 2008; Wang et al., 2008), India (UNEP, 2010, 2014), and the US (UNEP, 2010; Toole-O'Neil et al., 1999; US EPA, 1997; Srivastava et al., 2006; US EPA, 2002, 2010). All values are for pulverized coal combustion.

Control Strategy	Device Configuration	Hg Reduction Range (%)		
		China	India	United States
Plant efficiency improvements		0-7	0-7	0-7
Pre-combustion controls				
Coal cleaning		-	$13\text{-}40^{ns}$	$0\text{-}78^{b}$
Post-combustion controls				
PM Control	CS-ESP	$20\text{-}41^{b}$	$19\text{-}73^{ns}$	$0\text{-}81^{b}$; $0\text{-}22^{s}$
	FF	$10\text{-}80^{b}$	-	$63\text{-}93^{b}$; $53\text{-}88^{s}$
PM + SO$_2$ Control	CS-ESP+wFGD	$13\text{-}74^{b}$	-	$64\text{-}76^{b}$; $2\text{-}58^{s}$
	FF+wFGD	-	-	$62\text{-}99^{b}$
	FF+SDA	-	-	$97\text{-}99^{b}$; $0\text{-}48^{s}$
PM + SO$_2$ + NO$_x$ Control	SCR+CS-ESP+wFGD	-	-	¿90^{b}
	SCR+FF+SDA	-	-	$94\text{-}99^{b}$; 0-487\$^s\$
Hg specific Control	CS-ESP + ACI*	-	-	Up to 98^{b}; Up to 62^{s}

[b] Bituminous coal; [s] Sub-bituminous coal; [ns] Not Specified; [*] Capture depends on sorbent injection rate; CS-ESP: cold-side electrostatic precipitator; FF: fabric filter; wFGD: wet flue gas desulfurization; SDA: spray dry absorber; SCR: selective catalytic reduction; ACI: activated carbon injection

have been demonstrated to reduce emissions in the US by greater than 90% and up to 98% (Srivastava et al., 2006; UNEP, 2010). However, these technologies may have higher installation and operating costs than co-benefit approaches (Pacyna et al., 2010b). The most common configurations involve sorbent injection, typically activated carbon (ACI), in conjunction with a PM control device (UNEP, 2011). While this technology is commercially available, application is not yet widespread, particularly in Asia (Sloss, 2012a,b). However, recent stringent standards for mercury from utility boilers in the US and Canada may result in installation of ACI at many plants (Sloss, 2012a; US EPA, 2011b). Multi-pollutant systems specifically optimized to simultaneously capture mercury, SO$_2$, and NO$_x$ are currently in development, however are not yet widely commercially available (UNEP, 2011; Sloss, 2012a).

2.3.2 Technology scenarios

Table 2.2 presents the NAC, MF, and MS scenarios for China and India: technology configurations and values for f$_{capture}$, f$_{Hg^0}$ and f$_{Hg2/P}$.

No Additional Control (NAC) Scenario. For China, under the NAC scenario, we

Table 2.2: Technology scenarios. Coal characteristic assumptions are further described in Table A.1.

Scenario	Technologies and Techniques	$f_{capture}$ (%)	$f_{Hg(0)}$ (%)	$f_{Hg(II)}$ (%)	Notes
China					
No Additional Control (NAC)	CS-ESP+wFGD	69	78	22	Values from Wang et al. (2010)
Minamata-Flexible (MF)	SCR+CS-ESP+ wFGD	82	68	32	Values from UNEP iPOG tool (Krishnakumar et al., 2012) with coal characteristics from UNEP China report (UNEP, 2011)
Minamata-Strict (MS)	SCR+FF+ wFGD	90	68	32	Value for $f_{capture}$ based on interpretation of qualitative descriptions in Srivastava et al. (2006) and limited test data from US EPA (2002). In absence of speciation data, the same fractions as SCR+CS-ESP+wFGD are applied.
India					
No Additional Control (NAC)	CS-ESP	42	67	33	Value from UNEP India Report (UNEP, 2014)
Minamata-Flexible (MF)	Efficiency gains + Coal washing + CS-ESP	58	67	33	Estimates for removal efficiency from washing and ESP from UNEP (2014); estimates for efficiency gains from UNEP (2010) and India Central Electricity Authority (2012)
Minamata-Strict (MS)	Efficiency gains + Coal washing + CS-ESP+wFGD	70.5	93	7	Estimate from UNEP iPOG tool (Krishnakumar et al., 2012) with coal characteristics from UNEP (2014)

assume countrywide use of CS-ESP and wFGD technologies, leading to a capture efficiency of 69%, and a 78% and 22% breakdown of power sector emissions between $Hg(0)$ and $Hg(II)$. Installation of ESP technology at Chinese power plants has steadily grown since the 1980s, reaching $\approx$95% of nationwide capacity by 2003 (Wu et al., 2006). Installation of FGD systems began approximately a decade later, with the most rapid increases in coverage occurring over the past 10 years (from approximately 14 to 86% of plants between 2005 and 2010), to meet SO_2 reduction targets in Chinas 11th Five Year plan (Wu et al., 2006; Sloss, 2012a).

For India, the NAC scenario assumes countrywide CS-ESP only, with a capture efficiency of 42%, and speciation split of 67 and 33% between $Hg(0)$ and $Hg(II)$. Currently, there is limited air pollution regulation of stationary combustion sources in India (Sloss, 2012b; Mukherjee et al., 2009; Burger Chakraborty et al., 2013). Indian coal is relatively low sulfur (Sloss, 2012b; UNEP, 2014); consequently, SO_2 controls have not been a regulatory priority, with the exception of requirements for stack height, which do not capture SO_2, but lessen its local impacts (Sloss, 2012b; Mukherjee et al., 2009). However, emissions limits for particulates have been in place since 1981, resulting in widespread installation of ESP systems (Sloss, 2012b).

Minamata Flexible (MF) Scenario. Under the MF scenario, Chinese plants adopt SCR technology to control for NO_x, in addition to existing PM and SO_2 controls. Percent mercury reduction of control technology thus increases from 69% to 82%. There is a small shift towards divalent and particulate-bound emissions compared to NAC, as SCR promotes oxidation to these species. This scenario is based on existing policy. In 2011, the Chinese Ministry of Environmental Protection adopted the new Emission Standard of Air Pollutants for Thermal Power Plants (GB 13223-2011) for 2012-2017 (China MEP, 2011). Under this standard, newly constructed plants in China face more stringent emissions limits for sulfur dioxide, nitrogen oxides, and particulates, with tighter standards for existing plants being imposed in 2014 (China MEP, 2011)—though enforcement remains a potential challenge (Schreifels et al., 2012; Standaert, 2014). Achieving the new standards will require almost all plants to install FGD, SCR, and increased use of ESPs and FFs (UNEP, 2011; Sloss, 2012a). Addressing mercury pollution was also a stated goal in Chinas 12th Five Year Plan (2011-2015). This is reflected in the Emission Standard, which limits mercury emission to 30 μg/m^3. This limit is an order of magnitude weaker than limits in the US. However, the

stringency of requirements for SO_2, NO_x, and PM are likely to lead to large co-benefits for mercury (Sloss, 2012a; UNEP, 2011). Consequently, most plants will be well below the emission limit set for mercury without mercury specific control technologies—though results will depend on coal attributes and plant processes, which can vary plant to plant (Sloss, 2012a).

For India, the MF scenario assumes pre-combustion measures like plant efficiency gains and coal washing, in addition to ESP, increasing $f_{capture}$ from 42% to 58%. Since these measures are pre-combustion, we assume that the speciation breakdown does not change from NAC. In contrast to China, attributes of Indian coal and Indias domestic power sector may make "end-of-pipe" mercury control more costly for India, with fewer opportunities for alignment with existing domestic environmental policies. Domestically sourced coal provides ≈70% of Indias heat and electricity generation (Sloss, 2012b). Indian coal has a high reported ash content (UNEP, 2014), exacerbating already low plant efficiency and increasing mercury emissions per unit energy (Sloss, 2012b). Moreover, the mercury content of Indian coal may be highly variable (Mukherjee et al., 2009; Burger Chakraborty et al., 2013; Kumari, 2011; UNEP, 2014).

Recent regulatory efforts for CFPPs have focused on plant efficiency improvements. The Indian Central Pollution Control Board has capped the ash content of coals used for thermal power generation at 34%, with all plants required to comply by 2016 (India Central Pollution Control Board, 2014). This requirement has led to ongoing increases in coal washing and blending. Plans for capacity expansion have also highlighted the need to increase unit sizes, and adopt supercritical and integrated gasification combined cycle technology in new installations—technologies that can offer efficiency gains of up to ≈2 and 10% respectively (India Central Electricity Authority, 2012).

Minamata Strict Scenario. Under the MS scenario, we assume a further increase from 82% to 90% capture efficiency in China, based on substituting FF for ESP. The actions currently being undertaken by US CFPPs to comply with a recent US rule targeting mercury emissions illustrate possible mercury control strategies beyond co-benefits from standard PM, SO_2, and NO_x control. To comply with the new rule, US CFPPs are expected to double the application of FF, and increase ACI by 15-fold, compared to a baseline case including just air quality policies (US EPA, 2011b). We focus on the substitution of FF for ESP for our MS scenario for China, given that it is less costly than ACI (Pacyna et al.,

2010b).

For India, we assume an increase in mercury reduction from 58% to 70.5% through the introduction of FGD. With this technology, there is a large shift towards emissions in gaseous elemental form (93% of emissions). FGD is considered a logical next step in pollution control devices after those for PM control (Sloss, 2012b; Burger Chakraborty et al., 2013), and this scenario, and its associated capture efficiency, is consistent with existing proposed mercury emissions reduction strategies for India (Burger Chakraborty et al., 2013).

2.3.3 Emissions

Figure 2-1 shows projected emissions of total mercury (THg) under the three technology scenarios for the A1B (left) and B1 (right) energy scenarios. Colored bars represent emissions from the power sector, by species, while grey bars represent contributions from all other domestic anthropogenic sources (projected following Streets et al. (2009)). Emissions for India are shown in red, while those for China are shown in blue.

For India, under MF with an A1B development scenario, projected emissions are 24% lower than under NAC (468 vs. 619 Mg/y). Moving from MF to MS, emissions are an additional 26% (122 Mg/y) lower. All of these additional avoided emissions are as $Hg(II)$, with emissions of $Hg(0)$ roughly constant between MF and MS. Under all scenarios, projected 2050 Indian power sector emissions are large increases over 2010 estimated emissions (49 Mg/y) (Arctic Monitoring and Assessment Programme & United Nations Environment Programme, 2013).

In China, projected emissions are 36% lower under MF compared to NAC (156 vs. 247 Mg/y), and a further 33% (51 Mg/y) lower between MF and MS. Projected 2050 Chinese power sector emissions exceed the 2010 estimate (97 Mg/y) for all technology scenarios (Arctic Monitoring and Assessment Programme & United Nations Environment Programme, 2013), though under MS the increase (+8 Mg/y) is within the uncertainty range of the current estimate (which ranges up to 139 Mg/y).

We also consider the same technology scenarios to 2050 under a B1 development scenario. Assuming no technological change (NAC) but a global transition away from coal, we project that power sector emissions are 61% (150 Mg) lower relative to A1B in China, and 60% (370 Mg) lower in India. The difference between 2050 A1B and B1 total emissions, including

other sectors, under the NAC scenario is 525 Mg for China and 417 Mg for India.
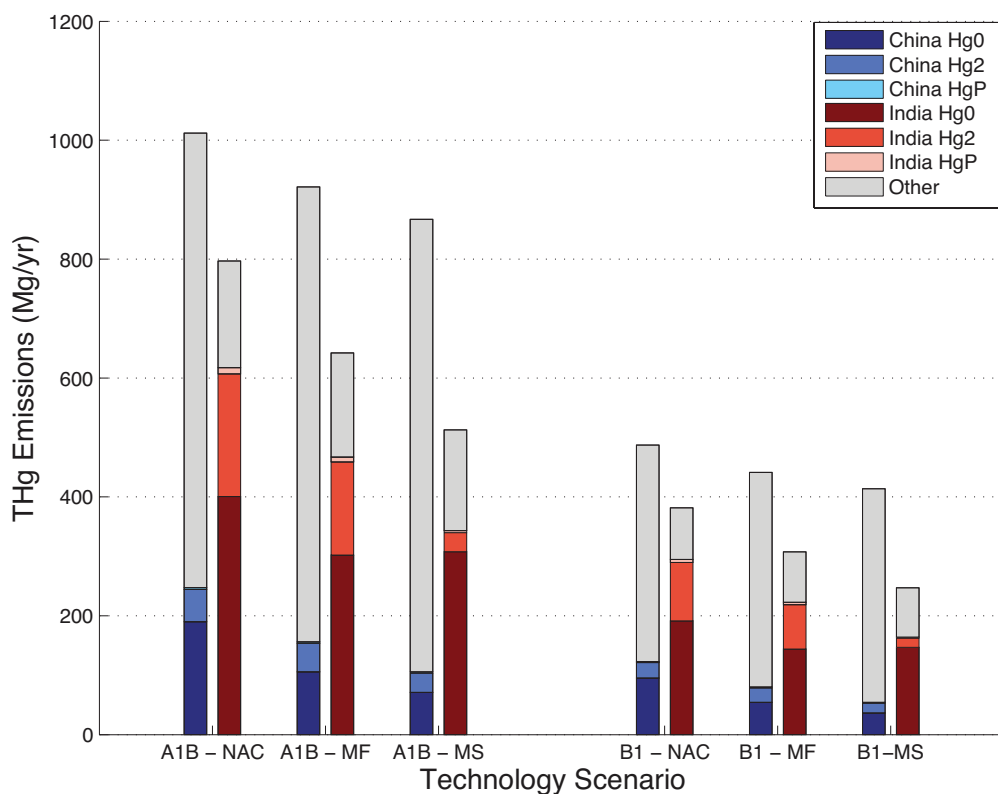
Figure 2-1: Projected 2050 emissions of THg (all species) under technology scenarios. Emissions for China (blue) and India (red) are presented under the three technology scenarios for the A1B and B1 development scenarios. Colored bars represent emissions from the power sector, by species. Grey bars represent contributions from all other domestic anthropogenic sources.

2.3.4 Impacts on deposition

Figure 2-2 shows modeled differences in annual gross deposition flux between technology scenarios (NAC - MF and MF - MS) for 2050 A1B. Spatially averaged values for the countries (total deposition mass/total area), along with their mass equivalents, are presented in Table A.2 . Differences in average annual deposition flux over China and India between NAC and MF are approximately 4.6% (2.1 $\mu g \cdot m^{-2}$) and 42% (13.4 $\mu g \cdot m^{-2}$) of modeled present-day

domestic deposition respectively (shown in Figure A-2). The differences between MF and MS are 4.8% (2.2 μg·m^{-2}) and 82% (26.1 μg·m^{-2}) of modeled present-day deposition in China and India.

We also find changes in modeled deposition flux in other regions. Average annual deposition fluxes over both the US and Europe change by approximately -5% of modeled present day annual average deposition between NAC and MS (-1.2 μg·m^{-2} and -0.7 μg·m^{-2}, respectively). We also examine deposition to ocean basins, given that mercury exposure for many global populations is through marine fish, and it is the predominant US exposure source (Sunderland, 2007; Oken et al., 2012). Moving from the NAC to the MS scenario changes deposition to the North Pacific by -6.6% of modeled present day levels (-1.0 μg·m^{-2}), and by -9.6% (-0.8 μg·m^{-2}) for the South Pacific/Indian Oceans. These source regions contribute the two largest shares to US mercury intake from commercial fish (Chen et al., 2012; Sunderland et al., 2010).

Assuming reduction of $Hg(II)$ to $Hg(0)$ in power plant plumes (IPR) results in simulated CFPP emissions contributing more strongly to extra-regional deposition. Simulated differences in annual gross deposition flux under the IPR assumption are shown in Figure A-4. With IPR, the deposition benefit of moving from NAC to MS decreases by 57% over India (from 39.5 to 17 μg·m^{-2}), while the benefit increases by 25% over the US (from 1.2 to 1.5 μg·m^{-2}), compared to the without IPR case. For China, the benefit of moving from NAC to MS increases by 9% compared to the without IPR case (from 4.3 to 4.8 μg·m^{-2}), because of reduced contributions from Indian emissions. The North Pacific and Atlantic oceans see larger benefits between NAC and MS compared to the without IPR case, while there is a small decrease in benefit for the South Pacific/Indian Ocean.

2.3.5 Sensitivity Analysis

We calculate the response ratio, α, the average percent change in mean deposition flux to a region due to a 1% change in capture efficiency or emissions speciation, based on $\pm20\%$ perturbations from the MF scenario. Based on this analysis, changes in assumed domestic capture efficiency in the power sector lead to larger responses in domestic deposition flux for India than China ($\alpha = 0.561$ and 0.382 respectively), reflecting India's larger share of total emissions from the power sector. We also find that speciation of power sector
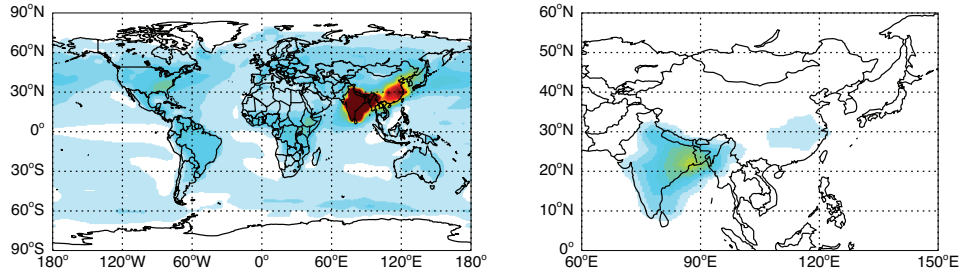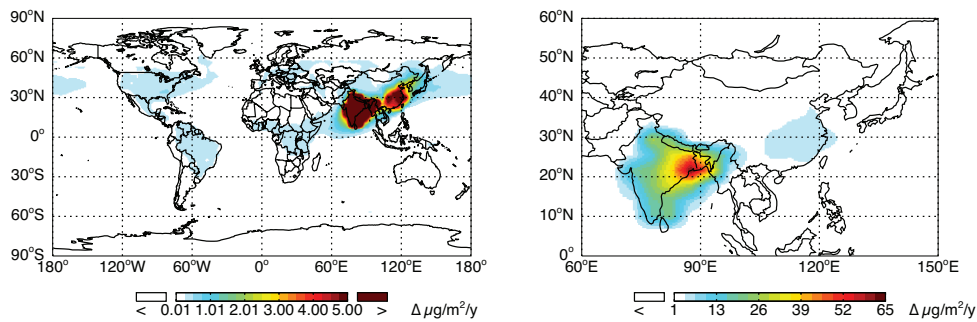
Figure 2-2: Modeled differences in annual gross deposition flux between technology scenarios for 2050 A1B. Note that the color bar for figures on the left hand side, showing global extent, saturate at a maximum of 5 μg·m^{-2}·y^{-1}, while right hand side figures, zoomed in on Asia, show a minimum of 1 μg·m^{-2}·y^{-1}. See Figure A-3 for a version with a log color scale.

emissions plays a larger role in domestic deposition in India compared to China ($\alpha = 0.663$ and 0.133), as most emitted Indian $Hg(II)$ is deposited domestically. For China, because Chinese emissions sources are predominantly in the east, and transport tends eastward, less of the benefit of reduced emissions is captured over the Chinese landmass itself. We find that a 1% change in Chinese and Indian power sector capture efficiencies lead to approximately equivalent responses in US deposition. The larger influence of transpacific transport of Chinese emissions on US deposition balances the fact that a 1% change in $f_{capture}$ India represents a larger absolute change in emissions. Additional discussion of sensitivity analysis results is provided in the SI. Tabulated α values are given in Table A.3.

We also relate deposition responses by mass to the mass change in total mercury emissions in each country, ΔTHg, and the mass conversion of emissions from $Hg(0)$ to $Hg(II)$, $\Delta Hg(0) \rightarrow Hg(II)$ in Table A.4.

2.4 Discussion and Policy Implications

We project that the magnitude of avoided emissions and deposition from implementing more effective control technologies for mercury in Asian CFPPs is large (as a fraction of current emissions levels), even when considering a scenario where the Minamata Convention's requirements for mercury are consistent with existing, non-mercury specific domestic pollution and energy policy plans (MF). Even such a flexibly designed Convention could lead to avoided emissions of 242 Mg in India and China in 2050 compared to a scenario with no additional technology, which is equivalent to $\approx$12% of the total global anthropogenic emissions in 2010 (UNEP, 2013a). More than 60% of these avoided emissions are from India, highlighting the importance of India's participation in any global requirements for CFPPs. Despite the fact that mercury reduction efficiencies in Chinese CFPPs are likely to be higher, the larger emissions avoided in India reflect the fact that power sector emissions contribute a larger fraction of total emissions in India across technology scenarios (67-78%) than in China (12-24%), where emissions from industrial processes represent the largest fraction of 2050 projected emissions.

The benefits of these avoided emissions in terms of avoided deposition are concentrated regionally in Asia, particularly in India, where the difference reaches a maximum of $\approx$30μg/m^2. However, deposition differences between the NAC and MF technology scenarios in the US and Europe, are also $\approx$5% of current deposition. Moreover, global benefits occur through avoided enrichment to oceans, particularly to the Pacific and Indian Oceans, from which many of the fish in the global seafood market are sourced (Sunderland, 2007; Chen et al., 2012). Qualitatively, this distribution of benefits is robust to the IPR assumption (see Figure A-4), though regions outside of Asia receive a larger share of benefits.

The definition of BAT for CFPPs under the Convention is being discussed in a technical working group, and will also be discussed during future conferences of parties. We show that these definitions can have a substantial impact on environmental mercury: differences in stringency of required control technologies (MF vs. MS) could result in emissions differences

in India and China of 173 Mg combined in 2050, which is roughly the total estimated emissions from India in 2005 (UNEP, 2008). Increased stringency in the definition of BAT could also avoid growth in power sector emissions over present-day levels in China, though this is unlikely for India. The majority of benefits of increased stringency in terms of deposition are captured by India and China, suggesting that there is a strong domestic incentive for these two countries to take further actions beyond a flexible BAT. In the model simulation, 73% by mass of the 173 Mg avoided emissions between MF and MS would have been deposited in India and China. This pattern is due to the speciation of modeled emissions reductions between MF and MS, as all Indian emissions reductions are in $Hg(II)$, forms that contribute most strongly to regional pollution.

For the power sector, the emissions gap between technology scenarios (A1B:NAC – A1B:MS) is comparable to, but smaller than, that between energy and development scenarios (A1B:NAC – B1:NAC), with the former being 94% of the latter for China, and 74% for India. This result underscores the importance of energy and development trajectories for mercury emissions. While moving towards more effective mercury control technologies in China and India can mitigate some of the emissions growth associated with aggressive increases in coal consumption for the power sector, avoiding coal consumption and transitioning towards less carbon-intensive energy sources is likely necessary for reducing emissions from present-day levels. Both Indian and Chinese governments have stated goals to increase renewable capacity, and to pursue demand-side management of electricity, for instance through energy efficiency targets (India Central Electricity Authority, 2012; Ministry of Environmental Protection China, 2013). A recent agreement between China and Russia on natural gas also suggests a shift towards Russian-exported gas-fired power plants in the future. Avoided consumption of coal could also be an important facet of a mercury emissions mitigation strategy.

Several assumptions made in the chemical transport modeling present opportunities for future exploration. Without locational data on Chinese and Indian power sector emissions, emissions were scaled uniformly across countries, based on 2005 spatial distributions (Pacyna et al., 2010a). This approach does not capture real spatial patterns, particularly as substantial new coal generation capacity will be built by 2050. The distribution of these new plants, particularly on the East-West axis, may have implications for transboundary transport to countries downwind; however, the present analysis still offers insight into global

distributional impacts, at the regional scale. Better spatial data could also provide further insight into the deposition patterns of divalent mercury within India and China.

Our projections suggest that under the Convention, Asian CFPP emissions will be avoided, but will likely increase from present-day, consistent with previous estimates that Minamata will result in avoided emissions increases (Selin, 2014b); however, total deposition benefits from these avoided emissions are likely to be larger than our estimates, which only take into account the impact of primary emissions changes. The mercury simulation used for this study does not completely account for the legacy impacts of anthropogenic emissions: primary anthropogenic emissions once deposited to terrestrial and aquatic ecosystems join a legacy pool of mercury that can continue to cycle through air, water, and land. Because primary emissions also enrich legacy pools in ocean and soil reservoirs, by 2050, re-emissions from these pools may contribute $> 50\%$ of global deposition (Amos et al., 2013; Sunderland & Selin, 2013). However, while the legacy anthropogenic contribution to total deposition by 2050 under an A1B scenario may be significant (Sunderland & Selin, 2013), we address here the difference between technology scenarios. The influence of legacy emissions is less substantial in the difference calculation, which is small compared to projected overall changes in the global mercury budget. Using a global box model developed by Amos et al. (2013, 2014), we estimate that accounting for legacy effects could increase deposition differences between NAC and MF and MF and MS by $\approx 30\%$ by mass (additional details are provided in Table A.5). Because legacy pools are likely to increase global background concentrations of mercury, their inclusion will not substantially change the modeled spatial patterns of deposition.

The effects of climate change could also have additional impacts on global mercury transport that we do not account for in this analysis. Climate change is likely to impact mercury biogeochemical cycling through increased volatilization from ocean and soil reservoirs (which will increase the influence from legacy mercury), increased plant respiration and wildfires, changes to oxidant concentrations, and changes to food webs, amongst other factors (Jacob & Winner, 2009; Booth & Zeller, 2005; Macdonald et al., 2005). Exploring the coupled effects of direct anthropogenic mercury emissions changes and climate change will be critical for understanding to what extent there will be a climate penalty on policy efforts to reduce mercury pollution (Jacob & Winner, 2009). These interactions between future energy and development trajectories, climate change, and mercury biogeochemical

cycling suggest that beyond technology choices for the power sector itself, a broader consideration of energy and development choices will be necessary to understand future mercury emissions trajectories and their impacts.

Chapter 3

Understanding factors influencing the detection of mercury policies in modelled Laurentian Great Lakes wet deposition

Abstract

We use chemical transport modelling to better understand the extent to which policy-related anthropogenic mercury emissions changes (a policy signal) can be statistically detected in wet deposition measurements in the Great Lakes region on the decadal scale, given sources of noise. In our modelling experiment, we consider hypothetical regional (North American) and global (rest of the world) step policy changes, consistent with existing policy efforts, that divide an eight-year period (Δ_{global} = -13%; $\Delta_{regional}$ = -30%). The magnitude of statistically significant (p<0.1) pre- and post-policy period wet deposition differences, holding all else constant except for the policy change, ranges from -0.4 to -2.4% for the regional policy and -0.9 to -2.5% for the global policy. We then introduce sources of noise— trends and variability in factors that are exogenous to the policy action—and evaluate the extent to which the policy signal can still be detected. We find that global trends in emissions of realistic magnitudes (for instance, due to economic activity) reduce the areas where the policy signal can be detected to the immediate vicinity of targeted emissions

sources, as these global trends dominate elsewhere. We find that interannual variability in emissions magnitude and speciation can reduce the magnitude of wet deposition differences between periods (but not their significance) by up to 35% for the regional policy, and up to 80% for the global policy. Interannual variability in meteorology has the largest effect, driving wet deposition differences between periods ±20%, far exceeding the magnitude of the policy signal. These results highlight the potential challenges of detecting statistically significant policy-related changes in Great Lakes wet deposition within the short-term.

3.1   Introduction

Mercury—a bioaccumulative toxin, particularly in its organic forms—poses risks to public health and the environment (Sundseth et al., 2017). Consequently, anthropogenic mercury emissions have been the target of policy action, from local to global scales (examples include Lake Superior Binational Program, 2012; US EPA, 2017; Selin & Selin, 2006; UNEP, 2013b). For instance, emissions in the United States and Canada have decreased by more than 75% since 1990, from 246 Mg/yr in 1990 to 55 Mg/yr in 2014 in the US (US EPA, 2016), and from 35.3 Mg/yr in 1990 to 6 Mg/yr in 2010 in Canada (Steffen, 2016) (see Supplementary Information Figure B-1). Domestic regulations targeting waste incineration (particularly in the US) and metals production (particularly in in Canada) contributed to steep declines in the 1990s, and since the mid-2000s, regulations targeting other air pollutants in addition to mercury have contributed to more modest decreases from the electricity generation sector (Zhang et al., 2016c; Castro & Sherwell, 2015). In the future, the United Nations Minamata Convention on Mercury, which is expected to come into force in 2017, may lead to reductions in emissions globally (Sundseth et al., 2017; Rafaj et al., 2013; Giang et al., 2015; Pacyna et al., 2016).

Coal combustion is the second largest source of anthropogenic mercury emissions globally, after artisanal and small scale gold mining (UNEP, 2013a), and given potential growth in energy demand from global economic development (Streets et al., 2009), decoupling energy production from mercury emissions is a potentially important part of mitigation efforts (Pacyna et al., 2016). Between the mid 2000s and mid 2010s, regulations targeting pollutant emissions from power plants in the US and Canada—for instance, the Clean Air Interstate Rule and its replacement, the Cross State Air Pollution Rule (Jaramillo & Muller, 2016),

and the Clean Air Mercury Rule and its replacement, the Mercury and Air Toxics Standards (US EPA, 2011a) in the US—led to the increased adoption of end-of-pipe air pollution control devices (Jaramillo & Muller, 2016). While many of these controls are not mercury-specific and target particulate matter (PM), SO_2, and NO_x, they also capture mercury as a co-benefit (UNEP, 2010). Globally, the UN Minamata Convention requires that parties apply best available techniques and best environmental practices for controlling mercury emissions from sources like coal-fired power plants, which includes co-benefit mercury capture from a range of air pollution control devices (UNEP, 2015). In China, adoption of these approaches in the electricity generation sector to address air quality concerns have already led to reductions in mercury emissions per unit coal (Zhang et al., 2015a; Zhao et al., 2015).

To what extent can these policy-related emissions decreases be detected in changes in mercury inputs to specific vulnerable ecosystems? In the Laurentian Great Lakes region, where mercury remains a concern for human and wildlife health (Evers et al., 2011a; Cain et al., 2011; Cohen et al., 2016), many community stakeholders in mercury management (including Indigenous communities and recreational anglers) are interested in whether these policies translate into decreases in atmospheric loadings of mercury to aquatic ecosystems, and ultimately, decreases in dietary human exposure from fish (Gagnon et al., 2013; Gagnon, 2014; Gagnon et al., 2017). As recent source attribution modelling studies have highlighted the importance of both local/regional and global anthropogenic sources for deposition in the Great Lakes basin (Cohen et al., 2016; Grant et al., 2014; Zhang et al., 2015b), this question is important not only for evaluating the effectiveness of historical and future policy efforts in North America at protecting human health in this region, but also for evaluating the potential impact of prospective policy actions elsewhere in the world in response to the Minamata Convention.

Recent studies report statistically significant declines in observed mercury wet deposition aggregated over North America between the mid-1990s and early 2010s (Zhang et al., 2016c; Cole et al., 2014; Weiss-Penzias et al., 2016), and that these long-term, large-scale declines have been driven by anthropogenic emissions changes (Zhang et al., 2016c; Zhang & Jaegle, 2013). However, spatially and temporally disaggregated trends within this larger spatio-temporal region show much heterogeneity (Weiss-Penzias et al., 2016; Butler et al., 2008; Prestbo & Gay, 2009; Lynam et al., 2016; Gratz et al., 2009; Risch et al., 2012). The

National Atmospheric Deposition Program (NADP) Mercury Deposition Network (MDN) collects weekly integrated wet deposition samples at monitoring sites in the US and Canada, with a continuous data record beginning in 1996 for the longest running sites (NADP, 2017). In an analysis of this monitoring data, Weiss-Penzias et al. (2016) found significant negative trends in wet deposition concentration in the majority of sites with data from 1997-2013 (ranging from -0.5 to -1.8 % per year), but that this fraction decreased substantially to 6% when considering only the more recent period of 2008-2013, when 30% of sites showed significant positive trends in wet deposition concentration. Regionally, positive trends were concentrated in the central and western areas of the continent, while negative trends were concentrated in the eastern areas (Weiss-Penzias et al., 2016). These results are consistent with previous analyses of MDN data: Prestbo & Gay (2009) found significant decreases in concentration in the range of -1 to -2% per year between 1996-2005 in the Northeast and Mid-Atlantic regions, but no significant trends in the upper Midwest (including Minnesota and Wisconsin) or lower Southeast, and Butler et al. (2008) found significant declines in the Northeast and Midwest (defined to include parts of the Ohio River Valley), but no trend in the Southeast from 1998-2005. Focusing on the Great Lakes region, Risch et al. (2012) reported small statistically significant decreases in Hg concentration between 2002 and 2008, but no significant trends in wet deposition, as decreases in concentration were coupled with increases in precipitation.

Several studies have advanced hypotheses to explain the spatial and temporal pattern of trends observed at North American monitoring sites—that is, the lack of significant negative trends and increasing prevalence of positive ones between the early 2000s and early 2010s, particularly in central and western regions of the US, when North American emissions continued to decline by approximately 50% during this period (see Figure B-1). These include: uncertainties in both magnitude and speciation of emissions inventories (Zhang et al., 2016c); decreasing influence of local/regional sources given increasing global background concentration of atmospheric mercury, driven by emissions growth in Asia (Weiss-Penzias et al., 2016); and meteorological and climatological variability (Gratz et al., 2009; Shah & Jaeglé, 2017). Zhang et al. (2016c) find better agreement between modeled and observed twenty-year (1990-2010) trends in elemental mercury and mercury wet deposition in North America and Europe after revising emissions inventories to take into account decreasing emissions from commercial products and artisanal and small scale gold

mining, and changes in flue gas speciation due to adoption of air pollution control devices. Weiss-Penzias et al. (2016), based on their interpretation of spatial patterns in observed wet deposition and concentration patterns, suggest that the recent positive trends in the central and western US may be due increases in the trans-pacific transport of mercury in tropospheric air masses, which have larger influences over these regions. Finally, Gratz et al. (2009) propose that interannual variability in local meteorology—particularly precipitation amount and type—can mask the influence of emissions in wet deposition concentration at a remote northeastern site. Shah & Jaeglé (2017), using a modeling approach, reach similar conclusions on the contribution of precipitation to variability in wet deposition, while also highlighting the importance of meteorological factors that affect the production and export of divalent mercury to free tropospheric air, like subtropical anticyclones.

The goal of this study is to use atmospheric modelling to better understand which of these hypotheszied factors affect the translation of policy-related emissions changes into changes in wet deposition (concentration and flux) in the Great Lakes region on a decadal scale, and to quantify their relative influence. In this work, we use modelling experiments to explore the extent to which variability and trends in these intervening factors, exogenous to policy action, can act as "noise" in the detection of a policy "signal" in monitored wet deposition in the Great Lakes. We consider these dynamics for both regional (North American) and global policy signals. We discuss how the results of this analysis can be used to help interpret observed trends, and the potential implications of these signal-to-noise challenges for policy monitoring and design, for instance in the context of the Minamata Convention.

3.2 Methods

3.2.1 Overall approach

We begin with an analysis of historical observations from 2005-2012, to replicate trends reported in the literature and evaluate the ability of the chemical transport model to capture spatial patterns and magnitudes of wet deposition over the Great Lakes region. Then, in our modelling experiment, we consider a hypothetical step policy change, consistent with existing policy efforts, requiring the application of increased air pollution control devices in

the electricity generation sector that divides this eight-year period, resulting in a four-year pre-policy period and a four-year post-policy period. We consider a regional policy, targeting North America (NA), and a policy that targets the rest of the world (ROW) separately, to evaluate the influence of regional and global "policy signals" on the Great Lakes region. We define "policy signal" as the percent difference between pre- and post-policy period for any given metric (precipitation weighted concentration, deposition, and precipitation). We first evaluate the strength of this signal over the region holding all else constant except emissions in the sector targeted by policy. We then introduce sources of noise—trends and variability in factors that are exogenous to the policy action—and evaluate the extent to which the policy signal can still be detected. These scenarios are summarized in Table 3.2 and the emissions resulting from these scenarios are summarized numerically in Tables 3.3 and 3.4 and visually in Figures 3-1 and 3-2. The scenarios are also described in detail in Section 3.2.4.

As has been noted in the literature, observed patterns and trends of wet deposition—particularly on the decadal scale—are highly dependent on which start and end dates are chosen (Weiss-Penzias et al., 2016); the goal of this analysis is to provide insight into the factors that lead to this variability. In this light, our focus on the eight year period of 2005-2012 is illustrative in that it is meant to illuminate the relative influence of sources of noise that operate on a decadal scale. Given this goal, while selecting a different eight-year window would have been possible, our focus on 2005-2012 is due to the richer availability of data (in monitoring, detailed sectoral emissions, air pollution control technology) during this period, which supports the development of "noise" scenarios that reflect real-world variability. Similarly, our focus on wet deposition (flux and concentration) as the metric of interest is due to the larger spatial and temporal coverage of wet deposition monitoring stations (compared to atmospheric mercury concentration) and its importance as a vector for mercury inputs into the Great Lakes (Lepak et al., 2015)—both factors that contribute to the continuing relevance of wet deposition observations as a means of evaluating past and future policy efforts.

Table 3.1: Technology standard policy change targeting emissions from electricity generation sector. Removal fractions and speciation profiles are based on data from Bullock & Johnson (2011), collected for the US EPA.

	Region	Technology	Emissions
Pre-Policy Period (4 years)	NA	ESP+FGD *Removal: 77.8%; Speciation: 92% Hg(0), 8% Hg(II)*	82 Mg/yr
	ROW	ESP *Removal: 29.4%; Speciation: 26% Hg(0), 74% Hg(II)*	1381 Mg/yr
		Policy Change	
Post-Policy Period (4 years)	NA	SDA+FF+SCR *Removal: 97.8%; Speciation: 49% Hg(0), 51% Hg(II)*	57 Mg/yr
	ROW	ESP+FGD *Removal: 77.8%; Speciation: 92% Hg(0), 8% Hg(II)*	1196 Mg/yr

3.2.2 Analysis of observed trends

To assess historical trends in the Great Lakes region between 2005 and 2012, we use MDN measurements of weekly integrated precipitation depth (mm), mercury concentration (ng/L), and calculations of mercury wet deposition flux (ng/m^2) based on these measurements(NADP, 2017). We define monitoring sites in the Great Lakes region broadly to include all sites in the eight states and the one province abutting the lakes (Minnesota, Wisconsin, Illinois, Indiana, Ohio, Michigan, Pennsylvania, New York, and Ontario), which are shown in Figure 3-3. For analysis of historical trends, and model-observation comparison, we consider only sites with >75% data for each year over this period.

The Seasonal Mann Kendall trend test (SMK) and Theil-Sen estimator of slope (Gilbert, 1987; Helsel & Hirsch, 2002; Burkey, 2006) were used to assess the significance, sign, and magnitude of trends in monthly means of precipitation weighted concentration, precipitation depth, and wet deposition flux, across years. The SMK is a non-parametric test for the presence of a monotonic trend commonly used for environmental monitoring time series with seasonal variation, and the Theil-Sen estimator is a non-parametric method of estimating the slope of the linear trend (Gilbert, 1987; Helsel & Hirsch, 2002). In our analysis, each month is treated as a separate "season," yielding 12 test statistics which are then combined to yield an annual statistic (Gilbert, 1987; Burkey, 2006). In all statistical analyses in this work, we define the threshold for significance as p<0.1.

