OHIO VALLEY-OZONE ALLEY

SMOG POLLUTION AND POWER PLANTS IN THE OHIO RIVER VALLEY: WHAT CAN BE DONE

The Ohio Environmental Council
Ohio Valley Environmental Coalition
Regional Coalition for Ohio Valley Environmental Restoration

February 2000

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The Ohio Environmental Council (OEC) is a statewide network of Ohio environmental and public health organizations. Founded in 1969, the OEC works to inform, unite, and assist the citizens of Ohio in protecting their health, environment, and natural resources.

Ohio Valley Environmental Coalition (OVEC) is a nonprofit organization that encompasses much of West Virginia and portions of southern Ohio and eastern Kentucky. OVEC's mission is to organize and maintain a diverse grassroots organization dedicated to the improvement and preservation of the environment through research, education, communication, and leadership development.

Regional Coalition for Ohio Valley Environmental Restoration (RECOVER) is a nonprofit organization dedicated to improving the environmental quality of the Mid-Ohio River Valley through awareness, education and cooperative action. RECOVER strives to achieve this while maintaining or improving the economic prosperity of the region, as well as promoting individual and community environmental stewardship.

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FOREWORD

Air pollution does not respect boundaries, not even in areas as large as a state. As individuals, we cannot fence off and enhance "our" air as we do, for example, our back yard gardens. There is no feasible way to have healthful air individually, so we have to work toward cleaner air collectively, making choices as a community. In 1970, national health-based standards for air quality were established, and subsequently ratified in 1977 and 1990. These standards define the maximum pollutant concentrations in outdoor air that are acceptable if the public health, including the health of sensitive groups such as children and people with asthma, is to be protected with an adequate margin of safety.

Throughout Ohio, 1,333,669 individuals, or 12 percent of the state's entire population, suffer from Chronic Lung Disease. In essence, air quality issues in Ohio are about quality of life. Dirty air damages health and results in the loss of life.

FACT: Children are more vulnerable to air pollution than adults because their respiratory defenses are not fully formed. Their airways are smaller and more likely to become blocked when irritated. They spend a lot of time outdoors, playing hard and breathing hard, and may not be aware of the effects of ozone exposure, even when their breathing is significantly reduced.

FACT: People with asthma, chronic bronchitis, emphysema, and other lung diseases already have breathing problems because of their illness, but when ozone exposure is present, they can end up needing immediate medical attention.

FACT: Many people, such as construction workers, police officers, farmers, and mail carriers, are exposed to ozone all day long. Many of these workers are unaware that they are part of a high-risk, sensitive group.

FACT: Cyclists, joggers, and others who engage in intensive or regular outdoor exercise around polluted urban areas also make up a large part of the most sensitive group.

The American Lung Association® of Ohio is dedicated to the prevention of lung disease and the promotion of health. We focus on multiple issues that impact lung disease, including air quality. We must never lose sight of the human side of what it means to live with lung disease. We must take every step possible to protect those who are at risk of the health consequences of dirty air, for when you can't breathe, nothing else matters!

Larry McAllister
President/CEO
American Lung Association of Ohio

Executive Summary

The Ohio River Valley is by popular impression a relatively clean place – substantially rural, with low congestion, and dotted with only a few mid-sized cities. Air pollution problems such as ozone smog (which causes lung damage, asthma attacks, and shortness of breath) are thought to be mainly an issue for East Coast cities such as New York and Boston.

A year-long study, coordinated by the Ohio Environmental Council in cooperation with the Ohio Valley Environmental Coalition and the Regional Coalition for Ohio Valley Environmental Restoration, has examined the validity of this impression using both new and recently available research. Work from researchers at the University of Michigan's Department of Atmospheric, Oceanic, and Space Sciences, the Harvard School of Planetary Studies, as well as projects of the consulting firms Earth Tech and Abt Associates, have yielded the results described in the following pages and summarized below.

Health impacts:

- ✓ Residents of the Ohio River Valley region suffered more than ***83,000 asthma attacks*** in the summer of 1997 because of smog pollution, as well as nearly two million cases of more minor respiratory problems. In addition, ***1,909 residents had to visit the emergency room, and 636 were hospitalized*** because of unhealthy air levels that year.

How does the air compare? Midwest versus Northeast:

- ✓ In 1998, people in Ohio River Valley communities such as Cincinnati and Marietta, Ohio, and Huntington, West Virginia, were exposed to dangerous levels of ozone smog ***more often than residents in East Coast cities – in some cases 75 percent more often than in Boston and New York.***

- ✓ 1997 data, the latest available, show that ***Ohio River Valley hospital admission rates for smog-related respiratory problems such as asthma attacks and emphysema were higher than East Coast admission rates.***

In short, the Ohio River Valley is blanketed with high smog levels for much of the year, making it a virtual "ozone alley."

Ohio Valley power plants and ozone smog:

- ✓ Emissions from older coal and oil-fired power plants contribute ***nearly 50 percent of smog pollution levels*** for areas such as Huntington, West Virginia, and Marietta, Ohio, in the River Valley.

- ✓ When added to naturally occurring background levels of ozone, ***Midwestern power plant pollution is enough to cause an exceedence of the air quality standard*** in the Ohio River Valley.

Solutions to the Ohio Valley's pollution:

This analysis of power plant emissions shows that persistent, high levels of ozone smog do not have to continue.

- ✓ ***Adding off-the-shelf, cost-effective emission controls to plants could substantially reduce health problems*** and suffering from Ohio Valley smog.

- ✓ ***Cleaning up or converting older plants to cleaner fuels would also reduce premature death*** in the region from fine soot, ***improve agricultural productivity, reduce toxic air emissions***, and help ***slow global warming***.

- ✓ Ironically, while much of the current policy debate about air pollution from power plants has focused on the impact of Midwestern plants on Northeast cities, this analysis demonstrates that the ***greatest harm to public health from Ohio Valley plants occurs in the Ohio Valley itself.***

In sum, air pollution in the Ohio River Valley compromises our health, our economy, and our quality of life. The technology exists to reclaim healthy air for the Ohio Valley. Therefore, we can and must take action now.

Methodology

This study is based on:

1. A recent study by Abt Associates (1999) that estimated the health impacts of ozone for several areas in the Eastern United States (see Appendix I).

2. A new analysis by researchers from the Harvard School of Planetary Studies (1999), conducted for this study, that identified cumulative hourly exposure to various levels of ozone in the Ohio River Valley and the Northeast (see Appendix II).

3 A new air pollution modeling analysis performed by Earth Tech under the supervision of researchers from the University of Michigan's Department of Atmospheric, Oceanic, and Space Sciences (see Appendix III).

Health Impacts of Ozone in the Ohio River Valley

Health Standards for Ozone

Since the late 1980s, more than 3,000 new studies have been published on the health and ecological effects of ozone. These studies, including those used in this analysis, helped form the basis of the decision by the United States Environmental Protection Agency (USEPA) in 1997 to revise the health standard for ozone.

Plateaus instead of peaks

The ozone standard, when it was last revised in 1979, was set at 120 parts per billion (ppb) for one hour of exposure. Many new health studies show that health effects occur at ozone levels lower than the previous standard and that the primary health threat comes from exposure to ozone over several hours rather than for short periods of time. Therefore, the agency moved from a standard focusing on ozone *peak*s, 120 ppb for one hour, to a standard focusing on *plateau*s, 80 ppb for eight hours.

Although the new standard has been remanded, or put on hold, by a federal appeals court, the court specifically agreed with scientific studies backing the need for a more protective standard. The USEPA's new standard was remanded on the judicial philosophy of "non-delegation," meaning the USEPA interpreted its mandate as an executive agency too broadly.

Ozone smog in our air can result in shortness of breath, trigger asthma attacks, and (for those suffering from chronic lung disorders) lead to trips to the emergency room and admissions to the hospital. A recent analysis by Abt Associates, a consulting firm for national and international health and environmental agencies, estimated the health impacts in 1997 due to ozone smog in the Eastern United States, based on numerous recent epidemiological studies.

According to the analysis, during the 1997 smog season (April-October), the residents of 32 counties in Ohio, West Virginia, and Kentucky bordering the upper Ohio River Valley experienced the following health impacts due to ozone smog levels (see Table I):

- ***636 were hospitalized***
- ***1,909 had to visit the emergency room***
- ***83,035 suffered asthma attacks***
- ***1,097,483 experienced minor symptoms***

"During the summer months, I see kids who can't play outside because the air pollution levels put their ability to breathe at risk."