Table 3.2: Descriptions of modeling experiments. For all experiments, the step policy change (described for the Policy Only experiment) is applied after year 4, leading to 4 year pre- and post-policy periods.

Modeling Experiment		Description	Emissions Years	Meteoro- logical Years
No Noise	Policy Only	See Table 3.1 for a full description of the policy scenario; all else held constant except for policy change	2005 x 8	2005 x 8
Trend	Energy and Economic Trends	Trends in energy and economic activity for all sectors, and control technology adoption in sectors not targeted by regulation	2005-2012	2005 x 8
	Product Emission Trends	Decreasing trend in additional source of Hg(0) emissions from commercial products	2005 x 8	2005 x 8
Variability	Removal Variability	Interannual variability in the removal fraction of air pollution control devices in the power generation sector	2005 x 8	2005 x 8
	Speciation Variability	Interannual variability in fraction Hg(0) of end-of-pipe emissions in the power generation sector	2005 x 8	2005 x 8
	Meteorological Variability	Interannual variability in meteorology (e.g. precipitation magnitude and type, wind patterns)	2005 x 8	2005-2012
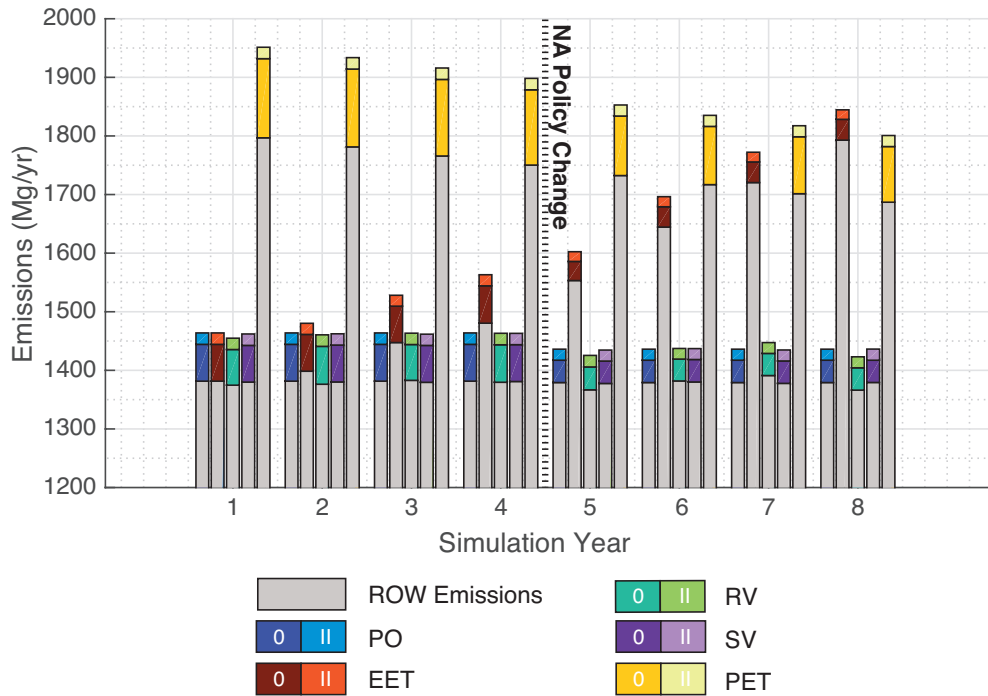
Figure 3-1: Summary of emissions by year under NA policy and noise scenarios. NA speciated emissions are shown in color (Hg0 = darker shade; Hg2/HgP = lighter shade), and ROW emissions, not speciated, are shown in grey. NA policy is implemented as a step change between 2008 and 2009. The "Meteorological Variability" and "Sectoral Trends + Meteorological Variability" scenarios are not shown as in the first, emissions are identical to "Policy Only," and in the second, emissions follow "Sectoral Trends." Note that the y-axis begins at 1200 Mg/yr.

3.2.3 Chemical Transport Modeling

Model description

To model mercury deposition, we use the GEOS-Chem (version 10-01; `http://acmg.seas.harvard.edu/geos/`) coupled atmosphere-ocean-land mercury simulation, which includes a 3-D atmosphere (Holmes et al., 2010), and 2-D land (Selin et al., 2008) and ocean modules (Soerensen et al., 2010). Globally, we use a horizontal resolution of 4° latitude x 5° longitude, while over North America (10° to 70° latitude, -140° to -40 ° longitude), we also use a finer 1/2° x 2/3° resolution, using a one-way nested-grid simulation developed by Zhang et al. (2012), with boundary conditions from the global simulation. The atmosphere is modeled with 47 vertical layers in both the global and nested simulations. In the at-
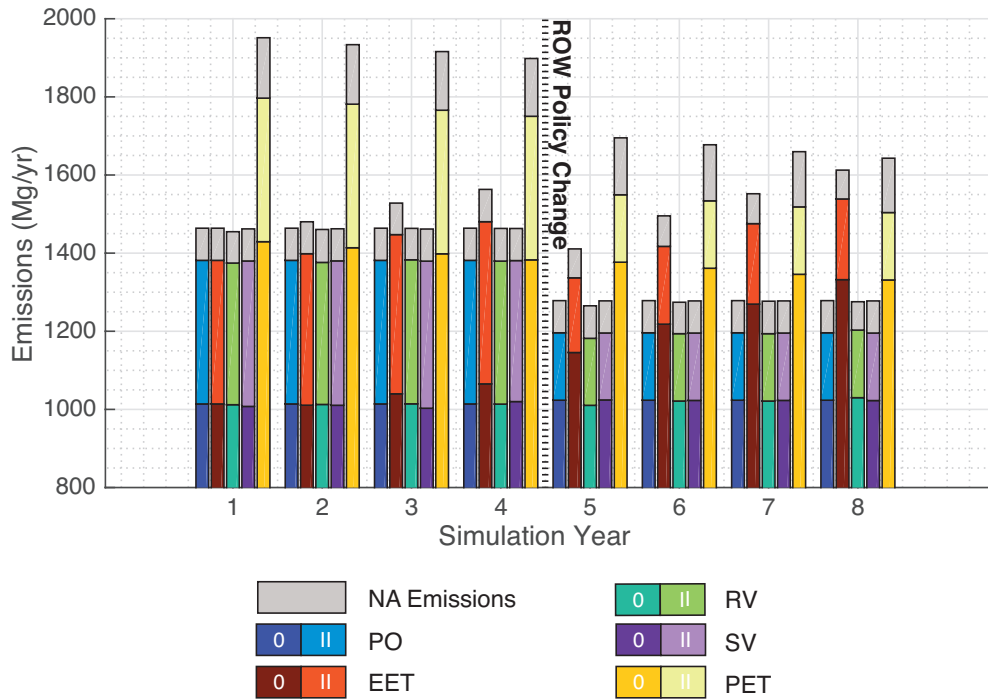
Figure 3-2: Summary of emissions by year under ROW policy and noise scenarios. ROW speciated emissions are shown in color (Hg0 = darker shade; Hg2/HgP = lighter shade), and NA emissions, not speciated, are shown in grey. ROW policy is implemented as a step change between 2008 and 2009. The "Meteorological Variability" and "Sectoral Trends + Meteorological Variability" scenarios are not shown as in the first, emissions are identical to "Policy Only," and in the second, emissions follow "Sectoral Trends." Note that the y-axis begins at 800 Mg/yr.

mosphere and ocean, the model tracks inorganic mercury in two forms: gaseous elemental mercury, Hg(0), and divalent mercury, Hg(II), which in the atmosphere is modeled with equilibrium partitioning between gas and particle-bound phase based on temperature and aerosol concentration (Amos et al., 2012). Oxidation of Hg(0) to Hg(II) in the atmosphere, and in-cloud reduction of Hg(II) to Hg(0) follow the mechanisms described in Holmes et al. (2010). Bromine is assumed to be the primary oxidant in a two-step process (Goodsite et al., 2004, 2012). Bromine concentrations are taken from a full-chemistry GEOS-Chem simulation described in Parrella et al. (2012). In-cloud reduction of Hg(II) to Hg(0) (see Pongprueksa et al., 2011) follows Holmes et al. (2010). Wet deposition, the metric of interest in this study, results from large-scale washout and rainout, and scavenging in moist convective updrafts of Hg(II), as described in Liu et al. (2001); Holmes et al. (2010); Amos

et al. (2012).

Meteorology

In this work, GEOS-Chem mercury simulations are driven by assimilated meteorological fields from the NASA Goddard Earth Observing System, Version 5 (GEOS-5.2.0; `http://gmao.gsfc.nasa.gov/GEOS/`). The temporal coverage of GEOS-5.2.0 is 2004 to 2012, with a native resolution of $1/2°$ x $2/3°$. GEOS-Chem mercury simulations using this meteorological data have been extensively compared to wet deposition and concentration measurements over the region of interest (including in Amos et al., 2012; Zhang et al., 2012, 2016c). In the remainder of the text, we use the term "meteorological year" to refer to the year with which meteorological data is associated (see Table 3.2). All simulations are initialized with a three year spin-up.

Emissions

Global anthropogenic mercury emissions are based on data from the Emission Database for Global Atmospheric Research (EDGAR) v4.tox2 inventory, which provides a time-series of spatially-resolved, speciated emissions at $0.1°$ x $0.1°$, from 1970 to 2012 (Muntean et al., 2014, 2017). This sectorally disaggregated inventory combines international activity data statistics, emissions factors, and data on control technology performance and adoption (Muntean et al., 2014, 2017). For the electricity generation sector—the sector targeted by policy in our model experiment—we begin with EDGAR emissions without end-of-pipe air pollution controls. Emissions from this sector, that are based on activity data and emissions factors without additional reductions from end-of-pipe control, are then modified according to the technology standard policy change scenario described in Section 3.2.4 and Table 3.1. In the remainder of the text, we use the term "emissions year" to refer to the year with which activity data, emissions factors, and technology specifications are associated. For the electricity generation sector, "emissions year" corresponds to activity data and emissions factors only, as technology specifications are set in the policy scenario.

55

3.2.4 Model Experiment

Policy scenarios

In our experiment, we model hypothetical policy change scenarios targeting the electricity generation sector, summarized in Table 3.1. We use a simplified policy treatment, assuming homogenous technology standards applied as a step change, to more easily diagnose the signal and noise dynamics that arise from introducing variability in policy implementation—which in our analysis is limited to variability in technology performance and resulting speciation from the sector-wide technology standard (see Section 3.2.4).

Our policy change scenarios for the electricity generation sector are modeled after existing policy efforts. To distinguish regional and global policy influences on the Great Lakes region, we consider NA and ROW policies separately. We apply a homogenous technology standard to the sector, requiring 100% adoption. For NA, in the pre-policy period, we assume the use of PM and SO_2 controls—specifically cold-side electrostatic precipitators (ESP) and wet flue gas desulfurization (FGD) in pulverized coal boilers. In the post-policy period, we assume a configuration of SO_2, PM, and NO_x controls with higher mercury removal—specifically, spray dry absorber (SDA), fabric filter (FF), and selective catalytic reduction (SCR). This shift is similar to the actions that some plants would undertake to comply with the Mercury and Air Toxics Standards (US EPA, 2011a). For ROW, in the pre-policy period, we assume the use of PM controls only, in the form of ESP. In the post-policy period, we assume the use of PM and SO_2 controls, through ESP and FGD. This technology shift is similar to the actions that plants have undertaken in China to comply with air quality regulations (Zhang et al., 2015a; Zhao et al., 2015).

The removal fractions and speciation profiles resulting from these technology standards are listed in Table 3.1. These values are based on emissions testing data collected by Bullock & Johnson (2011) for the US EPA. Normal distributions, truncated between 0 and 1, for removal fraction and fraction Hg(0) were fit for each configuration, with goodness of fit evaluated using the Kolomogorov-Smirnov test at 5% significance. Values shown in Table 3.1 represent the mean of the distribution.

Our simplified policy scenarios do not reflect the real-world complexity of the power generation sector, which is globally heterogenous and time-varying in fuel type, and plant and air pollution control technologies. Moreover, many air pollution policies targeting this

sector use performance standards/emission limits or market mechanisms, that allow for some flexibility in pollution control approach, accounting for local context (Keohane & Olmstead, 2016).

Simulations

Realistic time varying emissions and meteorology (Realistic). For the purposes of model-observation comparison, we conduct a simulation with realistic time varying emissions (using the EDGAR inventory) and meteorology (GEOS-5) between 2005 and 2012.

Policy only simulation (PO). We evaluate the strength of the policy signal—the difference between pre- and post-policy period wet deposition—holding all else constant except the technology standard in the power generation sector. Meteorological year 2005 and emissions year 2005 are therefore repeated throughout the eight year period. As shown in Figures 3-1 and 3-2, the resulting emissions are constant in each four year period, with a step change occurring between 2008 and 2009. For NA policy, ROW emissions remain at pre-policy levels in the post-policy period, and vice versa.

Energy and economic trends simulation (EET). We consider the effect of trends in emissions due to changes in energy and economic activity, which are exogenous to the policy, on the strength and significance of the policy signals in the Great Lakes region. Between 2005 and 2012, global anthropogenic emissions are estimated to have increased, due to increased activity in power generation, cement production, metals production, and artisanal and small-scale gold mining (though there is substantial uncertainty associated with this source category) (UNEP, 2013a; Muntean et al., 2014, 2017). These global inventories indicate that industrial activity in Asia in particular was a key driver of this growth (UNEP, 2013a; Muntean et al., 2014). In North America, emissions were estimated to be relatively stable between 2005 and 2008, while a combination of macroeconomic trends and regulation that affected energy and industrial activity contributed to lower emissions overall between 2009 and 2012 (Muntean et al., 2014, 2017; Weiss-Penzias et al., 2016). We use emissions years 2005-2012 from EDGAR, while repeating meteorological year 2005 throughout the eight year period.

Product emission trend simulation (PET). The use of mercury in commercial products has been hypothesized to be an often unaccounted for source of Hg(0) to the

atmosphere, with emissions peaking in the 1970s and declining since then (Horowitz et al., 2014). We evaluate the impact of a large, declining source of Hg(0) on policy signals, using product emission magnitudes and spatial distributions from Zhang et al. (2016c). Because Zhang et al. (2016c) provide inventories for 2000 and 2010, we linearly interpolate a decreasing trend in each of the geographic regions they define between 2005 and 2010, and extend this trend to 2012. Because this product emissions inventory was harmonized with a base inventory from Streets et al. (2011), it is possible that some emissions are double-counted when combined with the EDGAR inventory (for instance, from waste incineration). However, given the purpose of this simulation—to investigate the impact of a large, and declining source of Hg(0) emissions—we do not expect these inconsistencies to change our interpretation. Emission year and meteorological year 2005 are repeated throughout the eight years.

Removal variability simulation (RV). Variability in the performance of air pollution control devices can be due to variabilities in fuel characteristics and operating conditions (Wu et al., 2010; Zhao et al., 2015; Zhang et al., 2016b). To investigate the potential impact of such variability on the policy signal, we treat the removal fraction of each air pollution control configuration probabilistically each year. Rather than assuming a static removal fraction for each air pollution control configuration, we bootstrap a normal distribution for the population mean from the sample data from Bullock & Johnson (2011), described in Section 3.2.4, and randomly select the removal fraction for each year from this bootstrapped distribution. The distributions from Bullock & Johnson (2011) are shown in SI Figure B-2, and the parameterizations for distributions of the resulting population means are listed in Table B.1. Speciation is deterministic in this simulation. We hold emission year and meteorological year constant at 2005.

Speciation variability simulation (SV). The same procedure used to probabilistically generate removal fraction for each year and air pollution control configuration is applied to % Hg(0), using data from Bullock & Johnson (2011). (Note: % Hg(II) = 1 - % Hg(0)) Removal fraction is deterministic in this simulation. Emissions year and meteorological year are held constant at 2005.

Interannual meteorological variability simulation (MV). Interannual variability in meteorology—including in temperature, precipitation volume, and precipitation type— can impact mercury chemistry and transport, with implications for wet deposition (Gratz

et al., 2009; Shah & Jaeglé, 2017). We simulate meteorological years 2005-2012, while holding the emissions year constant at 2005, resulting in an identical emissions trajectory as in the "Policy Only" case.

Statistical analysis

For each modelled grid cell in the Great Lakes region, we evaluate the magnitude and statistical significance ($p < 0.1$) of the difference in wet deposition (precipitation weighted concentration, flux, precipitation) between pre- and post-policy periods using the seasonal Hodges-Lehmann (HL) estimator of difference and the seasonal Mann-Whitney-Wilcoxon (MWW) rank sum test (Helsel & Hirsch, 2002; Crawford et al., 1983). The MWW rank sum test and the HL estimator are non-parametric equivalents of a two sample t-test and difference of means, modified for seasonality (Helsel & Hirsch, 2002; Crawford et al., 1983). The HL estimator is the median value of all possible differences between observations from the first and second period. As in our use of the SMK trend test, each month is considered a separate season in our analysis.

3.3 Results

3.3.1 Observed trend

Figure 3-3 shows 2005-2012 trends in wet deposition, precipitation weighted concentration, and precipitation for MDN monitoring sites in the Great Lakes region, expressed as $\Delta\%$ per year. We find few significant trends in wet deposition over this period: significant negative trends at PA30 and PA47 are on the order of 2 to 3% per year—in one case, driven by a significant decrease in concentration—and a significant positive trend at ON07 is greater than 3% per year. Some non-significant decreasing trends in precipitation weighted concentration are observed downwind of major US emission sources in Ohio and Pennsylvannia. Non-significant increasing trends in wet deposition around the upper lakes during this period may be due to both increasing precipitation and increases in concentration. The spatial pattern of these results is consistent with findings in Risch et al. (2012) and Weiss-Penzias et al. (2016), though direct comparison is difficult due to differences in statistical methods and time periods.
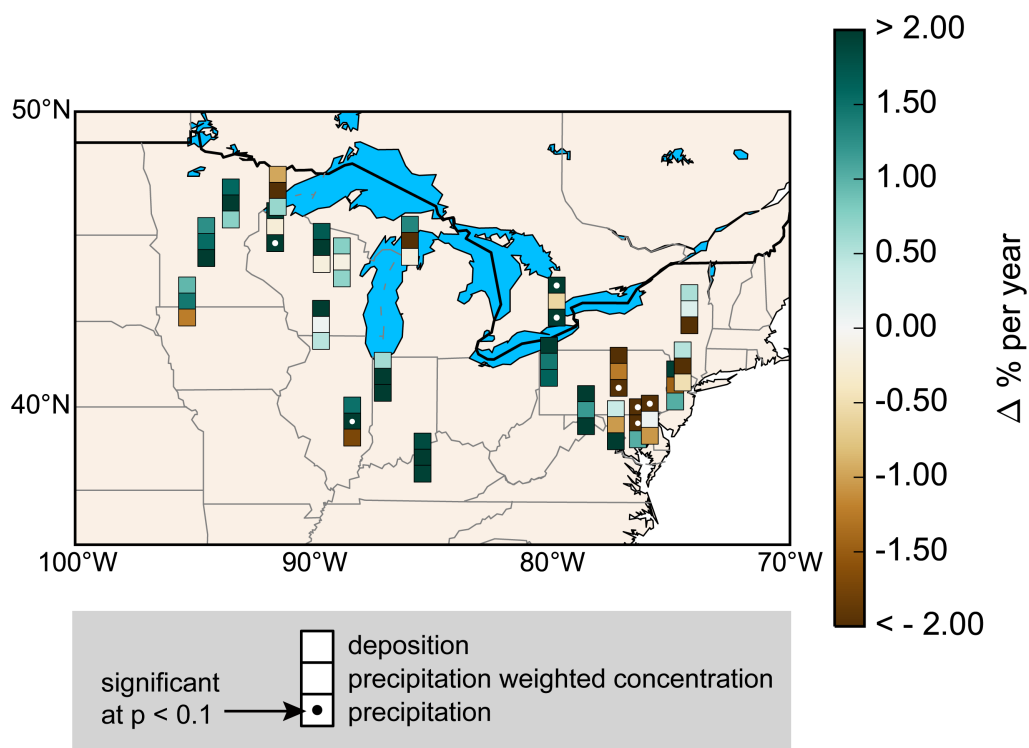
Figure 3-3: Observed 8 year (2005-2012) trend at MDN monitoring sites with $\geq 75\%$ data availability. For each site, the trend in wet deposition (top), precipitation weighted concentration (middle), and precipitation (bottom) are shown. Trends significant at p < 0.1 are indicated with a dot. We evaluate significance of trends using the Seasonal Mann-Kendall trend test, and quantify the magnitude of the trend using Theil-Sen's estimator of slope.

3.3.2   Model Evaluation

Figure 3-4 shows a comparison of modelled ("Realistic" simulation) and observed annual wet deposition averaged from 2010-2012, when the spatial coverage of observational data is greatest (NADP, 2017) and when underlying data in the EDGAR emissions inventory is most detailed (Muntean et al., 2014, 2017). The model reproduces the spatial pattern of annual wet deposition, with the highest values in the Ohio River Valley. Magnitudes are underestimated in the central US region (e.g. Nebraska and Kansas), contributing to lower average modelled wet deposition at MDN sites in the depicted region of 9.3 $\mu g/m^2$ compared to the MDN average of 10.2 $\mu g/m^2$.

An aggregated time series of monthly modelled and observed values is provided in SI
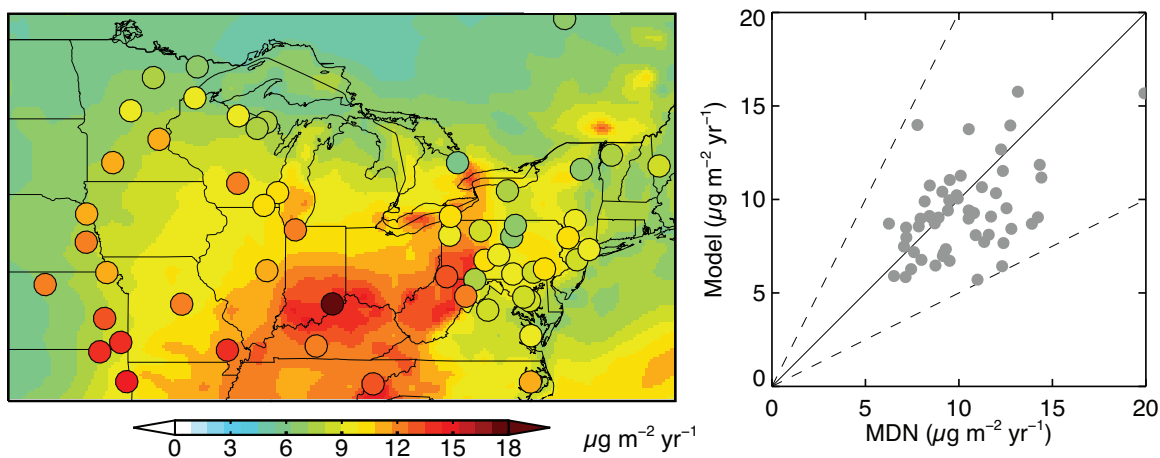
Figure 3-4: Comparison of modeled (background) and observed (filled circles) 2010-2012 average annual wet deposition.

Figure B-3. The Pearson correlation coefficient for the modelled and observed time series across the monthly site averages, r, is 0.41, with an individual site maximum of $r = 0.70$ and minimum of $r = 0.13$. Temporally, correlation between model and observations is stronger in the recent period of 2009-2012, when $r = 0.65$, while model predicted wet deposition magnitudes are biased low between 2005-2008. Lower estimates of emissions from key North American emission source categories, like coal combustion, in EDGAR compared to other inventories may contribute to this discrepancy (Muntean et al., 2014). While the model reproduces the general seasonal cycle at most sites, it underestimates summertime peaks, while wintertime values tend to be larger than those observed. Underestimates of precipitation in GEOS-5 in the Midwest and lower snow collection efficiency, compared to rain, of MDN samplers contribute to these biases (Zhang & Jaegle, 2013). SI Figure B-4 compares modelled and observed trends for 2005-2012, for sites with >75% data. The model predicts increasing trends South of the Great Lakes, and decreasing trends to the North and East. MDN observations indicate more sites with decreasing trends to the Southeast of the lakes, though regions with the strongest increasing trends are generally in agreement.

3.3.3 Policy Only Simulation

Figure 3-5 maps the pre-and post-policy emissions difference resulting from the NA and ROW technology standards. For the NA policy, the 30% decrease in emissions (25 Mg/y) occurs predominantly in the Northeastern US, where many coal-based power generating units are located to the South of the Great Lakes in the Ohio River Valley. Due to the nature of the prescribed air pollution control configuration, which promotes the oxidation of Hg(0) to Hg(II) and facilitates capture of this soluble form of mercury (Zhang et al., 2016b), these reductions are predominantly in the form of Hg(0) rather than Hg(II) (seen in Figure 3-1). Because Hg(0) is long-lived in the atmosphere, with an estimated lifetime of 0.5-1 year compared to a lifetime of days to weeks for Hg(II) (Holmes et al., 2006; Lindberg et al., 2007), these speciation differences have important implications for transport (Driscoll et al., 2013; Selin, 2009). The 13% emissions decrease (185 Mg/y) under ROW policy is predominantly in the form of Hg(II), as the adoption of FGD in addition to PM control increases the removal of gaseous oxidized mercury (Zhang et al., 2016b). These decreases are largest over East and South Asia, and Western Europe.

Simulated deposition differences in the PO simulation are shown in Figure 3-6. Figure B-5 in Appendix B plots differences in precipitation weighted concentration, with numerical results at MDN sites summarized in Table B.3. Note that here, Δs refer to the pre- and post-policy period difference, rather than a percent change per year. Table 3.5 gives a numerical summary of differences sampled at MDN site locations (that were active at any time during this period) for this and all subsequent simulations. Holding all else constant, the NA policy results in statistically significant decreases in deposition at all simulated grid cells in the region ranging from -0.4 to -2.4 %. The regions with strongest decreases trace the footprint of local power generation emission sources in the Ohio River Valley and the western edge of Lake Erie. The average difference in deposition at MDN sites due to policy is -1.01%. The relative spatial homogeneity of the policy difference is due to the speciation of the modeled emissions decrease (predominantly elemental mercury), resulting in a more diffuse impact on deposition.

The ROW policy also leads to statistically significant decreases in simulated wet deposition at all grid cells ranging from -0.9 to -2.5%, with an MDN site average of -1.71%. The magnitude of this decrease is even more spatially homogenous than for the NA policy,

reflecting the influence of ROW emission decreases on global background concentrations of mercury. The effect of ROW policy, in terms of $\Delta\%$ deposition is therefore weakest where the contribution of local emission sources to deposition is strongest—for instance, around metal smelting facilities near Lakes Ontario, Erie, and Michigan.
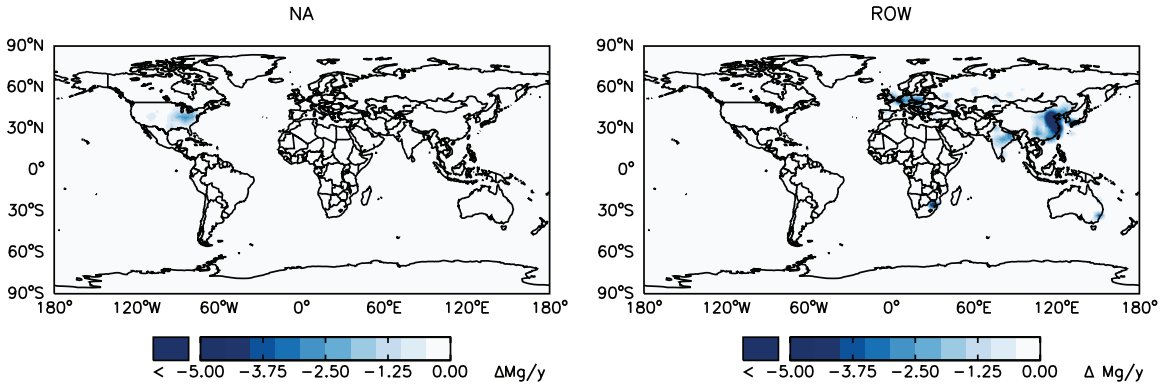
Figure 3-5: The spatial distribution of emissions changes associated with NA (left) and ROW (right) policy in Mg/y, between the pre- and post-policy period.

(a) NA Signal

(b) ROW signal
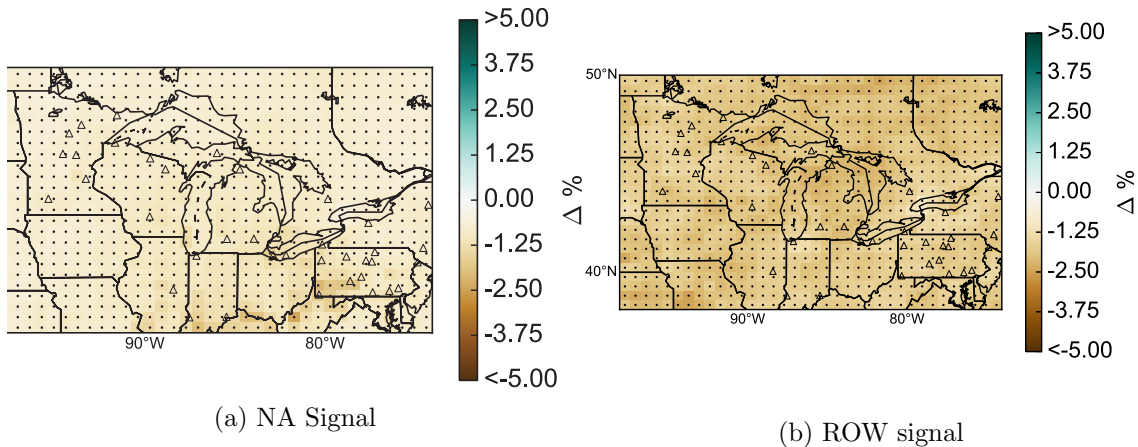
Figure 3-6: Change in deposition (%) between pre-policy and post-policy period, for Policy Only simulation. Grid cells with a significant (p<0.1) change are indicated with a dot. For reference, locations of monitoring stations are indicated with triangles.

3.3.4 Emission Trend Simulations

We consider two categories of emissions-related trends exogenous to the policy that may act as "noise" in detecting the policy signal in wet deposition: energy and economic activity trends (EET) that lead to globally increasing emissions, and product emission trends

(PET) leading to globally decreasing emissions. In our PET simulations, we add a linearly decreasing source of Hg(0) emissions from commercial products. These products result in an additional 488 Mg of Hg(0) in 2005, 15% of which is located in NA. Global emissions from products decrease by 150 Mg over the eight-year period.

In our EET simulation with NA policy, ROW emissions monotonically increase between 2005 and 2012 from 1381 to 1790 Mg/y (see Table 3.3). In NA, energy and economic trends drive emissions decreases overall between 2005 and 2012 (with a large drop between 2008 and 2009 due to the economic recession), leading to a larger emissions gap between pre- and post-policy periods, compared to the PO simulation. With the ROW policy (Table 3.4), total emissions decrease sharply between 2008 and 2009 due to decreases in Hg(II) emissions from the technology standard, however, by 2012, total emissions exceed the highest emissions year in the pre-policy period (1613 Mg/y in 2012 compared to 1563 Mg/y in 2008). For NA, even without policy, reduced activity in energy and other sectors leads to a decrease in emissions in the post-ROW-policy period

In the EET simulation, the area over which a statistically significant decrease is detected between periods is limited to the eastern portion of the Great Lakes region. The area surrounding Lake Superior is highly influenced by increasing global emissions, as seen in Figure 3-7, showing the $\Delta\%$ in wet deposition. Figure 3-7 also highlights the large impact of local emission sources on Great Lakes deposition: though total global emissions are monotonically increasing in the NA policy simulation, we find statistically significant decreases in Illinois, Michigan, Indiana, Ohio, Pennsylvania, New York, and Southern Ontario (an average -1.31% decrease at the 17 MDN sites with significant differences) driven by NA emission reductions. Comparison between the ROW and NA plots in Figure 3-7 suggests that the large differences simulated in Southern Ontario and upstate New York under both NA and ROW policies are due to decreasing activity in metals production (rather than the simulated NA power generation policy) that substantially reduce Hg(II) emissions from iron production facilities adjacent to Lake Ontario. The additional benefit of the NA policy targeting the power generation sector occurs in Indiana, Ohio, and the Northern areas of Kentucky and West Virginia (where fewer MDN sites are located). Results for precipitation weighted concentration are shown in Figure B-6 and Table B.3.

In the PET simulations, we see statistically significant decreases in wet deposition and precipitation weighted concentration at all grid cells in the modelled region between the pre-

and post-policy period, exceeding the magnitude of the PO simulation differences (Figures B-7 and B-8). For the NA policy, differences range from -1.5 to -4.9%, while for the ROW policy, differences range from -1.2 to -4.5%.The modelled differences are more reflective of the trend in commercial product emissions, which in this case, represents a less aggressive % decrease than the Policy Only simulation. This influence is also seen in the increased spatial homogeneity of differences in the NA policy simulation. The smaller fluctuations in emissions due to our simulated policy are harder to discern against a higher global background of atmospheric mercury, except in the immediate vicinity of local emissions sources.

(a) NA Signal

(b) ROW signal

Figure 3-7: Change in wet deposition (%) between pre-policy and post-policy period, for Energy and Economic Trends simulation. Grid cells with a significant (p<0.1) change are indicated with a dot. For reference, locations of monitoring stations are indicated with triangles.

3.3.5 Air Pollution Control Variability Simulations

We conduct two simulations that explore how variability in the performance of air pollution control devices in the regulated sector affects the pre- and post-policy wet deposition difference: one treats the removal fraction of pollution control as a probabilistic variable while holding speciation constant (RV), while the other considers the fraction of flue gas emissions that are in the form of Hg(0) probabilistically, while holding removal fraction constant (SV). The resulting variability in emissions *within* pre- and post-policy periods contributes to a reduced median difference *between* the periods. For removal fraction, this variability is primarily due to ESP and ESP+FGD, while for fraction Hg(0), SDA+FF+SCR is the

primary driver (see Figure B-2). As an example of the potential magnitude of real-world interannual variability in emissions, we include an analysis and comparison of the US EPA's Toxics Release Inventory and National Emissions Inventory in the Great Lakes region in the Supplemental Information (Section B.4).

Figure 3-8 shows resulting wet deposition differences for the RV simulation. Results for RV concentration and SV deposition and concentration are shown in Figures B-10-B-12. For both kinds of variability, and for both policies, though differences remain negative and statistically significant at all grid cells, the magnitude of the difference decreases (Figure 3-8 provides one example). For NA policy, removal fraction variability leads to a modelled difference of -0.84% at MDN sites (compared to -1.01% in the PO simulation, a reduction of 17%), with the largest simulated differences in the Ohio River Valley. This difference is larger than that modelled at MDN sites under ROW policy (-0.34%), reversing the result in the PO simulation (see Figure B-9). Relatively speaking, speciation variability has a larger dampening effect than removal variability on the NA policy signal (the difference at MDN sites is -0.66%, which is 35% smaller than in the PO simulation), while the opposite is true for the ROW policy signal. This result further emphasizes the importance of local emissions of Hg(II) to wet deposition in the Great Lakes region—even small variations in divalent mercury can weaken the effect of overall policy-related emissions decreases. In contrast, ROW policy affects Great Lakes wet deposition primarily through contributions to total atmospheric burden.

3.3.6   Interannual Meteorological Variability Simulation

We simulate the policy change while including the historical interannual meteorological variability from 2005-2012. Figures 3-9,3-10, and 3-11 show the pre-and post-policy period differences in wet deposition, concentration, and precipitation, respectively. Similar to the Energy and Economic Trends simulation, the resulting pattern of deposition differences has regions of positive and negative difference, ranging from <-20 to >20 %—a much larger spread than any other simulation. The number of grid cells showing statistically significant differences also decreases dramatically. These significant decreases are predominantly at higher latitudes and include areas in Ontario, Quebec, New York, and the Northern region of Minnesota. Large, but not significant, increases in deposition are simulated South of the
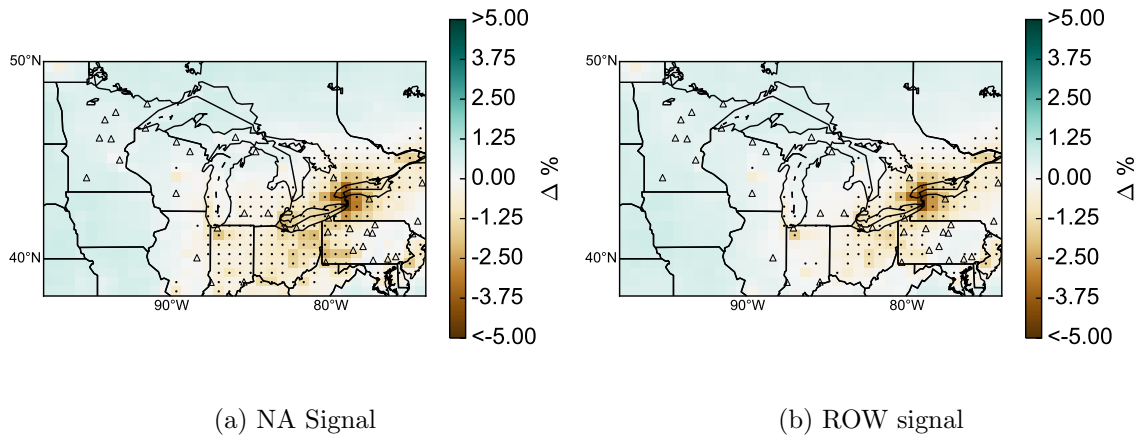
(a) NA Signal
(b) ROW signal
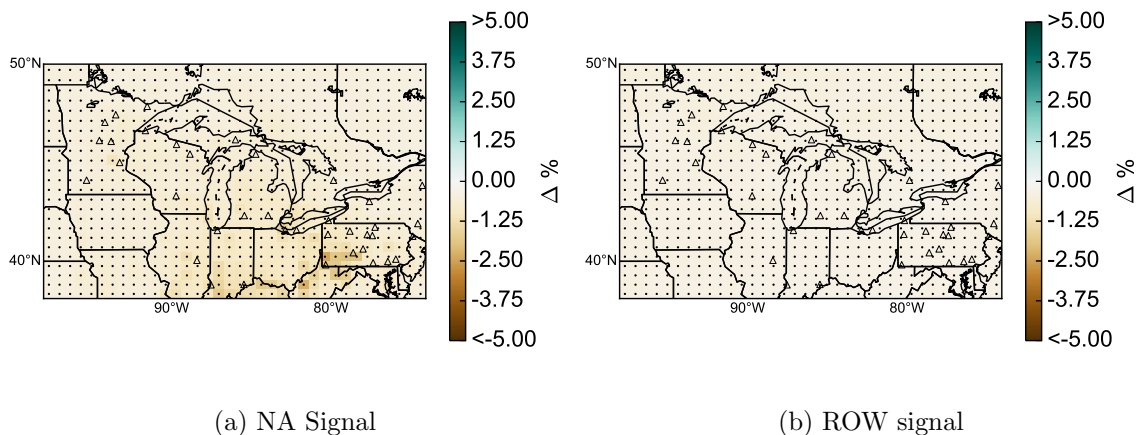
Figure 3-8: Change in wet deposition (%) between pre-policy and post-policy period, for removal fraction variability simulation. Grid cells with a significant ($p<0.1$) change are indicated with a dot. For reference, locations of monitoring stations are indicated with triangles.

Great Lakes.

The similarity of the results for the NA and ROW policy simulations with meteorological variability indicate that meteorological influence is larger than that of emissions—at least for the magnitude of emissions changes considered here. For instance, comparison of Figure 3-9, showing deposition changes, and Figure 3-11, showing precipitation changes, demonstrates that variability in precipitation volume alone can account for much of the simulated pattern in deposition change. Moreover, that the spatial pattern of deposition change in this simulation captures many of the features in Figure 3-3, our analysis of 2005-2012 trends at MDN sites, speaks to the extent to which interannual meteorological variability is a driving force in observed wet deposition.

3.4 Discussion and Implications for Policy Monitoring and Evaluation

Our modelling results highlight the potential challenges of detecting statistically significant policy-related changes in Great Lakes wet deposition within the short-term, given the magnitudes of realistic emissions changes and sources of confounding "noise," exogenous to the policy change. Our simplified policy scenario for the electricity generation sector results in 30% (25 Mg) and 13% (185 Mg) step decreases in emissions from NA and ROW, re-

(a) NA Signal

(b) ROW signal
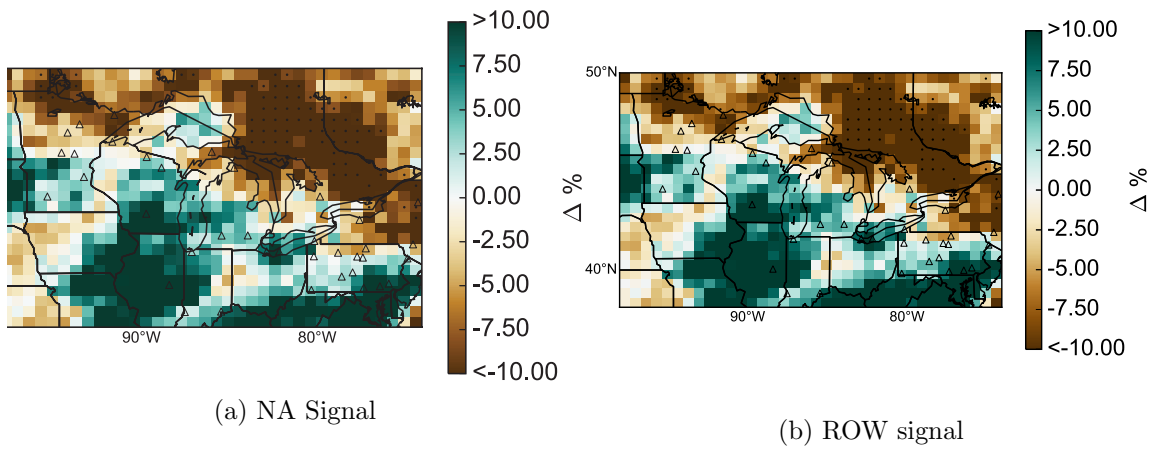
Figure 3-9: Change in wet deposition (%) between pre-policy and post-policy period, for interannual meteorological variability simulation. Note the larger color bar range of -10 to 10%, compared to the other plots. Grid cells with a significant (p<0.1) change are indicated with a dot. For reference, locations of monitoring stations are indicated with triangles.

(a) NA Signal

(b) ROW signal
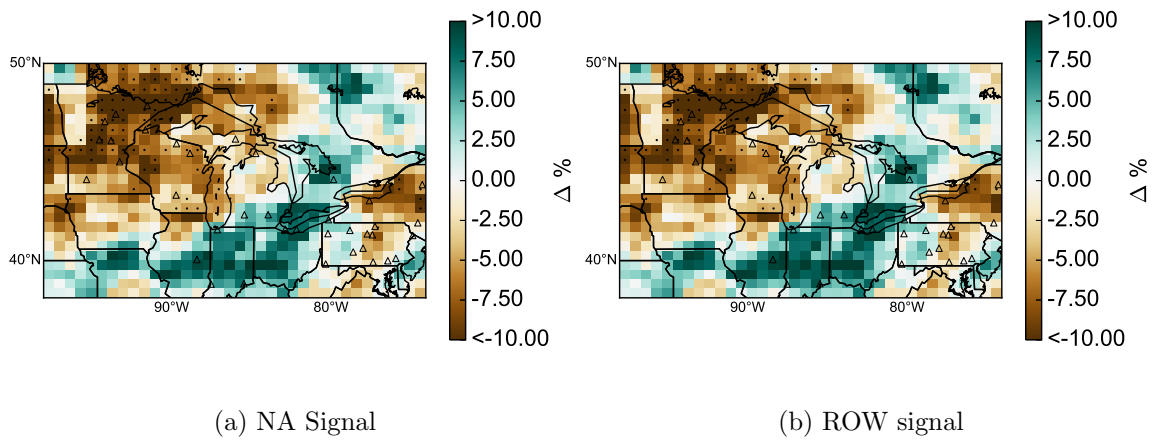
Figure 3-10: Change in precipitation weighted concentration (%) between pre-policy and post-policy period, for interannual meteorological variability simulation. Note the larger color bar range of -10 to 10%, compared to the other plots. Grid cells with a significant (p<0.1) change are indicated with a dot. For reference, locations of monitoring stations are indicated with triangles.
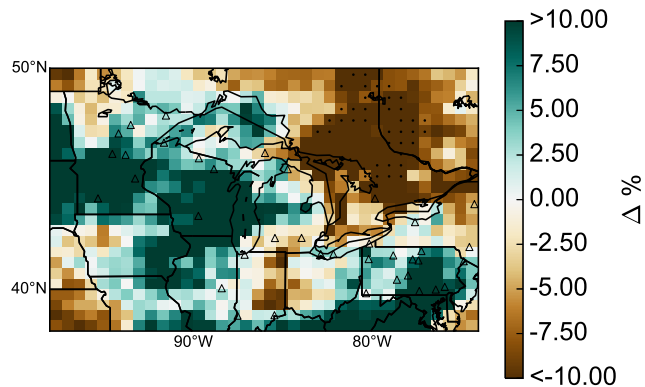
Figure 3-11: Change in precipitation volume (%) between pre-policy and post-policy period, for interannual meteorological variability simulation. Note the larger color bar range of -10 to 10%, compared to the other plots. Grid cells with a significant (p<0.1) change are indicated with a dot. For reference, locations of monitoring stations are indicated with triangles.

spectively, over successive four-year periods. These emissions decreases translate into pre- vs. post-policy deposition decreases ranging from -0.4 to -2.4% and -0.9 to -2.5% in the Great Lakes region, holding all else constant. The introduction of global trends in emissions with realistic magnitudes—based on energy and economic activity (increasing trend $\approx +50$ Mg/y) and commercial product emissions (decreasing trend ≈ -20 Mg/y)—reduces the areas where the policy signal can be detected to the immediate vicinity of targeted emissions sources, as these global trends dominate elsewhere. We find that the introduction of variability in emissions and meteorology can also obscure policy signals. In our simulations with variability in the magnitude and speciation of emissions, based on air pollution control test data, even a relatively small amount of year to year variability within pre- and post-policy periods reduced the magnitude of the simulated deposition difference between periods at MDN sites compared to the PO simulation—by up to 35% for the NA policy, and up to 80% for the ROW policy. Even more influential, however, is interannual meteorological variability, which drove deposition differences of more than ±20% in some areas of the Great Lakes in our simulation, greatly exceeding the changes associated with both the regional and global modelled policy. In the real world, these sources of "noise" that we have treated separately here, from exogenous trends in emissions to variability in technical

or natural systems, operate simultaneously, further complicating the task of attributing observed changes in deposition to specific policy-action.

Our emissions trend results point to the continued importance of North American policy for the Great Lakes region, even in the face of potentially increasing global background concentrations of mercury. As North American emissions represent a smaller fraction of the anthropogenic total with continued emissions growth elsewhere, global emissions may have a larger impact on regional wet deposition (Zhang & Jaegle, 2013). However, the results from our energy and economic trends simulation highlight the extent to which some areas of the Great Lakes region are influenced by local/regional sources, supporting results from monitoring campaigns (Lynam et al., 2016). The persistence of these areas is perhaps surprising: in our simulation of NA policy with global energy and economic trends, even though global emissions increases far outweigh regional policy-related emissions decreases ($\approx +400Mg$ vs. $\approx -30Mg$), statistically significant wet deposition decreases before and after policy on the order of -1% can still be detected in Indiana and Ohio. Our simulations indicate that strategic location of monitoring sites near emissions sources targeted by policy may compensate for noise from exogenous trends in emissions. For communities living in the Great Lakes then, where coal combustion, metals production, and incineration facilities are located, there remain opportunities to build on past progress in local/regional emissions decreases, to achieve further reductions in locally-driven wet deposition.

Another key finding from our work is that variability in emissions—potentially due to stochastic processes in social and technical systems—can greatly attenuate our ability to detect statistically significant trends or differences in wet deposition at monitoring sites. The large epistemic uncertainties—that is, uncertainty due to imperfect knowledge (Stern & Fineberg, 1996; Walker et al., 2003; Roy & Oberkampf, 2011)—in anthropogenic emissions inventories (in the range of $\pm 30\%$) are widely acknowledged to be a challenge for mercury modelling, monitoring, and policy evaluation (UNEP, 2013a). However, our simulations demonstrate that even if "true" emissions values are known, year-to-year variability in these emissions—in our simulations, driven by variability in air pollution control technology performance (Wu et al., 2010; Zhao et al., 2015; Zhang et al., 2016b), but potentially also from other sources, like fluctuations in economic activity—can dampen a policy effect. Because they are labor-intensive to produce, many emissions inventories are released at multi-year intervals, with users linearly interpolating between these years. However, these

assumed linear changes between data points may elide true interannual variability, resulting in larger and more statistically significant predicted effects in environmental concentrations and fluxes than can be actually observed. In the absence of continuous emissions monitoring for mercury, there may be a tradeoff between ensuring more accurate point estimates (i.e. reducing epistemic uncertainty), and better capturing temporal variability (i.e. quantifying aleatory uncertainty) (see Ambrose et al. (2015) for a comparison on TRI and NEI against plume measurements from six power plants). Our analysis indicates that both efforts are relevant for interpreting monitoring data.

These findings on emissions variability also have implications for chemical transport modelling. It is important to note that our simulations represent only single realizations of this emissions variability—these results therefore speak only to the ability to detect statistically significant differences, rather than quantify the full distribution of these differences. Although probabilistic emissions inventories for mercury have been developed (e.g., Wu et al., 2010; Zhao et al., 2015; Zhang et al., 2016b), the computational resource intensity of Eulerian chemical transport modelling can be prohibitive to fully-coupled emissions-chemistry probabilistic simulation. The application of computationally efficient means to quantify the resulting uncertainty in wet deposition due to emissions variability—for instance, response surface modelling (e.g., Ashok et al., 2013), adjoint or other sensitivity methods (e.g., Sandu et al., 2005; Henze & Seinfeld, 2007), and polynomial chaos expansion (e.g., Thackray et al., 2015)—would be a valuable next step.

Our results emphasize the large role of meteorology in explaining spatial and temporal variability in wet deposition in the Great Lakes region, particularly in comparison to anthropogenic emissions. Similar to studies exploring anthropogenic signal detection with respect to climate change (Santer et al., 2011), and O_3 (Barnes et al., 2016; Garcia-Menendez et al., 2017), these results indicate that distinguishing policy signals over meteorological variability in an 8 year observation record requires substantially larger emissions decreases than those modelled here (or alternatively, distinguishing policy signals of the size modelled here requires a substantially longer observation record). Future work addressing this topic can further clarify the mechanisms through which meteorology drives wet deposition variability, on an interannual and decadal scale. For instance, in addition to precipitation volume, Shah & Jaeglé (2017) find that meteorological processes affecting oxidation of global pools of Hg(0) in the mid and upper troposphere explain spatial variability in MDN.

In this work, we have evaluated several factors hypothesized in the literature to affect the translation of emissions mitigation policy into wet deposition changes, clarifying the nature and potential magnitude of their influence in the Great Lakes region in particular; however, there remain additional factors that merit further investigation. The results from our speciation variability simulation, and the large impact of this variability on the detection of regional policy in the Great Lakes region, suggest that a better understanding of mercury's atmospheric redox chemistry (Ariya et al., 2015; Mao et al., 2016; Horowitz et al., 2017), and potential meteorological and climatological drivers of its variability (Zhang et al., 2016a), can aid in the interpretation of monitoring data and attribution of global vs. local/regional policy signals. While we focus on atmospheric emissions, trends and variabilities in discharges to terrestrial and aquatic systems may have important effects as well due to mercury biogeochemical cycling (Amos et al., 2014). Finally, the endpoint of our analysis is atmospheric inputs into the Great Lakes ecosystem, yet the ultimate goal of much mercury mitigation policy is to prevent dietary mercury exposure from fish consumption (Evers et al., 2016). Understanding sources of "noise" in the translation of decreases in atmospheric inputs of mercury into changes in fish tissue concentration, and ultimately human exposure, is therefore a critical next step in this line of inquiry.

This work speaks to the severity of the signal-to-noise challenges for mercury monitoring in the Great Lakes, and provides support for taking them seriously in the design and evaluation of mercury policy. Our simulations illustrate the wide variety of wet deposition outcomes that could be consistent with policy adoption, given the influence of "noise." These results suggest that failing to see a decrease in wet deposition—for instance, in our interannual meteorological variability simulation or energy and economic trends simulation—does not indicate a failure in implementation of policy (indeed, our simulations assume 100% compliance). However, although all of our simulated deposition outcomes are consistent with successful policy implementation, they are not all consistent with successful policy outcomes. If the goal of policy is to reduce mercury inputs to vulnerable ecosystems within a decade—and ultimately, human exposure—further attention to the magnitude of noise, and how to design policy signals that overcome it, is necessary.

Table 3.3: Summary of emissions by simulation year (1-8) under NA policy. NA policy is implemented as a step change between years 4 and 5. Speciated emissions (0: elemental, II: divalent) are given for both NA and ROW.

		Pre-NA Policy								Post-NA Policy							
Sim. Year		1		2		3		4		5		6		7		8	
Speciation		0	II	0	II	0	II	0	II	0	II	0	II	0	II	0	II
PO	NA	62.8	19.5	62.8	19.5	62.8	19.5	62.8	19.5	38.1	18.8	38.1	18.8	38.1	18.8	38.1	18.8
	ROW	1014.0	367.5	1014.0	367.5	1014.0	367.5	1014.0	367.5	1014.0	367.5	1014.0	367.5	1014.0	367.5	1014.0	367.5
EET	NA	62.8	19.5	62.8	19.0	62.1	18.6	63.8	18.9	32.6	16.6	34.7	17.2	35.2	16.6	35.5	16.2
	ROW	1014.0	367.5	1010.8	387.7	1039.9	407.5	1065.3	415.1	1132.3	420.9	1204.3	440.1	1254.8	465.6	1317.2	475.6
PET	NA	135.0	19.5	132.8	19.5	130.6	19.5	128.4	19.5	101.6	18.8	99.4	18.8	97.2	18.8	95.0	18.8
	ROW	1429.3	367.5	1413.8	367.5	1398.3	367.5	1382.8	367.5	1364.9	367.5	1349.4	367.5	1333.9	367.5	1319.4	367.5
RV	NA	60.8	19.4	64.6	19.7	61.1	19.4	64.0	19.6	39.0	19.7	37.3	18.1	37.8	18.5	38.1	18.8
	ROW	1012.0	362.7	1012.7	363.6	1014.3	368.6	1013.6	366.1	1008.3	358.4	1012.4	369.5	1014.9	376.2	1008.2	358.1
SV	NA	62.6	19.5	63.0	19.2	62.9	19.2	62.8	19.3	38.2	18.8	38.4	18.6	38.2	18.7	38.0	19.0
	ROW	1007.7	372.3	1010.6	369.6	1003.0	376.6	1020.1	360.9	1005.9	371.7	1036.7	343.5	1007.8	369.9	1026.4	352.8
MV	NA	62.8	19.5	62.8	19.5	62.8	19.5	62.8	19.5	38.1	18.8	38.1	18.8	38.1	18.8	38.1	18.8
	ROW	1014.0	367.5	1014.0	367.5	1014.0	367.5	1014.0	367.5	1014.0	367.5	1014.0	367.5	1014.0	367.5	1014.0	367.5

Table 3.4: Summary of emissions by simulation year (1-8) under ROW policy. ROW policy is implemented as a step change between years 4 and 5. Speciated emissions (0: elemental, II: divalent) are given for both NA and ROW.

Sim. Year		Pre-ROW Policy								Post-ROW Policy							
		1		2		3		4		5		6		7		8	
Speciation		0	II	0	II	0	II	0	II	0	II	0	II	0	II	0	II
PO	NA	62.8	19.5	62.8	19.5	62.8	19.5	62.8	19.5	62.8	19.5	62.8	19.5	62.8	19.5	62.8	19.5
	ROW	1014.0	367.5	1014.0	367.5	1014.0	367.5	1014.0	367.5	1023.6	172.6	1023.6	172.6	1023.6	172.6	1023.6	172.6
EET	NA	62.8	19.5	62.8	19.0	62.1	18.6	63.8	18.9	56.8	17.3	60.2	17.9	59.3	17.2	57.0	16.8
	ROW	1014.0	367.5	1010.8	387.7	1039.9	407.5	1065.3	415.1	1145.6	191.2	1218.2	199.1	1269.6	205.9	1332.1	206.6
PET	NA	135.0	19.5	132.8	19.5	130.6	19.5	128.4	19.5	126.2	19.5	124.0	19.5	121.8	19.5	119.6	19.5
	ROW	1429.3	367.5	1413.8	367.5	1398.3	367.5	1382.8	367.5	1376.8	172.6	1361.3	172.6	1345.8	172.6	1331.3	172.6
RV	NA	60.8	19.4	64.6	19.7	61.1	19.4	64.0	19.6	63.7	19.6	61.1	19.4	63.6	19.6	53.0	19.5
	ROW	1012.0	362.7	1012.7	363.6	1014.3	368.6	1013.6	366.1	1010.5	171.5	1021.4	172.4	1021.4	172.4	1030.0	173.1
SV	NA	62.6	19.5	63.0	19.2	62.9	19.2	62.8	19.3	62.9	19.3	62.6	19.6	62.8	19.4	63.0	19.1
	ROW	1007.7	372.3	1010.6	369.6	1003.0	376.6	1020.1	360.9	1024.2	171.4	1023.0	172.6	1023.0	172.5	1022.7	172.9
MV	NA	62.8	19.5	62.8	19.5	62.8	19.5	62.8	19.5	62.8	19.5	62.8	19.5	62.8	19.5	62.8	19.5
	ROW	1014.0	367.5	1014.0	367.5	1014.0	367.5	1014.0	367.5	1023.6	172.6	1023.6	172.6	1023.6	172.6	1023.6	172.6

Table 3.5: Step change in Hg wet deposition ($\Delta\%$) between the pre-policy and post-policy period under different simulated scenarios. Significance ($p<0.1$) and size of the step change are calculated using the Mann-Whitney-Wilcoxon Seasonal Rank Sum Test and the Hodges-Lehmann Estimator of Difference. Values in the table represent the average change across all sites and just those with significant changes. The share of sites with a significant change is shown in brackets (% of all sites).

		Policy Signal	Removal Variability	Speciation Variability	Met. Variability	Energy and Economic Trends	Product Trends
NA	all sites	-1.01%	-0.84%	-0.66%	2.57%	-0.45%	-3.45%
	sig. sites	-1.01%	-0.84	-0.66%	-16.34%	-1.31%	-3.45%
		(100%)	(100%)	(100%)	(2.4%)	(41.4%)	(100%)
ROW	all sites	-1.71%	-0.34%	-1.62%	2.92%	-0.25%	-2.95%
	sig. sites	-1.71%	-0.34%	-1.62%	-16.31%	-1.23%	-2.95%
		(100%)	(100%)	(100%)	(2.4%)	(29.3%)	(100%)

Chapter 4

Creating and sustaining communities of concern for the long-term management of persistent pollutants

Abstract

This chapter explores the role that university-community partnerships can play in the long-term management of persistent pollutants through an empirical case study of the Superfund Research Program (SRP), a research program with a mandate to support the management of (often persistent) hazardous substances with a new emphasis on working with and for communities affected by these substances. It argues that because persistent pollutants implicate new places and people across time and space, their management requires the continued constitution of new *communities of concern*. It observes that community engagement in practice often supports this community building role, helping to trace pollutants across space and time and bringing together those potentially affected along those dimensions as a community of shared identity and interest. Community engaged research can therefore be a site at which knowledge about contamination and community identity are co-produced. Further, the case study illustrates the active work that university-based researchers and community-organizers engage in to help build communities of concern, particularly when

the reach of pollutants is large in time and space. I propose a conceptual framework for categorizing and assessing the roles that academic partners in particular can play to support the constitution of these communities. Through the application of this framework to the SRP, the chapter identifies potential challenges that university-based researchers may face in fulfilling these functions and suggests that a better understanding of the institutional conditions that enable these researchers to participate in this work of creating and sustaining communities of concern is necessary for the improved management of persistent pollutants.

4.1   Introduction

In 2012, the US Army Corps of Engineers began a thirty-year dredging project in the Indiana Harbor and Ship Canal in East Chicago. Though the waterway is designated an Area of Concern by the International Joint Commission due to its heavy historical contamination with toxic industrial chemicals and heavy metals (International Joint Commission, 2003, 2013), the dredging was primarily for navigational and economic reasons: Indiana Harbor and Ship Canal, which empties into Lake Michigan, was already one of the busiest in the region and dredging would allow larger barges access to local industry including steel mills and an oil refinery (Bienkowski, 2012; Lydersen, 2011). To the contrary, the dredging of the harbor and canal actually raised a host of environmental health and also environmental justice—East Chicago is a low-income area with predominantly Hispanic and African American residents—concerns from the surrounding community and environmental health experts (Bienkowski, 2012). Chief amongst these was the potential release from sediment of polychlorinated biphenyls (PCBs), a persistent bioaccumulative toxin classified by the the World Health Organization as a carcinogen (International Agency for Research on Cancer, 2016). Though banned by Congress in 1979, PCBs remained in high concentration in the surface sediment, acting as a continuing source of PCBs to water and air (Martinez et al., 2010). With potentially higher concentrations of PCBs in sub-surface sediment (Martinez & Hornbuckle, 2011), could dredging actually lead to increased human exposure to PCBs? And then, because remediation or treatment of dredged sediment is not considered financially feasible (National Research Council, 2007), would the long-term storage of the removed contaminated sediments at a Confined Disposal Facility less than half a mile from

the East Chicago Central High School lead to higher exposure by inhalation for students and other nearby residents (Marek et al., 2014)?

In this chapter, I take up the call of Gray-Cosgrove et al. (2015), who ask, "what depollution might look like given permanence." What does it mean to manage substances, like the PCBs discussed above, when they persist in the environment and when "remediation becomes an exercise in shifting materials in space rather than eliminating harm altogether" (Gray-Cosgrove et al., 2015)? Persistent does not mean static: persistent substances, because they are resistant to degradation, move through time and space, implicating new places and people (Diamond & Harrad, 2009). Persistent also does not mean uniform: as demonstrated in the case above, the distribution of toxic risks is often uneven, with burdens often falling disproportionately on racialized, low-income, or Indingenous communities (Bullard, 2000; Spears, 2014; Gagnon, 2016). The questions presented above then—who is affected, when, where, to what extent—are questions relevant for a variety of persistent toxic substances, whether they are human synthesized—like PCBs, pthalates, and perfluorinated compounds, the by-product of natural and anthropogenic combustion processes—like dioxin, dioxin-like compounds and polycyclic aromatic hydrocarbons (PAHs), or natural but mobilized by human activity—like arsenic, mercury, and asbestos.

This article focuses on the role that community-university partnerships can play in supporting the long-term stewardship of persistent hazardous substances. Interest in collaborative environmental governance practices that involve diverse stakeholders has grown due to their potential to increase the relevance, utility, legitimacy, and emancipatory possibilities of management efforts (Layzer, 2008; Dietz & Stern, 2008; Newig & Fritsch, 2009; Susskind et al., 2012)[1], and processes of collaborative knowledge production like co-learning (Loh, 2016), joint fact finding (Karl et al., 2007), community based participatory research (O'Fallon & Dearry, 2002; Minkler & Wallerstein, 2008), and action research more broadly (Greenwood & Levin, 2006) are often seen as a central component of these collaborative management practices (Susskind et al., 2012). Building on the work of Korfmacher et al. (2016), who identify the different roles that community-university partnerships can play at different stages in the process of social systems change (i.e. change in institutions, norms, practices, and policies) for improved environmental health, this article explores how the

---

[1]Empirical research continues to evaluate whether these approaches have delivered on this promise, and if not, why.

spatio-temporality of persistent toxics in coupled social, material, and ecological systems also creates different roles for community-university partnerships at different times and places. In particular, I highlight how the persistence of a pollutant—because it results in the implication of new places and people across time and space—requires the continued constitution of new *communities of concern*. Communities of concern may share identity and interests due to vulnerability to a certain pollutant ("affected communities"), or they may share identity and interests due to research and practice organized around that pollutant. I illustrate how community engaged research—a spectrum of knowledge production and mobilization activities about, for, and with communities impacted by persistent hazardous pollution (Vega et al., 2016)—can be a site at which these constitutive activities (Jasanoff, 2004a) occur, and propose a conceptual framework for understanding these functions while also identifying potential obstacles. I conclude by suggesting that the successful long-term stewardship of hazardous pollutants requires also the long-term stewardship of communities of concern, and reflect on institutional conditions that can support the work of creating and sustaining these communities.

To reach these ends, this article focuses empirically on a case study of the US National Institute of Environmental Health Sciences Superfund Research Program (SRP). Since 2011 the SRP, which funds multidisciplinary research centres that investigate methods to detect, assess, and prevent health effects from hazardous substances and contaminated sites, has required that grantees engage communities impacted by hazardous substances through a dedicated core (NIEHS, 2012; Landrigan et al., 2015).[2] I combine detailed analysis of the documentary content associated with all Centers and their engagement activities, including grant reporting, websites, publications, presentations, videos, town hall meeting minutes, and media reports, with fifteen semi-structured interviews ranging from 40 to 75 minutes with Center personnel (13 core leaders and coordinators) and community partners (2) across nine out of thirteen Centers funded as of January 2017.[3] An overview of these Centers is given in Table 4.2. These diverse materials were analyzed following a constructivist grounded theory approach (Charmaz, 2006), with situational analysis in particular used as a means of interrogating relational dynamics within these systems of human and non-human actors (Clarke, 2005; Clarke et al., 2016).

---

[2]For some Centers, engagement activities predate this requirement—including the Center that was involved in research on the Indiana Harbor and Ship Canal, described above.

[3]There are also four additional Centers that are in their "gap year"/"no cost extension" year.

In the next section of this chapter (Section 4.2), I begin by reviewing coordinating concepts related to persistent pollution in social-material-ecological systems, and community formation, drawing on literatures of Science and Technology Studies, Environmental Science, and Environmental Health Social Science. These are each rich bodies of scholarship, and in my brief review, my aim is to highlight concepts at the intersection of these spaces. The chapter then turns to the case study: I briefly review the history of the SRP, and community engagement in NIEHS funded research (Section 4.3.1), before discussing how communities of concern are formed across space, time, and the academy (Sections 4.3.3-4.3.5) and characterizing the roles that academic partners can play in this process. I conclude with reflections on implications for the effective long-term stewardship of hazardous substances (Section 4.4).

4.2   Background

4.2.1   Theorizing persistent pollutants

Environmental fate and transport of persistent pollutants

Table 4.1 outlines the environmental behaviour of three persistent pollutants, illustrating a spectrum of ways in which these substances are produced, move through the environment, and routes of human exposure (Diamond & Harrad, 2009; Selin, 2009, 2011; Rodan et al., 1999; Wania & Mackay, 1996; Farrington & Takada, 2014; Boethling et al., 2009; Scheringer et al., 2009; Kim et al., 2013; Wilcke, 2000; Douben, 2003; Punshon et al., 2017). These examples highlight the particular challenges presented by pollutants that are resistant to environmental and biological degradation: their capacity for long-range transport, their presence in multiple environmental media (air, water, soil/sediment), and their persistence in bodies. These characteristics expand the scope of who can be exposed, when, where, and through what means.

Persistent toxics of human concern are produced and mobilized in the environment both intentionally and as unintentional by-products and may be geogenic or anthropogenic in their origin. Even for substances that are naturally occurring, like arsenic and mercury, human activity may increase the mobilization of these substances, through practices like

79

coal combustion and mining. Large reservoirs of these pollutants can exist in the stock of products and materials that contain them, and in soils and sediments that are contaminated with them, even if these substances are no longer produced. Pollutants may be found in different proportions in different environmental media of soil/sediment, water, and air based on their chemical properties and forms. Substances can cycle between these different environmental compartments travelling long distances, but may be more or less mobile in these different media. Time-scales for biological or chemical transformation/degradation can also differ between these environmental compartments. For substances that do not degrade, or are highly resistant to environmental or biological degradation, long-term sinks are immobilization in deep mineral, soil, and sediment reservoirs.

Many persistent toxic substances are bioaccumulative, meaning that they can build up in higher concentrations in organisms than in surrounding environmental media. Some of these substances also biomagnify, leading them to be found in higher concentrations in the tissues of organisms higher up in food chains. Human exposure to these substances can occur through a variety of pathways, including consuming contaminated food and water, inhalation, and dermal contact. Substances that persist in the body can result in multi-generational exposure during gestation or breastfeeding.

Table 4.1: The environmental behavior of three persistent pollutants, drawing from Diamond & Harrad (2009); Selin (2009, 2011); Rodan et al. (1999); Wania & Mackay (1996); Farrington & Takada (2014); Boethling et al. (2009); Scheringer et al. (2009); Kim et al. (2013); Wilcke (2000); Douben (2003); Punshon et al. (2017).

	PCBs	PAHs	Arsenic
Production	Human synthesized compounds for intentional use in industry and building materials (banned in the US in 1979); also produced unintentionally as as by-product of some industrial processes	Unintentional by-product of incomplete combustion of organic material, anthropogenic or natural; Co-produced by geological processes creating fossil fuels (coal, oil)	Naturally occurring element in the earth's crust (20th most abundant); unintentionally mobilized during fossil fuel combustion and mining and metals production; intentionally used in some pesticides and preservatives
Reservoirs	Sediment, soil contaminated with industrial waste and PCB-containing materials still in-use act as continuing reservoirs	Creosote deposits; Sediment, soil contaminated with industrial waste	Soils, sediments, groundwater naturally enriched with arsenic; Mining waste; Arsenic-containing products
Persistence	Highly persistent	Moderately persistent	Cannot be degraded
Environmental mobility and multimedia transport	Soil: Not very mobile in soil due to adsorption, though less chlorinated compounds are more mobile; Water: Not very soluble, though less chlorinated compounds are more soluble; Air: Semi-volatile leading to volatization from water and soil and potential long-range transport in air	Soil: Not mobile when adsorbed to soils, but may be mobile when carried by dissolved organic matter; Water: Low aqueous solubility but highly mobile in water when adsorbed to particles in suspension Air: Volatility depends on molecular weight with lighter PAHs being more volatile, transport in the atmosphere is possible in the gas phase or adsorbed to particles	Soil: Mobility depends on chemical speciation, with some forms tightly bound to soil, and others mobile and bioavailable in soil pore water; Water: Solubility depends on chemical speciation and pH, but As(III) and As(V) widely found as groundwater contaminants; Air: local to regional atmospheric transport of particulate-bound arsenic
Bioaccumulation	Bioaccumulates in organisms (storage in fats) and biomagnifies up food chains	Bioaccumulates but does not strongly biomagnify	Bioaccumulates (particularly in plants) but does not biomagnify
Routes of human exposure, physical health impacts	Consumption (typically in fish); Inhalation, Dermal contact	Inhalation; Consumption (food grown in contaminated soil; grilled, roasted, or smoked food); Cigarette smoking; Dermal contact	Drinking water; Consumption (crop plants, like rice); Inhalation
Long-term sinks and storage	Disposal via incineration is possible, but may form more toxic products; Remediation through chemical treatment is possible, but costly; long-term confinement in storage facilities or burial in deep sediments	Lower weight PAHs can be degraded microbially while larger PAHs are typically removed through immobilization in deep sediments	Burial in deep terrestrial reservoirs

Challenges of slowness

In the previous section, I outlined the ways in which persistent substances move through space and time, emphasizing the agency of the material (Bennett, 2009). It is important to also emphasize that while linked to material properties, categorizations of matter as persistent are also socially negotiated. We define persistence on human time scales that make sense for human management. For instance, Selin & Eckley (2003) have discussed how the concept of Persistent Organic Pollutants (POPs) as a target for regulation emerged as a science-policy boundary object (Guston, 2001), with physical screening criteria for POPs created in the context of transboundary environmental co-operation. The time-scales associated with the substances I have discussed above—their degradation or long-term immobilization in the environment, their health effects—are often multi-generational. These time-scales are *slow*, on human terms (Gray-Cosgrove et al., 2015).

Nixon (2011) has used the term "slow violence" to describe the challenges and unequal burdens of toxic exposures, particularly for poor communities in the global South: "By slow violence, I mean a violence that occurs gradually and out of sight, a violence that is dispersed across time and space, an attritional violence that is typically not viewed as violence at all" (p. 2). For Nixon (2011), this slowness is related to imperceptibility, and the challenge of making this violence visible is part of its insidiousness. Certainly, the large temporal and spatial scale that persistent pollutants operate on present technical and scientific challenges for "seeing" their effects—for instance, linking chronic, low-dose environmental exposure to complex health effects like cancer through observational study can be challenging for statistical reasons and require decades of study (Pearce et al., 2015). However, as Murphy (2006) emphasizes, scientific, technical, political, and legal systems interact to produce "regimes of imperceptibility" that can render certain harms, in certain bodies, less visible—as well as privileging certain ways of seeing and knowing environmental health risk (Corburn, 2005; Brown & Mikkelsen, 1990; Ottinger & Cohen, 2011).

"Slow disaster" is a related concept employed in growing body of STS work (discussed in Fortun et al., 2017). Fortun et al. (2017) discuss the important analytical and practical implications of "[e]xpanding [t]ime [f]rames of [d]isaster" (p. 1009). By conceptualizing as disasters too the continuing contamination from toxic waste at the Union Carbide plant in Bhopal in the multiple decades since the gas leak in 1984 (Mukherjee, 2016), or the

everyday radiation exposure experienced by workers and their families in the "atomic cities" surrounding plutonium plants in the US and Russia, which by the end of the Cold War had cumulatively released an amount of radioactive isotopes equivalent to four Chernobyl disasters (Brown, 2013), these scholars highlight the need to prepare for and respond to chronic, slowly evolving traumas, just as we do catastrophic ones.

Long-term stewardship

The difficulty, or sometimes impossibility, of remediating or destroying persistent substances that in some cases are already ubiquitously found in the environment means that ongoing and active management is required to store, immobilize, and monitor these substances to minimize their potential harm. What might long-term stewardship entail? Two ways of thinking about long-term stewardship come from scholarship and practice on waste management and natural resources management. Though the targets and purposes of management efforts are different in these two contexts, work in both emphasizes that the need for management is ongoing, that management is both a social and a technical challenge, and that moving forward under uncertainty requires continuous processes of doing, learning, and changing.

Gray-Cosgrove et al. (2015) discuss the concept of perpetual care in the context of nuclear waste, which includes substances with half-lives that can range from thousands to billions of years. Containment of these wastes on these time-scales pushes on current technological, organizational, and epistemological limits, and system failure—at some point—is expected (National Research Council, 2000; Perrow, 1984). In this context, the importance of engaging social and institutional processes that enable adaptation, continuity, and communication (including through social technologies like stories and monuments) becomes apparent: "perpetual care includes political organization, infrastructural dedication, and an ethical framework to unite them" (Gray-Cosgrove et al., 2015, p. 8).

In the domain of natural resources management, the challenge of long-term stewardship is not cast as one of managing harm, but of managing benefit: how do pluralistic communities sustainably live with and benefit from resources like fisheries and forests when the complex dynamics of (socio-)ecological systems are often not well understood (Holling, 1978; Walters, 1986). Since emerging in the 1970s (Holling, 1978), adaptive management, involv-

ing iterative cycles of policy as experimentation, systematic monitoring, and assessment (Walters, 1986; Lee, 1999), has become an influential methodological paradigm. Further, given pluralism in values, interests, and ways of knowing, there has been an emphasis on *collaborative* adaptive management that brings together multiple stakeholders in an effort to achieve more legitimacy, stability and improved ecological outcomes in managed systems (Susskind et al., 2010, 2012; Stringer et al., 2006).

In both examples discussed above, while there is wide agreement in theory on the need for long-term collaborative and adaptive processes for waste and natural resource management, the application of these principles in practice often becomes a site of intense controversy. Debates over nuclear waste siting at Yucca Mountain (MacFarlane, 2003; Jasanoff & Kim, 2009) and ambivalence on the outcomes to date of many collaborative adaptive management programs (Layzer, 2008; Susskind et al., 2012; Westgate et al., 2013) illustrate that the details matter: how adaptation is institutionalized, who participates, and to what extent. Increased empirical attention then to how processes like collaborative knowledge production play out and what they do in the context of the long-term stewardship of socio-technical-ecological systems is necessary.

4.2.2 Theorizing community

Defining community

Community is a polyvalent, dynamic, and powerful concept. It implies "deep, horizontal comradeship" (Anderson, 1983, p. 50)—something shared. Minkler & Wallerstein (2012), reviewing a breadth of sociological literature summarize definitions of community as: spatial units linked to geographic place, "units of patterned social interaction," "symbolic units of collective identity," and social units that mobilize politically for change (p. 40).

In practice, these definitions of community often layer and overlap. For instance, in his book on communities facing toxic exposure, Edelstein (2004) writes: "I will use the term 'contaminated community' to refer to any residential area located within or proximate to the identified boundaries for a known exposure to pollution. Whether or not residents share a similar political, geographic, or social environment, the discovery of a toxic threat provides a basis for a new and shared identity that effectively defines a community of interest among those residing within this toxic territory" (p. 9). Edelstein's contaminated communities are

simultaneously communities of place, identity (as affected by a toxic threat), and interests. Communities of place, communities of identity, and communities of interests are often linked (Obama, 2012), and those concerned with environmental justice are explicitly interested in the intersections of place and race, class, immigration-status, and indigeneity amongst other dimensions (Corburn, 2005; Baron et al., 2009).

Finally, understanding community as relational is fundamental to the practice of community organizing and building (Walter & Hyde, 2012). Further, Walter & Hyde (2012), who see community as a complex and dynamic system rather than a functional unit, argue that "community has to do not just with engagement in relationship but, ultimately, with the quality of the relationship. Calling something community does not necessarily make it so. There can be greater or lesser degrees of 'communityness'" (p. 83). In this light, the importance of self-definition becomes clear; indeed, in her work on black feminist thought, Hill Collins (1990) emphasizes that consciousness and self-definition as a community can make community a locus for self-determination and empowerment.[4]

Co-producing community

How do communities form, particularly around environmental health concerns? Marres (2005, 2007) offers an interpretation of John Dewey's *The Public and Its Problems* for the STS community that focuses on the role of issues in community formation as part of a larger political process. Dewey (1927), in making his case for participatory democracy in an increasingly technological and technocratic age (the 1920s for Dewey, but this characterization remains true today), defines publics as communities that form to solve collective problems—or as summarized by Marres (2005, 2007), "no issue, no public." Dewey (1927) writes that when these issues "are in turn realized in thought and sentiment, recognition of them reacts to remake the conditions out of which they arose" (p. 54). In her interpretation of Dewey, Marres (2005) emphasizes the distinction between these communities of political interest and existing social communities: "if the issue is to be addressed, those who are jointly implicated in the issue must organise a community. What the members of a public share is that they are all affected by a particular affair, but they do not already belong to the same community: this is why they must form a political community, if the issue that

---

[4]At the same time, Hill Collins (1990) firmly resists essentialism, arguing for "heterogenous collectivity" (p. 101).

affects them is to be dealt with" (p. 9).