Dr. William Hardie, M.D., Pulmonologist
Children's Hospital, Cincinnati, Ohio

Table I

Breakdown of ozone smog-related health problems in the Ohio River Valley, by county, 1997

Source: Abt Associates

Kentucky-River Valley Counties				
County	**Respiratory Admissions**	**Total ER Visits**	**Minor symptoms**	**Asthma attacks**
Boone	7	22	17,748	1,314
Boyd	16	47	26,052	1,927
Bracken	3	8	4,004	313
Campbell	5	15	9,091	690
Greenup	8	25	16,170	1,193
Kenton	41	122	78,898	5,943
Lewis	4	11	5,831	456
Mason	5	16	7,716	605
Pendleton	3	10	5,852	457
Subtotal	**92**	**276**	**171,362**	**12,898**
Ohio- River Valley Counties				
County	**Respiratory Admissions**	**Total ER Visits**	**Minor symptoms**	**Asthma attacks**
Adams	7	21	10,607	846
Athens	12	37	28,772	1,897
Belmont	19	58	24,706	1,965
Brown	9	26	14,486	1,131
Clermont	29	87	65,647	4,850
Columbiana	33	98	46,702	3,662
Gallia	8	24	14,989	1,138
Hamilton	223	668	401,188	30,356
Jefferson	29	87	38,594	2,978
Lawrence	19	56	32,819	2,516
Meigs	7	21	10,897	858
Monroe	4	13	6,025	471
Scioto	21	63	32,657	2,565
Washington	16	47	25,491	1,918
Subtotal	**436**	**1306**	**753,580**	**57151**
West Virginia-River Valley Counties				
County	**Respiratory Admissions**	**Total ER Visits**	**Minor symptoms**	**Asthma attacks**
Brooke	5	15	7,316	549
Cabell	29	86	46,009	3,398
Hancock	9	26	12,062	905
Jackson	6	18	10,548	800
Marshall	12	35	17,994	1,371
Mason	5	15	8,478	647
Pleasants	2	7	3,555	272
Tyler	3	9	4,444	344
Wayne	6	19	11,294	854
Wetzel	6	18	8,981	694
Wood	27	80	41,863	3,153
Subtotal	**110**	**328**	**172,544**	**12987**
ORV Total	**636**	**1,909**	**1,097,483**	**83,035**

These ozone-related health impacts are defined as follows:

Respiratory Hospital Admissions – Hospital admissions for pneumonia, chronic obstructive pulmonary disease (COPD), asthma, and other ailments aggravated by ozone occur because a reaction to ozone is so intense that a high level of medical care is required for one or several days or, in some cases, much longer. The average hospital stay for these illnesses is 5.8 days.[1]

Total Emergency Room (ER) Visits – ER visits for pneumonia, COPD, asthma, and other respiratory ailments aggravated by ozone. ER visits focus on ensuring that victims are able to regain control of their breathing, although the most severe cases are often admitted to the hospital.

Asthma Attacks – The number of days that juvenile and adult asthmatics suffer one or more asthma attacks. Ozone is a known asthma trigger. Asthma attacks can be seriously debilitating and can lead to other problems, such as the inability to work or attend school.

Minor Symptoms – Symptoms in adults include cough, sore throat, and head cold. Ozone irritates the throat and lung tissues and makes even the healthiest individuals more at risk for such problems as coughs and colds.

Methodology of Health Estimates

The following chart illustrates the methodology used for this analysis:

National Respiratory Statistics → Apportion to state and local areas → Apply 1997 ozone season data → Apply concentration/ response factors based on studies → Health endpoint estimate

National health statistics were apportioned by state and metropolitan area. The 1997 ozone season (April-October) data from the United States Environmental Protection Agency's (USEPA) Aerometric Information Retrieval System (AIRS) were applied by examining levels of ozone pollution and the number of high ozone days. Finally, the percentage of health problems attributable to ozone was established through a synthesis of numerous epidemiological studies. The result is an estimate of ozone-related health effects by state and metro area. While these numbers are estimates, they establish ozone pollution as a pervasive health problem and hint at the magnitude of ozone-related health risks.

Abt Associates, the consulting firm under contract with the USEPA to analyze air pollution damages, authored the report. Abt Associates' Environmental Research Area conducted extensive health analysis for the USEPA in support of the 1997 revisions to both the ozone and the particulate matter National Ambient Air Quality Standards. Abt Associates also prepared the health and economic analyses for EPA's 1997 Report-to-Congress, *The Benefits and Costs of the Clean Air Act: 1970 to 1990.*

[1] National Center for Health Statistics, Annual Admissions Inventory, 1994.

How Does the Air Compare?
Midwest vs. Northeast

Ozone Smog Levels in the Midwest Can Pose Greater Health Risks Than Those in the Northeast

In the past, emphasizing the maximum peak ozone level at a given hour led to a focus on cities such as New York City and Boston as the prime victims of ozone smog in the Eastern United States. These areas tend to experience the highest peak levels in the Eastern half of the country, such as a 139 parts per billion (ppb) reading in Portland, ME in the summer of 1998. Peak levels of this magnitude are not normally reached in the Midwest.

However, areas of the Midwest, including the Ohio River Valley, can experience higher plateaus of ozone for periods of longer duration than the Northeast. In short, levels of unhealthy air in the Ohio River Valley can be sustained for longer periods of time than in the Northeast.

As the rationale for moving to an eight-hour ozone standard suggests (see box on p. 5), longer exposures pose a more potent health threat. Residents of the Ohio River Valley can suffer greater threats to their health from levels of smog in the air than residents of the Northeast.

Northeast Peaks are Higher

Examinations of ozone levels in 1998 indicate that Northeastern cities, such as Portland, ME, experience higher peak levels of air pollution than cities in the Ohio River Valley (Figure I). In this case, Portland had the highest, 139 ppb, versus the low Midwest peak of 125 ppb of Marietta, OH.

(**Note:** The three Ohio River Valley cities have higher peaks than New York and Boston.)

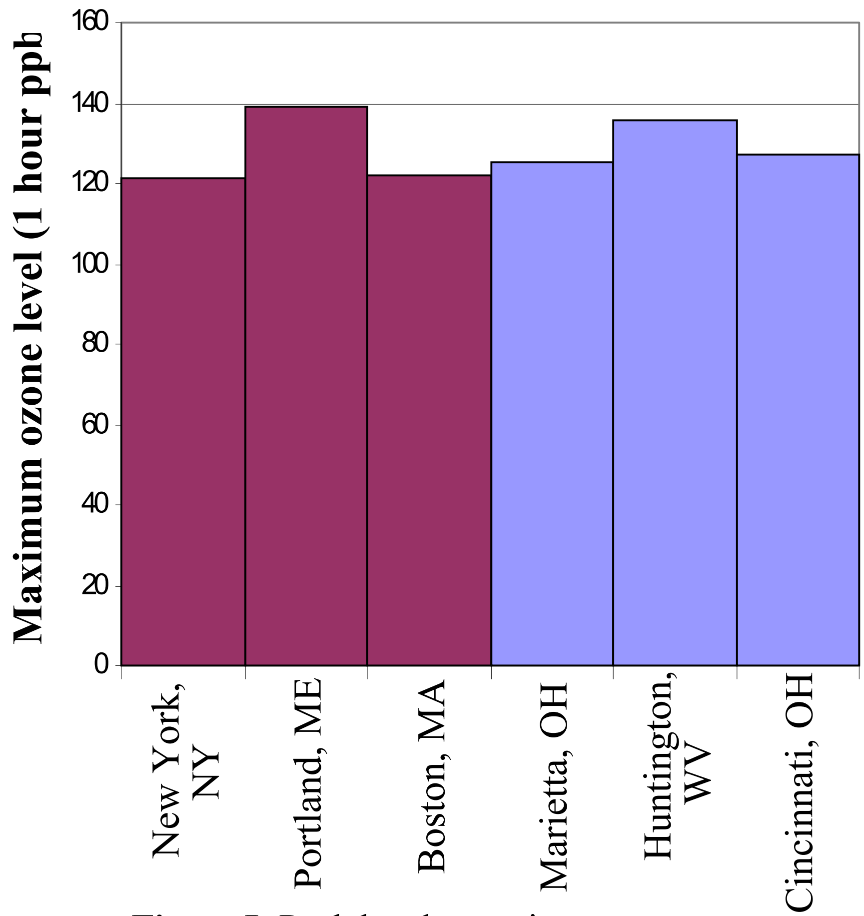


Figure I: Peak levels, maximum ozone value, 1998 smog season

Source: USEPA AIRS database

Ohio Valley Plateaus are Higher for a Longer Duration

A different picture emerges from the examination of the total hours of unhealthy air experienced in the Northeast and the Ohio River Valley (Figure II). In this case, Huntington, WV experienced the highest number of hours, 191, and Portland, ME, the lowest with 41 hours. The Ohio River Valley can experience bad air quality for longer periods of time than the Northeast.