Building on this conception of issue-based publics, others have explored the *things*—the matters of concern (Latour, 2004)—around which these political communities form (Latour, 2007; Hird et al., 2014; Mukherjee, 2016). Mukherjee (2016) introduces the concept of a chemical public, ontologically heterogeneous (including human and non-human actors) and transnational in nature, to understand advocacy efforts for government recognition of chronic toxicity in Bhopal. Hird et al. (2014) use a case study of waste management in a small Canadian city to explore how landfills become and remain public matters of concern. They argue that waste stops being of public concern when it is governed in certain ways, highlighting the discursive (meaning-making) and material (in this case, leaks and spills) work that goes into making landfills issues again.

How does an identity as "jointly implicated" (Marres, 2005) in some issue emerge and how does this identity evolve? Another strand of literature focuses on the role that knowledge plays in identity formation, particularly in the context of health and illness. For instance, Rabinow's (1992) biosociality concept brings attention to "the kinds of socialities and identities that are forming around new sites of knowledge" (Gibbon & Novas, 2008, p. 3) with advances in genetics and medicine: "chromosome 17, locus 16,256, site 654,376 allele variant with a guanine substitution" (Rabinow, 1992, p.244) may become a site around which groups form a shared identity, explore what it means to live with a genetic diagnosis or predisposition, and mobilize collectively for treatment and research. One way to think about this phenomenon is co-production—briefly, the idea that "the ways in which we know and represent the world (both nature and society) are inseparable from the ways in which we choose to live in it" (Jasanoff, 2004b, p.2). By understanding the relationship between scientific knowledge and social order as interdependent and mutually constitutive, a co-productionist perspective brings our attention to both who participates to (co)produce knowledge and in which social structures, and how that knowledge itself produces social structures (Susskind & Elliott, 1983). In Rabinow's (1992) example then, genetic sequencing does not just make new knowledge about genetic categorizations, but makes new groups of people who are defined by those categorizations. To return to Marres (2005) and Dewey (1927), an issue like exposure to a contaminant must be made known and "realized in thought and sentiment" before a community can organize around it.

This sense of "making up people" (Hacking, 1986) underscores the power of scientific

knowledge and classification structures that order the world (Bowker & Star, 1999). It also underscores the power of who gets to participate in producing this knowledge that is deemed authoritative for (policy) action. In the domains of environment and health, technical (and typically quantitative) experts who are located in the academy or in the state often command this cognitive authority (Wynne, 1992; Rayner, 2003; Jasanoff, 1991; Beck, 1992). However, the "popular epidemiology" efforts of residents in Love Canal and Woburn, Massachusetts (Brown & Mikkelsen, 1990; Brown, 1992), "street science" conducted by a low-income community in Brooklyn (Corburn, 2005), and citizen science efforts to monitor and police air quality standards through "bucket brigades" in California and Louisiana (O'Rourke & Macey, 2003; Ottinger, 2010) demonstrate that in many cases, lay rather than expert communities are proactive in identifying environmental health risks and patterns of impacts. Though these community-initiated efforts may lead to subsequent expert or hybrid inquiry that addresses these potential harms, in other cases technical experts, their practices, and the larger social structures that they sit in have also played a role in silencing and perpetuating communities' health concerns (see examples in Edelstein, 2004; Cohen & Ottinger, 2011; Murphy, 2006). As a result, when these lay communities organize to identify as harmed by pollution, sometimes marshalling diverse ways of knowing, it is as much a political act as it is one of knowledge-making.

Another mechanism through which lay communities participate in knowledge production that implicates them is through what Hacking (1995) describes as looping effects: people are reflexive beings, and once "made up," can also work to remake themselves (Hacking, 1995). Health social movements form around collective illness identity (Brown et al., 2004), and patient groups can mobilize to shape and participate in the process of knowledge production itself (Rabeharisoa & Callon, 2004; Epstein, 2008; Frickel et al., 2010). Increasingly in environmental health justice movements, groups organize to also engage in broader social critique challenging the structural drivers and inequalities that contribute to environmental exposures and illness (Brown et al., 2004). These examples offer reminders that while the power to produce knowledge and "make up people" (as disease groups, as implicated by pollution, as publics) is often concentrated with experts (university-based, government, professional) and policy-makers, that once named, collectivities can "appropriate, resist, and transform these roles and identities" (Felt & Fochler, 2010, p.219).

87

Building and organizing community

Although the literature discussed above outlines how knowledge production can be the site around which shared identity forms, knowledge production alone is not enough to make a strong community (or to use the phrase of Walter & Hyde (2012), a high degree of 'communityness'). Following Minkler & Wallerstein (2012), I will use the term *community organizing* to refer to processes "by which community groups are helped to identify common problems or change targets, mobilize resources and develop and implement strategies to reach their collective goals" (Minkler & Wallerstein, 2012, p.37), and *community building* to describe processes that emphasize capacity building and fostering shared identity, whether or not task-oriented.[5] These definitions focus on empowerment and self-determination, suggesting that the roles for outsiders—whether they are organizing, public health, urban planning, social work or academic professionals (Stoecker, 2013; Corburn, 2009; Susskind & Ozawa, 1984)—in working with communities are primarily as enablers and facilitators of the conditions under which self-determination and empowerment are possible (Labonte, 1989, 1993). Some scholars and practitioners, like Walter & Hyde (2012), include these professionals as part of their conception of community as a dynamic system. In both cases though, whether working with or as part of communities to address environmental health disparities, the idea of focusing on issues that communities themselves collectively identify as salient—often summarized in Nyswander's (1956) phrase of "starting where the people are" (qtd. in NIH et al., 2011)—is seen as central to the success in reducing these disparities.

Figure 4-1, adapted from Minkler (2012), presents a typology of approaches to community organizing and building. Figure 4-1 situates older and newer approaches to these practices along two dimensions: one which spans consensus/collaboration (emphasizing community capacity and "power with," illustrated by Rubin & Rubin (2008)) to conflict/advocacy (emphasizing social change and "challenging power over," illustrated by Alinsky (1971)); and the other which spans a focus on community assets and strengths to a focus on community needs and deficits (Walter & Hyde, 2012). This two dimensional space emphasizes that in practice, strategies for community organizing (like leadership development or coalition

---

[5]Though the term community organizing was first used in the US in the late 19th century in the context of the settlement house movement, the roots of these practices are diverse and complex (Minkler & Wallerstein, 2012); for instance, Garvin & Cox (2001); DeFilippis et al. (2010) offer histories of community organizing in the US that emphasize the work of people of color and women who are often not included in the cannon of critical milestones in community work.

building) can combine different aspects of these approaches in varying degrees, often in complementary ways (Rothman, 2007). For instance, as discussed previously, Hill Collins's (1990) work on black feminist thought emphasizes social change that operates through strengthening community identity. Although this diagram does not fully unpack the theory and practice of community organizing and building, and its challenges (e.g., trust, representation, pluralism, power imbalances) and opportunities (DeFilippis et al., 2010; Labonte, 1989, 1993; Israel et al., 2010), it offers a useful vocabulary.

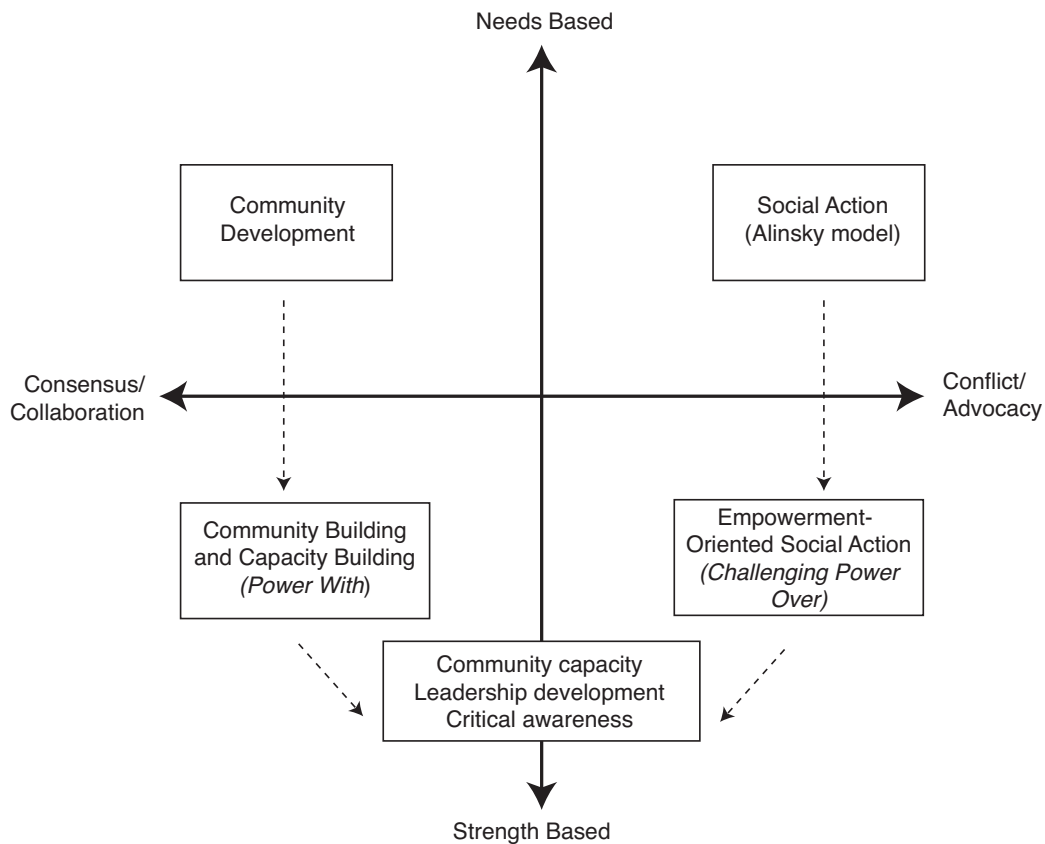Figure 4-1: Typology of community organizing and building, adapted from Minkler & Wallerstein (2012). Minkler & Wallerstein (2012) summarize older and newer approaches to community organizing practice.

4.2.3   Communities of concern

Given this theoretical background, what I will call *communities of concern* in this chapter are issue-based communities, though these communities may overlap and intersect with

other forms of community (e.g., geographic). Knowledge production—driven by expert and/or lay investigators—is critical to the constitution of these communities because it is through these processes that individuals identify as implicated by the issue. However, knowledge production is just one part of broader community building and organizing efforts that are required to address the issue. I use *communities of concern* rather than *contaminated communities* (Edelstein, 2004) to focus on how the matters of concern that bind these communities—persistent pollutants—implicate not only those potentially vulnerable to their effects but also those experts and practitioners who are professionally responsible for their study or management (as in Mukherjee's (2016) chemical publics). I propose that these communities of concern are therefore necessary for the long-term management of persistent pollutants (which requires social structures and not just technology) with the understanding that concern can be located in different places and people over time.

4.3 Case Study of the Superfund Research Program

This case study of community-engagement in the Superfund Research Program explores how we might bring these two lines of thinking—one on persistent pollutants and what is required to manage them, and the other on communities and what is required to make them—together to better address the environmental health and justice challenges that persistent pollutants present. Community-engaged research can be a site at which knowledge and identities are co-produced. Importantly, in the context of the SRP, community-engaged research becomes a space where not only identities as "communities impacted by hazardous substances" (NIEHS, 2015b) begin to be constituted, but where those communities, through self-defining what impact means, constitute what relevant knowledge is. Further, when it comes to persistent pollutants that "jointly implicate" (Marres, 2005) new people across space and time, this generative role may in fact be a critical one to which community-university partnerships (and the potential resources and continuity that come with them) can contribute. After all, perpetual care requires communities to do the caring. Yet, the examples from the SRP show that building communities of concern requires the participation of a diverse set of actors—partnerships between (insider) organizers in affected communities, researchers, policy-makers, and other stakeholders—as well as time and effort to build knowledge, trust, relationships, and capacity. Building on previous efforts to characterize

the diverse roles that academic partners can play in in supporting communities and to develop conceptual frameworks that facilitate the design and evaluation of these academic-community partnerships (Korfmacher et al., 2016; Trochim et al., 2011; NIH et al., 2011; Ahmed & Palermo, 2010; Esmail et al., 2015), I propose a conceptual framework and set of process evaluation questions (CDC, 2009) that focuses specifically on how academic partners can support these constitutive functions of identifying risk, building identity, building capacity, and enabling organizing. After illustrating this framework with examples from the SRP, I discuss potential challenges and funding and academic incentive structures that may ameliorate them.

4.3.1 History of the SRP

The US Congress passed the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) in 1980 in response to growing public recognition that improper hazardous waste disposal was a serious risk to the environment and public health (Landrigan et al., 2015). This awareness was spurred by the public discovery that homes and a school had been sited atop a former hazardous waste disposal site where approximately 21 000 tons of chemical waste had been dumped—the now infamous Love Canal in New York—resulting in miscarriages, birth defects, and other health effects in residents (Newman, 2016), as well as the discovery of a 23 hectare site strewn with thousands of drums of chemical waste in Kentucky—the Valley of the Drums (Landrigan et al., 2015). CERCLA is more commonly called the Superfund Act because it created a trust fund for the cleanup of contaminated sites when the responsible polluting party could not be identified or could not pay.[6]

The Superfund Amendments and Reauthorization Act (SARA) of 1986 added new enforcement tools and more clearly delineated roles for federal, State, Tribal, and citizen stakeholders (Vig & Kraft, 2016), including the creation of the Hazardous Substances Basic Research and Training Program (at the time, abbreviated as the Superfund Basic Research Program) within the NIEHS (Suk, 1995). The scale and complexity of existing cleanup efforts at that time underscored the need for better understandings of the environmental and human health impacts of often complex mixtures of hazardous persistent substances, and better methods to detect, assess, and remediate them. The Congressional mandate for

---

[6]Originally, the fund was financed through a tax on the petroleum and chemical industries, however, in the 1986 reauthorization of the Act, the source was shifted to general tax revenue.

the university-based research program was therefore to support the activities of the Environmental Protection Agency (EPA) and Agency for Toxic Substances and Disease Registry (ATSDR) in managing hazardous waste through research that focused on: "development of methods and technologies to detect hazardous substances in the environment; advanced techniques for the detection, assessment, and evaluation of the effects on human health of hazardous substances; methods to assess the risks to human health presented by hazardous substances; and basic biological, chemical, and physical methods to reduce the amount and toxicity of hazardous substances in the environment" (Suk, 1995, 3). While the SRP operates a variety of granting mechanisms, including individual, multi-project center, and small business grants, I will focus only on the multi-project centers (Superfund Research Centers, or SRCs).

Over its now thirty year history, the SRP has evolved. Although it still responds to the same Congressional mandate (and has had continuity in leadership under its founding and current director, William Suk), over time its research portfolio has become less focused on specific Superfund sites (for instance, a number of Centers do work across-borders in Bangladesh and Mexico, and focus on hazardous substances and exposure pathways that are less directly linked to industrial waste), greater emphasis has been placed on ensuring that research has real-world relevance for all stakeholders (in 2009 it was even renamed from the Superfund Basic Research Program to simply the Superfund Research Program) and that this research reaches these stakeholders in a more timely fashion (Morris et al., 2011), and the composition of the university-based researchers in Centers has become more multi-disciplinary (including, toxicologists, biomedical engineers, epidemiologists, health clinicians, environmental engineers and scientists, anthropologists, sociologists, and economists). Today, each SRC requires a minimum of two biomedical research projects, a minimum of two environmental science or engineering research projects, and cores dedicated to administration and integration, training of graduate students, research translation (required starting in 2003), and most recently, community engagement (required starting in 2010).

This recent shift to emphasize community engagement reflects a larger trend in NIEHS as a whole. At its outset, the NIEHS, with its less clinical orientation than other Institutes within the National Institutes for Health, operated under a model of "basic science," with an emphasis on toxicology, informing policy (Hawkins, 1987; Lichtveld et al., 2016). However,

beginning the in the 1990s, NIEHS developed an increasingly public and community health orientation with a focus on addressing environmental health disparities—much of which is credited to grassroots environmental justice (EJ) activism and the leadership of Kenneth Olden, who served as the Director between 1991 and 2005 (Lichtveld et al., 2016; Landrigan et al., 2015; Green & Mercer, 2001; Baron et al., 2009; Finn & Collman, 2016; Matz et al., 2016). As one SRC Community Engagement Core (CEC) leader described:

> So he [Kenneth Olden] was responsive to the EJ [environmental justice] issues there and he just was also very engaged in public health in general and helped to shape what would later become the Institute's idea of, "we do environmental public health," you know, which is different than the National Cancer Institute. They would not say that "We do cancer public health." They would say, "We do cancer research, we try to find the causes and the cure for cancer." So having this sort of practical and applied public health, outward facing community engagement approach really got its head start at that point, and then it got laid onto... existing programs like Superfund... so it diffused from initially a very focused RFA [request for application] around EJ and CBPR [community based participatory research] to take over more parts of the agency's portfolio. (Interview 1, CEC Core Lead)

4.3.2 Community engagement in practice

In its RFA for SRCs, the NIEHS (2015b), outlines the objectives of mandatory community engagement cores (CECs) as directing best practices in the engagement of communities impacted by hazardous substances through bidirectional interactions, ultimately to empower these affected communities to participate in efforts to reduce "the amount and toxicity" of these substances where they live, work, learn, and play. Felt et al. (2016) argue that funding schemes provide "scripts," outlining the roles of actors and the spaces in which they operate, but that actors necessarily redefine and reinterpret these scripts within their specific local contexts. Indeed, translating concepts like "community engagement" from a "nascent idea in someone's head" (Interview 13, RT Core Lead), to funding requirement, to practice has been an ongoing process of negotiation and dialogue:

> Now you know the terminology has evolved even as SRP has evolved. I think

they've [NIEHS] had to learn what those words mean for themselves because in the first years of them asking for these things I think they were having a hard time — I remember us having these discussions of what do they really want, you know. So I think that everyone is kind of coming along in this evolution of what that means. (Interview 3, RT Core Lead)

SRCs work across the US in dramatically different geographical, social, ecological, and institutional contexts (see Table 4.2). The enactment of community engagement in the context of the SRP is also particularly interesting because, as alluded to in the history above, programs like the SRP register historical tensions between an emphasis on basic research, and an applied and problem-oriented mandate to serve and work as partners with both federal agencies (and others) responsible for managing hazardous substances and communities affected by them. For instance, several informants characterized the SRP as a program with strength and tradition in "bench science" (basic science typically conducted at the laboratory bench). While those working across the environmental health sciences share a joint mission of improving human health—or as one informant summarized, "nobody becomes a toxicologist just because they like to kill rats" (Interview 5)—different subdisciplines within the field may emphasize different topics of inquiry (molecules, cells, animals, technology, people), different ways of knowing, different understandings of what rigorous inquiry is, and different skill sets, depending on laboratory, clinical, epidemiological, or social science training. In some SRCs, work with communities impacted by hazardous substances predated the official CEC requirement, through research translation and (voluntary) community outreach and engagement cores, however for others the CEC addition represented a very new "grafting on onto a set of basic researchers" (Interview 6, CEC Core Lead). For some SRP researchers then, working in a community-based context may be entirely new, and the "laying on" of this community orientation may involve interdisciplinary exchange (and potentially friction) and blending of disciplinary norms (compared to NIEHS programs that were conceived with community participation at the outset, like the Breast Cancer and the Environment Research Program).

Community engagement in research has come to describe a wide spectrum of activities for and with communities. At the strongly participatory end of the spectrum, community based participatory research (CBPR) emphasizes linking research and action, capacity building through research, and the role of communities as active and equal co-researchers

from issue selection to mobilization of results (O'Fallon & Dearry, 2002; Israel et al., 2010; Loh, 2016). As Wallerstein & Duran (2008) outline, this approach to research that is gaining traction in work on (environmental) health has theoretical, historical, and practice roots in the broader tradition of participatory action research (Greenwood & Levin, 2006; Flyvberg, 2001; Lewin, 1948; Freire, 2000). This orientation to research is often motivated by a desire for improved research relevance and utility (a strong component of work that draws from Lewin, 1948), but also for emancipatory aims, addressing historical and ongoing power imbalances in research that have been harmful for affected communities (a strong component of work that draws from Freire, 2000).[7] Importantly, participatory action research represents an epistemological and methodological departure from traditional social science approaches with its emphasis on contextually "actionable" rather than "generalizable" knowledge (Flyvberg, 2001; Flyvbjerg et al., 2012).[8]

However, engagement can also refer to other bi-directional activities, including: research that is initiated by community concerns and guided by community input (for instance, through a community advisory board) but where community members are not fully co-researchers; capacity building; and outreach and education (sometimes community-initiated) to support affected communities in their organizing around environmental health issues and participation in hazardous waste management decisions (NIEHS, 2015a; NIH et al., 2011).

In practice, community engagement in SRCs also reflects this spectrum of possibilities. Some activities are explicitly CBPR (e.g., Ramirez-Andreotta et al., 2015; Martinez et al., 2017; Hoover, 2013), while others hew closer to outreach, education, and community capacity building. Table 4.3 provides examples of CEC activities that fall along a spectrum of participation. Categories in this spectrum are adapted from the work of Morgan & Lifshay (2012); Arnstein (1969); Chess et al. (1988); Davidson et al. (1998), and range from no participation (Academic Partner Leads) to community initiated and directed inquiry (Community Leads). Note that each SRC conducts a range of engagement activities, often

---

[7]Indeed, many communities that are disproportionately affected by environmental contaminants have also experienced historical research trauma (Smith, 2012; Potts & Brown, 2005; Morello-Frosch et al., 2015)—that is research practices that reinforce existing health disparities and reinscribe power structures that contribute to environmental injustice.

[8]Flyvberg (2001); Flyvbjerg et al. (2012) use the Aristotelean term *phronesis* to connote this knowledge for doing, which is one of three virtues of knowledge: analytical or scientific (*episteme*); technical or craft (*techne*), and practical wisdom (*phronesis*). It should be noted that this emphasis on phronesis remains controversial in many corners of the academy.

involving different degrees of community participation, so the examples given for Centers are not representative of entire CEC portfolios. Given the number of SRCs, the examples given for each category are also not exhaustive, but illustrate the diversity of approaches that are taken.

Many of those involved in CECs in the SRP describe CBPR as a gold standard, while also acknowledging that CBPR in some SRC contexts is not (yet) possible. Some argue that CBPR, because it involves equitable partnership between universities and communities, is preconditioned on self-identified communities with the resources (time, money, energy) and capacity to engage in research, and sufficient trust between academic and community partners, which may take years to form (Israel et al., 1998). In this light, CBPR is possible when existing communities (and associated community-based organizations or representatives)— be they related to geography, race, ethnicity, immigration status, socio-economic status, employment, length of residency in a place, shared interest in environmental or health issues, or several of these dimensions simultaneously[9]—see themselves as a community (or communities) impacted by environmental health risk. This is more likely to be the case when contamination is known and salient—for instance, through proximity to existing or newly declared Superfund sites (e.g., those discussed in Hoover, 2017; Clapp et al., 2016), in the aftermath of a leak or spill (e.g., the Gold King mine waste water spill in 2015), or when development uncovers previously unknown contamination (e.g., Senier et al., 2008; Martinez et al., 2010).

However, as discussed in Section 4.2.1, for many persistent pollutants affected communities may be spatially and temporally distributed in ways that are not readily apparent.

---

[9]These layered and intersecting aspects of community are well illustrated by this comment on community engagement surrounding an asbestos Superfund site: "I mean Ambler is a small town and historically it's been kind of defined by administrative boundaries. But the problem is not confined to the administrative boundary of Ambler. And so kind of depending upon what question we're asking I think we talk about community in different kinds of ways... So one example is in terms of the epidemiology of mesothelioma... that's allowed us to draw a circle around you know sort of where the affected—where people who were affected by mesothelioma live. And that's one way of defining the community affected by asbestos. But that's really only defining the community in terms of a health effect. Within Ambler and then the Greater Ambler Area there are a couple of different communities I think that would define themselves as communities. There's a fairly middle class upper middle class community of people who live in the general area who see themselves as the community. There's also a fairly—small but I would say sizeable—African-American group that see themselves as a community as well. And then there's a fair number of Italian, people of Italian descent who came to work in the factories back in the early part of the 20th century and then their families have stayed and then a fair number of Irish folks that came to work in the factories as well. At the same time when the factory closed, this was pretty much a one horse town, and so when the factory closed a lot of people moved out and a lot of renters and more transient people starting in the 90s started to move in and the old timers see themselves as the real community and don't see the newcomers as part of the community..." (Interview 8, University of Pennsylvania CEC Core Co-Lead)

Scholar-practitioners like Stoecker (1999) have argued that community-building can be a component of community-engaged research and that these projects can "us[e] the research to bring people together and build skills and relationships" (p. 847) in less well-organized communities (while also acknowledging that "the tragedy of conflating the organizer and researcher roles... is that only a few academics are good organizers," p. 846). For some CECs then, the process of engaging communities "affected by hazardous substances" (NIEHS, 2015b) involve less participatory activities, including research about communities, and (collaborative) outreach and education efforts that support community organizing and building practice around a shared identity as impacted by pollutants. As one interviewee explained, in describing the process of forming a spatially distributed epidemiological cohort of expecting mothers who may be at risk for pre-term birth due to environmental exposures:

> If I work with people at a contaminated site that has been discovered and there's a community group that develops around that, that needs help with research or getting the government to do whats needed to get wells tested or something. There you have a clear site, you've got a group of people already. Here you're developing a cohort from scratch. (Interview 1, CEC Core Lead)

In the preceding quotation, two categories of activities are described: those that support established communities of concern in working towards systems change, and those that support the building of communities of concern that may not already be well established. Past efforts to develop conceptual and evaluative frameworks to facilitate the design and evaluation of community-university partnerships (Korfmacher et al., 2016; Esmail et al., 2015; Ahmed & Palermo, 2010; NIH et al., 2011) have predominantly focused on the former set of activities. Recently, for instance, Korfmacher et al. (2016) have characterized the different roles that academic partners can play in supporting community partners in driving system (institutions, norms, policies) change, arguing that different functions can be more or less appropriate at different stages of the process (see Figure 4-2). Further, they propose a set of evaluation questions (CDC, 2009; NIH et al., 2011) that map onto these functions, for use in formative and process assessments. However, this framework often takes the community of concern as an established entity. How might we understand the functions of engagement when its focus is supporting the constitution of these communities, as a precursor to system change efforts?

In the following sections, I first illustrate the different ways in which community engagement activities support community building around persistent pollutants across space, time, and the academy, using examples from the SRP. I then propose a framework for characterizing these activities and apply it to assess the SRP more broadly, discussing common patterns and potential barriers to success.

Figure 4-2: Stages of systems change, excerpted from Korfmacher et al. (2016).

Table 4.2: Overview of currently funded Superfund Research Centers.

Center	Name	Grant Period	CEC Period	Hazardous Substances Studied	Community partners
Boston University	Receptor-Based Developmental and Reproductive Toxicity of Superfund Chemicals	1995-2017	2005-2017	PCE, flame retardants	community advocacy and environmental justice organizations (including Alternatives for Community & Environment, Toxics Action Center, and the Science & Environmental Health Network), municipal health boards and departments in Massachusetts, residents living in proximity to contaminated sites, EJ communities
Brown	Toxicant Exposure in Rhode Island: Past, Present, and Future	2005-2020	2005-2020	TCE, PCE, nanomaterials	community organizations serving/representing RI residents impacted by Superfund and Brownfield sites (Blackstone River Watershed Council, Woonasquatucket River Watershed Council, Environmental Justice League of RI, UPP Arts, RI Indian Council, Narragansett Tribe)
Columbia	Health Effects and Geochemistry of Arsenic and Manganese	2000-2017	2012-2017	arsenic, manganese	Private well users in Maine and New Jersey exposed to arsenic and other contaminants
Dartmouth	Sources and Protracted Effects of Early Life Exposure to Arsenic and Mercury	1995-2019	2014-2019	arsenic, mercury	private well users, consumers of food products of concern, parents and expecting parents, students at high schools near Superfund sites in Northern New England
Michigan State University	Environmental, Microbial, and Mammalian Biomolecuar Responses to AhR Ligands	1989-2017	2013-2017	dioxin, dioxin-like compounds	Michigan Tri-Cities (Saginaw, Midland, Bay City) community
Northeastern	Puerto Rico Testsite for Exploring Contamination Threats	2010-2019	2014-2019	contaminants contributing to preterm birth, TCE, pthalates	participants in PROTECT study cohort (over 1000 women), residents in areas where a groundwater study is being conducted, community serving organizations (March of Dimes, Citizens of the Karst), Puerto Rico Environmental Quality Committee, Municipality of Barceloneta, four Federally Qualified Health Centers
Oregon State University	PAHs: New Technologies and Emerging Health Risks	2009-2018	2009-2018	PAHs	three Native American Tribes in Pacific Northwest (Confederated Tribes of the Umatilla Indian Reservation, the Swinomish Indian Tribal Community, Samish Indian Nation)

Institution	Project Title			Contaminants	Community
University of Arizona	Hazardous Waste Risk and Remediation in the US Southwest	1990-2017	2000-2017	hardrock mining metals, arsenic, tailings	communities impacted by mining with a focus on Hispanic communities in Arizona-Sonoma border region, Tribal communities through Tribal Community Colleges, and Health Representatives (including Navajo Nation)
UC San Diego	Detection and Models of Toxicant Exposure	2000-2017	2005-2017	triclosan, cadmium, arsenic, endocrine disruptors	community serving and grassroots environmental organizations support communities in the San Diego-Tijuana city-region (including Casa Familiar and Alter Terra)
University of Iowa	Semi-volatile PCBs: Sources, Exposures, Toxicities	2006-2020	2006-2020	PCBs	partner schools in East Chicago, IN and Columbus Junction, IA, community advisory boards in both cities
University of Kentucky	Nutrition and Superfund Chemical Toxicity	1997-2019	2005-2019	PCBs, TCE	communities in Eastern and Western Kentucky interested in nutrition-based intervention programs (e.g., five county senior centers, community in Dayhoit, KY near Superfund site)
University of Pennsylvania	Asbestos Fate, Exposure, Remediation, and Adverse Health Effects	2014-2018	2014-2018	asbestos	stakeholder groups for communities surrounding Ambler and BoRit Superfund sites including EJ communities, through a stakeholder advisory board
University of Washington	Effects-Related Biomarkers of Environmental Neurotoxic Exposures	1987-2017	2015-2017	cadmium, manganese, arsenic	communities affected by hazardous waste sites in Washington, Oregon, Idaho, and Alaska, Northwest Toxic Communities Coalition, Duwamish River Cleanup Coalition, communities involved in field studies

Table 4.3: Examples of SRC community engagement activities that span a spectrum of participation. The spectrum of participation in the context of community-engaged research is adapted from the work of Morgan & Lifshay (2012); Arnstein (1969); Chess et al. (1988); Davidson et al. (1998), and ranges from no participation (Academic Partner Leads) to community initiated and directed inquiry (Community Leads). Note that the examples given in each category are not exhaustive. Further, each SRC conducts a range of engagement activities, often involving different degrees of community participation, so the examples given for Centers are not representative of entire CEC portfolios. This table spans two pages.

	Definition	Examples from SRC activities
Community Leads	Community initiated and directed inquiry	–
Power Sharing	Community and university partners define, design, and conduct research together	Explicitly CBPR projects: the Namaus (fish) Project between the Narragansett Tribe and the Brown SRC to evaluate exposure to contaminants through fish and identify possible sources to support tribal leaders in management; toxicant survey and environmental health needs assessment in the Tijuana River watershed between Alta Terra (Tijuana-based NGO),UCSD SRC, and UABC; the Dewey-Humboldt Gardenroots citizen science project addressing potential exposure through vegetable gardening in residential areas near Superfund mining site with UA SRC; smoked fish metabolism study between CTUIR Tribal members and OSU SRC (involving clear material and data sharing agreements)
Comprehensive Consultation	Comprehensive and ongoing input from the community to guide inquiry	Community Advisory Boards and Stakeholder Advisory Boards that provide ongoing input on overall center research activity or specific research efforts like community-based air monitoring studies for PCBs in New Bedford Harbor, MA, East Chicago, IN, and Columbus Junction, IA
Limited Consultation	Specific and periodic input requested from the community to guide inquiry	Focus groups and interviews conducted with the PROTECT epidemiological cohort of women (the center investigates environmental influences on preterm birth) providing opportunities for participants to express their preferences, and interests in terms of how their exposure data is reported back and how samples are collected
Continued on next page		

Collaboratively Inform and Educate	Community partners are co-educators with university partners in informing the community about the topic of inquiry	University partners develop training modules for promotoras de salud (community health advocates) in a train-the-trainer model (originally focused on Hispanic communities in Arizona-Sonora border, but work has expanded to address the impacts of mining on tribal lands with tribal community colleges and universities); Outreach through art with community activists and artists (art installations and events that draw from place-based history in Rhode Island; plays that draw from oral histories from the community and explore dimensions of asbestos' impact in Ambler, PA; a documentary film project on pre-term birth); Developing materials that community organizers can use (e.g., sample interventions and communication materials for well-water testing)
Inform and Educate	University-based researchers conduct outreach and education activities with the community about the topic of inquiry (which may be in response to community requests)	Nutrition-based educational intervention programs responding to community requests for more information on the potentially protective effects of diet given environmental contaminants exposure in Eastern and Western Kentucky; Development of awareness materials like brochures that are tailored to the affected community (in terms of language, examples); Developing fact sheets to respond to community questions in the wake of pollution events (e.g., Gold King Mine Spill)
Academic Partner Leads	University-based researchers initiate and direct inquiry about the affected community	Ethnographic research to understand lived experiences of toxicity and stigma for communities impacted by an asbestos Superfund site in Ambler, PA and residents surrounding a floodplain contaminated with industrial waste in Michigan Tri-Cities area; Historical research looking at past land-use to identify potential sites of contamination

4.3.3 Creating communities of concern across space

Persistent pollutants often act as groundwater contaminants. Three examples addressed in SRCs are arsenic in naturally enriched soils, industrial solvents, like trichloroethylene (TCE), and plasticizers like phthalates. Even if they are not strongly water soluble, contaminated soils or areas of natural enrichment can act as a continuing source, resulting over time in potentially harmful concentrations in drinking water. However, the fate and transport of pollutants through aquifers is often complex and unpredictable (Padilla et al., 2011), particularly when the interaction between the distribution of human activity resulting in contamination and environmental processing of those contaminants is not fully understood. For instance, even when property owners of sites with known contamination clean up those properties, one investigator explained, "the problem is the contamination has spread beyond the property boundaries, underground and so people don't know about this" (Interview 2,

CEC Core Lead). Further, "it really shows that there is this large systemic set of processes that is ongoing and operating to produce and distribute risk or hazards across an urban and metropolitan and even rural landscape... It's spreading" (Interview 2, CEC Core Lead).

The challenge for engagement then is to bring together "a community that may be more geographically dispersed... but is held together by some of these characteristics that make them particularly vulnerable to [these pollutants]" (Interview 4). The difficulty of this challenge is magnified when groundwater contamination is odorless, colorless, and tasteless— when it is not easily rendered visible, in the idiom of Nixon (2011). Referring to the awareness that communities abutting a Superfund site have of shared risk, particularly if health effects are geographically clustered, one informant said, "I think our approach to community engagement is wanting to build that up because we didn't have as much of a tangible, you know, Superfund site. People [here] don't think that much about Superfund sites, I guess, is one way to say it. But we have... omnipresent chemicals that are impacting people" (Interview 4).

Households using private wells for drinking water, and therefore who may be vulnerable to exposure, may not see themselves as a community with shared identity or interest, and so for many SRCs working in this context, engagement efforts focus on identifying geographic areas that may be at risk, raising awareness, and developing tools that enable community organizing. One former CEC Core Lead described efforts to identify "community champions" (Interview 7) in towns who could act as point persons for well water testing days and catalyze further organizing. A toolkit for community organizers, based on learnings from these experiences, was also prepared and shared online. In one town, these outreach and education efforts spurred the town conservation commission to design and implement a pilot project to increase water testing rates through an awareness campaign and test-kit distribution, with the support of the SRC CEC and state agencies (Paul et al., 2015). Under the pilot program, the number of water samples tested was more than three times as large as the total from the six previous years (Paul et al., 2015). Working with communities that rely on private wells to develop and disseminate best practices and guidelines has also been the focus of a community-engaged research project in the Puerto Rico Testsite for Exploring Contamination Threats (PROTECT) SRC (Vega et al., 2016).

In addition to providing tools and support to enable traditional community organizing activities, SRCs are exploring online tools as a means of bringing together geographically

dispersed communities of shared vulnerability. One CEC is interested in the potential of interactive databases and mapping tools that allow residents to explore potential pollutant sources in their local neighbourhoods, which may then create the opportunity to develop more participatory community-university research questions. The PROTECT SRC is developing a smartphone application for its epidemiological cohort of expecting mothers—another geographically dispersed community that may share vulnerability to groundwater contamination—with the hope that it will provide not only a means for data report-back, but a forum for discussion and formation of relationships within the cohort: "Our idea is to actually build communities so that they can be either virtual communities or where they can actually meet in person and say, you know, let's form a support group around these issues" (Interview 1, CEC Core Lead).

4.3.4 Creating communities of concern across time

Even when pollutants remain in the same place, for instance, in reservoirs of contaminated sediments or soils, their persistence through time can also require the constitution of new communities of concern. One reason is that over time, the presence of these pollutants can be forgotten. This is particularly true in urban industrial settings, where land use can shift from industrial to residential to commercial—what Elliott & Frickel (2013) call "industrial churning." Knowledge of historical contamination, if ever documented, may be easily lost: "I mean that's the thing about urbanization. Everything's—land is precious and it gets used and it gets used for other things. And then within a generation, previous uses disappear and nobody remembers. But the contaminants probably don't. They're still there, you know, in one way or another" (Interview 2, CEC Core Lead). One community activist working on removing PCBs from schools (Osterberg & Scammell, 2016) described the challenges of tracking a contaminant that may have been used in building materials like caulking over decades of school infrastructure: "You have to actively test for it" (Interview 14, community group partner), a process that requires a formal request for sampling and then subsequent laboratory analysis which is typically not part of routine school maintenance.

Understanding the full extent of communities that may be impacted therefore requires "excavating history" (Interview 15, community group partner). For one SRC community partner, the leader of a community-based environment and arts education organization,

this process involved leading students to compile seventy oral histories with residents of the ethnically diverse neighbourhoods in the watershed and former workers at the facilities that surrounded the pond (Valk & Ewald, 2013). A CEC trainee has since built upon this deep, place-based knowledge in a collaborative pilot project to create a historical database that combines information from archival records and directories with ecological data to explore how potential (forgotten) industrial sources may have intersected with places in the pond's watershed. The histories of these pollutants are deeply entangled with the history of people and place, and need to be built from the ground up (Valk & Ewald, 2013; Elliott & Frickel, 2013).