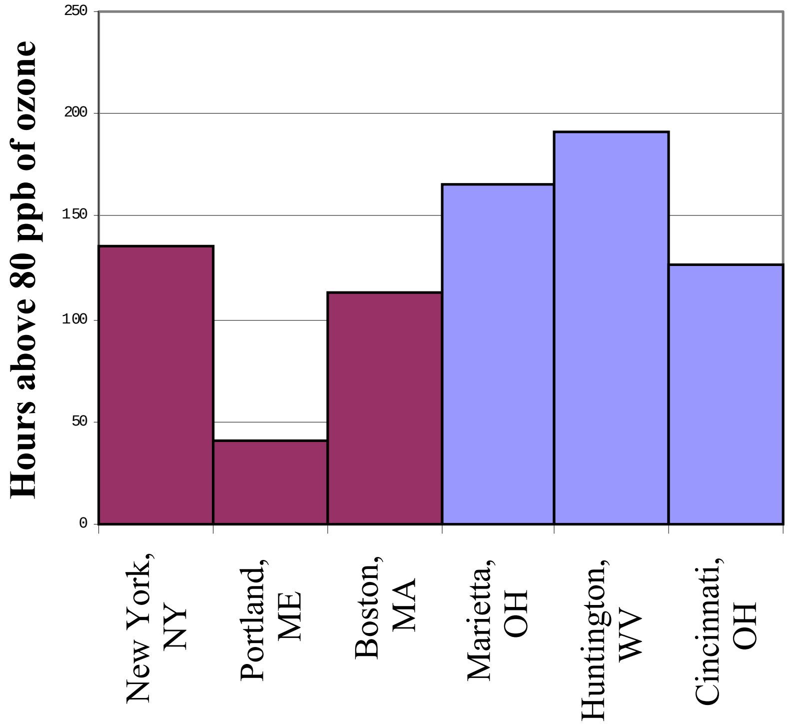


Figure II: Total hours of unhealthy air, April-October 1998

Source: Arlene Fiore, Harvard School of Planetary Science, 1999

Midwest Ozone Levels Can Pose Greater Per Capita Health Risks

Consistent with the most recent health research, which recognizes that longer-term exposures to smog can pose a more significant health risk than high-peak exposures, the Abt work shows that Midwesterners face a greater health threat than their Northeastern counterparts. When the relative rates of three distinct ozone-induced health effects (respiratory hospital admissions, emergency room visits, and asthma attacks) are compared, Midwesterners experience a greater rate of ozone-related health effects than residents of the Northeast. For example, the communities of the Ohio River Valley have a greater per capita risk of hospitalization from ozone than do residents of the large Northeast cities that are typically viewed as the smog capitals of the Eastern United States (see Figure III). Similarly, per capita emergency room visits by residents of several Midwestern states exceed those found in several Northeastern states (see Figure IV). Likewise, the Midwest as a region (OH, MI, IN, IL, KY, WV) suffers greater per capita risk from ozone-induced asthma attacks than the Northeast (NY and New England) (see Figure V).

While these conclusions in no way minimize the problem of air pollution in the Northeast, they do put into perspective the level of harm experienced by residents of the Ohio River Valley.

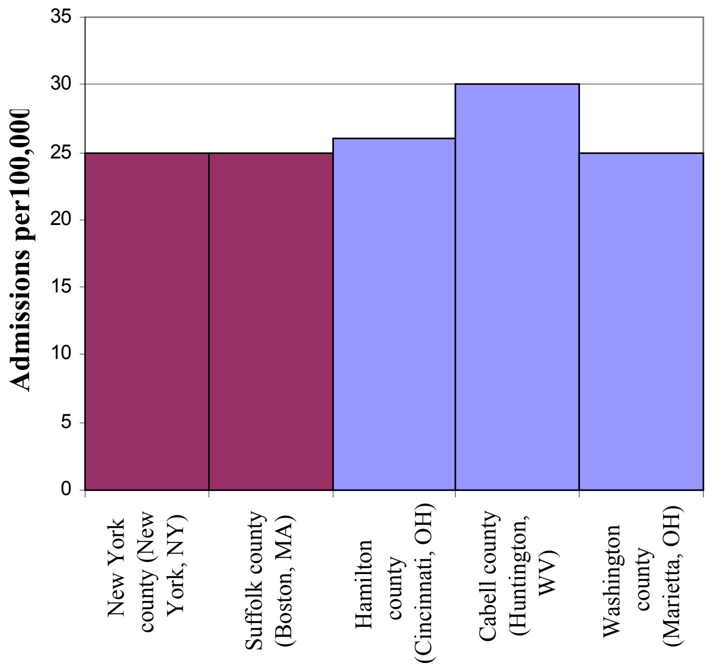
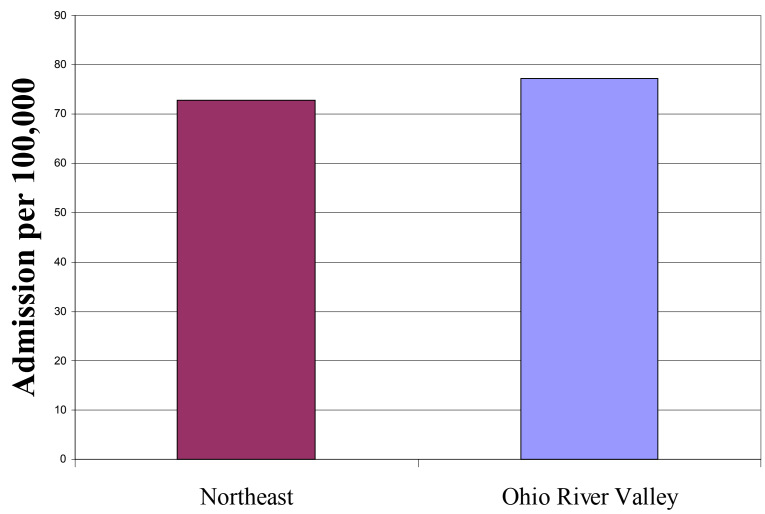


Figure III: Per capita respiratory hospital admissions due to ozone are higher for cities in the Ohio River Valley than for those in the Northeast

Source: Abt Associates and MSB Energy Associates

Figure IV: Per capita respiratory hospital admissions due to ozone smog in the Ohio River Valley are as high or higher than Northeastern states

Source: Abt Associates and MSB EnergyAssociates
Note: Northeast includes Maine, Massachusetts, Connecticut, New Hampshire, New York, Vermont, and Rhode Island. Ohio River Valley includes Ohio, Kentucky, and West Virginia



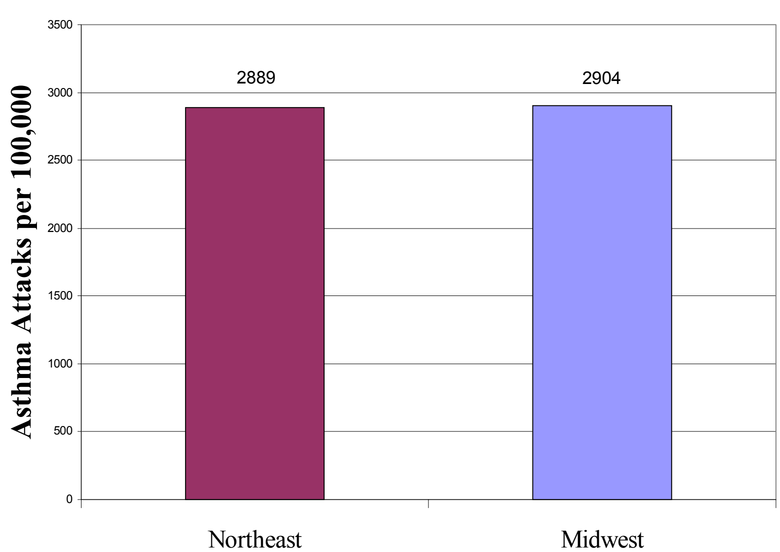


Figure V: Midwestern states suffer ozone-related asthma comparable to those in Northeastern states

Source: Abt Associates and MSB Energy Associates
Note: Northeast includes Maine, Massachusetts, Connecticut, New Hampshire, New York, Vermont, and Rhode Island. Midwest includes Michigan, Ohio, Illinois, Indiana, Kentucky, and West Virginia.

Power Plants and the Ohio Valley's Smog Problem

Ozone levels in the Ohio River Valley are impacted by power plants within the region, as well as by upwind power plants. Power plants in the Valley emit a total of 1,042,805 tons of nitrogen oxide, equivalent to nitrogen oxide emissions from 53 million cars. Adding the emissions from upwind power plants results in 2,849,190 million tons of this pollution, or the equivalent of emissions from 146 million cars.

This combination of emissions has negative ramifications on air quality in the region. Air pollution modeling of the region indicates that power plants contribute as much as 50 percent of the ozone pollution in the region that is above normal background levels (Figure VI). In areas such as Marietta, OH, Huntington, WV, and Pittsburgh, PA, the impact ranges from 40-50 percent.

How Ozone Smog Is Formed

Ozone or smog is formed when nitrogen oxide (NOx) combines with volatile organic chemicals (VOCs) in the presence of sunlight. Sources of nitrogen oxide include fossil fuel-burning power plants, motor vehicles, and diesel engines. VOCs are emitted by a variety of sources, including motor vehicles, dry cleaners, printers, industrial processes, and natural sources such as pine trees.

Background ozone – "Global" background ozone is the concentration that would be present throughout the northern hemisphere, even in remote locations (such as the mid-Pacific Ocean) due to the cycle of the Earth's climate. In the Northern Hemisphere, this global background is 40 ppb.

Regional ozone –"Regional" background ozone refers to ozone that has been chemically produced from upwind pollution sources, and then has traveled for one day or more to the region of interest. Regional ozone events typically produce a region of near-uniform high ozone extending over 500 miles or more. Urban emissions, power plants, and widely distributed emissions in rural areas all contribute to regional ozone. This often represents the major impact of power plants on ozone air quality.

Local ozone – "Local" input from nearby urban areas or large power plants represents ozone that has been produced during a specific day, typically leading to maximum ozone concentrations in the afternoon. In a rural location removed from direct input from urban centers or power plants, the local input (today's ozone) is indistinguishable from the regional background (yesterday's ozone). Urban areas and power plants both generate "plumes" that typically have higher ozone than the regional background.

Air Pollution Modeling

Air pollution modeling for ozone is the process of using a computer model to analyze how temperature, humidity, and air pollution combine to produce smog. The Ozone Transport Assessment Group, through a study on the transport of power plant emissions sponsored by 37 states, developed sophisticated databases on several smog events for the Eastern United States. For the analysis in this report, we have used data for the July 20th, 1991 event because it was representative of a typical unhealthy "ozone smog episode" for the Ohio River Valley region. The consulting firm Earth Tech conducted the modeling analysis, under the supervision of the University of Michigan's Department of Atmospheric, Oceanic, and Space Sciences.

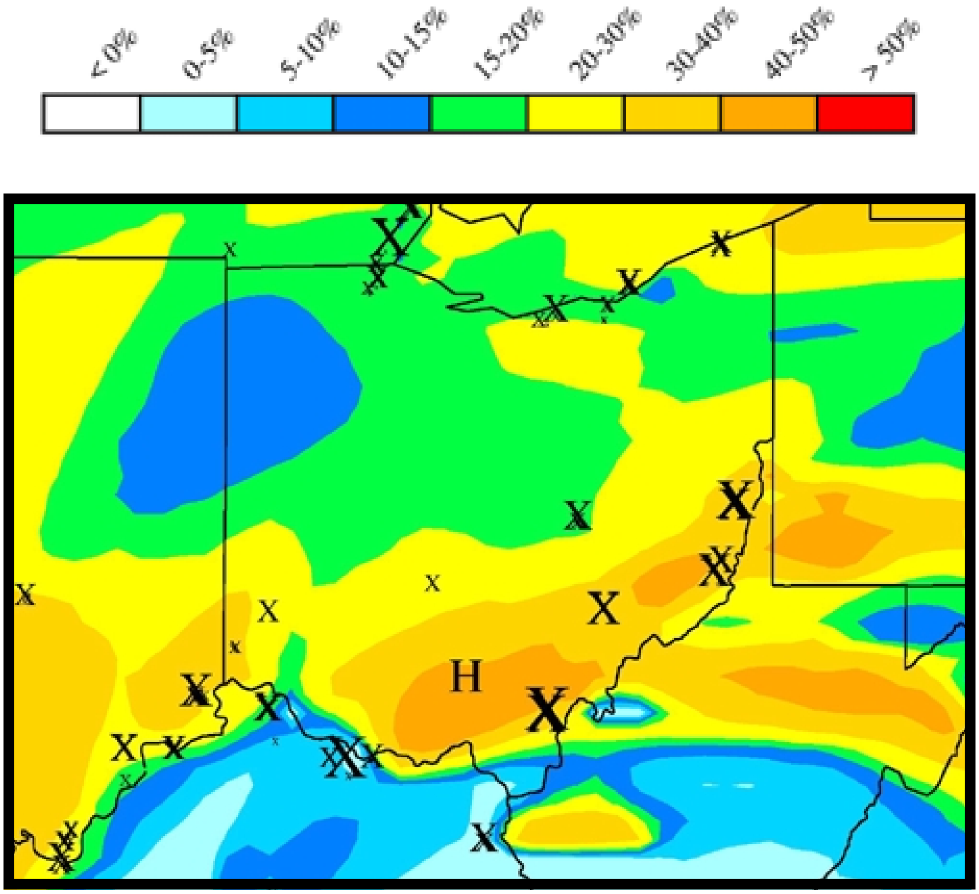


Figure VI: Contribution of power plants to ozone smog above background level (in percent)
Source: Earth Tech.

Power plants can contribute up to 50 percent of the unhealthy air on a "Bad Air" day in the Ohio River Valley.

Figure VI above, a plot of the model's output, describes the percentage of ozone levels that power plants are responsible for creating above the 40 ppb backgrond. The x's in the graph indicate the location and relative size of area power plants. In the majority of the area of the Ohio River Valley, the power plant contribution ranges from 30-50 percent. The power plants affecting the area are located in Ohio, West Virginia, Kentucky, Indiana, Illinois, and Missouri.

The Legacy of Air Pollution: the History of Power Plant Emissions

Many of the power plants in the Ohio River Valley, as well as upwind areas, were built in the 1940s-1960s. A provision of the Clean Air Act exempts these older plants from the most up-to-date pollution control requirements. Comparatively, a "grandfathered" power plant is allowed to emit up to ten times the level of pollution as a modern power plant.

This exemption, created by Congress in 1977, was justified under the assumption that many of the plants were ending their natural life span and would soon be replaced with cleaner power generators. However, most of these old plants have outlived their expected retirement dates by decades and continue to emit pollution at levels equal to or greater than when they were built.

The Impact of Power Plants on the Ohio River Valley and the Midwest

While Ohio Valley power plants degrade local air quality, the combined emissions of Midwestern power plants can have a devastating impact. The difference can be determined by hypothetically eliminating the emissions of various power plants and observing the estimated air quality improvement in a given area. The four graphs below show how different sets of power plants impact air quality. The impact is seen by the drop in ozone (averaged for an 8-hour period) that occurs when plant emissions are hypothetically eliminated.

Scenario 1: Pollution from Eight Plants in the Valley Despoils the Immediate Area

Eight power plants found in the Ohio River Valley emit more than 309,493 tons of ozone-forming nitrogen oxide annually. The pollution from these eight plants alone can create 25-50 percent of the ozone levels above background needed to trigger an eight-hour standard exceedence (80 ppb for an 8 hour average) violation in the Valley, e.g., levels in Cincinnati, OH or Covington, KY of 10-20 ppb.

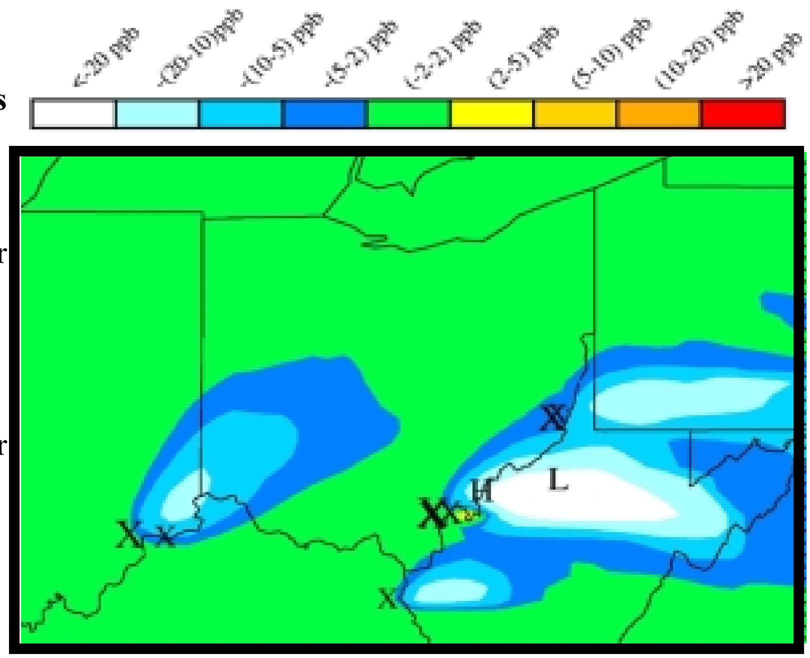


Figure VII.

Source: Earth Tech

Scenario 2: Add Six More Plants and the Impacts Extend to Bordering States

Fourteen plants emit 679,995 tons of nitrogen oxide each year. The combined impact of these plants reaches well beyond local communities, including roughly two-thirds of Ohio and West Virginia, the area of Kentucky bordering Ohio, throughout Pennsylvania, and beyond.