When awareness of the presence of contamination does not fade over time, maintaining concern can equally be a challenge. Contaminants like PCBs, dioxins, and dioxin-like compounds are highly persistent in sediment and built infrastructure. Without targeted removal and disposal, large reservoirs can persist for many decades. In some areas that have long histories of industrial contamination, long-time residents have known about the presence of these pollutants for decades (Zhuang et al., 2016). In the case described by Zhuang et al. (2016), when the extent of contamination first became known in the 1980s, it became an issue around which the local community organized, ultimately resulting in the undertaking of long-term remediation efforts (funded by the polluting party, Dow Chemical, and overseen by the EPA and Michigan Department of Environmental Quality). However, remediation processes at the floodplain-scale can themselves be slow, even multi-decadal affairs. Zhuang et al. (2016) report that watershed residents are not currently very actively engaged in the remediation process (e.g., providing public input on cleanup plans and schedule, or requesting monitoring). In the interim, environmental concentrations of pollutants may remain unattenuated in unremediated areas.

Active organizing by impacted community members around an issue with this life-span, suggested one CEC Core lead, is a challenge:

> Issues like this—problems like this that become issues—that receive attention, and thus are issues, they have—they have life spans. They don't go on forever and they get displaced by other things... And I think that from a community member's perspective also there's a lot to be active about and concerned about. And one topic might be the right thing to be engaged and concerned about at a particular point in time but then, if the problem is adequately being dealt with,

105

probably your attention moves on to something else and that probably is a very productive thing. (Interview 6)

Further, Zhuang et al. (2016) hypothesize that in addition to the normalizing effect of time and active remediation efforts, when the polluting party remains a major employer offering economic value to the community, this benefit may temper concern over environmental health risks (a dynamic also described by Brown (2013) in her historical account of residents of plutonium processing complexes).

Though community concern can wane over time, university-based researchers may still see a role for their own continued engagement:

> You know when a public issue stops being an issue of concern for a community, when that happens, it can become—attractive—to employees of that company or for that kind of stakeholder to believe that things are OK. And maybe they are and maybe they're not. But there is a strong interest from a number of people in this watershed, and certainly from the polluting company, there's a real strong desire to believe that everything is taken care of. (Interview 6, CEC Core Lead)

In this context, university-based researchers may understand their roles as challenging or confirming stakeholder assumptions (e.g., on relevant routes of exposure and homesteading), and focusing the attention of stakeholders on potential risks that may otherwise be ignored. At the same time, community engagement may focus on outreach and education that supports environmental health literacy and community capacity for engagement in the remediation process (public information sessions, developing curricula for schools), and importantly, which builds relationships that enables community members to call on university-based researchers in a consultative role if required (as described by Stoecker, 1999).

4.3.5 Creating communities of concern across the academy

The previous two sections focused on how CECs work to form new communities of concern around persistent pollutant issues through space and time in potentially affected communities; however, the implication of new affected communities can implicate new people in the academy as well. One example of the constitution of new academic communities of concern is the increasing involvement of university-based researchers from the social sciences in

transdisciplinary communities of environmental health scholarship and practice (see Valk & Ewald, 2013; Elliott & Frickel, 2013; Hoover et al., 2015; Matz et al., 2016; Cordner et al., 2012). The methods and skills of these disciplines, from historical inquiry to ethnography to ethical reflexivity, are often useful in the engagement activities undertaken by CECs, contributing not only to the "social science *of* environmental health" but also "social science *with* environmental health" (Matz et al., 2016, p.352). Further, all informants spoke of how collaboration between training (supporting student trainees), research translation, and community engagement cores was contributing to the next generation of transdisciplinary researchers often with experience working at the bench, in the field, and with communities to drive change. Indeed, many individuals currently in leadership positions in SRCs are themselves former trainees. In this sense, the community engagement activities described previously not only (co)produce affected communities, but academic communities of environmental health that are action-oriented and methodologically diverse.

New academic communities of concern are also forming in response to the perceived need to speak out collectively, not just as individual researchers, on issues of policy relevance for (newly identified) affected communities. As one investigator described,

> I am a strong believer that research is relational... So in a sense some of these efforts are number one, to come up with useful documents that synthesize the science for policy, but number two, to create communities of science— scientists who can harness those connections to speak out in a more collective voice than just each of them. (Interview 3)

One example of these dynamics is related to research on dietary exposure to inorganic arsenic. While exposure to inorganic arsenic in drinking water is a well-known hazard, the extent of dietary exposure was not well characterized (Gilbert-Diamond et al., 2011; Jackson et al., 2012). Research from the Dartmouth SRC identifying certain food products as significant sources of dietary arsenic, like rice and rice-based sweeteners often used in first foods for young children, prompted concern amongst consumers of these foods (including disease group organizations like the New England Celiac Organization), and expecting parents/parents of young children. Engagement with these communities highlighted the dearth of regulatory guidance on arsenic in food (compared to water) for potentially vulnerable communities, prompting the Dartmouth SRC to sponsor the Collaborative on Food with

Arsenic and associated Risk Regulation (C-FARR).[10] Other examples of similar academic communities of concern that have formed around SRP-relevant hazardous substances, often with explicit goals of synthesizing knowledge for perceived community and policy needs, are the International PCB Workshop (Osterberg & Scammell, 2016; Hunt et al., 2016) and International Conference on Mercury as a Global Pollutant (Driscoll et al., 2013).

One community activist described why she thought the need for scientific communities to speak out collectively, in actionable ways, was so acute for science-based advocacy: "People always say 'I'm not a scientist, but...' because if you *are* a scientist, that holds a lot of weight...There's power in that." (Interview 14, community group partner) For activists like her then, these synthesis activities are important because of the cognitive authority that science, and the scientists that are seen to speak for it, commands in the media, and judicial, and regulatory contexts.

4.3.6 A conceptual framework for academic roles in supporting the building of communities of concern

Table 4.4 presents a set of five functions that university-based researchers can perform through community engagement to support the building of communities of concern around persistent pollutant issues and associated process evaluation questions. The emphasis on the role of the academic partner is not to suggest that these functions cannot be fulfilled by non-academic partners, including members of affected communities, community-serving organizations, or other civil society actors. Rather, it highlights that through programs like the SRP, academic partners often have resources (e.g., financial, expertise, time) within the context of a long-lived institutional structure that make them well-placed to do so.

Table 4.4: Framework to assess the academic role in supporting the building of communities of concern.

Function	Process evaluation questions
Identifying risk	What role did the academic partner play in identifying places and people affected or potentially affected by hazardous substances?
Building identity (as vulnerable to risk)	What role did the academic partner play in raising awareness and providing actionable knowledge about shared vulnerability?
Continued on next page	

---

[10]For examples of C-FARR output, see Cubadda et al. (2017); Nachman et al. (2017); Punshon et al. (2017); Taylor et al. (2017); Davis et al. (2017) for synthesis papers resulting from C-FARR included in a special issue of *Science of the Total Environment* and `http://www.dartmouth.edu/~arsenicandyou/` for an online resource for vulnerable communities, produced in partnership with SRCs at Columbia University, University of Arizona, University of North Carolina at Chapel Hill, University of California, Berkeley, and University of Kentucky.

Building environmental health literacy and capacity (university-based researchers)	How did the academic partner work to better understand the real-world needs of affected communities to improve their capacity to support them?
Building environmental health literacy and capacity (affected community)	How did the academic partner support the capacity of affected communities to respond to risk vulnerability (e.g., participate in decision-making about remediation or exposure reduction?
Enabling community organizing	What tools or approaches did the academic partner use to facilitate communities in organizing themselves around the issue of concern?

Identifying risk. As discussed in Section 4.2.2, knowledge production is central to the constitution of issue based communities. For persistent pollutants, the complex nature of their processing through social, technical, environmental, and biological systems (see Section 4.2.1) means that identifying places and people affected or potentially affected by these substances can be challenging, often requiring new research efforts (such as environmental and exposure monitoring). Further, identifying risk to support community building requires not just theoretical insights, but the linking of these theoretical insights to specific places and people.

Building identity (as vulnerable to risk). Once shared vulnerability to risk is identified (e.g., based on geography, based on physical/biological characteristics, based on diet), university-based researchers can support awareness raising and the dissemination of information about risk to build a shared identity in affected communities. This can take the form of outreach efforts (e.g., at community events, through media, educational campaigns).

Building environmental health literacy and capacity. In addition to providing knowledge about shared vulnerability, university-based researchers can help build capacity to act on this knowledge to respond to this vulnerability. However, environmental health literacy is a "two way street" (Interview 13)—university-based researchers can work to improve their own capacity to support affected communities by better understanding their real-world needs, while also supporting the capacity of affected communities to participate in decision-making on remediation or exposure reduction. Capacity building for university-based researchers may include opportunities for two-way exchange with affected communities, but also potentially research about affected communities (e.g., oral histories, survey research on risk attitudes). Supporting the capacity of affected communities to participate in decision-making on hazardous substances may include training and supporting scientific literacy,

including through participation in the scientific process, as well as creating opportunities for skills and leadership development more broadly (e.g., health-protective cooking skills).

Enabling community organizing. Academic partners can facilitate affected communities and (insider) organizers within them to organize themselves around the issue of concern, providing resources, tools, and spaces and platforms (physical, virtual) for organizing, while respecting the self-determination of affected communities. Academic partners can also support the organizing of academic and professional communities that are implicated by the issue of concern, convening these stakeholders through conferences, consortia, for collective voice.

4.3.7 Applying the framework

Figures 4-3 and 4-4 apply the framework described in the previous section to the SRP more broadly, focusing on the 9 out of 13 SRCs identified as having a strong focus on community constitutive functions (represented as A to I). Figure 4-3 indicates whether a SRC is fulfilling or attempting to fulfill a function that supports building a community of concern, and if so, the level of community participation in those activities. Figure 4-4 offers a preliminary assessment of the extent to which these SRCs have succeeded in performing these functions to date, using a qualitative ordinal scale that ranges from no evidence of activity to strong evidence of success, based on a review of available documentary evidence. Because the different SRCs operate on different timelines and within different contexts (see Table 4.2), the purpose of the evaluation is not to directly compare the centers to each other, but rather to highlight emergent patterns across the SRCs, working towards an understanding of barriers to success. In addition, the inherent selection bias of considering only funded SRCs should be noted—all funded centers as of 2017 have succeeded in a review process that includes explicit consideration of community engagement efforts.

Overall, Figure 4-3 affirms that constitutive functions with a stronger emphasis on community empowerment (building capacity in the affected community, enabling organizing) are more likely achieved with strategies that emphasize increased bidirectional interaction between university-based researchers and affected communities. However, Figure 4-3 also suggests that university-driven inquiry as part of community engagement, often involving social science research, can contribute to useful efforts to identify risk (for specific places

110

and people—which may not necessarily be a strong focus of SRC research projects) and to build environmental health literacy and capacity for university-based researchers. Forms of this research that are inherently more interactive, like oral histories, may also contribute to building identity, further highlighting the importance of social science methods (in addition to environmental or biomedical research about exposure and vulnerability) in identifying risk and understanding community needs. Identifying risk can also involve research with participatory elements for non-academic partners, even when the affected community is not yet well-organized. In some cases, this occurs when study participants participate in some aspects of the study itself, like sample collection. (This level of participation is distinct from CBPR, where community and academic partners participate more equally in all aspects of research design, implementation, and analysis.) Similarly, more traditional outreach and education efforts can be useful in building identity, and depending on the content of these efforts and the platform of delivery (e.g., creating new physical or virtual spaces for this interaction), can contribute to building capacity in affected communities and enabling organizing.

Figure 4-4 suggests that to date, SRCs have been most successful at fulfilling constitutive functions related to identifying risk, building identity, and building the capacity of university-based researchers to work with communities. In particular, CECs have been successful at identifying risk for specific places and people, sometimes, but not always, in collaboration with SRC research projects, and often utilizing social science methods alongside those from biological and environmental sciences. This finding suggests that knowledge that supports community building may involve distinct research efforts that build upon and extend overarching SRC research goals. However, SRCs have been less successful to date at building capacity in affected communities and enabling organizing. Further, as shown in Figure 4-3, these functions are not uniformly pursued by all SRCs. Several factors contribute to this uneven performance: perceived tension between "meeting communities where they are" and initiating concern; perceived tension between beneficence and communicating risk; divided communities; distrust; and length of time of engagement efforts.

When community engagement focuses on community building, it can come with its own set of ethical challenges. Indeed, community engaged activities, in general, can foreground ethical dilemmas because they differ from traditional research practice and the existing institutionalized ethical standards that govern them (Cordner et al., 2012; Cordner & Brown,

	Academic Partner Leads	Inform and Educate	Collaboratively Inform and Educate	Limited Consultation	Comprehensive Consultation	Co-researching
Enabling community organizing		D, H	A, C, D, F, I	G		C
Building capacity (affected community)		B, E, I	C, D	D		A, C, F
Building capacity (university-based researchers)	A, B, C, D, E, F, G, H, I		C	B, D, G		
Building identity	C, D	A, B, C, D, E, F, G, H, I	C, F, H			
Identifying risk	A, B, C, D, E, F, G, H, I					A, C, F

Figure 4-3: Process evaluation summary. The chart indicates whether a SRC is fulfilling (or attempting to fulfill) a function that supports building a community of concern, and if so, the level of community participation in those activities, for the 9 out of 13 SRCs identified as have a strong focus on community constitutive functions as part of their CEC (represented as A to I). The x axis is arranged with increasing levels of community participation from left to right, while the y axis is arranged with constitutive functions with increasing community empowerment from bottom to top.

2014; Morello-Frosch et al., 2015; Harding et al., 2012). CEC personnel attributed their desire to proceed slowly and carefully with identity building and organizing in part due to their awareness of these challenges. In the examples presented, those engaged in CE activities often voiced concerns about balancing between meeting communities where they were (Nyswander, 1956) (which aligns with the empowerment and self-determination principles of community-participatory work) and activities that focused on informing and educating (or as summarized by Stoecker (1999, p.843) "Can an academic adopt an initiator approach that is truly empowering?"). Time, energy, and resources (which are often in short supply, particularly for EJ communities) are required to be concerned about an issue and to act on
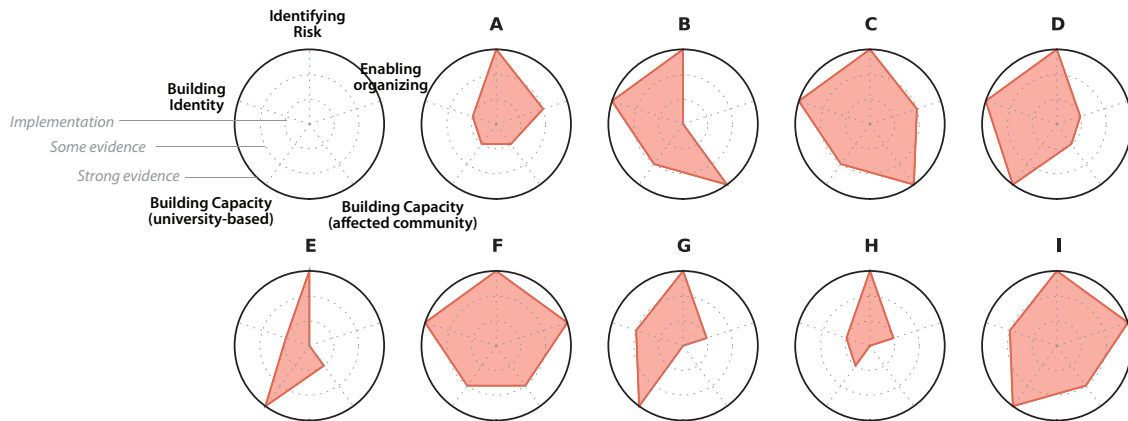
Figure 4-4: Process evaluation summary displaying a preliminary assessment of the extent to which SRCs A-I have succeeded in preforming these functions to date. The top left entry is a legend for the subsequent figures: the five spokes correspond to five constitutive functions, while concentric radii represent an ordinal scale of no evidence of activity, evidence of implementation of activities, some evidence of success, and strong evidence of success.

it, and in addition to issues surrounding hazardous substances or specific hazardous sub-stances, "there are other things to be focused on too, I'm afraid" (Interview 6, CEC Core Lead), related to environmental health or otherwise.

Another critical tension that interviewees articulated was between beneficence and com-munication (sharing information about communities' potential exposure with them). Could the labelling of a community as potentially "affected by a hazardous substance," particu-larly when there remains substantial scientific uncertainty about the possible exposure and its human health impacts, cause more harm to the community? Could concern be undue or unwarranted? Taken together, these concerns, and the resultant hesitancy on the part of those engaged in community engaged work at times to move forward on awareness building, capacity building, and enabling organizing, signal that increased attention and guidance on ethical report back procedures in environmental health is necessary (Ohayon et al., 2017; Hoover et al., 2015; Morello-Frosch et al., 2015).

One area where several informants felt particular care and reflection was required was the topic of dietary exposures. Because food is so linked with nutrition, livelihoods, and culture, informants argued that exposure reduction strategies (at least successful ones) were seldom straight-forward exhortations to avoid a food product. One informant explained, "If you're going to measure hazard in a food you also have to understand how you are going to

translate that risk message in a way that is culturally appropriate and does not cause greater harm" (Interview 5). Another discussed the challenge of creating concern about diet when scientific consensus about what to do given that concern—how much is safe to eat—was still developing. Finally, there were concerns that particularly in small communities where commercial food production takes place, research about the potential for contamination, even if ultimately none is found, could be permanently damaging to livelihoods.

Others highlighted different material and psychological disincentives to identifying as a "contaminated community" (Edelstein, 2004): "People don't want to know they're having these problems" (Interview 2, CEC Core Lead). Knowledge of contamination could have impacts for property values (see Senier et al., 2008, for one example), which several CEC leads reported as also being an important dimension of impact for community partners, in addition to human health. Studies have elucidated the potential psychological and affective consequences of identifying as a contaminated community (Zhuang et al., 2016; Clapp et al., 2016; Edelstein, 2004). One informant explained that their surveys and focus groups with residents living in areas at risk of groundwater contamination indicated that reasons for resistance to well-water testing included wanting to avoid retrospective guilt for exposing their families. As expressed by one survey participant:

> I mean it's shocking information. Here you are, you're raising some kids, your [sic] making sure they're in car seats, you're buckling them up every day, you're giving them good food, you think you're doing everything you can... And then you discover you've been feeding them poisonous water for ten years, and you can imagine how that makes you feel. (Survey participant qtd. in Evans-Brown (2014))

These affective and psychological consequences of knowledge of exposure are also an important dimension of community well-being (Edelstein, 2004).

Some researchers involved in CE acknowledged the need to reflect on how engagement efforts fit into existing tensions between and within stakeholder groups, and potential implications for community integrity. For instance, one informant described the mixed attitudes in the community with regards to the potentially polluting party:

> And it was the economic driver of that region...You had a sort of pride... and this whole, you know, putting forth the effort for the country kind of thing.

And I think that stood even in the face of some worker exposures that were later discussed... "We have great pride in our contribution to the nation. We're also concerned about our health." And then as time goes on that split keeps pulling wider and wider and wider. (Interview 13)

Recognition of power and resource imbalances within the ecosystem of stakeholders also contributed to a desire to proceed carefully with communicating new knowledge about potentially impacted groups.

Supporting community building and organizing activities is a long-term process that requires trust. Many communities disproportionately impacted by hazardous substances have historically seen limited benefit from research that has been extractive, and relationships with academic institutions that have been intermittent (e.g. Smith, 2012; Potts & Brown, 2005; Herising, 2005; Absolon & Willett, 2005). As one informant summarized,

You have a general distrust of people who want to come in and take things out. And those things might be coal or trees or any variety of things, or it might be my knowledge and my wisdom as a resident of this place. It might be cultural appropriation, it might be knowledge appropriation. And we have had a history in scientific research of drive by research in these communities. (Interview 13)

In communities that have experienced a long history of contamination, and where remediation has been contested, slow, or incomplete, distrust may apply not only to university-based researchers but to other stakeholders as well, like federal and state agencies and industry (Hoover, 2017). In contexts where distrust is high, and where affected communities' desire to engage with academic partners may be limited, interviewees suggested that significant time may be required to build relationships and trust, beginning with outreach before advancing to more sustained partnerships.

These challenges are not unique to the SRP (Cordner & Brown, 2014; Cordner et al., 2012). Taken together, they also indicate that those involved in community engagement activities see community health and well-being as multifaceted in nature—going beyond simply the "amount and toxicity" of hazardous substances (NIEHS, 2015a). They also indicate a belief (or hope) that despite these (ethical) complexities, new knowledge, and new identities, can ultimately be a source of empowerment and improved environmental health for affected communities.

4.4   Creating and sustaining communities of concern

This article began by asking what the long-term management of persistent pollutants might entail. Through an empirical case study of a research program with a mandate to support the management of hazardous substances—many of which are persistent—and a new emphasis on working with and for communities affected by these substances, it considered what role these university-community partnerships could play. Though community engagement was observed to encompass a range of activities, several programs focusing on persistent pollutants were conducting activities that seemed to be supporting community building: using research about communities to trace pollutants across space and time, and outreach and education activities (some of which were collaborative with members of affected communities) to enable (insider) community organizers to bring together those potentially affected along those dimensions as a community of shared interest. In parallel, these engagement activities are contributing to the organization of new academic communities. This constitutive role suggests that community engagement can be a site at which knowledge about contamination and community identity as jointly implicated by contamination are co-produced. Further, the case study clarified the different functions that university-based researchers may perform in support of community building (identifying risk, building identity, building environmental health literacy and capacity, enabling organizing) when the reach of pollutants is large in time and space. With these findings in mind, this article ends by reflecting on the larger institutional conditions under which university-based researchers can participate in work that supports the building of communities of concern around persistent pollutants.

Academic-community partnerships may play a valuable role in supporting the creation of communities of concern because of their ability to mobilize resources (e.g., financial, expertise-related) to monitor and detect the spread of pollutants. Though communities, particularly those that are "fence-line" or "front-line," have often been the first to identify and assess environmental health harms, pushing (or trying to push) experts and governments into action (see cases described by Brown & Mikkelsen, 1990; Corburn, 2005; O'Rourke & Macey, 2003; Edelstein, 2004; Ottinger & Cohen, 2011, as well as recent events in Flint, Michigan), interviews with community group activists and other literature (Hoover et al., 2015) have underscored the tremendous amounts of time, energy, and resources that may be required for communities to actively monitor and discover contamination when the

reach of persistent pollutants can be so wide in space and time. Universities command resources like funding, physical working space, and cognitive authority in many contexts that can support affected communities in these discovery activities, and subsequent organizing (Hoover et al., 2015). However, writing in the Austrian context, Felt et al. (2016) have questioned whether transformative transdisciplinary partnerships between universities and communities can exist without broader and more radical changes to "knowledge regimes" that currently prioritize efficiency of classical research outputs, and typically envision research on discrete 3-5 year time periods. Both of these conditions may also be challenges for academic-community partnerships in the US.

Existing literature and interviews with university-based researchers and community partners emphasize how continuity of the academic-community partnership is critical. Yet, sustaining these relationships is seldom an explicit target of funding: "So though it's not necessarily resourced through Superfund or other kinds of grants to maintain long term relationships, you know, that's what you end up doing with a lot of your resources" (Interview 13, CEC Core Lead). Sustaining these relationships if there is a break between major funding cycles is a particular challenge. This may represent a funding gap that dedicated intra-university or foundation grants could fill.

The tension between devoting resources to sustaining existing partnerships with affected communities and devoting resources to building new relationships with potentially affected communities is also a challenge. Some researchers conducting CE work expressed concern that grant reviewers may not see the value in sustained partnerships, putting greater value on establishing new relationships. Yet, both are important and necessary for the long-term management of persistent pollutants. Exploring the potential for new digital tools and emerging digital socialities to enable the scaling-up of engagement, as some SRCs have begun to do, may be one way of addressing this tension.

The sustainability of academic-community partnerships also requires that community engagement (and action research more broadly) be recognized as valuable work within the academic incentive structure. Early- and mid-career researchers interviewed were particularly cognizant of the need to "appeal to the tenure process" while delivering on what they felt were their ethical responsibilities to the communities they worked with: "I would say that the academic institution is slow to recognize the scholarly work associated with translational and community engaged and community based participatory research and public

participatory research processes... I think there is a scholarly act. Community engagement and research translation are not off the cuff things." (Interview 12, RT Core Lead) The limited budget allotted to cores (and not to research projects) under the SRP may contribute to the perception that these activities are differently valued. Nevertheless, academic culture is changing: "I would say that the role of the professor is changing and should be changing to incorporate more science communication engagement and translational work" (Interview 12, RT Core Lead). Another researcher spoke of the "tremendous amount of weight" (Interview 5) NIEHS has had as a funding body in shaping the culture of academic communities where there had been limited grassroots movement from researchers to engage in more community-partnered work: "You know, at the end of the day, you have to write to the RFA." Funding mechanisms embed values and norms, and attention to their design (including through scholarship) is therefore critical.

These structural changes may support academic-community partnerships that can create and sustain communities of concern for persistent pollutants. However, communities of concern may not be enough. The founder of a community-based environment and arts organization described the pond that had inspired her activism as a "rich but troubled place" (Interview 15, community partner). She spoke of the pond as a place of unexpected natural beauty within the city, despite the fact that contamination had severely limited the ways in which the surrounding community could interact with it. But, she said, "What do you do with a polluted pond? You can't turn your back on it. Instead, you ask, what are the things that *are* possible here?" Her question suggests that as much as we need machineries to manage the potential harms of these pollutants, we also need machineries to imagine in a collective sense what desirable futures might be (Jasanoff & Kim, 2009). What we need then, are not just communities of concern, but also *communities of possibility* that work to collaboratively (re)imagine what it means to meaningfully live, learn, work, and play in landscapes that have been permanently changed by human activity.

Appendix A

Supplemental Information for Chapter 2

A.1 Interview details

Our semi-structured interviews were conducted as part of a broader project on how key emitter countries approached the negotiation of the emissions article in the Minamata Convention. The interview instrument involved the following topics, with additional follow-up questions based on interviewee responses: country negotiating positions, coalitions between parties during the negotiations, and linkages between mercury and other environmental issues. In this context, domestic source controls in China and India emerged as key factors in the negotiating positions of these two countries, and the final treaty outcome. Our follow-up questions therefore focused on: (1) which political and technical (technology, geology) factors have contributed to domestic pollution control trends; (2) the extent and direction of influence between domestic pollution policy and the international treaty negotiations leading up to and during the negotiations; and, (3) interactions between air pollution, energy, and mercury policy, both on domestic and international fronts. Interviewees were also asked how these interactions could evolve in the future, given the treaty outcome. Insights from the interviews were then used in conjunction with literature review (including peer reviewed sources, and technical reports from governments, and international agencies) to develop technology scenarios that reflect global trends in air pollution and energy policy re-

lated to mercury emissions. We use these multiple data sources to triangulate any findings, verifying insights from interviews with documentary evidence.

Interviews were conducted with the following individuals:

- Senior developed country negotiator, January 2014

- Coal expert knowledgeable about India, January 2014

- Three United Kingdom negotiators, January 2014

- Chinese negotiator, January 2014

- Negotiator from the European Union, January 2014

- Senior developed country negotiator, February 2014

- Indian power sector expert, February 2014

- US negotiator, March 2014

- US negotiator, March 2014

- UNEP Programme Officer in Chemicals Branch, April 2014

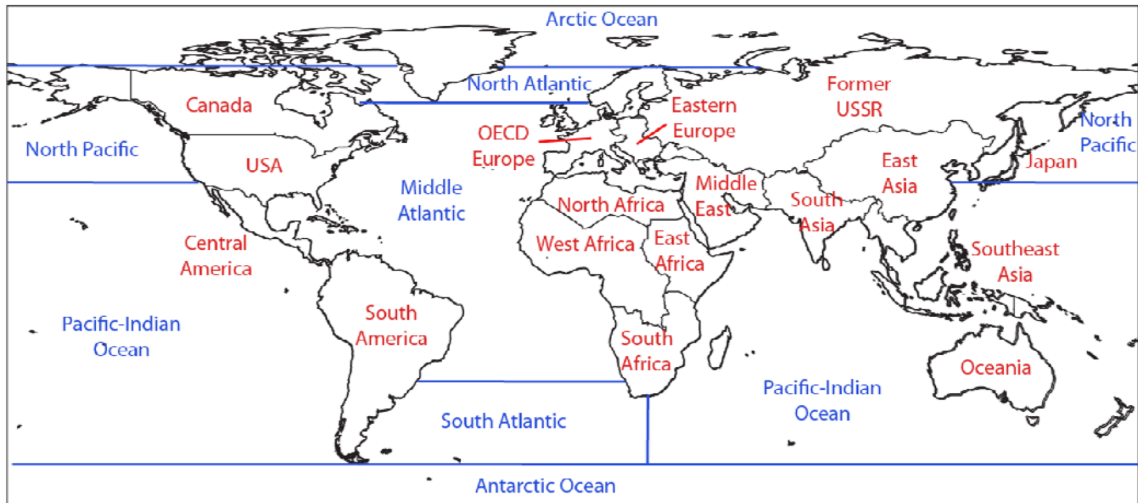## A.2   Chemical transport modelling regions

Figure A-1: Regions used for development of emissions scaling factors, based on (Streets et al., 2009) and (Soerensen et al., 2010), from (Corbitt et al., 2011).

A.3 Additional review of available Hg control technologies and techniques

A.3.1 Pre-combustion controls

Coal beneficiation, which involves both coal washing and specific treatment to reduce mercury content, can lead to reductions of up to 70% (UNEP, 2010). Blending with lower mercury coals, and additives to increase halogen content, when combined with appropriate post-combustion controls, can lead to mercury removal of up to 80% (UNEP, 2010). The presence of halogens promotes the oxidation of Hg^0 to Hg^{2+} in combustion or post-combustion (Pacyna et al., 2010b; UNEP, 2010). Oxidized mercury, because it is water soluble, is more easily captured by post-combustion controls. Halogen additives may be particularly beneficial for sub-bituminous coals, which tend to have both lower halogen content than bituminous coals, and more alkaline compounds that competitively inhibit mercury oxidation by halogens (Pacyna et al., 2010b; UNEP, 2010).

A.3.2 Post-combustion controls

ESP and FF are the most commonly applied commercial PM control devices, however, the penetration of ESP is much higher worldwide (Sloss, 2012a,b; Srivastava et al., 2006). Average capture efficiencies for cold-side ESP in the US have been reported as 29% and

3% for bituminous and sub-bituminous coals respectively, and as high as 63% and 18% (Srivastava et al., 2006). Similar values have been reported in China for bituminous coals: up to 42% with an average of 30% (Wang & Shen, 2000; Wang et al., 2010, 2009; Wu et al., 2010; Zhang et al., 2008; Zhou et al., 2008; Zhu et al., 2002). For India, the range of reported values for bituminous coals is slightly larger, from 19% to 73% (UNEP, 2014). Previously, it has been hypothesized that because of the high ash and low sulfur content on Indian coals, ESP systems may not capture mercury as effectively (Sloss, 2012b). Evidence from a recent UNEP study focusing on emissions from Indian plants suggests that a higher proportion of mercury remains in the fly ash in Indian plants, which may counteract this effect (UNEP, 2014). The higher collection efficiencies for bituminous coals may be due to their higher sulfur content, which creates fly ash that is more easily collected by ESPs (UNEP, 2010). Generally FFs are more effective PM and mercury co-benefit control devices than ESPs (Wang et al., 2008). FFs capture a larger fraction of gaseous mercury, leading to reported total mercury capture efficiencies greater than 90% for bituminous coals in the US, and an average of 52% in China (Srivastava et al., 2006; US EPA, 2002; Wang et al., 2009; Zhang et al., 2008).

PM control in combination with desulfurization systems like wFGD can increase mercury collection for both bituminous and sub-bituminous coals. Because this combination is most effective at capturing Hg^P and Hg^{2+}, resulting emissions are predominantly in gaseous elemental form (Wu et al., 2010). Results from a field study of ESP + wFGD in the US demonstrated an average of approximately 70% reduction, and a maximum of 74% for bituminous coals (Srivastava et al., 2006), and results from a recent US Environmental Protection Agency (EPA) information collection request (ICR) fall in a similar range (US EPA, 2010). For sub-bituminous coals, the average reduction was recorded as only approximately a quarter that of the higher-rank coals, though values from the recent ICR tend higher (Srivastava et al., 2006; US EPA, 2010). Values from China also show a similar range (Wang et al., 2010, 2009). The combination of FF + wFGD can be more effective, as mercury oxidation can occur across the filter, facilitating collection downstream. Reported values in the US have been as high as 98% for bituminous coals (US EPA, 2002). Similarly, the addition of SCR can increase oxidation and improve mercury capture by FGD. For instance, combinations of PM, sulfur, and NOx control have been shown to average above 90% mercury reductions (Srivastava et al., 2006; US EPA, 2002).

A.4 Coal Characteristics

Table A.1: Average domestic coal characteristics. Values presented are averages of limited numbers of samples, presented in the reference studies. Percentages are by mass. We report only bituminous coal characteristics for China, because the majority of power generation coal is of this rank (Wu et al., 2010; Streets et al., 2005). For India, samples were a mix of bituminous and sub-bituminous coals, fed to power plants.

Country	Ash (%)	Moisture (%)	S (%)	Cl+Br (ppm)	Mercury (ppm)	HHV (J/g)	Reference
India	34.94	6.31	0.55	153	0.14	18590	(UNEP, 2014)
China	13.57	6.8	0.53	271	0.045	27590	(UNEP, 2011)

A.5 Deposition Data

Table A.2: Gross deposition statistics. Spatially averaged 2050 deposition fluxes $\mu g \cdot m^{-2} \cdot y^{-1}$ (and total masses ($Mg \cdot y^{-1}$)) for India, China, the US, and selected oceans are tabulated for each technology scenario. These simulations assume an A1B energy and development trajectory (Streets et al., 2009), and no in-plume reduction of power plant emissions. Note that total changes in deposition (by mass) exceed changes in anthropogenic emissions because anthropogenic emissions changes also result in changes to re-emissions from land, snow, and oceans in the model.

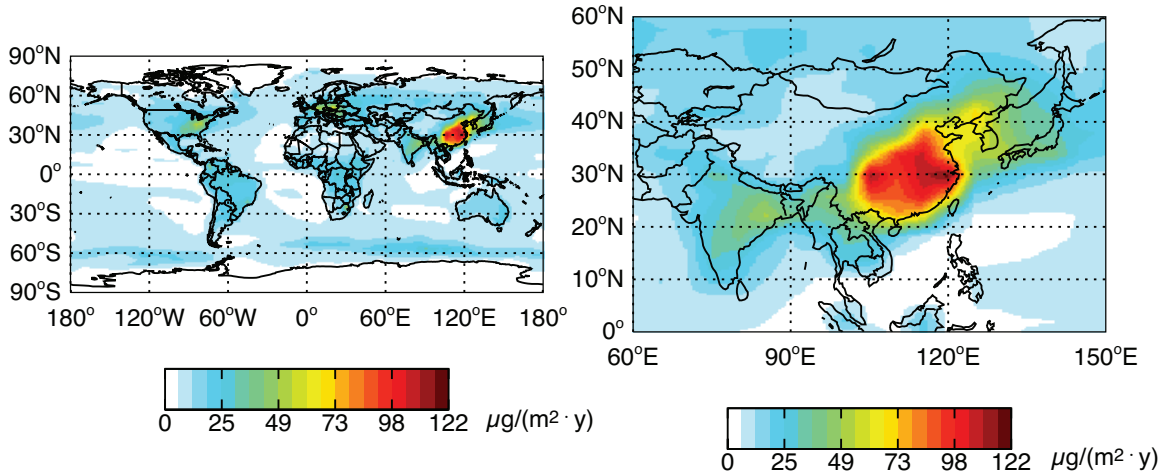| Tech Scenario	China	India	US	N. Pacific	S. Pacific/ Indian	Atlantic
NAC	49.5	96.6	31.8	19.0	11.4	12.7
	(462)	(371)	(283)	(573)	(2106)	(869)
MF	47.4	83.3	30.9	18.3	10.9	12.3
	(442)	(320)	(275)	(552)	(2022)	(839)
MS	45.2	57.1	30.6	18.0	10.6	12.1
	(421)	(219)	(272)	(541)	(1960)	(829)
NAC - MF	2.14	13.4	0.9	0.7	0.5	0.4
	(20.0)	(51.3)	(8.02)	(21.4)	(83.7)	(29.6)
MF - MS	2.23	26.1	0.3	0.3	0.3	0.2
	(20.8)	(100)	(2.9)	(10.3)	(61.9)	(10.0)

Figure A-2: Gross deposition (of all species) under a "present day" inventory, based on 2006 totals from (Streets et al., 2009), spatially located according to (Pacyna et al., 2010a).

A. Δ Gross Deposition, NAC - MF
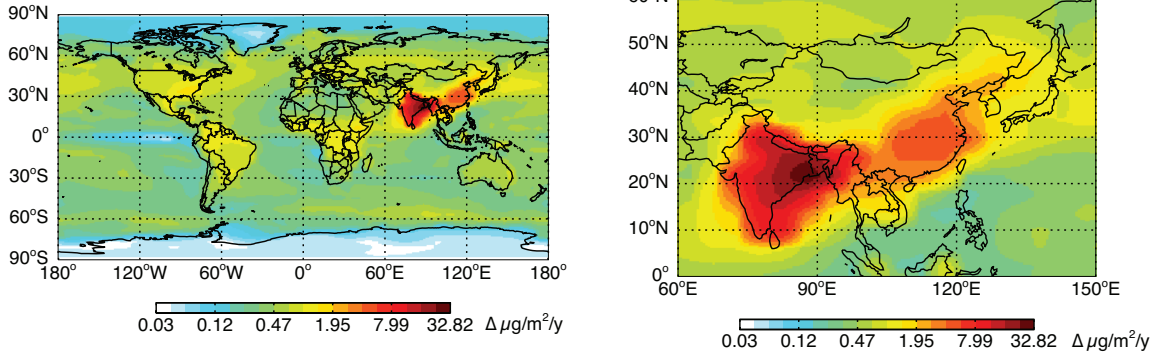
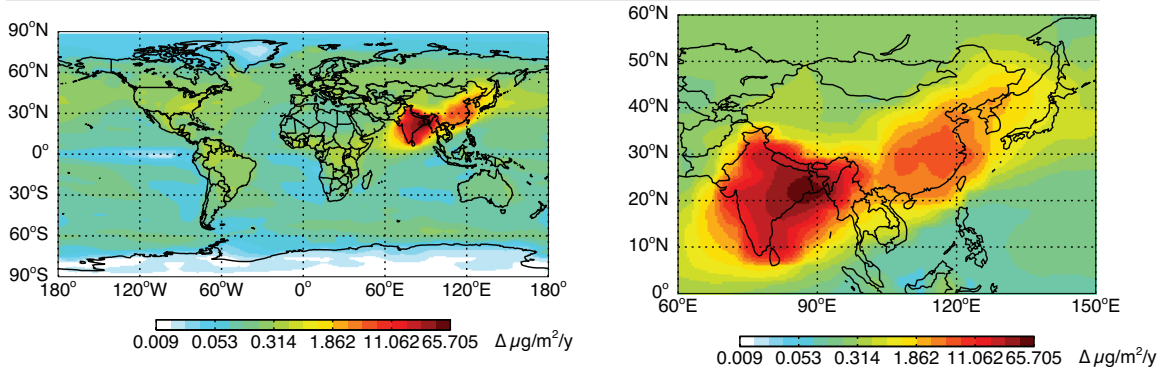**B. Δ Gross Deposition, MF - MS**

Figure A-3: Difference in 2050 gross deposition (of all species) between technology scenarios, under base case assumptions. Deposition differences are shown in log scale to better illustrate spatial variation.