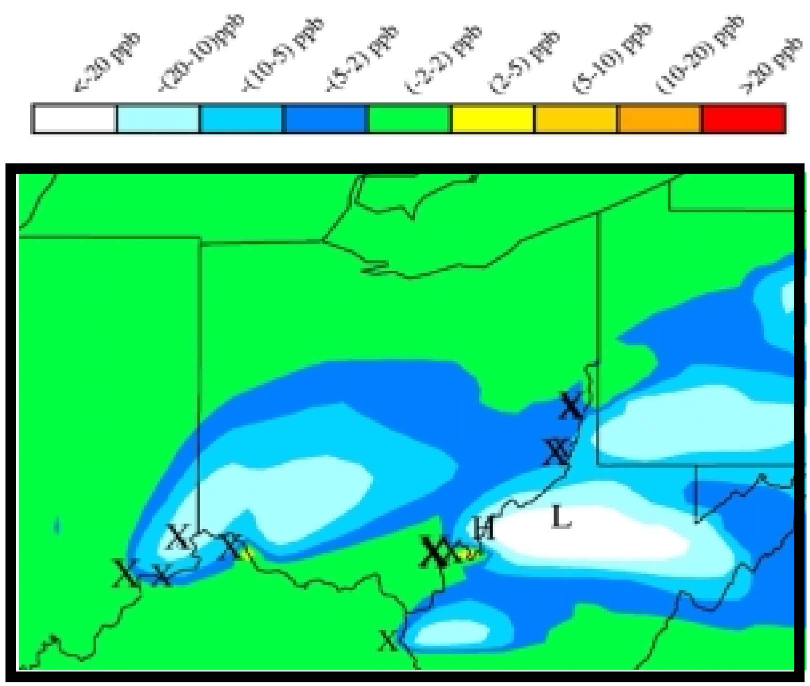


Figure VIII.

Source: Earth Tech

Scenario 3: All Ohio River Valley Power Plants Overwhelm Neighboring States

When we examine the impact of all plants in Ohio, as well as border plants in Kentucky and West Virginia, the combined effect covers the entire upper Ohio River Valley. In total, these plants emit more than 1,042,805 tons of nitrogen oxide per year.

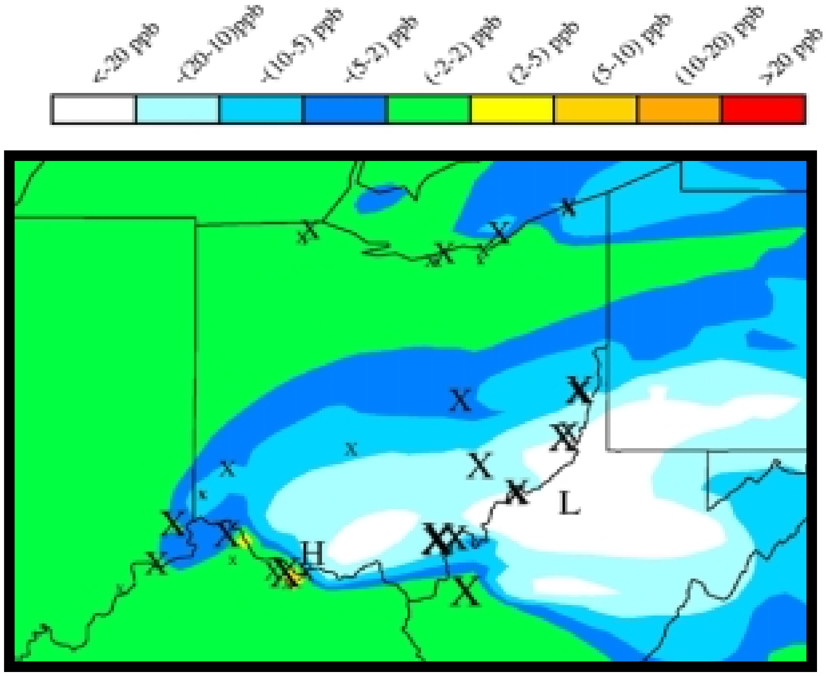


Figure VIII.

Source: Earth Tech

Scenario 4: Including Upwind Midwestern Plants Triggers Automatic Violations of Air Quality Standards

Finally, pollution from all Midwestern power plants blankets the entire Midwest and affects the Ohio River Valley most intensely.

The total impact of Midwestern power plants on the region is overwhelming. Emissions from these plants totals 2,849,190 tons of nitrogen oxide per year. For areas such as the Pennsylvania/Ohio or West Virginia/Ohio border, the impact on air quality approaches 40 ppb of ozone, equivalent to all the ozone needed to push background levels beyond the health standard.

(**Note:** The letter L in northeastern West Virginia indicates the maximum ozone drop in the region of 40 ppb)

When added to background levels, Midwestern power plant pollution alone is enough to cause an exceedence of the air quality standard.

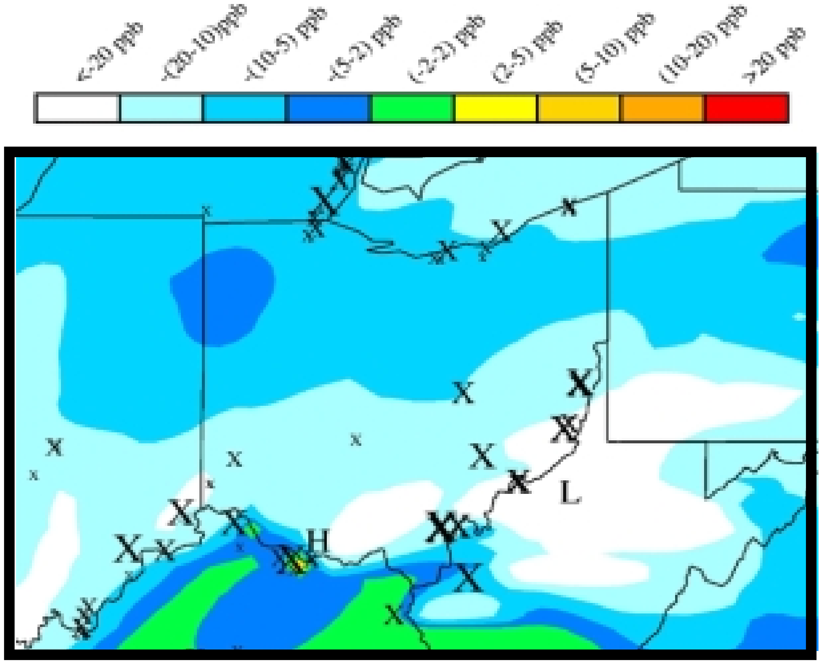


Figure IX.

Source: Earth Tech

Clean Power Equals Clean Air for the Ohio River Valley

Significant reductions in emissions from power plants that affect the Ohio River Valley can be achieved in a variety of ways. Some plants can be upgraded with the most up-to-date pollution control technology available, Selective Catalytic Reduction (SCR), which would reduce nitrogen oxide pollution by 85 percent. Others can be replaced with more efficient and cleaner natural gas plants, which would cut emissions by nearly 97 percent. An additional strategy is to displace some of the energy generated by current plants with power from cleaner, renewable sources, such as solar or wind power. Finally, energy efficiency is the most cost-effective method of reducing emissions.

Figure X below shows the ozone smog levels (averaged over an 8 hour period) during the episode of June 20, 1991. Areas in yellow, orange, or red are in violation of the national health standard for smog. These smog levels occur under the current, lax emission standards applied to the Midwest's "grandfathered" power plants. The current average nitrogen oxide emission rate for these plants is more than twice that required of a coal plants planned and built in the last two decades.