A. Δ Gross Deposition, NAC - MF

B. Δ Gross Deposition, MF - MS

< 0.01 1.01 2.01 3.00 4.00 5.00 > Δ µg/m²/y
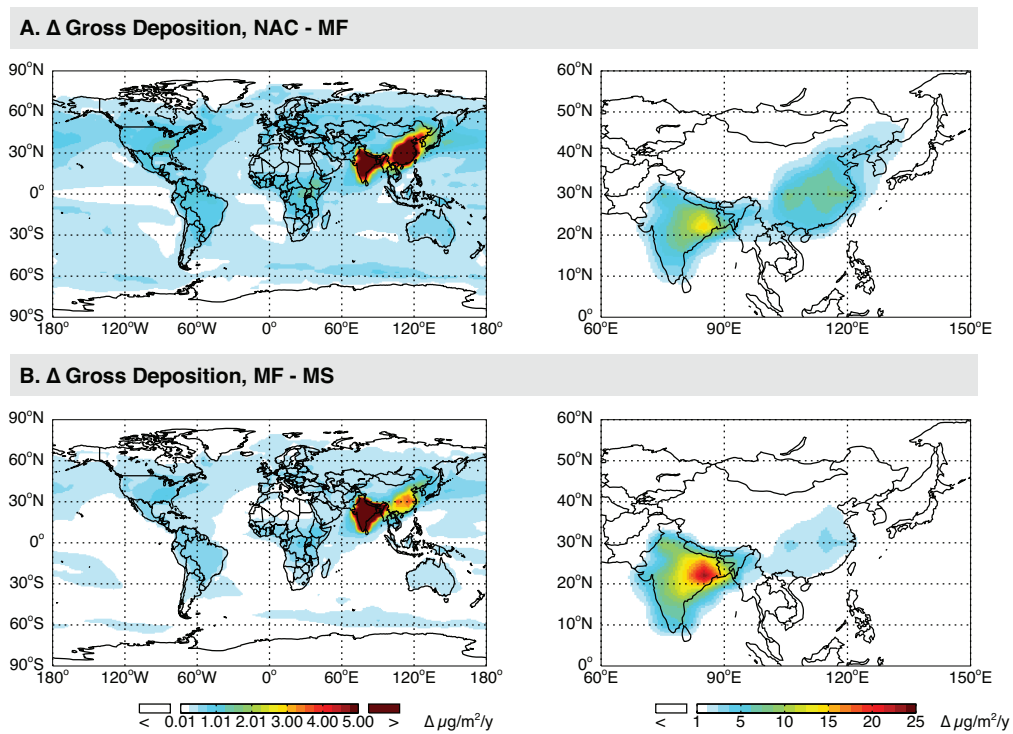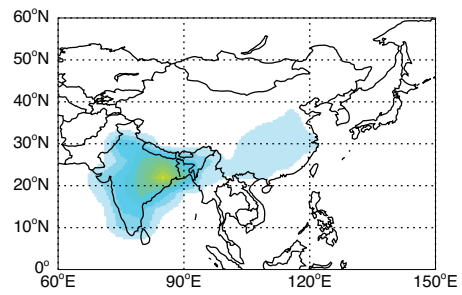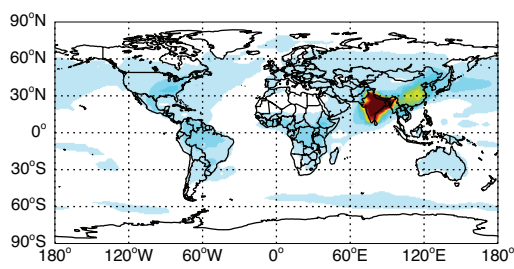
< 1 5 10 15 20 25 Δ µg/m²/y

Figure A-4: Difference in 2050 gross deposition (of all species) between technology scenarios, under the IPR assumption.

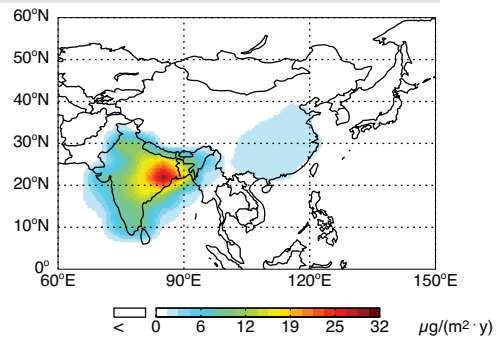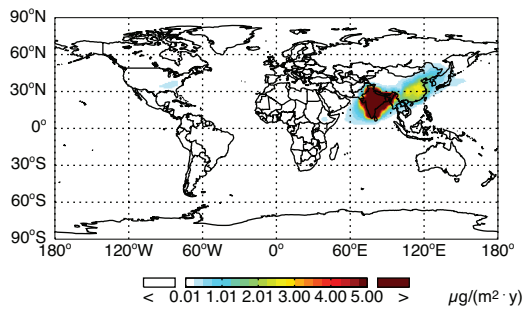**A. Δ Gross Deposition, NAC - MF**

B. Δ Gross Deposition, MF - MS

Figure A-5: Difference in 2050 gross deposition (of all species) between technology scenarios, under a B1 energy and development scenario (Streets et al., 2009).

A.6   Sensitivity analysis

We explore the sensitivity of our deposition results to our assumptions of $f_{capture\ China}$, $f_{capture\ India}$, $f_{Hg0\ China}$, $f_{Hg0\ India}$. We perturb each of the above variables by $\pm 20\%$ from the MF scenario, and evaluate the impacts on deposition. Table A.3 presents the response ratios, α, calculated based on this sensitivity analysis. These α values represent the % change in deposition flux corresponding to a 1% change in each perturbed variable, based on the 20% perturbation. From this sensitivity analysis, we can also calculate deposition responses by mass: $\beta = \frac{\Delta dep}{\Delta THg}$ and $\gamma = \frac{\Delta dep}{\Delta Hg0 \rightarrow Hg2}$, where dep is deposition over a region, by mass, ΔTHg is a change in total mercury emissions mass, holding speciation constant, and $\Delta Hg0 \rightarrow Hg2$ is a mass conversion of Hg0 to Hg2. These values are presented in Table A.4.

We estimate that approximately 90% of the deposition decrease in China between NAC and MF are due to changes in China ($\approx 60\%$ from capture efficiency, and $\approx 30\%$ from speciation), with the remaining 10% attributed to changes in Indian capture efficiency. For India, less than 5% of deposition changes between NAC and MF are due to changes in China, with the remaining portion due to changes in Indian capture efficiency. Between MF and MS, we estimate that approximately 20% of the deposition changes in China are due to changes in India (a negative contribution to deposition from changes in capture efficiency, and a small positive contribution from changes in speciation). In contrast, for India, changes in China only account for 0.7% of modeled deposition decreases between MF and MS. The majority (60%) of the modeled deposition decrease in India is due to changes in $f_{Hg0\ India}$, which increases by approximately 38% between MF and MS.

We use these response ratios to explore how the range of reported performance (in terms of $f_{capture}$ and f_{Hg0}) within each scenario affects our estimates of deposition. We consider a low estimate case, where we use the low end of the ranges for both parameters and countries for each technology scenario, and a high estimate case, where we use the high end of the range. Table A.5 shows the estimated ranges for $f_{capture}$ and f_{Hg0} under the technology scenarios. We then use deposition response ratios by mass, from Table A.4, to estimate changes in deposition to India and China, shown in Table A.6.

Under the low case, we find that projected power sector emissions decreases between the NAC and MS scenarios (NAC - MS) for India are slightly larger (+9 Mg) than the

base case; however, projected deposition decreases are close to 3 times as large, as a much larger fraction of reduced emissions is assumed to be in divalent form. For China, under the low case, projected emissions decreases between NAC and MS are about 3 times as large as in the base case, resulting in deposition benefits that are approximately twice as large. While the benefits of adopting more effective control technologies are larger if we assume low performance, absolute deposition is projected to be between 8 and 22% higher in China (across technology scenarios) and between 38 and 80% higher in India, compared to our base case performance assumptions. Under the low case and an A1B energy and development scenario, projected 2050 total and power sector emissions increase absolutely over present day (2010) estimated emissions for both countries, under all technology scenarios (Arctic Monitoring and Assessment Programme & United Nations Environment Programme, 2013).

Under the high case, projected emissions decreases between NAC and MF are smaller than the base case for India (-121 Mg), and slightly larger than the base case for China (+12 Mg). Resulting deposition changes to China between NAC and MF are approximately equivalent in magnitude between the base and high cases; however, their distribution shifts such that the majority of deposition benefits are achieved between NAC and MF. Under the high performance case, the assumed capture efficiency of the SCR + CS-ESP + wFGD technology suite (MF) is already close to 100% (97%), leading to a smaller marginal gain from switching to from CS-ESP to FF (MS). The Indian deposition benefit between NAC and MF decreases by approximately 50% compared to the base case. Absolute deposition is projected to be between 6 and 10% lower (across technology scenarios) in China, and between 13 and 25% lower in India, than in the base case. With high performance assumptions, 2050 power sector emissions from China, with an A1B energy and development trajectory, are lower than the 2010 estimate (97 Mg (Arctic Monitoring and Assessment Programme & United Nations Environment Programme, 2013)) under the MF and MS scenarios. For India, projected total and power sector emissions under high performance assumptions exceed estimated present day emissions, under all technology scenarios.

Table A.3: Deposition response ratios (α). We calculate the % change in average deposition flux for the tabulated regions given a 1% increase in $f_{capture\ China}$, $f_{capture\ India}$, $f_{Hg0\ China}$, $f_{Hg0\ India}$, based on 20% perturbations around the MF scenario. A negative value implies that the direction of change in the deposition variable is opposite that of the perturbed variable (i.e. An increase in $f_{Hg0\ India}$ decreases average Indian deposition flux).

Perturbed Var.	India	China	US	N. Pacific	S. Pacific/ Indian	Atlantic
$f_{capture\ China}$	-0.022	-0.382	-0.037	-0.158	-0.289	-0.115
$f_{capture\ India}$	-0.561	-0.045	-0.032	-0.079	-0.420	-0.124
$f_{Hg0\ China}$	0.003	-0.133	0.006	-0.019	0.014	0.020
$f_{Hg0\ India}$	-0.663	0.002	0.016	0.038	-0.049	0.065

Table A.4: Deposition response by mass (β, γ). We calculate the change in deposition by mass, given a unit change in THg, and $Hg0 \rightarrow Hg2$. A negative value implies that the direction of change in the deposition variable is opposite that of the perturbed variable.

Perturbed Var.	ΔIndia	ΔChina
ΔTHg_{China}	0.013	0.305
ΔTHg_{India}	0.318	0.035
$\Delta Hg0 \rightarrow Hg2_{China}$	-0.010	0.536
$\Delta Hg0 \rightarrow Hg2_{India}$	0.706	-0.002

Table A.5: Ranges for technology scenario performance, in terms of $f_{capture}$ and f_{Hg0}.

Tech Scenario	$f_{capture}$ China	f_{Hg0} China	$f_{capture}$ India	f_{Hg0} India	Sources
NAC	13% - 75%	75% - 100%	19% - 73%	27% - 67%	China: Zhu et al. (2002); Wang et al. (2010); Wang & Shen (2000); Zhang et al. (2008); Wang et al. (2009); Zhou et al. (2008); Wang et al. (2008); Wu et al. (2010) India: UNEP (2014); Wu et al. (2010)
MF	65% - 97%	60% - 80%	31% - 85%	27% - 67%	China: United Nations Environment Programme (2010); Krishnakumar et al. (2012) India: UNEP (2014); United Nations Environment Programme (2010); India Central Electricity Authority (2012)
MS	80% - 99%	60% - 80%	49% - 90%	79% - 100%	China: Srivastava et al. (2006); United Nations Environment Programme (2010); US EPA (2002) India: Krishnakumar et al. (2012)

Table A.6: Range of 2050 power sector emissions (Mg THg) and spatially averaged deposition fluxes $\mu g \cdot m^{-2} \cdot y^{-1}$ (and total masses ($Mg \cdot y^{-1}$)) for India and China, calculated using deposition response ratios from Table A.4, and performance ranges from Table A.5.

Estimate type	Tech Scenario	China power emissions	India power emissions	China deposition	India deposition
Low	NAC	594.6	835.4	60.5 (563.4)	174.6 (670.5)
	MF	264.7	722.2	53.0 (493.5)	155.8 (598.2)
	MS	169.6	552.5	48.9 (455.7)	78.6 (301.7)
	NAC - MF	329.9	113.2	7.5 (69.9)	18.8 (72.3)
	MF - MS	95.1	169.7	4.1 (37.8)	77.2 (296.6)
High	NAC	201.3	326.1	44.8 (417.5)	72.13 (277.0)
	MF	61.7	212.9	43.1 (401.2)	62.3 (239.3)
	MS	49.0	165.8	42.6 (396.7)	49.7 (190.9)
	NAC - MF	139.6	113.2	4.5 (16.3)	9.8 (37.7)
	MF - MS	12.7	47.1	0.5 (4.5)	12.6 (48.4)

A.7 Legacy anthropogenic impacts

We use the seven reservoir, global biogeochemical box model for mercury cycling developed by Amos et al. (Amos et al., 2013, 2014), and available online at `http://bgc.seas.harvard.edu/models.html`, to approximately estimate the extent to which the deposition difference results reported in this study underestimate legacy anthropogenic effects. In contrast to GEOS-Chem v9-02, which uses slab oceans and soil, the Amos et al. (Amos et al., 2013, 2014) model fully couples an atmospheric, three oceanic (surface, subsurface, and deep), and three terrestrial (fast, slow, and armored) reservoirs. The fully coupled, multi-compartment ocean and soil reservoirs allow us to investigate how the multi-decadal emissions trajectory from present-day to 2050 impacts legacy pools of mercury in ocean and soil, from which mercury can be re-emitted.

The model is initialized with natural emissions, and then driven from the pre-anthropogenic period to 2050 using historical anthropogenic emissions (Streets et al., 2011), and projected 2050 A1B emissions (Streets et al., 2009), with totals adjusted to match the NAC, MF, and MS technology scenarios. The emissions trajectory from 2008 to 2050 is assumed to be linear.Following Amos et al. (Amos et al., 2013), we calculate legacy anthropogenic contributions to 2050 deposition as total deposition minus primary anthropogenic emissions and steady state natural emissions (calculated without anthropogenic forcings). Deposition results are presented in Table A.7.

Table A.7: Estimates of legacy anthropogenic contributions to total deposition for 2050 A1B, based on global biogeochemical box model (Amos et al., 2013, 2014).

Technology Scenario	Total Deposition (Mg)	Primary Anthro. (Mg)	Legacy Anthro.(Mg)
NAC	9268	3773	5213
MF	8955	3533	5140
MS	8733	3363	5088
NAC - MF	313	240	73
MF - MS	222	170	52

Appendix B

Supplemental Information for Chapter 3

B.1 Additional emissions information
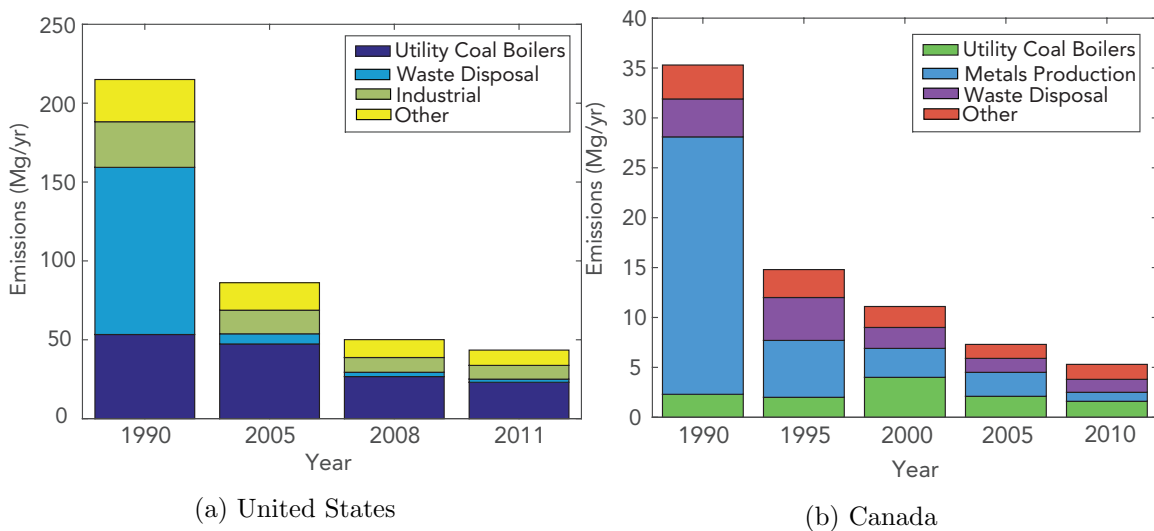

Figure B-1: Emissions trends in the US and Canada. Data for the US is from US EPA (2016). Data from Canada is from Steffen (2016).
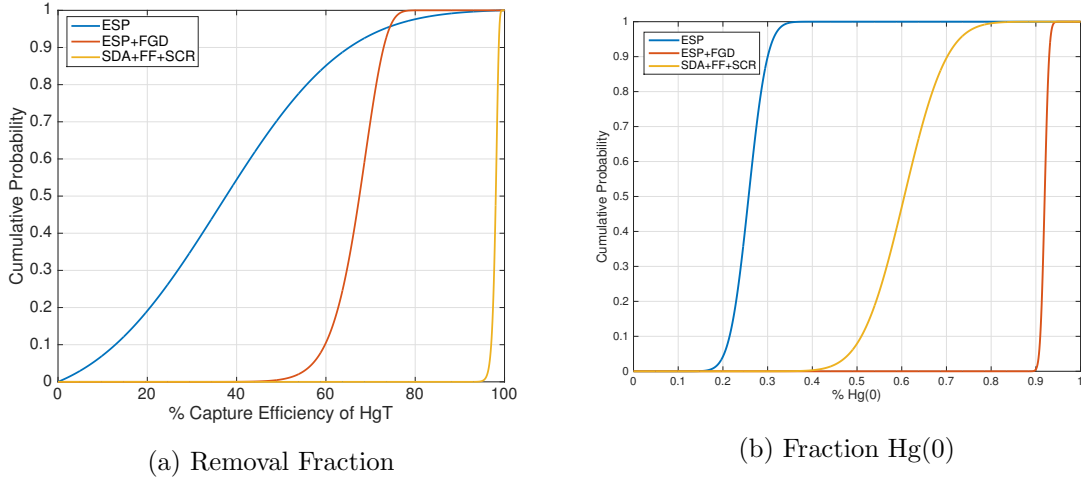
(a) Removal Fraction

(b) Fraction Hg(0)

Figure B-2: Cumulative distribution functions of removal fraction and fraction Hg(0) for different air pollution control configurations at the individual plant level, using data from Bullock & Johnson (2011).

Table B.1: Parameterization for variability in the population mean of removal fraction and fraction Hg(0) in flue gas, for air pollution control technology configurations. Parameters μ and σ are for normal distributions.

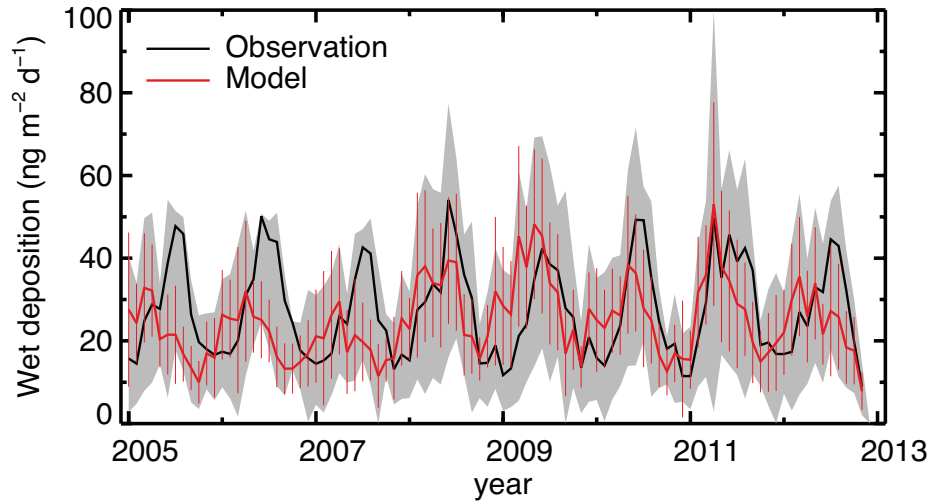|	μ	σ
Removal Fraction		
ESP	0.294	0.039
ESP+FGD	0.778	0.017
SDA+FF+SCR	0.974	0.007
Fraction Hg(0)		
ESP	0.258	0.033
ESP+FGD	0.920	0.008
SDA+FF+SCR	0.606	0.075

B.2 Model-Observation Comparison

Figure B-3: Model-observation comparison of monthly mean wet deposition fluxes from 2005-2012. Mean values from MDN sites are shown in black, with grey shading representing the standard deviation of individual sites. Modelled values are shown in red, with red bars indicating the standard deviation in individual sites. The Pearson correlation coefficient $r = 0.41$ for the entire period, while $r = 0.65$ in the recent period of 2010-2012.
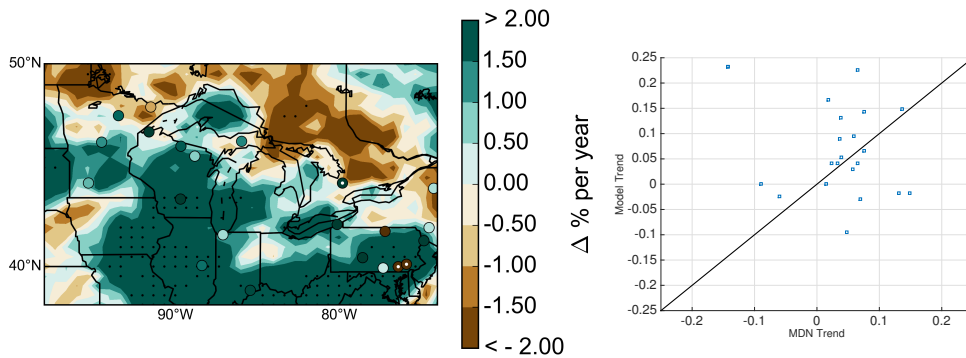

Figure B-4: Comparison of modeled (background) and observed (filled circles) 8 year trends (2005-2012) in wet deposition, using SMK trend test and Theil-Sen estimator of slope. Monitoring sites and model grid cells with a significant trend (p<0.1) are indicated with a dot.

B.3 Additional wet deposition results

(a) NA Signal　　　　　　　　　　　(b) ROW signal

Figure B-5: Change in precipitation weighted concentration (%) between pre-policy and post-policy period in Policy Only simulation. Grid cells with a significant (p<0.1) change are indicated with a dot. For reference, locations of monitoring stations are indicated with triangles.

(a) NA Signal　　　　　　　　　　　(b) ROW signal
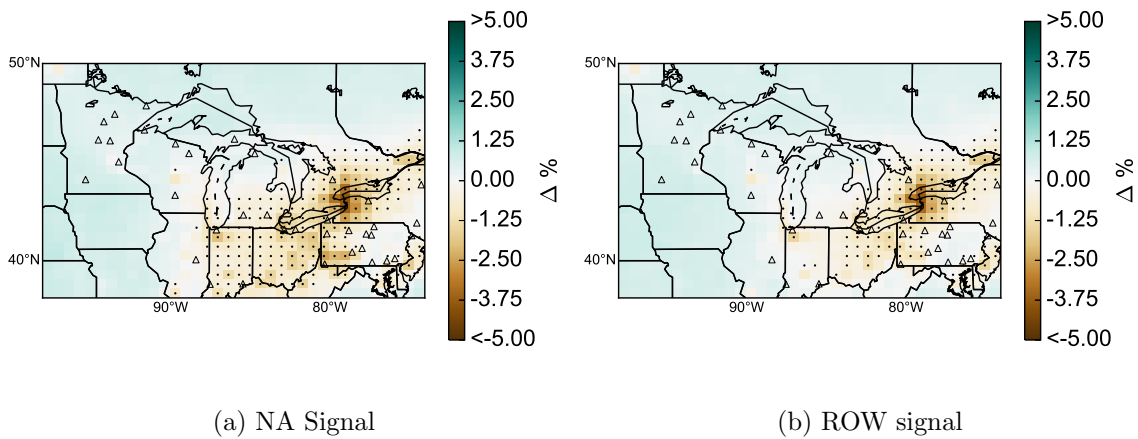
Figure B-6: Change in precipitation weighted concentration (%) between pre-policy and post-policy period in Energy and Economic Trends simulation. Grid cells with a significant (p<0.1) change are indicated with a dot. For reference, locations of monitoring stations are indicated with triangles.

(a) NA Signal

(b) ROW signal

Figure B-7: Change in wet deposition (%) between pre-policy and post-policy period in Product Emissions Trend simulation. Grid cells with a significant (p<0.1) change are indicated with a dot. For reference, locations of monitoring stations are indicated with triangles.
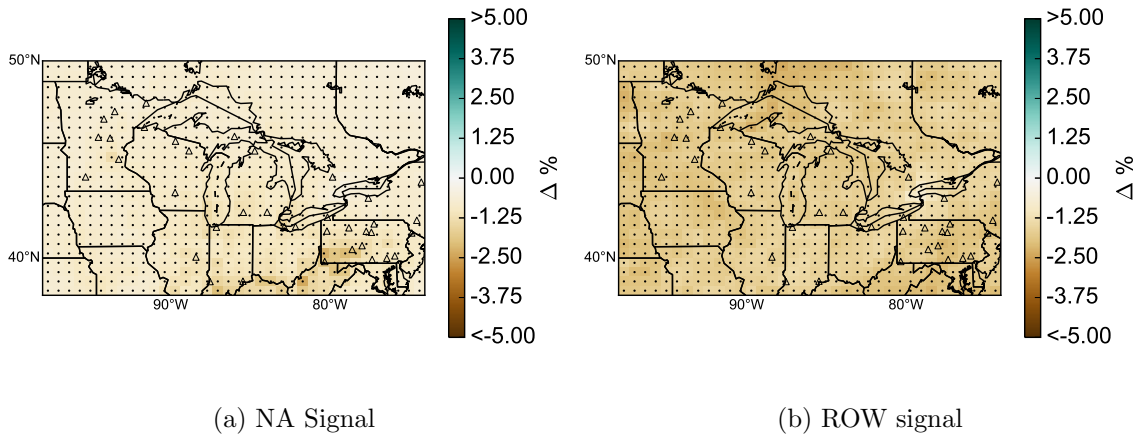

(a) NA Signal

(b) ROW signal
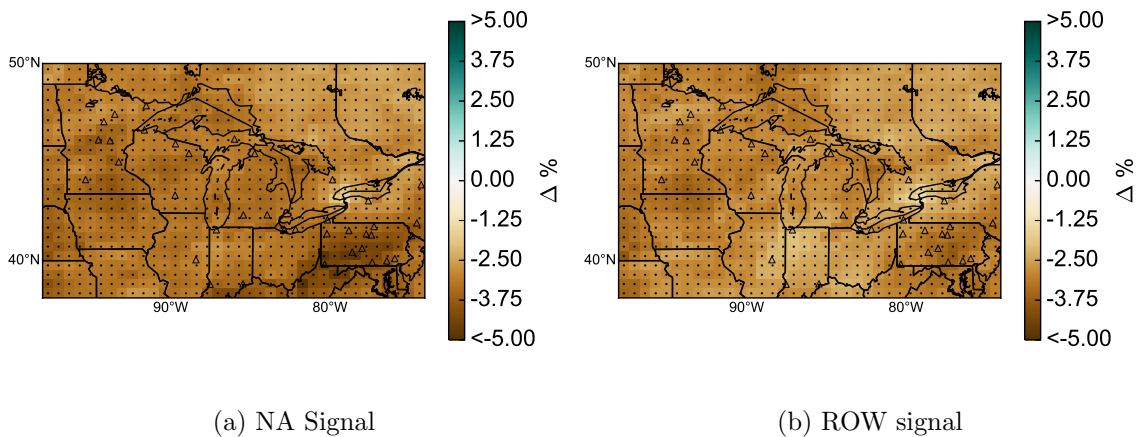
Figure B-8: Change in precipitation weighted concentration (%) between pre-policy and post-policy period in Product Emissions Trends simulation. Grid cells with a significant (p<0.1) change are indicated with a dot. For reference, locations of monitoring stations are indicated with triangles.

(a) Policy Only

(b) Removal Variability
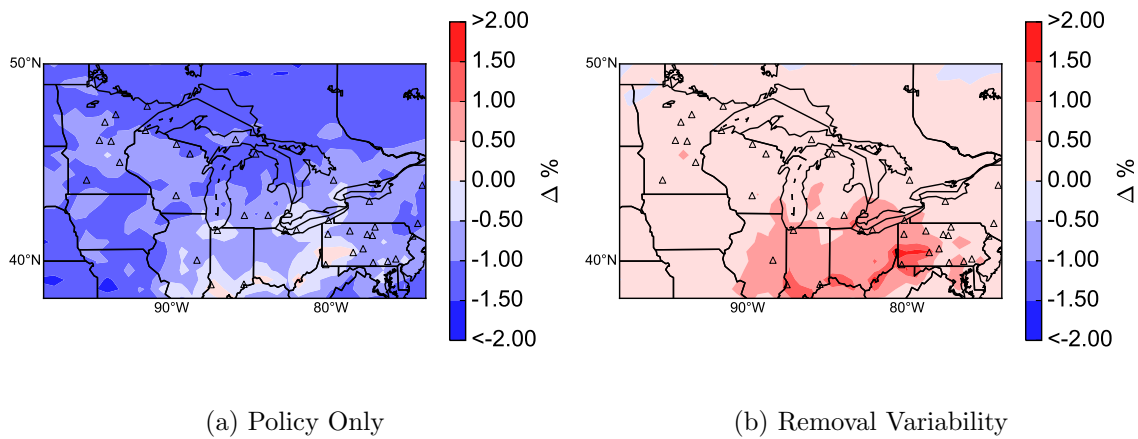
Figure B-9: Comparison of ROW Signal and NA signal (ROW-NA) in wet deposition, for Policy Only and Removal Variability simulations. Because all values are negative, regions that are red indicate areas where the NA signal is stronger.

(a) NA Signal

(b) ROW signal
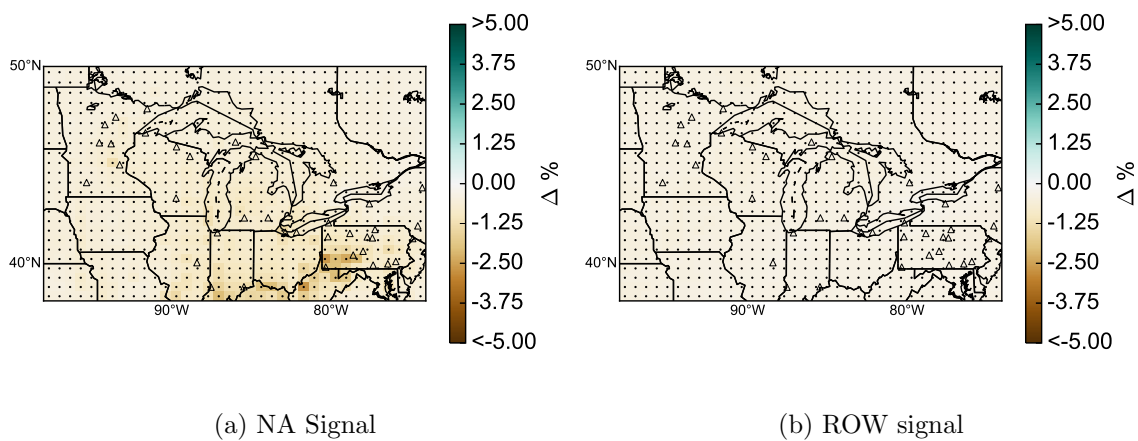
Figure B-10: Change in precipitation weighted concentration (%) between pre-policy and post-policy period in Removal Variability simulation. Grid cells with a significant (p<0.1) change are indicated with a dot. For reference, locations of monitoring stations are indicated with triangles.
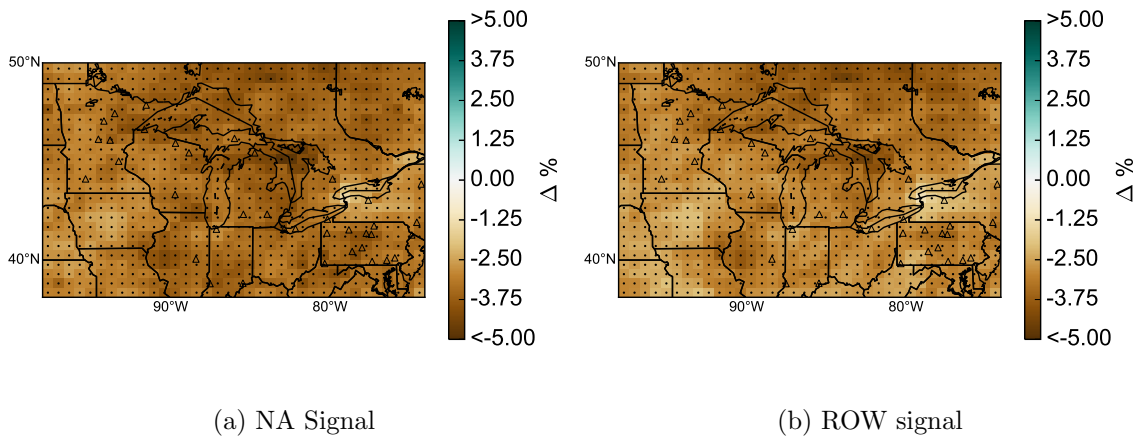
(a) NA Signal   (b) ROW signal
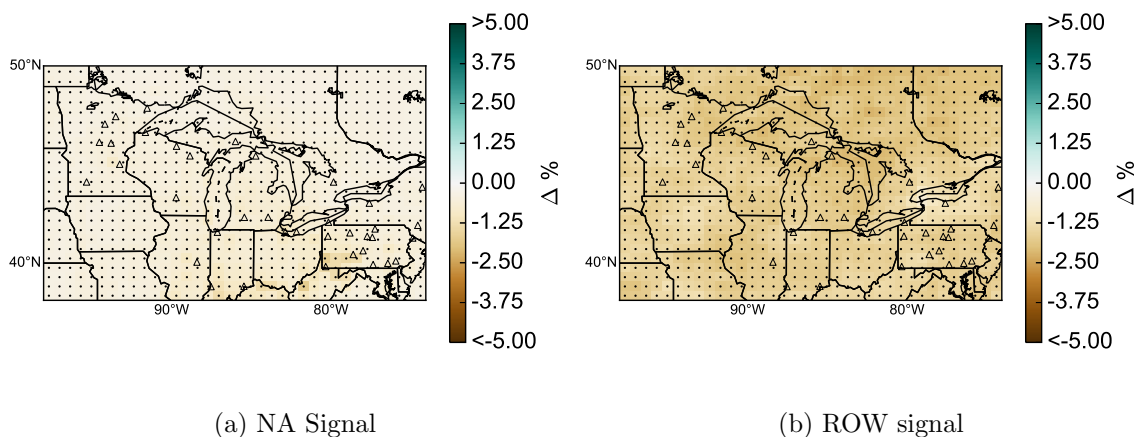
Figure B-11: Change in wet deposition (%) between pre-policy and post-policy period in Speciation Variability simulation. Grid cells with a significant (p<0.1) change are indicated with a dot. For reference, locations of monitoring stations are indicated with triangles.

(a) NA Signal   (b) ROW signal
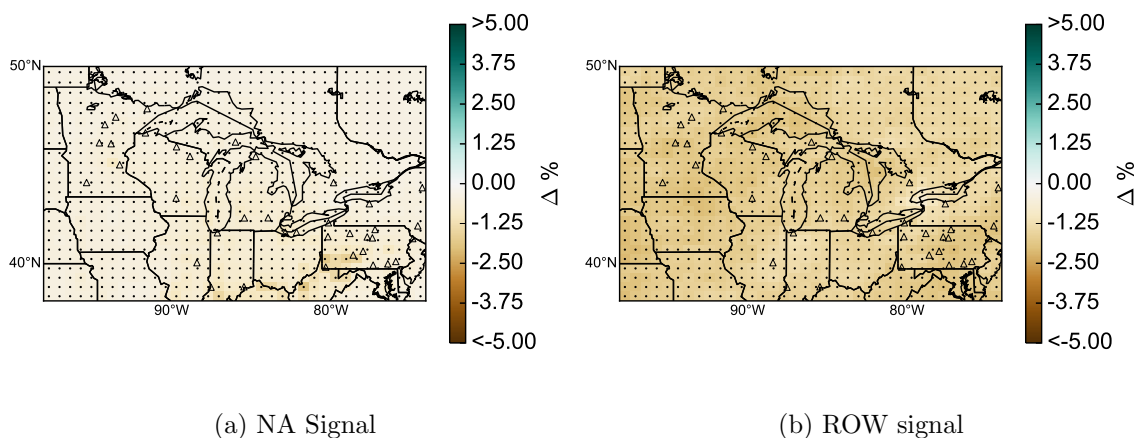
Figure B-12: Change in precipitation weighted concentration (%) between pre-policy and post-policy period in Simulation Variability simulation. Grid cells with a significant (p<0.1) change are indicated with a dot. For reference, locations of monitoring stations are indicated with triangles.

B.4 Emissions Inventory Comparison
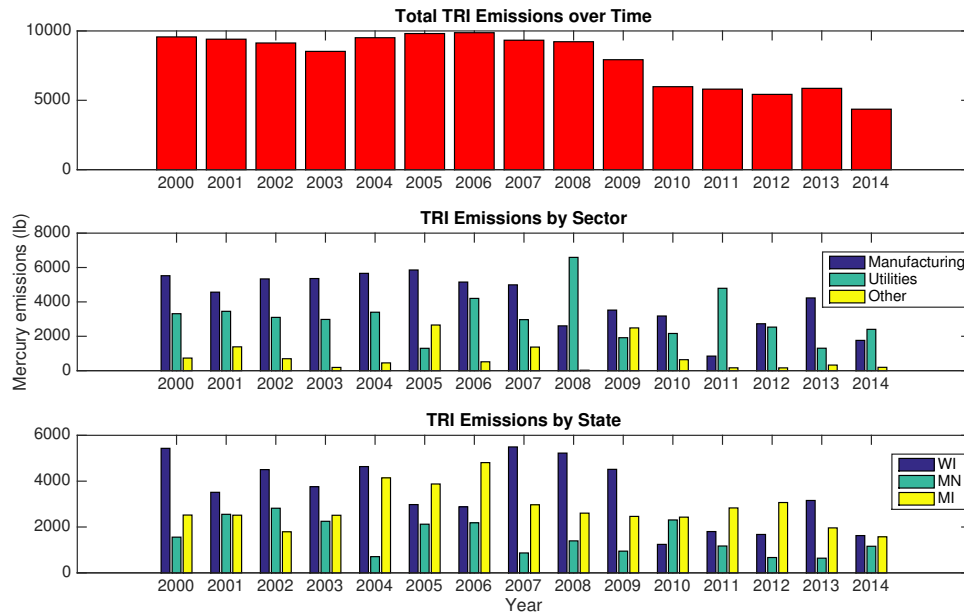
Figure B-13: TRI time series for Lake Superior States, broken down by sector and state.

Table B.2: NEI and TRI totals for fossil fuel emissions by state. 1999-2000 numbers are taken from Murray & Holmes (2004), which used the 1999 NEI and 2000 TRI inventories. The 1999-2000 numbers only represent coal-fired power plants, while the 2008 and 2011 numbers are for all fossil fuel power generation.

	1999-2000	2008	2011
Michigan			
NEI	3094	2134	1936
TRI	3010	2990	2580
% diff	-2.71	40.09	33.25
Minnesota			
NEI	1265	1207	924
TRI	1497	1254	906
% diff	18.34	3.70	-2.01
Wisconsin			
NEI	2263	2313	1134
TRI	2114	1456	1263
% diff	-6.58	-58.85	11.36

Table B.3: Step change in precipitation weighted concentration ($\Delta\%$) between the pre-policy and post-policy period under different simulated scenarios. Significance (p<0.1) and size of the step change are calculated using the Mann-Whitney-Wilcoxon Seasonal Rank Sum Test and the Hodges-Lehmann Estimator of Difference. Values in the table represent the average change across all sites and just those with significant changes. The share of sites with a significant change is shown in brackets (% of all sites).

		Policy Signal	Removal Variability	Speciation Variability	Met. Variability	Energy and Economic Trends	Product Trends
NA	all sites	-1.03%	-0.87%	-0.68%	-1.28%	-0.47%	-3.58%
	sig. sites	-1.03%	-0.87	-0.68%	-9.27%	-1.36%	-3.58%
		(100%)	(100%)	(100%)	(19.5%)	(41.5%)	(100%)
ROW	all sites	-1.74%	-0.36%	-1.66%	-0.95%	-0.23%	-2.98%
	sig. sites	-1.74%	-0.36%	-1.66%	-9.36%	-1.19%	-2.97%
		(100%)	(100%)	(100%)	(17%)	(29.3%)	(100%)

Bibliography

Absolon, K., & Willett, C. (2005). Putting Ourselves Forward: Location in Aboriginal Research. In L. Brown, & S. Strega (Eds.) *Research as Resistance: Critical, Indigenous, and Anti-oppressive Approaches*, (pp. 97–126). Toronto: Canadian Scholar's Press.

Ahmed, S. M., & Palermo, A. G. S. (2010). Community engagement in research: Frameworks for education and peer review. *American Journal of Public Health*, *100*(8), 1380–1387.

Ahn, S.-J., & Graczyk, D. (2012). Understsanding Energy Challenges in India: Policies, Players and Issues. Tech. rep., International Energy Agency, Paris, France.

Alinsky, S. D. (1971). *Rules for Radicals: A Practical Primer for Realistic Radicals*. New York: Random House.

Ambrose, J. L., Gratz, L. E., Jaffe, D. a., Campos, T., Flocke, F. M., Knapp, D. J., Stechman, D. M., Stell, M., Weinheimer, A., Cantrell, C., & Mauldin, R. L. (2015). Mercury Emission Ratios from Coal-Fired Power Plants in the Southeastern U.S. during NOMADSS. *Environmental Science & Technology*, (p. 150710154020002).

Amos, H. M., Jacob, D. J., Holmes, C. D., Fisher, J. A., Wang, Q., Yantosca, R. M., Corbitt, E. S., Galarneau, E., Rutter, A. P., Gustin, M. S., Steffen, A., Schauer, J. J., Graydon, J. A., Louis, V. L. S., Talbot, R. W., Edgerton, E. S., Zhang, Y., & Sunderland, E. M. (2012). Gas-particle partitioning of atmospheric Hg(II) and its effect on global mercury deposition. *Atmospheric Chemistry and Physics*, *12*(1), 591–603.

Amos, H. M., Jacob, D. J., Kocman, D., Horowitz, H. M., Zhang, Y., Dutkiewicz, S., Horvat, M., Corbitt, E. S., Krabbenhoft, D. P., & Sunderland, E. M. (2014). Global biogeochemical implications of mercury discharges from rivers and sediment burial. *Environmental science & technology*, *48*(16), 9514–22.

Amos, H. M., Jacob, D. J., Streets, D. G., & Sunderland, E. M. (2013). Legacy impacts of all-time anthropogenic emissions on the global mercury cycle. *Global Biogeochemical Cycles*, *27*, 1–12.

Anderson, B. (1983). *Imagined communities: Reflections on the origin and spread of nationalism*. New York: Verso Books, 2006 ed.

Arctic Monitoring and Assessment Programme, & United Nations Environment Programme (2013). Technical Background Report to the Global Mercury Assessment 2013. Tech. rep., Arctic Monitoring and Assessment Programme, United Natios Environment Programme.

Ariya, P. a., Amyot, M., Dastoor, A., Deeds, D., Feinberg, A., Kos, G., Poulain, A., Ryjkov, A., Semeniuk, K., Subir, M., & Toyota, K. (2015). Mercury Physicochemical and Biogeochemical Transformation in the Atmosphere and at Atmospheric Interfaces: A Review and Future Directions. *Chemical Reviews*, (p. 150430165241003).

Arnstein, S. R. (1969). A Ladder Of Citizen Participation. *Journal of the American Institute of Planners*, *35*(4), 215–224.

Ashok, A., Lee, I. H., Arunachalam, S., Waitz, I. A., Yim, S. H. L., & Barrett, S. R. H. (2013). Development of a response surface model of aviation's air quality impacts in the United States. *Atmospheric Environment*, *77*, 445–452.

Barnes, E. A., Fiore, A. M., & Horowitz, L. W. (2016). Detection of trends in surface ozone in the presence of climate variability. *Journal of Geophysical Research: Atmospheres*, (2), n/a–n/a.

Baron, S., Sinclair, R., Payne-Sturges, D., Phelps, J., Zenick, H., Collman, G. W., & O'Fallon, L. R. (2009). Partnerships for Environmental and Occupational Justice : Contributions to Research , Capacity and Public Health. *American Journal of Public Health*, *99*(3), S517–S525.

Beck, U. (1992). *Risk Society: Towards a New Modernity*. London: Sage.

Bellanger, M., Pichery, C., Aerts, D., Berglund, M., Castano, A., Cejchanová, M., Crettaz, P., Davidson, F., Esteban, M., Exley, K., Fischer, M. E., Gurzau, A. E., Halzlova, K., Katsonouri, A., Knudsen, L. E., Kolossa-Gehring, M., Koppen, G., Ligocka, D., Miklavcic, A., Reis, M. F., Rudnai, P., Tratnik, J. S., Weihe, P., Budtz-Jørgensen, E., & Grandjean, P. (2013). Economic benefits of methylmercury exposure control in Europe: Monetary value of neurotoxicity prevention. *Environmental Health*, *12*(1), 3.

Bennett, J. (2009). *Vibrant matter: A political ecology of things*. Duke University Press.

Bienkowski, B. (2012). Dredging could unleash PCBs in Indiana Community. *Scientific American*.

Biester, H., Bindler, R., Martinez-Cortizas, A., & Engstrom, D. R. (2007). Critical Review Modeling the Past Atmospheric Deposition of Mercury Using Natural Archives. *Environmental science & technology*, *41*(14), 4851–4860.

Boethling, R., Fenner, K., Howard, P., Klecka, G., Madsen, T., Snape, J. R., & Whelan, M. J. (2009). Environmental Risk Assessment of Human Pharmaceuticals in the European Union: A Caso Study with the B-Blocker Atenolol. *Integrated Environmental Assessment and Management*, *6*(1), 514–523.

Booth, S., & Zeller, D. (2005). Mercury, Food Webs, and Marine Mammals: Implications of Diet and Climate Change for Human Health. *Environmental Health Perspectives*, *113*(5), 521–526.

Bowker, G. C., & Star, S. L. (1999). *Sorting Things Out: Classification and Its Consequences*. November. Cambridge: MIT Press.

Brown, K. (2013). *Plutopia: Nuclear families, atomic cities, and the great soviet and American plutonium disasters*. Oxford University Press, USA.

Brown, P. (1992). Popular epidemiology and toxic waste contamination: lay and professional ways of knowing. *Journal of Health and Social Behavior*, *33*(3), 267–281.

Brown, P., & Mikkelsen, E. J. (1990). *No Safe Place: Toxic Waste, Leukemia, and Community Action*. Berkeley, CA: University of California Press.

Brown, P., Zavestoski, S., Mayer, B., Morello-Frosch, R., & Altman, R. G. (2004). Embodied health movements : new approaches to social movements in health. *Sociology of Health & Illness*, *26*(1), 50–80.

Bullard, R. D. (2000). *Dumping in Dixie: Race, class, and environmental quality*, vol. 3. Westview Press Boulder, CO.

Bullock, D., & Johnson, S. (2011). Electric Generating Utility Mercury Speciation Profiles for the Clean Air Mercury Rule. Tech. rep., US EPA, Research Triangle Park, NC.

Burger Chakraborty, L., Qureshi, A., Vadenbo, C., & Hellweg, S. (2013). Anthropogenic mercury flows in India and impacts of emission controls. *Environmental science & technology*, *47*(15), 8105–13.

Burkey, J. (2006). A non-parametric monotonic trend test computing Mann-Kendall Tau, Tau-b, and Sen's Slope written in Mathworks-MATLAB implemented using matrix rotations. URL `http://www.mathworks.com/matlabcentral/fileexchange/authors/23983{%}0A`

Butler, T. J., Cohen, M. D., Vermeylen, F. M., Likens, G. E., Schmeltz, D., & Artz, R. S. (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*(7), 1582–1592.

Cain, A., Morgan, J. T., & Brooks, N. (2011). Mercury policy in the Great Lakes states: past successes and future opportunities. *Ecotoxicology (London, England)*, *20*(7), 1500–11.

Castro, M. S., & Sherwell, J. (2015). Effectiveness of Emission Controls to Reduce the Atmospheric Concentrations of Mercury. *Environmental Science & Technology*, *49*, 14000–14007.

CDC (2009). Developing Process Evaluation Questions. Tech. Rep. 4, Centers for Disease Control and Prevention.

Charmaz, K. (2006). *Constructing grounded theory: a practical guide through qualitative analysis*. London, UK: Sage Publications.

Chen, C., Driscoll, C. T., Lambert, K. F., Mason, R. P., Rardin, L. R., Schmitt, C. V., Serrell, N. S., & Sunderland, E. M. (2012). *Sources to Seafood: Mercury Pollution in the Marine Environment.*. Hanover, NH: Toxic Metals Superfund Research Program, Dartmouth College.

Chen, L., Wang, H. H., Liu, J. F., Tong, Y. D., Ou, L. B., Zhang, W., Hu, D., Chen, C., & Wang, X. J. (2014). Intercontinental transport and deposition patterns of atmospheric mercury from anthropogenic emissions. *Atmospheric Chemistry and Physics*, *14*(18), 10163–10176.

145

Chess, C., Hance, B. J., & Sandman, P. M. (1988). Improving Dailogue with Communities: A Short Guide for Government Risk Communication. Tech. rep., New Jersey Department of Environmental Protection.

China MEP (2011). Emissions Standard of Air Pollutants for Thermal Power Plats (GB 13223-2011). Tech. rep., Ministry of Environmental Protection.

Clapp, J. T., Roberts, J. A., Dahlberg, B., Berry, L. S., Jacobs, L. M., Emmett, E. A., & Barg, F. K. (2016). Realities of environmental toxicity and their ramifications for community engagement. *Social Science and Medicine*, *170*, 143–151.

Clarke, A. (2005). *Situational analysis: Grounded theory after the postmodern turn*. Sage.

Clarke, A. E., Friese, C., Washburn, R., & Others (2016). *Situational Analysis in Practice: Mapping Research with Grounded Theory*. Routledge.

Cohen, B., & Ottinger, G. (2011). Introduction: Environmental Justice and the Transformation of Science and Engineering. In G. Ottinger, & B. R. Cohen (Eds.) *Technoscience and Environmental Justice: Expert cultures in a grassroots movement*, chap. 1, (pp. 1–14). Cambridge, MA: MIT Press. URL http://mitpress.universitypressscholarship.com/view/10.7551/mitpress/9780262015790.001.0001/upso-9780262015790

Cohen, M., Artz, R., Draxler, R., Miller, P., Poissant, L., Niemi, D., Ratté, D., Deslauriers, M., Duval, R., Laurin, R., Slotnick, J., Nettesheim, T., & McDonald, J. (2016). Modeling the atmospheric transport and deposition of mercury to the Great Lakes. *Elementa: Science of the Anthropocene*, *4*, 000118.

Cole, A. S., Steffen, A., Eckley, C. S., Narayan, J., Pilote, M., Tordon, R., Graydon, J. A., St Louis, V. L., Xu, X., & Branfireun, B. A. (2014). A Survey of Mercury in Air and Precipitation across Canada: Patterns and Trends. *Atmosphere*, *5*, 635–668.

Corbitt, E. S., Jacob, D. J., Holmes, C. D., Streets, D. G., & Sunderland, E. M. (2011). Global source-receptor relationships for mercury deposition under present-day and 2050 emissions scenarios. *Environmental science & technology*, *45*(24), 10477–84.

Corburn, J. (2005). *Street Science: Community Knowledge and Environmental Health Justice*. Cambridge, MA: MIT Press.

Corburn, J. (2009). *Toward the healthy city: people, places, and the politics of urban planning*. Cambridge, MA: MIt Press.

Cordner, A., & Brown, P. (2014). Moments of Uncertainty: Ethical Considerations and Emerging Contaminants. *Sociological Forum*, *28*(3), 1–22.

Cordner, A., Ciplet, D., Brown, P., & Morello-Frosch, R. (2012). Reflexive Research Ethics for Environmental Health and Justice: Academics and Movement-Building. *Soc Mov Stud*, *11*(2), 161–176.

Crawford, C. G., Slack, J. R., Hirsch, R. M., & Peck, D. L. (1983). Nonparametric Tests for Trends in Water-Quality Data Using the Statistical Analysis System. Tech. rep., United States Geological Survey.

Crump, K. S., Kjellström, T., Shipp, a. M., Silvers, a., & Stewart, a. (1998). Influence of prenatal mercury exposure upon scholastic and psychological test performance: benchmark analysis of a New Zealand cohort. *Risk analysis : an official publication of the Society for Risk Analysis*, *18*(6), 701–13.

Cubadda, F., Jackson, B. P., Cottingham, K. L., Van Horne, Y. O., & Kurzius-Spencer, M. (2017). Human exposure to dietary inorganic arsenic and other arsenic species: State of knowledge, gaps and uncertainties. *Science of The Total Environment*, *579*, 1228–1239.

Davidson, P. W., Myers, G. J., Cox, C., Axtell, C., Shamlaye, C., Sloane-reeves, J., Cernichiari, E., Needham, L., Choi, A., Wang, Y., Berlin, M., & Clarkson, T. W. (1998). Effects of Prenatal and Postnatal Methylmercury Exposure From Fish Consumption on Neurodevelopment: Outcomes at 66 Months of Age in the Seychelles Child Development Study. *Journal of the American Medical Association*, *280*(8).

Davis, M. A., Signes-Pastor, A. J., Argos, M., Slaughter, F., Pendergrast, C., Punshon, T., Gossai, A., Ahsan, H., & Karagas, M. R. (2017). Assessment of Human Dietary Exposure to Arsenic through Rice. *Science of The Total Environment*, *586*, 1237–1244.

DeFilippis, J., Fisher, R., & Shragge, E. (2010). *Contesting community: The limits and potential of local organizing*. New Brunswick, NJ: Rutgers University Press.

Dellinger, J., Dellinger, M., & Yauck, J. S. (2012). Mercury Exposure in Vulnerable Populations: Guidelines for Fish Consumption. In M. S. Bank (Ed.) *Mercury in the Environment: Pattern and Process*, chap. 14, (pp. 289–300). Berkeley: University of California Press.

Depew, D. C., Basu, N., Burgess, N. M., Campbell, L. M., Devlin, E. W., Drevnick, P. E., Hammerschmidt, C. R., Murphy, C. a., Sandheinrich, M. B., & Wiener, J. G. (2012). Toxicity of dietary methylmercury to fish: derivation of ecologically meaningful threshold concentrations. *Environmental toxicology and chemistry*, *31*(7), 1536–47.

Dewey, J. (1927). *The Public and Its Problems: An Essay in Political Inquiry*. University Park, PA: Pennsylvania State University Press, 2012 ed.

Diamond, M., & Harrad, S. (2009). The Chemicals that will not Go Away: Implications for Human Exposure to Reservoirs of POPs. *Persistent Organic Pollutants*, (pp. 241–270).

Diamond, M. L., de Wit, C. A., Molander, S., Scheringer, M., Backhaus, T., Lohmann, R., Arvidsson, R., Bergman, k., Hauschild, M., Holoubek, I., Persson, L., Suzuki, N., Vighi, M., & Zetzsch, C. (2015). Exploring the planetary boundary for chemical pollution. *Environment International*, *78*, 8–15.

Dietz, T., & Stern, P. C. (Eds.) (2008). *Public Participation in Environmental Assessment and Decision Making*. National Academies Press.

Douben, P. E. (Ed.) (2003). *PAHs: An Ecotoxicological Perspective*. West Sussex: Wiley. URL `http://dx.doi.org/10.1002/0470867132.ch11`

Driscoll, C. T., Mason, R. P., Chan, H. M., Jacob, D. J., & Pirrone, N. (2013). Mercury as a global pollutant: Sources, pathways, and effects. *Environmental Science and Technology*, *47*(10), 4967–4983.

Edelstein, M. R. (2004). *Contaminated Communities: Coping with residential toxic exposure*. Boulder, CO: Westview Press, 2nd ed.

Edgerton, E. S., Hartsell, B. E., & Jansen, J. J. (2006). Mercury speciation in coal-fired power plant plumes observed at three surface sites in the southeastern U.S. *Environmental science & technology*, *40*(15), 4563–70.

Elliott, J. R., & Frickel, S. (2013). The Historical Nature of Cities: A Study of Urbanization and Hazardous Waste Accumulation. *American Sociological Review*, *78*(4), 521–543.

Epstein, S. (2008). Patient groups and health movements. In E. J. Hackett, O. Amsterdamska, M. Lynch, & J. Wacjman (Eds.) *The Handbook of Science and Technology Studies*, (pp. 499–539). Cambridge, MA: MIT Press, 3rd ed.

Esmail, L., Moore, E., & Rein, A. (2015). Evaluating patient and stakeholder engagement in research: moving from theory to practice. *Journal of Comparative Effectiveness Research*, *4*(2), 133–145.

Evans-Brown, S. (2014). 50,000 N.H. Wells At Risk of High Arsenic, Negative Health Impacts. URL `http://nhpr.org/post/50000-nh-wells-risk-high-arsenic-negative-health-impacts{#}stream/0`

Evers, D. C., Keane, S. E., Basu, N., & Buck, D. (2016). Evaluating the effectiveness of the Minamata Convention on Mercury: Principles and recommendations for next steps. *Science of The Total Environment*.

Evers, D. C., Wiener, J. G., Basu, N., Bodaly, R. a., Morrison, H. a., & Williams, K. a. (2011a). Mercury in the Great Lakes region: bioaccumulation, spatiotemporal patterns, ecological risks, and policy. *Ecotoxicology (London, England)*, *20*(7), 1487–99.

Evers, D. C., Wiener, J. G., Driscott, C. T., Gay, D. A., Basu, N., Monson, B. A., Lambert, K. F., Morrison, H. A., Morgan, J. T., Williams, K. A., & Soehl, A. G. (2011b). Great Lakes Mercury Connections: The Extent and Effects of Mercury Pollution in the Great Lakes Region. Tech. rep., Biodiversity Research Institute - BRI 2011-18, Gorhan, ME.

Farrington, J., & Takada, H. (2014). Persistent Organic Pollutants (POPs), Polycyclic Aromatic Hydrocarbons (PAHs), and Plastics: Examples of the Status, Trend, and Cycling of Organic Chemicals of Environmental Concern in the Ocean. *Oceanography*, *27*(1), 196–213.

Felt, U., & Fochler, M. (2010). Machineries for Making Publics: Inscribing and De-scribing Publics in Public Engagement. *Minerva*, *48*(3), 219–238.

Felt, U., Igelsböck, J., Schikowitz, A., & Völker, T. (2016). Transdisciplinary Sustainability Research in Practice: Between imaginaries of collective experimentation and entrenched academic value orders. *Science, Technology & Human Values*, *41*(4), 732–761.

Finn, S., & Collman, G. (2016). The Pivotal Role of the Social Sciences in Environmental Health Sciences Research. *NEW SOLUTIONS: A Journal of Environmental and Occupational Health Policy*, *26*(3), 389–411.

Flyvberg, B. (2001). *Making Social Science Matter: Why social inquiry fails and how it can succeed again*. Cambridge, UK: Cambridge University Press.

Flyvbjerg, B., Landman, T., & Schram, S. (2012). *Real social science: Applied phronesis*. Cambridge, UK: Cambridge University Press.

Fortun, K., Knowles, S. G., Choi, V., Jobin, P., Matsumoto, M., de la Torre III, P., Liboiron, M., & Murillo, L. F. R. (2017). Researching Disaster from an STS Perspective. In U. Felt, R. Fouche, C. A. Miller, & L. Smith-Doerr (Eds.) *The Handbook of Science and Technology Studies*, chap. 34, (pp. 1003–1028). Cambridge, MA: MIT Press, fourth ed.

Freire, P. (2000). *Pedagogy of the Oppressed*. New York: Bloomsbury, 30th anniv ed.

Frickel, S., Gibbon, S., Howard, J., Kempner, J., Ottinger, G., & Hess, D. J. (2010). Undone Science : Charting Social Movement and Civil Society Challenges to Research Agenda Setting. *Science, Te*, *35*(4), 444–473.

Gagnon, V. S. (2014). A Talking Circles Event: Synthesis and Community Brief. Tech. rep., Michigan Technological University, Keweenaw Bay Ojibwa Community College, Baraga, MI. URL `http://asep.mtu.edu/Workshops/2014TalkingCircle/Synthesis{_}and{_}Community{_}Brief{_}FINAL{_}August{_}2014.pdf`

Gagnon, V. S. (2016). Ojibwe Gichigami (Ojibwa's Great Sea): an intersecting history of treaty rights, tribal fish harvesting, and toxic risk in Keweenaw Bay, United States. *Water History*, *8*(4), 365–384.

Gagnon, V. S., Gorman, H., Morrison, M., Norman, E. S., Perlinger, J., & Xchange, D. I. N. E. (2013). 2013 Community and Partner Workshop Proceedings: Managing impacts of global transport of atmosphere-surface exchangeable pollutants in the context of global change. Tech. rep., Michigan Technological University, Houghton, MI. URL `http://asep.mtu.edu/Workshops/2013-11-15/Proceedings{_}Nov-15-2013{_}ASEP{_}Workshop{_}Final.pdf`

Gagnon, V. S., Gorman, H. S., & Norman, E. S. (2017). Power and politics in research design and practice : Opening up space for social equity in interdisciplinary , multi-jurisdictional, and community-based research. *International Journal of Community Research*.

Garcia-Menendez, F., Monier, E., & Selin, N. E. (2017). The role of natural variability in projections of climate change impacts on U.S. ozone pollution. *Geophysical Research Letters*, (pp. n/a—-n/a).

Garvin, C., & Cox, F. (2001). A History of Community Organizing since the Civil War with Special Reference to Oppressed Communities. In J. Rothman, J. Erlich, & J. Tropman (Eds.) *Strategies of Community Intervention*, (pp. 65–100). Itasca, IL: F.E. Peacock Publishers, 6th ed.

Giang, A., & Selin, N. E. (2016). Benefits of mercury controls for the United States. *Proceedings of the National Academy of Sciences*, *116*(2), 286–291.

Giang, A., Stokes, L. C., Streets, D. G., Corbitt, E. S., & Selin, N. E. (2015). Impacts of the Minamata Convention on mercury emissions and global deposition from coal-fired power generation in Asia. *Environmental Science & Technology*, *49*(9), 5326–5335.

Gibbon, S., & Novas, C. (2008). Introduction: Biosocialities, genetics and the social sciences: Making biologies and identities. In S. Gibbon, & C. Novas (Eds.) *Biosocialities, Genetics and the Social Sciences*, (pp. 1–18). New York: Routledge.

Gilbert, R. O. (1987). *Statistical Methods for Environmental Pollution Monitoring.* New York: Van Norstrand Reinhold Company. URL `http://www.osti.gov/scitech/servlets/purl/7037501`

Gilbert-Diamond, D., Cottingham, K. L., Gruber, J. F., Punshon, T., Sayarath, V., Gandolfi, A. J., Baker, E. R., Jackson, B. P., Folt, C. L., & Karagas, M. R. (2011). Rice consumption contributes to arsenic exposure in US women. *Proceedings of the National Academy of Sciences of the United States of America*, *108*(51), 20656–20660.

Goodsite, M. E., Plane, J. M. C., & Skov, H. (2004). A Theoretical Study of the Oxidation of Hg 0 to HgBr 2 in the Troposphere. *Environmental Science & Technology*, *38*(6), 1772–1776.

Goodsite, M. E., Plane, J. M. C., & Skov, H. (2012). Correction to A Theoretical Study of the Oxidation of Hg0 to HgBr2 in the Troposphere. *Environmental Science and Technology*, *46*(3), 5262.

Grandjean, P., Weihe, P., White, R. F., Debes, F., Araki, S., Yokoyama, K., Murata, K., Sørensen, N., Dahl, R., & Jørgensen, P. J. (1997). Cognitive deficit in 7-year-old children with prenatal exposure to methylmercury. *Neurotoxicology and teratology*, *19*(6), 417–28.

Grant, S. L., Kim, M., Lin, P., Crist, K. C., Ghosh, S., & Kotamarthi, V. R. (2014). A simulation study of atmospheric mercury and its deposition in the Great Lakes. *Atmospheric Environment*, *94*, 164–172.

Gratz, L. E., Keeler, G. J., & Miller, E. K. (2009). Long-term relationships between mercury wet deposition and meteorology. *Atmospheric Environment*, *43*(39), 6218–6229.

Gray-Cosgrove, C., Liboiron, M., & Lepawsky, J. (2015). The Challenges of Temporality to Depollution and Remediation. *Surveys and Perspectives Integrating Environment and Society*, *8*(1), 1–13.

Green, L. W., & Mercer, S. L. (2001). Can Public Health Researchers and Agencies Reconcile the Push From Funding Bodies and the Pull From Communities? *American Journal of Public Health*, *91*(12), 1926–1929.

Greenwood, D. J., & Levin, M. (2006). *Introduction to action research: Social research for social change*. SAGE publications.

Guston, D. H. (2001). Boundary Organizations in Environmental Policy and Science : An Introduction. *Science, Technology, & Human Values*, *26*(4), 399–408.

Hacking, I. (1986). Making Up People. In T. C. Heller, M. Sosna, & D. E. Wellbery (Eds.) *Reconstructing Individualism: Autonomy, Individuality, and the Self in Western Thought*, (pp. 222–236). Stanford: Stanford University Press.

Hacking, I. (1995). The Looping Effects of Human Kinds. In D. Sperber, D. Premack, & A. James Premack (Eds.) *Causal cognition: a multi-disciplinary debate*, chap. 12, (pp. 351–383). New York: Oxford University Press.

Harding, A., Stone, D., Neill, C. O., Berger, P., Harris, S., Harper, B., & Donatuto, J. (2012). Conducting Research with Tribal Communities: Sovereignty, Ethics, and Data-Sharing Issues. *Environmental Health Perspectievs*, *120*, 6–10.

Hawkins, T. R. (1987). A history of progress: NIEHS, the first 20 years (1966 to 1986). *Environmental Health Perspectives*, *75*, 7–10.

Helsel, D., & Hirsch, R. (2002). Statistical methods in water resources. In *Techniques of Water-Resources Investigations of the United States Geological Survey: Book 4, Hydrologic Analysis and Interpretation*, (p. 323). United States Geological Survey. URL `http://water.usgs.gov/pubs/twri/twri4a3/`

Henze, D. K., & Seinfeld, J. H. (2007). Development of the adjoint of GEOS-Chem. *Atmospheric Chemistry and Physics*, *7*, 2413–2433.

Herising, F. (2005). Interrupting Positions: Critical Thresholds and Queer Pro/Positions. In L. Brown, & S. Strega (Eds.) *Research as Resistance: Critical, Indigenous, and Anti-oppressive Approaches*, (pp. 127–152). Toronto: Canadian Scholar's Press.

Hill Collins, P. (1990). *Black Feminist Thought: Knowledge, Consciousness, and the Politics of Empowerment*. New York: Routledge.

Hird, M. J., Lougheed, S., Rowe, R. K., & Kuyvenhoven, C. (2014). Making waste management public (or falling back to sleep). *Social Studies of Science*, *44*(3), 441–465.

Holling, C. (Ed.) (1978). *Adaptive environmental assessment and management*. Wiley-Interscience.

Holmes, C. D., Jacob, D. J., Corbitt, E. S., Mao, J., Yang, X., Talbot, R., & Slemr, F. (2010). Global atmospheric model for mercury including oxidation by bromine atoms. *Atmospheric Chemistry and Physics*, *10*(24), 12037–12057.

Holmes, C. D., Jacob, D. J., & Yang, X. (2006). Global lifetime of elemental mercury against oxidation by atomic bromine in the free troposphere. *Geophysical Research Letters*, *33*(August), 1–5.

Hoover, A. G. (2017). Sensemaking, stakeholder discord, and long-term risk communication at a US Superfund site. *Reviews on environmental health*, *32*(1-2), 165–169.

Hoover, E. (2013). Cultural and health implications of fish advisories in a native American community. *Ecological Processes*, *2*(1), 1–12.

Hoover, E., Renauld, M., Edelstein, M. R., & Brown, P. (2015). Social Science Collaboration with Environmental Health. *Environmental Health Perspectives*, *123*(11), 1100–1106.

Horowitz, H. M., Jacob, D. J., Amos, H. M., Streets, D. G., & Sunderland, E. M. (2014). Historical Mercury Releases from Commercial Products: Global Environmental Implications. *Environmental Science and Technology*, *48*, 10242–10250.

Horowitz, H. M., Jacob, D. J., Zhang, Y., Dibble, T. S., Slemr, F., Amos, H. M., Schmidt, J. A., Corbitt, E. S., Marais, E. A., & Sunderland, E. M. (2017). A new mechanism for atmospheric mercury redox chemistry: Implications for the global mercury budget. *Atmos. Chem. Phys. Discuss.*, *2017*(2), 1–33.

Hunt, G., Stegeman, J., & Robertson, L. (2016). PCBs: exposures, effects, remediation, and regulation with special emphasis on PCBs in schools. *Environmental Science and Pollution Research*, *23*(3), 1971–1974.

Hynes, A. J., Donohoue, D. L., Goodsite, M. E., & Hedgecock, I. M. (2009). Our Current Understanding of Major Chemical and Physical Processes Affecting Mercury Dynamics in the Atmosphere and at the Air-Water/Terrestrial Interfaces. In R. Mason, & N. Pirrone (Eds.) *Mercury Fate and Transport in the Global Atmosphere*, chap. 14, (pp. 427–457). Boston, MA: Springer US. URL `http://www.springerlink.com/index/10.1007/978-0-387-93958-2`

India Central Electricity Authority (2012). National Electricity Plan: Volume 1 - Generation. Tech. Rep. January, Ministry of Power, India, Central Electricity Authority.

India Central Pollution Control Board (2014). Thermal Power Plants Environmental Regulations. URL `http://cpcb.nic.in/upload/Latest/Latest{_}88{_}Notification{_}0422014.pdf`

International Agency for Research on Cancer (2016). Polychlorinated biphenyls and polybrominated biphenyls. Tech. rep., World Health Organization, Lyons, France.

International Joint Commission (2003). Status of Restoration Activities in Great Lakes Areas of Concern : A Special Report. Tech. Rep. April, International Joint Commission.

International Joint Commission (2013). Assessment of Progress Made Towards Restoring and Maintaining Great Lakes Water Quality Since 1987. Tech. rep., International Joint Commission, Ottawa, Canada, Washington D.C, USA.

Israel, B. A., Coombe, C. M., Cheezum, R. R., Schulz, A. J., McGranaghan, R. J., Lichtenstein, R., Reyes, A. G., Clement, J., & Burris, A. (2010). Community-based participatory research: A capacity-building approach for policy advocacy aimed at eliminating health disparities. *American Journal of Public Health*, *100*(11), 2094–2102.

Israel, B. A., Schulz, A. J., Parker, E. A., & Becker, A. B. (1998). Review of community-based research: assessing partnership approaches to improve public health. *Annual review of public health*, *19*, 173–202.

Jackson, B. P., Taylor, V. F., Punshon, T., & Cottingham, K. L. (2012). Arsenic concentration and speciation in infant formulas and first foods. *Pure Appl Chem*, *84*(2), 215–223.

Jacob, D. J., & Winner, D. A. (2009). Effect of climate change on air quality. *Atmospheric Environment*, *43*(1), 51–63.

Jaramillo, P., & Muller, N. Z. (2016). Air pollution emissions and damages from energy production in the U.S.: 2002-2011. *Energy Policy*, *90*(March), 202–211.

Jasanoff, S. (1991). Acceptable Evidence in a Pluralistic Society. In D. G. Mayo, & R. D. Hollander (Eds.) *Acceptable evidence: science and values in risk management*, chap. 2, (pp. 29–47). New York: Oxford University Press.

Jasanoff, S. (2004a). Ordering Knowledge, Ordering Society. In *States of Knowledge: The co-production of science and social order*, chap. 2. Cambridge, MA: Routledge.

Jasanoff, S. (2004b). The idiom of co-production. In S. Jasanoff (Ed.) *States of Knowledge: The co-production of science and social order*, chap. 1. New York: Routledge.

Jasanoff, S., & Kim, S. H. (2009). Containing the atom: Sociotechnical imaginaries and nuclear power in the United States and South Korea. *Minerva*, *47*(2), 119–146.

Jiang, G.-B., & Shi, J.-B. (2006). Mercury Pollution in China: An overview of the past and current sources of the toxic metal. *Environmental Science and Technology*, *40*(12), 3672–3678.

Karagas, M. R., Choi, A. L., Oken, E., Horvat, M., Schoeny, R., & Kamai, E. (2012). Evidence on the Human Health Effects of Low-Level Methylmercury Exposure. *Environmental Health Perspectives*, *120*(6), 799–806.

Karl, H. A., Susskind, L. E., & Wallace, K. H. (2007). A Dialogue, not a Diatribe: Effective Integration of Science and Policy through Joint Fact Finding. *Environment: Science and Policy for Sustainable Development*, *49*(1), 20–34.

Keohane, M. N. O., & Olmstead, S. M. (2016). *Markets and the Environment*. Island Press.

Kim, K.-H., Jahan, S. A., Kabir, E., & Brown, R. J. C. (2013). A review of airborne polycyclic aromatic hydrocarbons (PAHs) and their human health effects. *Environment international*, *60*(October), 71–80.

Korfmacher, K. S., Pettibone, K. G., Gray, K. M., & Newman, O. D. (2016). Collaborating for Systems Change: A Social Science Framework for Academic Roles in Community Partnerships. *NEW SOLUTIONS: A Journal of Environmental and Occupational Health Policy*, *26*(3), 429–457.

Krishnakumar, B., Niksa, S., Sloss, L., Jozewicz, W., & Futsaeter, G. (2012). Process Optimization Guidance (POG and iPOG) for Mercury Emissions Control. *Energy & Fuels*, *26*(8), 4624–4634.

Kumari, R. (2011). Emission Estimate of Passport-Free Heavy Metal Mercury from Indian Thermal Power Plants and Non-Ferrous Smelters. Tech. rep., Toxics Link.

Labonte, R. (1989). Community empowerment: the need for political analysis. *Canadian Journal of Public Health*, *80*(2), 87–88.

Labonte, R. (1993). Community Development and Partnerships. *Canadian Journal of Public Health*, *84*(4), 237–240.

Lake Superior Binational Program (2012). Lake Superior Zero Discharge Demonstration Program and Critical Chemical Reduction Milestones. URL https://www.epa.gov/sites/production/files/2015-11/documents/lake-superior-zero-discharge-demonstration-program-2012-8pp.pdf

Landis, M. S., Ryan, J. V., Ter Schure, A. F. H., & Laudal, D. (2014). The Behavior of Mercury Emissions from a Commercial Coal-Fired Power Plant: The Relationship Between Stack Speciation and Near-Field Plume Measurements. *Environmental science & technology*, *48*(22), 13540–13548.

Landrigan, P. J., Wright, R. O., Cordero, J. F., Eaton, D. L., Goldstein, B. D., Hennig, B., Maier, R. M., Ozonoff, D. M., Smith, M. T., & Tukey, R. H. (2015). The NIEHS Superfund Research Program : 25 Years of Translational Research for Public Health. *Environmental Health Perspectievs*, *123*(10), 909–919.

Latour, B. (2004). Why Has Critique Run out of Steam ? From Matters of Fact to Matters of Concern. *Critical Inquiry*, *30*(Winter), 225–248.

Latour, B. (2007). Turning around Politics: A Note on Gerard de Vries' Paper. *Social Studies of Science*, *37*(5), 811–820.

Layzer, J. A. (2008). *Natural experiments: ecosystem-based management and the environment*. MIT Press.

Lee, K. N. (1999). Appraising Adaptive Management. *Ecology and Scoiety*, *3*(2), 1–16.

Lepak, R. F., Yin, R., Krabbenhoft, D. P., Ogorek, J. M., Dewild, J. F., Holsen, T. M., & Hurley, J. P. (2015). Use of Stable Isotope Signatures to Determine Mercury Sources in the Great Lakes. *Environmental Science and Technology Letters*, *2*(12), 335–341.

Lewin, K. (1948). *Resolving Social Conflicts and Field Theory in Social Science*. Washington DC: American Psychological Association, 1997 ed.

Lichtveld, M., Goldstein, B., Grattan, L., & Mundorf, C. (2016). Then and now: lessons learned from community-academic partnerships in environmental health research. *Environmental Health*, *15*(117), 1–4.

Lindberg, S., Bullock, R., Ebinghaus, R., Engstrom, D., Feng, X., Pirrone, N., Prestbo, E., Seigneur, C., & Fitzgerald, W. (2007). Mercury in Deposition A Synthesis of Progress and Uncertainties in Attributing the Sources of Mercury in Deposition. *AMBIO: A Journal of the Human Environment*, *36*(1), 19–33.

Liu, H., Jacob, D. J., Bey, I., & Yantosca, R. M. (2001). Constraints from 210Pb and 7Be on wet deposition and transport in a global three-dimensional chemical tracer model driven by assimilated meteorological fields. *Journal of Geophysical Research*, *106*(D11), 109–128.

Loh, P. (2016). Collaborations for Environmental Justice : Toward a Transformative Co-Learning Model. *NEW SOLUTIONS: A Journal of Environmental and Occupational Health Policy*, *26*(3), 412–428.

Lohman, K., Seigneur, C., Edgerton, E., & Jansen, J. (2006). Modeling mercury in power plant plumes. *Environmental science & technology*, *40*(12), 3848–54.

Lydersen, K. (2011). A Toxic River Improves, but Still Has Far to Go. *New York Times*.

Lynam, M. M., Dvonch, J. T., Barres, J. A., Landis, M. S., & Kamal, A. S. (2016). Investigating the impact of local urban sources on total atmospheric mercury wet deposition in Cleveland, Ohio, USA. *Atmospheric Environment*, *127*, 262–271.

Macdonald, R. W., Harner, T., & Fyfe, J. (2005). Recent climate change in the Arctic and its impact on contaminant pathways and interpretation of temporal trend data. *The Science of the total environment*, *342*(1-3), 5–86.

MacFarlane, A. (2003). Underlying Yucca Mountain: The Interplay of Geology and Policy in Nuclear Waste Disposal. *Social Studies of Science*, *33*(5), 783–807.

Mao, H., Cheng, I., & Zhang, L. (2016). Current understanding of the driving mechanisms for spatiotemporal variations of atmospheric speciated mercury: A review. *Atmospheric Chemistry and Physics*, *16*(20), 12897–12924.