The Problem

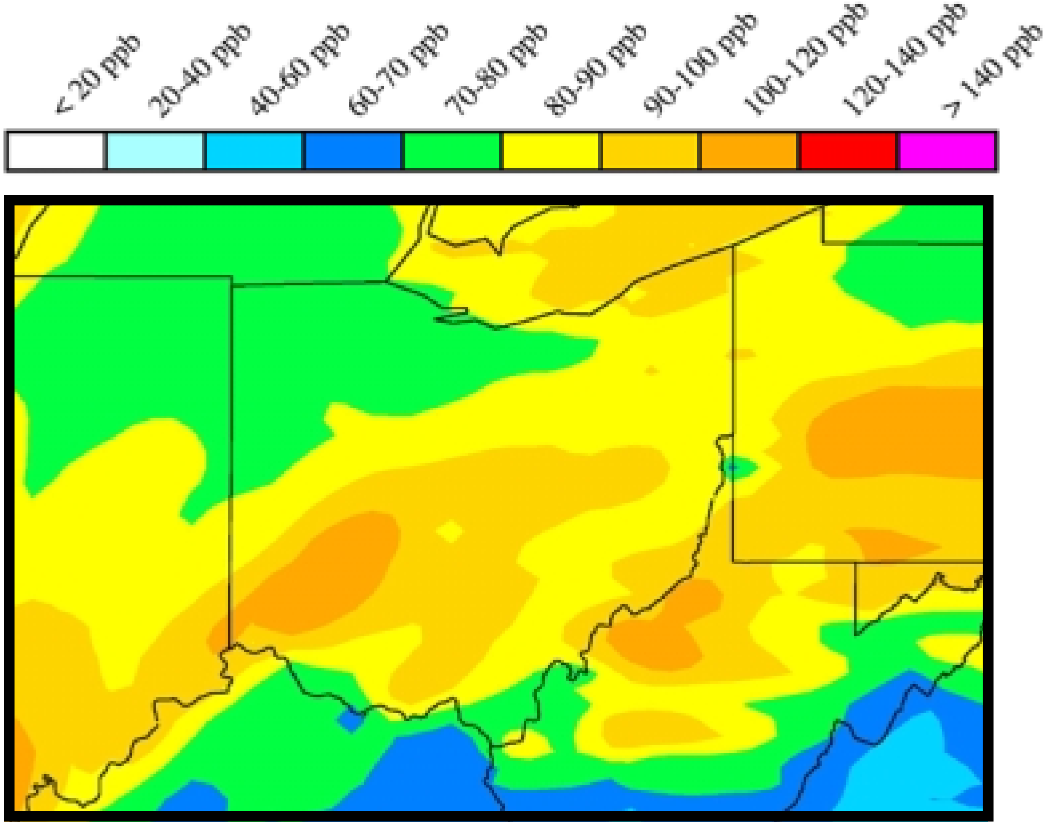


Figure X.

Source: Earth Tech

Ozone Levels in the Midwest on a Typical "Bad Air" Day

USEPA Smog Reduction Plan Greatly Outperforms The Power Companies' Proposal In Cleaning Up Ozone Alley

A number of strategies to reduce ozone pollution in the region are under consideration. The following discussion compares the effectiveness of strategies proposed by the USEPA to the proposal advanced by a group of obstructionist power and coal companies known as the Midwest Ozone Group (MOG), which includes American Electric Power and the Ohio Valley Coal Company. This latter position was ultimately adopted by several states: Ohio, West Virginia, Virginia, Alabama, Michigan, and Kentucky.

"Half-Measure" Proposal by Power Companies

Figure XI illustrates the control strategy proposed by the MOG. According to this plan, reductions would be achieved by retrofitting some plants with older, less effective pollution control equipment.

USEPA's Strategy Blocked in Court by Several Power Companies and States

Figure XII illustrates the control strategy of using the most up-to-date pollution control equipment. This approach would create twice the level of emission reductions as the industry plan.

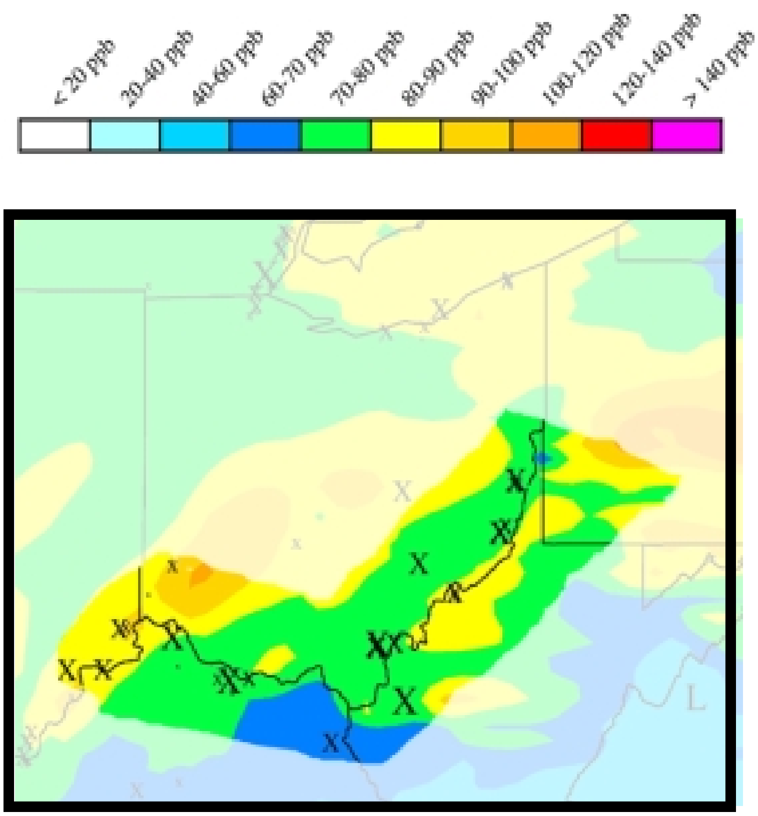


Figure XI.
Source: Earth Tech

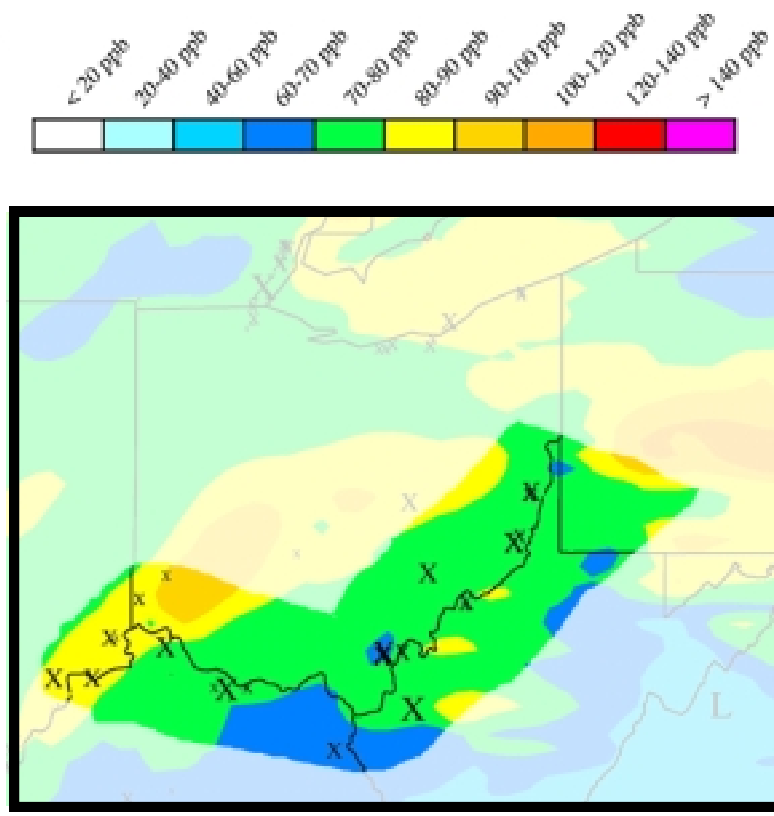


Figure XII.
Source: Earth Tech

The proposal advanced by the power industry and recalcitrant states fails to solve the ozone problem in the eastern half of the Ohio River Valley. By contrast, under the USEPA proposal, the great majority of the Valley achieves compliance.

The Problem

Power Generation Today

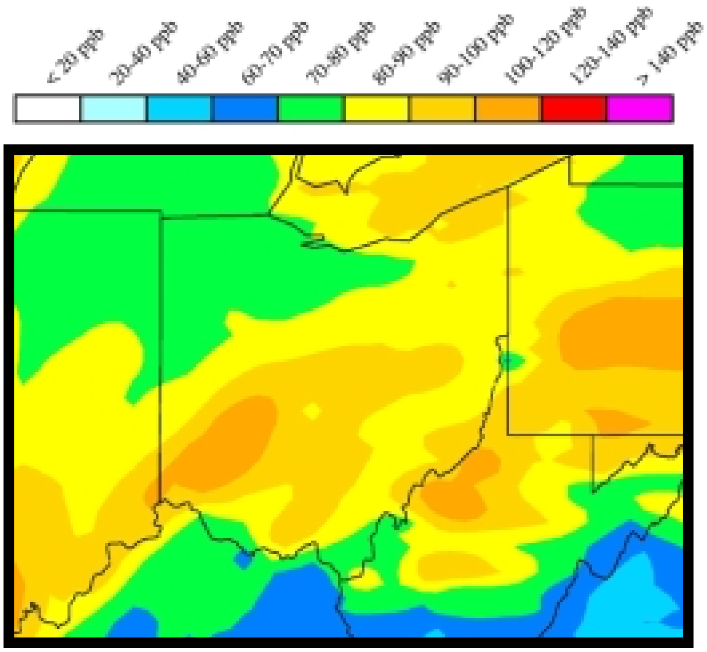


Figure XIII.
Source: Earth Tech

The Solution

Ohio's Clean Power Future

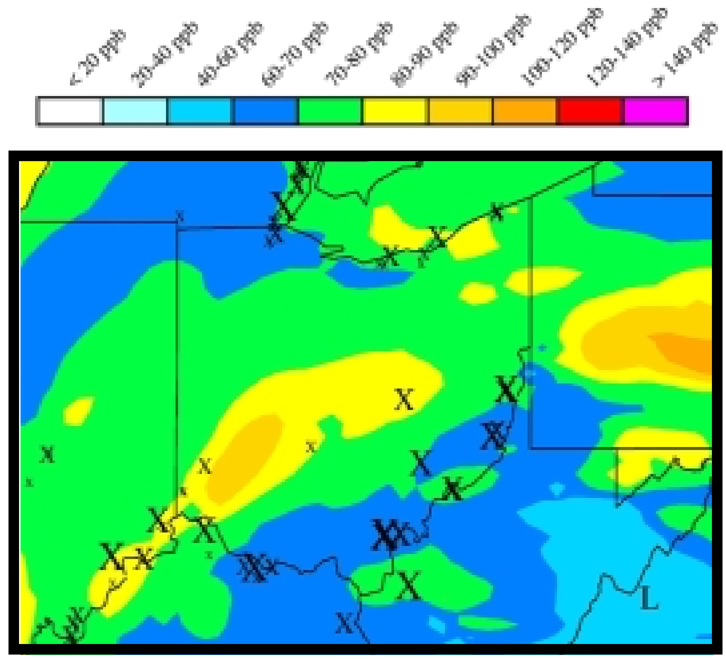


Figure XIV.
Source: Earth Tech

Power plants are the largest source of smog-causing pollution in the Ohio River Valley. While reducing emissions from other sources, such as vehicles and heavy diesel engines is important, no method of reducing smog pollution in the Ohio River Valley is as effective as cleaning up power plant emissions.

If policymakers have the courage to take steps that will protect public health, they will require older coal plants to meet the same pollution control standards as modern plants. This would dramatically reduce smog pollution. In addition, since old coal plants are only required to meet lax environmental standards in comparison to new coal and gas plants, "de-grandfathering" the old plants would make the newer cleaner plants more competitive. Finally, requiring the same standards for new and old plants would put high sulfur Eastern coal on a more competitive footing with low sulfur western coal, since all modern plants have to have pollution control devices for sulfur dioxide.

A lasting solution to the Ohio River Valley's air pollution problem entails moving toward a future in which energy is achieved through cleaner natural gas, renewable energy sources, and increased efficiency and conservation measures. Natural gas can serve as an abundant energy source for centuries, given the current rate of advances in technology and exploration, according to an analysis by the Environmental Law Institute.[2] In addition, according a recent study by the Union of Concerned Scientists, the Midwest could meet about 12 percent of its energy needs by the year 2010 from renewable energy.[3] Finally, a study by the American Council for an Energy Efficient Economy indicates that Ohio could reduce its energy needs by 26 percent by the year 2010 through investments in energy efficiency.[4]

[2] How Abundant? Assessing the Estimates of Natural Gas Supply. Environmental Law Institute (1999)
[3] Powerful Opportunity: Making Renewable Electricity the Standard. Union of Concerned Scientists (1999)
[4] Energy Efficiency as an Investment in Ohio's Future. American Council for an Energy Efficient Economy (1994)

Opportunities

Action for Public Officials

Currently, there are four opportunities for policymakers to make effective cuts in power plant pollution. They are:

Congressional action: The most effective way to reduce the air pollution threat to Ohio Valley residents would be Congressional action to repeal the "grandfather loophole," which allows old, dirty power plants to emit ozone-forming nitrogen oxide and deadly soot-forming sulfur dioxide at many times the emissions rate of newer plants. In addition, this step would assure that other areas throughout the United States that are harmed by these emissions would also experience improvements in their air quality. Currently, several bills are pending in Congress that would "de-grandfather" old, dirty plants.

USEPA's Summer Smog Rule: Implementation of the pollution reductions required by USEPA's Nitrogen Oxide State Implementation Plan (NOx SIP) would result in significant reductions of the emission of nitrogen oxides in 22 states. Reduction of the total emissions in all states would be comparable to installing up-to-date nitrogen oxide control equipment in the states' combined fleet of power plants. Allowing Ohio River Valley plants to purchase emission reduction credits from plants in other areas could, however, reduce the air quality benefits of this proposal to Ohio River Valley residents.

The utility sponsored Midwest Ozone Group has offered a counterproposal, calling for power plant cuts only half as deep as those contained in the USEPA plan. As mentioned above, six out of the 22 states covered by the USEPA rule have adopted the industry proposal. Several of these states, together with power companies, challenged the rule in the Washington, D.C. Circuit Court of Appeals, where it remains pending review.

USEPA Petitions: Eight Northeastern states have filed petitions with USEPA calling for the clean up of Midwestern power plants in order to reduce ozone levels in the Northeast. USEPA recently granted these transport petitions, ordering the clean up of 257 power plants, including 86 in the Midwest. However, the scope of the petition does not have the full reach of the 22-statewide USEPA smog plan and, for example, will not result in reductions from Illinois power plants that are also degrading the Ohio Valley's air quality.

Enforcement actions: Several Midwestern and national environmental organizations, the U.S. Department of Justice on behalf of USEPA, and five Northeastern states have filed enforcement actions against the owners of several "grandfathered" power plants. These actions allege that power plant owners have violated the law by failing to retrofit their plants with pollution control equipment when other major, life-extending investments were made. Cleaning up the plants named in the action would provide some benefit to residents living in the Ohio River Valley. However, the legal process will undoubtedly be a long one, and could result in a patchwork, rather than a comprehensive, solution.

Cost for Cleaner Power

Reducing smog pollution from power plants: The USEPA estimates that the cost of the NOx SIP would be about $1 a month for the average residential consumer household, or about $12 per year.[5] This figure is based on a cost of $1,700 for every ton of nitrogen oxide emissions that is reduced.

Reduction of nitrogen oxide emissions from power plants is the most cost-effective method of curbing this smog-causing pollutant. Comparing the cost, in dollars per ton, of different methods to reduce nitrogen oxide yields the following results:

- ♦ Power plants - $1,700 dollars per ton reduced
- ♦ Vehicle inspection and maintenance programs - $2,600 dollars per ton reduced
- ♦ Reformulated gas (ethanol or MBTE) - $3,600 dollars per ton reduced [6]

Clearly, power plants are the largest source of smog-causing nitrogen oxide emissions in the Ohio River Valley, and cleaning up the plants is the cheapest method of reducing related air pollution.

Reducing multiple pollutants: Our current regulatory system is designed to address one pollutant at a time. This can be the least cost-effective way to reduce pollution.

The best solution is to reduce power plant nitrogen oxide emissions within a broad, integrated emissions reduction strategy. Such an approach would address at least nitrogen oxide, sulfur dioxide, carbon dioxide, and air toxic emissions, as well as releases of toxic pollutants contained in the solid wastes and discharges to surface waters from power plants. Several recent analyses by USEPA and others suggest that such an approach will provide considerably more clean air benefits per dollar of clean up investment than a control strategy focused solely on power plant nitrogen oxide emissions.[7]

Taking action to comprehensively reduce all of these power plant emissions would provide owners of "grandfathered" power plants with greater certainty regarding long-term planning. Currently, owners individually add, and pay for, pollution control equipment as pollution regulations become finalized. Rather than pay for nitrogen oxide today, sulfur dioxide tomorrow, and mercury the next day, with future uncertainty about additional pollutants, a multi pollutant control strategy would add more certainty to the process. Owners could choose the most cost-effective capital investments to control multiple pollutants, either pollution control equipment, efficiency investments, or shifting to cleaner natural gas plants.

As mentioned above, USEPA estimates that implementing its NOx SIP will cost each residential consumer household about $1 per month. However, comprehensive power plant emissions reductions could be achieved for about $2-3 per month.[8]

5. USEPA NOx SIP Call final rule, regulatory impact analysis.
6. USEPA proposed NOx SIP Call rule-Technical Analysis, 1997
7. USEPA has conducted several analyses of multi-pollutant power plant emissions control strategies under the CAPI program. The most recent of these analyses, conducted in April 1999, found that "total costs of a set of (control) actions is less than a piecemeal approach."
8. See EPA's April 1999 CAPI analysis.

Action for Citizens

Encourage Action by State and Federal Officials: Citizens should encourage state and federal officials to take immediate steps to reduce power plant pollution. Citizens can contact their members to the state legislature, members of Congress, state and federal environmental protection agencies, their Governor, and the President (see Appendix IV for details).

Choose Cleaner Energy: As states continue to deregulate, both citizens and communities have the choice for choosing cleaner power. On January 1st, 2001, citizens of Ohio will be able to choose their power provider. Ohio communities will be able to purchase power for their citizens. West Virginia and Kentucky are in the process of deregulating their electric utilities.

Where communities and citizens have a choice, they should choose cleaner power. If states are still in the process of deregulating, communities and citizens need to first ensure that all customers have the ability to choose electricity providers, and then make the choice for cleaner power.