Marek, R. F., Thorne, P. S., Dewall, J., & Hornbuckle, K. C. (2014). Variability in PCB and OH-PCB Serum Levels in Children and Their Mothers in Urban and Rural U.S. Communities. *Environmental Science & Technology*, *48*, 13459–13467.

Marres, N. (2005). Issues spark a public into being: A key but often forgotten point of the Lippmann-Dewey debate. In B. Latour, & P. Weibel (Eds.) *Making Things Public: Atmospheres of Democracy*, chap. 3, (pp. 208–217). Cambridge, MA: MIT Press. URL `http://s3.amazonaws.com/ academia.edu.documents/6405848/marres{_}lippmann-dewey{_}mtp.pdf? AWSAccessKeyId=AKIAIWOWYYGZ2Y53UL3A{&}Expires=1492203257{&}Signature= 4{%}2B7uqMiLqueTtohthAZOq5sWvUQ{%}3D{&}response-content-disposition= inline{%}3Bfilename{%}3DIssues{_}spark{_}a{_}public{_}`

Marres, N. (2007). The Issues Deserve More Credit: Pragmatist Contributions to the Study of Public Involvement in Controversy. *Social Studies of Science*, *37*(5), 759–780.

Martinez, A., Hadnott, B. N., Awad, A. M., Herkert, N. J., Tomsho, K., Basra, K., Scammell, M. K., Heiger-Bernays, W., & Hornbuckle, K. C. (2017). Release of Airborne Polychlorinated Biphenyls from New Bedford Harbor Results in Elevated Concentrations in the Surrounding Air. *Environmental Science & Technology Letters*, (p. acs.estlett.7b00047).

Martinez, A., & Hornbuckle, K. C. (2011). Record of PCB congeners , sorbents and potential toxicity in core samples in Indiana Harbor and Ship Canal. *Chemosphere*, *85*(3), 542–547.

Martinez, A., Wang, K., & Hornbuckle, K. C. (2010). Fate of PCB congeners in an industrial harbor of lake Michigan. *Environmental Science and Technology*, *44*(8), 2803–2808.

Matz, J., Brown, P., & Brody, J. (2016). Social Science-Environmental Health Collaborations: An Exciting New Direction. *NEW SOLUTIONS: A Journal of Environmental and Occupational Health Policy*, *26*(3), 349–358.

McKelvey, W., & Oken, E. (2012). Mercury and Public Health: An Assessment of Human Exposure. In M. S. Bank (Ed.) *Mercury in the Environment: Pattern and Process*, chap. 13, (pp. 267–288). Berkeley: University of California Press.

Mergler, D., Anderson, H. A., Chan, L. H. M., Mahaffey, K. R., Murray, M., Sakamoto, M., & Stern, A. H. (2007). Methylmercury exposure and health effects in humans: a worldwide concern. *Ambio*, *36*(1), 3–11.

Milford, J. B., & Pienciak, A. (2009). After the clean air mercury rule: prospects for reducing mercury emissions from coal-fired power plants. *Environmental science & technology*, *43*(8), 2669–73.

Ministry of Environmental Protection China (2013). The State Council Issues Action Plan on Prevention and Control of Air Pollution Introducing Ten Measures to Improve Air Quality. URL `http://english.mep.gov.cn/News{_}service/infocus/ 201309/t20130924{_}260707.htm`

Minkler, M. (Ed.) (2012). *Community Organizing and Community Building for Health and Welfare*. New Brunswick, NJ: Rutgers University Press, 3rd ed.

Minkler, M., & Wallerstein, N. (Eds.) (2008). *Community-Based Participatory Research for Health: From Process to Outcomes*. San Francisco: Jossey-Bass, 2nd ed.

Minkler, M., & Wallerstein, N. (2012). Improving Health through Community Organization and Community Building: Perspectives from Health Education and Social Work. In M. Minkler (Ed.) *Community Organizing and Community Building for Health and Welfare*, chap. 3, (pp. 37–58). New Brunswick, NJ: Rutgers University Press, 3rd ed.

Morello-Frosch, R., Varshavsky, J., Liboiron, M., Brown, P., & Brody, J. G. (2015). Communicating results in post-Belmont era biomonitoring studies: Lessons from genetics and neuroimaging research. *Environmental Research*, *136*, 363–372.

Morgan, M. A., & Lifshay, J. (2012). A Ladder of COmmunity Participation in Public Health. In M. Minkler (Ed.) *Community Organizing and Community Building for Health and Welfare*, chap. Appendix 4. New Brunswick, NJ: Rutgers University Press, 3rd ed.

Morris, Z. S., Wooding, S., & Grant, J. (2011). The answer is 17 years, what is the question: understanding time lags in translational research. *Journal of the Royal Society of Medicine*, *104*(12), 510–520.

Moya, J. (2004). Overview of Fish Consumption Rates in the United States. *Human and Ecological Risk Assessment: An International Journal*, *10*(6), 1195–1211.

Mukherjee, A. B., Bhattacharya, P., Sarkar, A., & Zevenhoven, R. (2009). Mercury Emissions from Industrial Sources in India and its Effects in the Environment. In R. Mason, & N. Pirrone (Eds.) *Mercury Fate and Transport in the Global Atmosphere*, chap. 4, (pp. 81–112). New York: Springer US. URL `http://link.springer.com/10.1007/978-0-387-93958-2`

Mukherjee, R. (2016). Toxic Lunch in Bhopal and Chemical Publics. *Science, Technology & Human Values*, *41*(5), 849–875.

Muntean, M., Janssens-Maenhout, G., Son, S., Giang, A., Selin, N. E., Zhong, H., Zhao, Y., Olivier, J. G., Guizzardi, D., & Dentener, F. (2017). Evaluating EDGARv4 mercury emission speciation ex-post scenarios and their impacts on modelled global and regional wet deposition patterns.

Muntean, M., Janssens-Maenhout, G., Song, S., Selin, N. E., Olivier, J. G. J., Guizzardi, D., Maas, R., & Dentener, F. (2014). Trend analysis from 1970 to 2008 and model evaluation of EDGARv4 global gridded anthropogenic mercury emissions. *The Science of the total environment*, *494-495*, 337–50.

Murphy, M. (2006). *Sick building syndrome and the problem of uncertainty: Environmental politics, technoscience, and women workers*. Duke University Press.

Murray, M., & Holmes, S. A. (2004). Assessment of mercury emissions inventories for the Great Lakes states. *Environmental Research*, *95*(3), 282–297.

Nachman, K. E., Ginsberg, G. L., Miller, M. D., Murray, C. J., Nigra, A. E., & Pendergrast, C. B. (2017). Mitigating dietary arsenic exposure: Current status in the United States and recommendations for an improved path forward. *Science of The Total Environment*, *581-582*, 221–236.

NADP (2017). National Atmospheric Deposition Program. URL `http://nadp.sws.uiuc.edu/`

National Research Council (2000). *Long-term institutional management of US Department of Energy legacy waste sites*. National Academies Press.

National Research Council (2007). *Sediment dredging at Superfund megasites: Assessing the effectiveness*. Washington, DC: National Academies Press.

Newig, J., & Fritsch, O. (2009). Environmental governance: participatory, multi-level - and effective? *Environmental Policy and Governance*, *19*(3), 197–214.

Newman, R. S. (2016). *Love Canal: A Toxic History from Colonial Times to the Present*. Oxford University Press.

NIEHS (2012). Celebrating 25 Years of the Superfund Research Program. Tech. rep., National Institute of Environmental Health Sciences.

NIEHS (2015a). Suggested Research and Activities for the Superfund Hazardous Substance Research and Training Program Center Grants (P42). Tech. rep., National Institute for Environmental Health Sciences. URL `https://www.niehs.nih.gov/research/supported/centers/srp/assets/docs/srp{_}funding{_}opps{_}suggested{_}research{_}and{_}activities{_}508.pdf`

NIEHS (2015b). Superfund Hazardous Substance Research and Training Program (P42) RFA-ES-15-019. URL `https://grants.nih.gov/grants/guide/rfa-files/RFA-ES-15-019.html`

NIH, CDC, & ATSDR (2011). *Principles of Community Engagement,*. 2nd ed. URL `http://www.atsdr.cdc.gov/communityengagement/pdf/PCE{_}Report{_}508{_}FINAL.pdf`

Nixon, R. (2011). *Slow Violence and the Environmentalism of the Poor*. Harvard University Press.

NRC (2000). *Toxicological Effects of Methylmercury*. Washington, DC: National Research Counci, National Academy Press.

Nyswander, D. (1956). Education for health: Some principles and their application. *Health Education Monographs*, *14*, 65–70.

Obama, B. (2012). Why Organize? Problems and Promise in the Inner City. In M. Minkler (Ed.) *Community Organizing and Community Building for Health and Welfare*, chap. 2, (pp. 27–31). New Brunswick, NJ: Rutgers University Press, 3rd ed.

O'Fallon, L. R., & Dearry, A. (2002). Community-based participatory research as a tool to advance environmental health sciences. *Environmental Health Perspectives*, *110*, 155–159.

Ohayon, J. L., Cousins, E., Brown, P., Morello-Frosch, R., & Brody, J. G. (2017). Researcher and institutional review board perspectives on the benefits and challenges of reporting back biomonitoring and environmental exposure results. *Environmental Research*, *153*, 140–149.

Oken, E., Choi, A. L., Karagas, M. R., Mariën, K., Rheinberger, C. M., Schoeny, R., Sunderland, E., & Korrick, S. (2012). Which Fish Should I Eat? Perspectives Influencing Fish Consumption Choices. *Environmental Health Perspectives*, *120*(6), 790–798.

O'Rourke, D., & Macey, G. P. (2003). Community Environmental Policing: Assessing New Strategies of Public Participation in Environmental Regulation. *Journal of Policy Analysis and Management*, *22*(3), 383–414.

Osterberg, D., & Scammell, M. K. (2016). PCBs in schools—where communities and science come together. *Environmental Science and Pollution Research*, *23*(3), 1998–2002.

Ottinger, G. (2010). Buckets of Resistance: Standards and the Effectiveness of Citizen Science. *Science, Technology & Human Values*, *35*(2), 244–270.

Ottinger, G., & Cohen, B. R. (2011). *Technoscience and environmental justice: Expert cultures in a grassroots movement*. Cambridge, MA: MIT Press.

Pacyna, E., Pacyna, J., Sundseth, K., Munthe, J., Kindbom, K., Wilson, S., Steenhuisen, F., & Maxson, P. (2010a). Global emission of mercury to the atmosphere from anthropogenic sources in 2005 and projections to 2020. *Atmospheric Environment*, *44*(20), 2487–2499.

Pacyna, J. M., Sundseth, K., Pacyna, E. G., Jozewicz, W., Munthe, J., Belhaj, M., & Aström, S. (2010b). An Assessment of Costs and Benefits Associated with Mercury Emission Reductions from Major Anthropogenic Sources. *Journal of the Air & Waste Management Association*, *60*(3), 302–315.

Pacyna, J. M., Travnikov, O., Simone, F. D., Hedgecock, I. M., Sundseth, K., Pacyna, E. G., Steenhuisen, F., Pirrone, N., Munthe, J., & Kindbom, K. (2016). Current and future levels of mercury atmospheric pollution on a global scale. *Atmospheric Chemistry and Physics*, *16*(19), 12495–12511.

Padilla, I., Irizarry, C., & Steele, K. (2011). Historical contamination of groundwater resources in the north coast karst aquifers of Puerto Rico. *Revista dimension*, *3*, 7–12.

Parrella, J. P., Jacob, D. J., Liang, Q., Zhang, Y., Mickley, L. J., Miller, B., Evans, M. J., Yang, X., Pyle, J. A., Theys, N., & Van Roozendael, M. (2012). Tropospheric bromine chemistry: Implications for present and pre-industrial ozone and mercury. *Atmospheric Chemistry and Physics*, *12*(15), 6723–6740.

Paul, M. P., Rigrod, P., Wingate, S., & Borsuk, M. E. (2015). A Community-Driven Intervention in Tuftonboro, New Hampshire, Succeeds in Altering Water Testing Behavior. *Journal of environmental health*, *78*(5), 30–9.

Pearce, N., Blair, A., Vineis, P., Ahrens, W., Andersen, A., Anto, J. M., Armstrong, B. K., & Baccarelli, A. A. (2015). Commentary IARC Monographs : 40 Years of Evaluating Carcinogenic Hazards to Humans. *Environ Health Perspect*, *123*(6), 507–514.

Perlinger, J. A., Gorman, H. S., Norman, E. S., Obrist, D., Selin, N. E., Urban, N. R., & Wu, S. (2016). Measurement and Modeling of Atmosphere-Surface Exchangeable Pollutants (ASEPs) To Better Understand their Environmental Cycling and Planetary Boundaries. *Environmental Science & Technology*, *50*, 8932–8934.

Perrow, C. (1984). *Normal Accidents: Living with High-Risk Technologies*. Princeton, New Jersey: Princeton University Press, 1999 ed.

Pongprueksa, P., Lin, C. J., Singhasuk, P., Pan, L., Ho, T. C., & Chu, H. W. (2011). Application of CMAQ at a hemispheric scale for atmospheric mercury simulations. *Geoscientific Model Development Discussions*, *4*(3), 1723–1754.

Potts, K., & Brown, L. (2005). Becoming an Anti-oppressive Researcher. In L. Brown, & S. Strega (Eds.) *Research as Resistance: Critical, Indigenous, and Anti-oppressive Approaches*. Toronto: Canadian Scholar's Press.

Prestbo, E. M., & Gay, D. a. (2009). Wet deposition of mercury in the U.S. and Canada, 1996-2005: Results and analysis of the NADP mercury deposition network (MDN). *Atmospheric Environment*, *43*(27), 4223–4233.

Punshon, T., Jackson, B. P., Meharg, A. A., Warczack, T., Scheckel, K., & Guerinot, M. L. (2017). Understanding arsenic dynamics in agronomic systems to predict and prevent uptake by crop plants. *Science of The Total Environment*, *581-582*(December 2016), 209–220.

Rabeharisoa, V., & Callon, M. (2004). Patients and scientists in French muscular dystrophy research. *States of knowledge. The co-production of science and social order*, (pp. 142–160).

Rabinow, P. (1992). Artificiality and Enlightenment: From Sociobiology to Biosociality. In J. Crary, & S. Kwinter (Eds.) *Incorporations*, (pp. 234–252). New York: Zone Books.

Rafaj, P., Bertok, I., Cofala, J., & Schöpp, W. (2013). Scenarios of global mercury emissions from anthropogenic sources. *Atmospheric Environment*, *79*, 472–479.

Ramirez-Andreotta, M. D., Brusseau, M. L., Artiola, J., Maier, R. M., & Gandolfi, a. J. (2015). Building a co-created citizen science program with gardeners neighboring a superfund site: The Gardenroots case study. *International public health journal*, *7*(1), 1–18.

Ranco, D. J., O'Neill, C. a., Donatuto, J., & Harper, B. L. (2011). Environmental Justice, American Indians and the Cultural Dilemma: Developing Environmental Management for Tribal Health and Well-being. *Environmental Justice*, *4*(4), 221–230.

Rayner, S. (2003). Democracy in the age of assessment: reflections on the roles of expertise and democracy in public-sector decision making. *Science and Public Policy*, *30*(3), 163–170.

Remme, U., Trudeau, N., Graczyk, D., & Taylor, P. (2011). *Technology Development Prospects for the Indian Power Sector*. Ph.D. thesis, International Energy Agency, Energy Technology Policy Division.

Rice, G. E., Hammitt, J. K., & Evans, J. S. (2010). A probabilistic characterization of the health benefits of reducing methyl mercury intake in the United States. *Environmental science & technology*, *44*(13), 5216–24.

Risch, M. R., Gay, D. A., Fowler, K. K., Keeler, G. J., Backus, S. M., Blanchard, P., Barres, J. A., & Dvonch, J. T. (2012). Spatial patterns and temporal trends in mercury concentrations, precipitation depths, and mercury wet deposition in the North American Great Lakes region, 2002-2008. *Environmental Pollution*, *161*(October), 261–271.

RIVM (2001). IMAGE 2.2 implementation of the SRES scenarios A comprehensive analysis of emissions, climate change and impacts in the 21st century.

Rockstrom, J., Steffen, W., Noone, K., Persson, A., Chapin, F. S. I., Lambin, E. F., Lenton, T. M., Scheffer, M., Folke, C., Schellnhuber, H. J., Nykvist, B., de Wit, C. A., Hughes, T., van der Leeus, S., Rodhe, H., Sorling, S., Snyder, P. K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R. W., Farby, V. J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., & Foley, J. A. (2009). A safe operating space for humanity. *Nature*, *461*(September), 472–475.

Rodan, B., Pennington, D. W., Eckley, N., & Boethling, R. S. (1999). Screening for Persistent Organic Pollutants : Techniques To Provide a Scientific Basis for POPs Criteria in International Negotiations. *Environmental Science & Technology*, *33*(20), 3482–3488.

Roman, H. a., Walsh, T. L., Coull, B. a., Dewailly, É., Guallar, E., Hattis, D., Mariën, K., Schwartz, J., Stern, A. H., Virtanen, J. K., & Rice, G. (2011). Evaluation of the cardiovascular effects of methylmercury exposures: current evidence supports development of a dose-response function for regulatory benefits analysis. *Environmental health perspectives*, *119*(5), 607–14.

Rothman, J. (2007). Multi Modes of Intervention at the Macro Level. *Journal of Community Practice*, *15*(4), 11–40.

Roy, C. J., & Oberkampf, W. L. (2011). A comprehensive framework for verification, validation, and uncertainty quantification in scientific computing. *Computer Methods in Applied Mechanics and Engineering*, *200*(25-28), 2131–2144.

Rubin, H. J., & Rubin, I. S. (2008). *Community Organizing and Development*. Pearson, 4th ed.

Sandu, A., Daescu, D. N., Carmichael, G. R., & Chai, T. (2005). Adjoint sensitivity analysis of regional air quality models. *Journal of Computational Physics*, *204*(1), 222–252.

Santer, B. D., Mears, C., Doutriaux, C., Caldwell, P., Gleckler, P. J., Wigley, T. M. L., Solomon, S., Gillett, N. P., Ivanova, D., Karl, T. R., Lanzante, J. R., Meehl, G. A., Stott, P. A., Taylor, K. E., Thorne, P. W., Wehner, M. F., & Wentz, F. J. (2011). Separating signal and noise in atmospheric temperature changes: The importance of timescale. *Journal of Geophysical Research Atmospheres*, *116*(22), 1–19.

Scheringer, M., Jones, K. C., Matthies, M., Simonich, S., & van de Meent, D. (2009). Multimedia Partitioning, Overall Persistence, and Long-Range Transport Potential in the Context of POPs and PBT Chemical Assessments. *Integrated environmental assessment and management*, *5*(4), 557–576.

Schreifels, J. J., Fu, Y., & Wilson, E. J. (2012). Sulfur dioxide control in China: policy evolution during the 10th and 11th Five-year Plans and lessons for the future. *Energy Policy*, *48*, 779–789.

Selin, H. (2014a). Global Environmental Law and Treaty-Making on Hazardous Substances: The Minamata Convention and Mercury Abatement. *Global Environmental Politics*, *14*(1), 1–19.

Selin, H., & Eckley, N. (2003). Science, Politics, and Persistent Organic Pollutants: The Role of Scientific Assessments in International Environmental Co-operation. *International Environmental Agreements: Politics, Law and Economics*, *3*, 17–42.

Selin, N. E. (2009). Global Biogeochemical Cycling of Mercury: A Review. *The Annual Review of Environment and Resources*, *34*(1), 43–63.

Selin, N. E. (2011). Science and strategies to reduce mercury risks: a critical review. *Journal of environmental monitoring*, *13*(9), 2389–99.

Selin, N. E. (2014b). Global change and mercury cycling: Challenges for implementing a global mercury treaty. *Environmental toxicology and chemistry*, *33*(6), 1202–1210.

Selin, N. E., & Jacob, D. J. (2008). Seasonal and spatial patterns of mercury wet deposition in the United States: Constraints on the contribution from North American anthropogenic sources. *Atmospheric Environment*, *42*(21), 5193–5204.

Selin, N. E., Jacob, D. J., Yantosca, R. M., Strode, S., Jaeglé, L., & Sunderland, E. M. (2008). Global 3-D land-ocean-atmosphere model for mercury: Present-day versus preindustrial cycles and anthropogenic enrichment factors for deposition. *Global Biogeochemical Cycles*, *22*(GB2011), 1–13.

Selin, N. E., & Selin, H. (2006). Global Politics of Mercury Pollution: The Need for Multi-Scale Governance. *Review of European Community & International Environmental Law*, *15*(3), 258–269.

Senier, L., Hudson, B., Fort, S., Hoover, E., Tillson, R., & Brown, P. (2008). Brown Superfund Basic Research Program: A multistakeholder partnership addresses real-world problems in contaminated communities. *Environmental Science and Technology*, *42*(13), 4655–4662.

Shah, V., & Jaeglé, L. (2017). Surface deposition of oxidized mercury dominated by production in the upper and middle troposphere. *Atmospheric Chemistry and Physics Discussions*, (March), 1–41.

Sloss, L. (2012a). *Legislation, standards and methods for mercury emissions control*. Ph.D. thesis, International Energy Agency Clean Coal Centre.

Sloss, L. (2012b). *Mercury emissions from India and South East Asia*. Ph.D. thesis, International Energy Agency Clean Coal Centre.

Smith, L. T. (2012). *Decolonizing Methodologies: Research and indigenous peoples*. New York: Zed Books, 2nd ed.

161

Soerensen, A. L., Sunderland, E. M., Holmes, C. D., Jacob, D. J., Yantosca, R. M., Skov, H., Christensen, J. H., Strode, S. A., & Mason, R. P. (2010). An improved global model for air-sea exchange of mercury: high concentrations over the North Atlantic. *Environmental science & technology*, *44*(22), 8574–80.

Spears, E. G. (2014). *Baptized in PCBs: race, pollution, and justice in an all-American town*. UNC Press Books.

Srivastava, R. K., Hutson, N., Martin, B., Princiotta, F., & Staudt, J. (2006). Control of mercury emissions from coal-fired electric utility boilers. *Environmental science & technology*, *40*(5), 1385–1393.

Standaert, M. (2014). China Outlines Environmental Action in War' on Air, Water and Soil Pollution. URL `http://www.bna.com/china-outlines-environmental-n17179882762/`

Steffen, A. (2016). Canadian Mercury Science Assessment: Summary of Key Results. Tech. rep., Environment and Climate Change Canada, Toronto. URL `https://www.ec.gc.ca/mercure-mercury/default.asp?lang=En{&}n=32909A5D-1`

Stern, P. C., & Fineberg, H. V. (Eds.) (1996). *Understanding Risk: Informing Decisions in a Democratic Society*. National Academies Press.

Stoecker, R. (1999). Are Academics Irrelevant?: Roles for Scholars in Participatory Research. *American Behavioral Scientist*, *42*(5), 840–854.

Stoecker, R. (2013). *Research methods for community change: A project-based approach*. Thousand Oaks: Sage Publications.

Streets, D., Hao, J., Wu, Y., Jiang, J., Chan, M., Tian, H., & Feng, X. (2005). Anthropogenic mercury emissions in China. *Atmospheric Environment*, *39*(40), 7789–7806.

Streets, D. G., Devane, M. K., Lu, Z., Bond, T. C., Sunderland, E. M., & Jacob, D. J. (2011). All-time releases of mercury to the atmosphere from human activities. *Environmental science & technology*, *45*(24), 10485–91.

Streets, D. G., Zhang, Q., & Wu, Y. (2009). Projections of global mercury emissions in 2050. *Environmental science & technology*, *43*(8), 2983–8.

Stringer, L., Dougill, A., Fraser, E., Hubacek, K., Prell, C., & Reed, M. (2006). Unpacking participation in the adaptive management of socialecological systems: a critical review. *Ecology and Society*, *11*(2), 39.

Suk, W. A. (1995). The NIEHS Superfund Basic Research Program: Overview and Areas of Future Research Directions. *Environmental Health Perspectives*, *103*(1), 3–5.

Sunderland, E. M. (2007). Mercury exposure from domestic and imported estuarine and marine fish in the U.S. seafood market. *Environmental health perspectives*, *115*(2), 235–42.

Sunderland, E. M., Corbitt, E. S., Cossa, D., Evers, D., Friedli, H., Krabbenhoft, D. P., Levin, L., Pirrone, N., & Rice, G. (2010). Impacts of Intercontinental Mercury Transport

on Human & Ecological Health. In N. Pirrone, & T. Keating (Eds.) *Hemispheric Transport of Air Pollution 2010, Part B. Mercury*, chap. 5, (pp. 145–178). Geneva, Switzerland: United Nations Economic Comission for Europe, Task Force on Hemispheric Transport of Air Pollution.

Sunderland, E. M., & Selin, N. E. (2013). Future trends in environmental mercury concentrations: implications for prevention strategies. *Environmental health : a global access science source*, *12*, 2.

Sundseth, K., Pacyna, J., Pacyna, E., Pirrone, N., & Thorne, R. (2017). Global Sources and Pathways of Mercury in the Context of Human Health. *International Journal of Environmental Research and Public Health*, *14*(1), 105.

Susskind, L., Camacho, A. E., & Schenk, T. (2010). Collaborative Planning and Adaptive Management in Glen Canyon: A Cautionary Tale. *Columbia Journal of Environmental Law*, *35*(1), 1–53.

Susskind, L., Camacho, A. E., & Schenk, T. (2012). A critical assessment of collaborative adaptive management in practice. *Journal of Applied Ecology*, *49*(1), 47–51.

Susskind, L., & Elliott, M. (1983). *Paternalism, Conflict, and Coproduction: Learning from Citizen Action and Citizen Participation in Western Europe*. New York: Springer Science+Business Media.

Susskind, L., & Ozawa, C. (1984). Mediated Negotiation in the Public Sector: The Planner as Mediator. *Journal of Planning Education and Research*, *4*(1), 5–15.

Task Force on Hemispheric Transport of Air Pollution (2010). Hemispheric Transport of Air Pollution 2010. Tech. Rep. 10, United Nations Economic Comission for Europe, New York and Geneva.

Taylor, V., Goodale, B., Raab, A., Schwerdtle, T., Reimer, K., Conklin, S., Karagas, M. R., & Francesconi, K. A. (2017). Human exposure to organic arsenic species from seafood. *Science of The Total Environment*, *580*, 266–282.

Thackray, C. P., Friedman, C. L., Zhang, Y., & Selin, N. E. (2015). Quantitative Assessment of Parametric Uncertainty in Northern Hemisphere PAH Concentrations. *Environmental Science & Technology*, (p. 150722110224003).

Tong, Y., Eichhorst, T., Olson, M. R., Rutter, A. P., Shafer, M. M., Wang, X., & Schauer, J. J. (2014). Comparison of heterogeneous photolytic reduction of Hg(II) in the coal fly ashes and synthetic aerosols. *Atmospheric Research*, *138*, 324–329.

Toole-O'Neil, B., Tewalt, S., Finkelman, R., & Akers, D. (1999). Mercury concentration in coalunraveling the puzzle. *Fuel*, *78*(1), 47–54.

Trochim, W., Kane, C., Graham, M. J., & Pincus, H. A. (2011). Evaluating translational research: A process marker model. *Clinical and Translational Science*, *4*(3), 153–162.

UNEP (2002). Global Mercury Assessment 2002. Tech. rep., United Nations Environment Programme Chemicals Branch, Geneva, Switzerland.

UNEP (2008). The Global Atmospheric Mercury Assessment: Sources, Emissions and Transport. Tech. rep., UN Environment Programme Chemicals Branch, Geneva, Switzerland.

UNEP (2010). Process Optimization Guidance for Reducing Mercury Emissions from Coal Combustion in Power Plants. Tech. Rep. November, UN Environment Programme Division of Technology, Industry and Economics, Chemical Branch, Geneva, Switzerland.

UNEP (2011). *Reducing mercury emissions from coal combustion in the energy sector*. Ph.D. thesis, Tsinghua University, Beijing.

UNEP (2013a). Global Mercury Assessment 2013: Sources, Emissions, Releases and Environmental Transport. Tech. rep., UNEP Chemicals Branch, Geneva, Switzerland.

UNEP (2013b). Minamata Convention on Mercury - Text and Annexes. Tech. rep., United Nations Environment Programme, Geneva, Switzerland. URL `http://www.mercuryconvention.org/Portals/11/documents/Booklets/MinamataConventiononMercury{_}booklet{_}English.pdf`

UNEP (2013c). Report of the intergovernmental negotiating committee to prepare a global legally binding instrument on mercury on the work of its fifth session. Tech. rep., United Nations Environmentl Programme, Geneva, Switzerland. URL `http://www.unep.org/hazardoussubstances/Portals/9/Mercury/Documents/INC5/5{_}7{_}REPORT{_}ADVANCE.pdf`

UNEP (2014). *Assessment of the Mercury Content in Coal fed to Power Plants and study of Mercury Emissions from the Sector in India*. Ph.D. thesis, Central Institute of Mining & Fuel Research, Geneva, Switzerland.

UNEP (2015). Guidance on Best Available Techniques and Best Environmental Practices to Control Mercury Emissions from Coal-fired Power Plants and Coal-fired Industrial Boilers. Tech. rep., Expert Group on BAT/BEP. URL `http://www.mercuryconvention.org/Portals/11/documents/BAT-BEPdraftguidance/Coal{_}burning{_}power{_}stations{_}and{_}industrial{_}boilers.pdf`

United Nations Environment Programme (2010). Process Optimization Guidance for Reducing Mercury Emissions from Coal Combustion in Power Plants. Tech. Rep. November, Division of Technology, Industry and Economics, Chemicals Branch, Geneva, Switzerland.

US EPA (1997). Mercury Study Report to Congress Volume II: An Inventory of Anthropogenic Mercury Emissions in the United States. Tech. rep., US EPA Office of Air Quality Planning and Standards and Office of Research and Development.

US EPA (2002). Control of mercury emissions from coal-fired electric utility boilers: interim report including errata dated 3-21-02. Tech. Rep. April, US Environmental Protection Agency, Office of Air Quality Planning and Standards.

US EPA (2010). Information Collection Request for National Emission Standards for Hazardous Air Pollutants (NESHAP) for Coal- and Oil-fired Electric Utility Steam Generating Units. URL `http://www.epa.gov/ttn/atw/utility/utilitypg.html`

US EPA (2011a). 2010 Biennial National Listing of Fish Advisories Fact Sheet. URL `http://water.epa.gov/scitech/swguidance/fishshellfish/fishadvisories/upload/technical{_}factsheet{_}2010.pdf`

US EPA (2011b). Regulatory Impact Analysis for the Final Mercury and Air Toxics Standards. Tech. rep., Environmental Protection Agency, Office of Air Quality and Planning Standards, Research Triangle Park, NC.

US EPA (2016). 2014 National Emissions Inventory , version 1 Technical Support Document. Tech. rep., US Environmental Protection Agency, Office of Air Quality Planning and Standards. URL `http://www.epa.gov/ttn/chief/net/2011nei/2011{_}nei{_}tsdv1{_}draft2{_}june2014.pdf`

US EPA (2017). What EPA is Doing to Reduce Mercury Pollution, and Exposures to Mercury. URL `https://www.epa.gov/mercury/what-epa-doing-reduce-mercury-pollution-and-exposures-mercury`

Valk, A., & Ewald, H. (2013). Bringing a hidden pond to public attention: Increasing impact through digital tools. *Oral History Review*, *40*(1), 8–24.

van Vuuren, D. P., & Carter, T. R. (2013). Climate and socio-economic scenarios for climate change research and assessment: reconciling the new with the old. *Climatic Change*, *122*(3), 415–429.

Vega, C. M. V., Brown, P., Murphy, C., Figueroa, A., Cordero, J., & Alshawabkeh, A. (2016). Community Engagement and Research Translation in Puerto Rico's Northern Karst Region: The PROTECT Superfund Research Program. *NEW SOLUTIONS: A Journal of Environmental and Occupational Health Policy*, *26*(3), 475–495.

Vig, N. J., & Kraft, M. E. (Eds.) (2016). *Environmental Policy: New Directions for the 21st Century*. Thousand Oaks: CQ Press, 9th ed.

Vijayaraghavan, K., Karamchandani, P., Seigneur, C., Balmori, R., & Chen, S.-Y. (2008). Plume-in-grid modeling of atmospheric mercury. *Journal of Geophysical Research*, *113*(D24), D24305.

Walker, W., Harremoes, P., Rotmans, J., van der Sluijs, J., van Asselt, M., Janssen, P., & von Krauss, M. (2003). Defining Uncertainty: A Conceptual Basis for Uncertainty Management in Model-Based Decision Support. *Integrated Assessment*, *4*(1), 5–17.

Wallerstein, N., & Duran, B. (2008). The Theoretical, Historical, and Practice Roots of CBPR. In M. Minkler, & N. Wallerstein (Eds.) *Community-Based Participatory Research for Health: From Process to Outcomes*, chap. 2, (pp. 25–46). San Francisco: Jossey-Bass.

Walter, C. L., & Hyde, C. A. (2012). Community Building Practice: An Expanded Conceptual Framework. In M. Minkler (Ed.) *Community Organizing and Community Building for Health and Welfare*, chap. 5, (pp. 78–90). New Brunswick, NJ: Rutgers University Press, 3rd ed.

Walters, C. J. (1986). *Adaptive management of renewable resources*. Macmillan Publishers Ltd.

Wang, Q., & Shen, W. (2000). Estimation of Mercury Emission from Coal Combustion in China. *Environmental science & technology*, *34*(13), 2711–2713.

Wang, S. X., Zhang, L., Li, G. H., Wu, Y., Hao, J. M., Pirrone, N., Sprovieri, F., & Ancora, M. P. (2010). Mercury emission and speciation of coal-fired power plants in China. *Atmospheric Chemistry and Physics*, *10*(3), 1183–1192.

Wang, Y., Duan, Y., Yang, L., Zhao, C., Shen, X., Zhang, M., Zhuo, Y., & Chen, C. (2009). Experimental study on mercury transformation and removal in coal-fired boiler flue gases. *Fuel Processing Technology*, *90*(5), 643–651.

Wang, Y.-j., Duan, Y.-f., Yang, L.-g., Jiang, Y.-m., Wu, C.-j., Wang, Q., & Yang, X.-h. (2008). Comparison of mercury removal characteristic between fabric filter and electrostatic precipitators of coal-fired power plants. *Journal of Fuel Chemistry and Technology*, *36*(1), 23–29.

Wania, F., & Mackay, D. (1996). Tracking the Distribution of Persistent Organic Pollutants. *Environmental Science & Technology*, *30*(9), 390–396.

Weiss-Penzias, P. S., Gay, D. A., Brigham, M. E., Parsons, M. J., Gustin, M. S., & Schure Amout ter (2016). Trends in mercury wet deposition and mercury air concentrations across the U.S. and Canada. *Science of the Total Environment*, (In press), 1–11.

Weiss-Penzias, P. S., Gustin, M. S., & Lyman, S. N. (2011). Sources of gaseous oxidized mercury and mercury dry deposition at two southeastern U.S. sites. *Atmospheric Environment*, *45*(27), 4569–4579.

Westgate, M. J., Likens, G. E., & Lindenmayer, D. B. (2013). Adaptive management of biological systems: A review. *Biological Conservation*, *158*, 128–139.

Wilcke, W. (2000). Global patterns of polycyclic aromatic hydrocarbons (PAHs) in soil. *Journal of plant nutrition and soil science*, *163*, 157–166.

Wu, Y., Streets, D. G., Wang, S. X., & Hao, J. M. (2010). Uncertainties in estimating mercury emissions from coal-fired power plants in China. *Atmospheric Chemistry and Physics*, *10*, 2937–2947.

Wu, Y., Wang, S., Streets, D. G., Hao, J., Chan, M., & Jiang, J. (2006). Trends in Anthropogenic Mercury Emissions in China from 1995 to 2003. *Environmental Science & Technology*, *40*(17), 5312–5318.

Wynne, B. (1992). Misunderstood misunderstanding : social identities and public uptake of science. *Public Understanding of Science*, *1*, 281–304.

Yang, A., & Cui, Y. (2012). Global Coal Risk Assessment: Data Analysis and Market Research. Tech. Rep. November, World Resources Institute.

Zhang, H., Holmes, C., & Wu, S. (2016a). Impacts of changes in climate, land use and land cover on atmospheric mercury. *Atmospheric Environment*, *141*, 230–244.

Zhang, L., Wang, S., Wang, L., Wu, Y., Duan, L., Wu, Q., Wang, F., Yang, M., Yang, H., Hao, J., & Liu, X. (2015a). Updated Emission Inventories for Speciated Atmospheric Mercury from Anthropogenic Sources in China. *Environmental Science & Technology*, (p. 150216151320004).

Zhang, L., Wang, S. X., Wu, Q. R., Wang, F. Y., Lin, C.-J., Zhang, L. M., Hui, M. L., & Hao, J. M. (2016b). Mercury transformation and speciation in flue gases from anthropogenic emission sources: a critical review. *Atmospheric Chemistry and Physics*, *15*(22), 32889–32929.

Zhang, L., Zhuo, Y., Chen, L., Xu, X., & Chen, C. (2008). Mercury emissions from six coal-fired power plants in China. *Fuel Processing Technology*, *89*(11), 1033–1040.

Zhang, X., Rygwelski, K. R., Rowe, M. D., Rossmann, R., & Kreis, R. G. (2015b). Global and regional contributions to total mercury concentrations in Lake Michigan water. *Journal of Great Lakes Research*.

Zhang, Y., Jacob, D. J., Horowitz, H. M., Chen, L., Amos, H. M., Krabbenhoft, D. P., Slemr, F., Louis, V. S., & Sunderland, E. M. (2016c). Observed decrease in atmospheric mercury explained by global decline in anthropogenic emissions. *Proceedings of the National Academy of Sciences of the United States of America*, *113*(3), 526–531.

Zhang, Y., & Jaegle, L. (2013). Decreases in mercury wet deposition over the united states during 2004-2010: Roles of domestic and global background emission reductions. *Atmosphere*, *4*(2), 113–131.

Zhang, Y., Jaeglé, L., van Donkelaar, A., Martin, R. V., Holmes, C. D., Amos, H. M., Wang, Q., Talbot, R., Artz, R., Brooks, S., Luke, W., Holsen, T. M., Felton, D., Miller, E. K., Perry, K. D., Schmeltz, D., Steffen, A., Tordon, R., Weiss-Penzias, P., & Zsolway, R. (2012). Nested-grid simulation of mercury over North America. *Atmospheric Chemistry and Physics*, *12*(14), 6095–6111.

Zhao, Y., Zhong, H., Zhang, J., & Nielsen, C. P. (2015). Evaluating the effects of China's pollution controls on inter-annual trends and uncertainties of atmospheric mercury emissions. *Atmospheric Chemistry and Physics*, *15*(8), 4317–4337.

Zhou, J., Zhang, L., Luo, Z., Hu, C., He, S., Zheng, J. M., & Cen, K. F. (2008). Study on mercury emission and its control for boiler of 300 MW unit. *Thermal Power Generation*, *37*(4), 22–27.

Zhu, Z., Xu, L., Tan, Y., Zhang, C. L., Li, Y. G., Zhang, D. L., Wang, Q. J., Pan, L. H., & Ke, J. X. (2002). Research on characteristics of mercury distribution in combustion products for a 300MW pulverized coal fired boiler. *Power Engineering*, *22*(1), 1594–1597.

Zhuang, J., Cox, J., Cruz, S., Dearing, J. W., Hamm, J. A., & Upham, B. (2016). Environmental Stigma: Resident Responses to Living in a Contaminated Area. *American Behavioral Scientist*, *60*(11), 1322–1341.