Conclusion

Power plants that line the Ohio River Valley create a blanket of smog that covers the region. As a result, the Valley suffers from unhealthy air comparable to, or worse than, levels seen in cities such as Boston or New York. Those who are most at risk, children with asthma and the elderly, are the hardest hit by this poor air quality, but we are all effected.

The technology exists to start reclaiming clean air for the Valley in a cost-effective manner. Delaying action will only create additional unnecessary suffering. The time to act is now. In the words of the American Lung Association, "when you can't breathe, nothing else matters."

Appendix I

Health Analysis Methodology -Abt Associates, 1999

SECTION 1. OZONE AIR QUALITY ESTIMATION

This section describes the methods used to model ozone concentrations, from which reductions in health effect incidences are then estimated. Sub-section 1 briefly describes ozone formation. Sub-section 2 describes the available ozone monitoring data, and Sub-section 3 discusses the extrapolation of these data to areas without ambient air quality monitors.

1.1 OZONE MONITORING DATA

Ozone concentrations were considered only in the 37 eastern states that participated in the Ozone Transport Assessment Group study plus the District of Columbia (known as the OTAG region) for the period April through October, 1997. Hourly ozone concentrations for 1997 were extracted from the Aerometric Information Retrieval System (AIRS) and put into a single AMP350-format datafile. For the ozone data, a monitor record was considered to be complete if data were available for 50 percent of days in a given season (April 1-October 31 in this analysis). A monitor day was considered valid if 75 percent or more of the hours between 9am and 8:59pm were available[9]. There were 698 unique ozone monitoring locations found in AIRS, of which 687 passed the completeness criteria.

1.2 MODELING AMBIENT OZONE

Ambient ozone levels in 1997 were modeled using the Criteria Air Pollutant Modeling System (CAPMS), a population-based system for modeling exposures to criteria air pollutants. CAPMS has been used extensively to estimate air pollution control benefits in the United States. As a first step in the modeling process, CAPMS divides the OTAG region into eight kilometer by eight kilometer grid cells, and then estimates air quality for each cell.

1.2.1 Interpolation of Air Quality Monitoring Data to CAPMS Grid Cell Centers

Modeling air quality throughout the U.S. poses some problems, since air quality data are only available from limited monitor sites. We needed a method to extrapolate to unmonitored locations, in order to estimate the impact of air pollution on the health and welfare effects considered in this analysis[10]. Given the available air monitoring data, we extrapolated from all available monitor locations to a grid of eight km by eight km population-based grid-cells throughout the OTAG region, using a Voronoi Neighbor Averaging (VNA) spatial interpolation procedure.[11]

Exhibit 1-1: VNA Spatial Interpolation

The VNA procedure interpolates air quality estimates from air quality monitors to the center of each population grid-cell. The VNA procedure is a generalization of planar interpolation. Rather than limit the selection of monitors to, for example, three, VNA identifies the set of monitors that best "surrounds" the center of each grid-cell. The result of VNA is illustrated in Exhibit 2-1. VNA determines the set of monitors that best surrounds the grid-cell by identifying which monitor is closest (considering both angular direction and horizontal distance) in each direction from the grid-cell center. Each selected monitor will likely be the closest monitor for multiple directions. The set of monitors found using this approach forms a polygon around the grid-cell center. For each grid cell, CAPMS calculates the distance to each of a set of monitors surrounding that grid cell. Monitors close to the grid cell are assumed to yield a more accurate air quality description of that grid cell, and are given a larger weight when calculating the average air quality for that grid cell. Conversely, monitors that are further away contribute less to the average. After determining the final set of surrounding monitors, the grid-cell's air quality level is calculated as an inverse, distance-weighted average of the air quality levels at the selected monitors. Note that the air quality inputs to CAPMS must be in the form (averaging time) required by the concentration-response (C-R) functions being used. For example, a C-R function relating respiratory hospital admissions to daily maximum ozone concentrations requires that daily maximum ozone concentrations be available at CAPMS grid cell centers. Although the input ozone data must be in the form of daily maxima, the input data need not be at CAPMS grid cell centers. Given any set of location-specific air quality data, CAPMS interpolates the corresponding air quality values at each CAPMS grid cell center. To reduce computational effort, CAPMS typically "bins" air quality data being used to calculate air levels, and bins the resulting air quality at CAPMS grid cell centers.

9. The choice of using at least 75 percent of the hours between 9:00am and 8:59pm is consistent with the criterion defined in the Code of Federal Regulations (U.S. Office of the Federal Register, 1995, Part 50 Appendix H).

10. Appendix A has a map of the location of ozone monitors in the U.S. The map shows that some areas of the country do not have many ozone monitors in close proximity to each other.

11. Interpolation between monitors is conducted using the same method as used by Abt Associates (1998) for the NOx SIP call analysis; previously termed the "convex polygon" method, it is more accurately described as Voronoi Neighbor Averaging (VNA) spatial interpolation, which will be used throughout this document.

1.2.2 Binning Input Air Quality Data

To reduce computational time and effort when estimating the change in health effects associated with daily pollution levels, CAPMS approximates a season's (or year's) worth of daily pollutant concentrations at each input location (i.e., location for which air quality data are available) by n "bins" of pollutant concentrations. For the ozone analysis n = 20, so that each bin represents five percent of the daily pollutant concentrations in the ozone season, April through October in this analysis. Each bin is set at the midpoint of the percentile range it represents. For n = 20 and a season's worth of ozone observations, the first bin represents the first (lowest) five percent of the distribution of 214 pollutant concentrations at the given location, and is set at the 2.5th percentile value; the second bin represents the next five percent of the distribution of daily values, and is set at the 7.5th percentile value, and so on. Each of the twenty bins therefore represents 10.7 (=214/20) days. Interpolation of air quality levels at CAPMS grid cell centers is based on these input location-specific bins, so that the seasonal incidence changes in each grid cell are calculated for twenty pollutant concentrations (the 20 bins of air quality) rather than for 214 pollutant concentrations. The resulting incidence change is then multiplied by 10.7 to reconstruct an entire season's worth of incidence change in the CAPMS grid cell.

SECTION 2. CALCULATING OZONE-RELATED ADVERSE HEALTH EFFECTS

This appendix describes methods used to estimate the adverse health effects associated with ambient ozone concentrations present throughout the U.S. Sub-section 1 discusses the CAPMS approach to calculating point estimates of health effects. Sub-section 2 discusses issues affecting the estimation of health effects. Sub-section 3 discusses the epidemiological studies used to estimate incidence estimates for each of the health effects. Section 4 discusses the uncertainty in the estimation of air pollution-related health effects.

2.1 CAPMS APPROACH TO CALCULATING ADVERSE HEALTH EFFECTS

In each eight kilometer by eight kilometer grid cell in the OTAG region, CAPMS estimates the incidence of adverse health associated with the presence of air pollutants. The regional incidence estimate (or the estimate within an individual state or county) is then calculated as the sum of grid-cell-specific incidence estimates.

2.1.1 Calculation of Point Estimates of Incidence Changes

For this analysis, CAPMS estimates the adverse health effects attributable to levels of 1997 ambient ozone. To make this estimation, CAPMS interpolates the air quality at the CAPMS grid cell center. Since the daily values have been binned at the monitors, the resulting air quality data at the CAPMS grid cell center are also binned. It then accesses the selected C-R functions being used, the required baseline incidence rates, and the grid cell population. CAPMS then calculates the incidence associated with 1997 ozone levels for each adverse health effect for which a C-R function has been accessed. For example, if the functional form of the C-R relationship is log-linear (the most commonly used form), the relationship between a change in pollutant level[12], e.g., ΔO_3, and the change in incidence of the health effect, Δy, is:

$$\Delta y \ \mathit{population} * \mathit{baseline\ incidence\ rate} * \ [e^{\beta \Delta O3} - 1]$$

where β is the coefficient of O_3 in the C-R function. The changes in incidence corresponding to the changes in air quality in each of the 20 bins of pollutant concentrations are summed and multiplied by 10.7 (the number of days per bin) to produce a seasonal ozone-related incidence estimate in that grid-cell. The resulting seasonal incidence estimate is stored, and CAPMS proceeds to the next grid cell, where the above process is repeated. Total ozone-related incidence (or the incidence that occurs in any designated geographical area) is calculated at the end of the process by summing the grid cell-specific estimates.

12. The change in this case is simply equal to the 1997 ozone level, since the effect of all ozone is estimated.

2.1.2 Calculation of Distributions of Incidence Estimates

The C-R functions used in this analysis are simply estimates of the relationship between ozone and adverse health effects. To reflect the uncertainty surrounding pollutant coefficients in the C-R functions used, CAPMS can produce a *distribution* of possible incidence estimates attributed to ozone for each adverse health effect.

To produce this distribution, CAPMS uses both the estimate of the pollutant coefficient (β) and the standard error of the coefficient estimate to produce a normal distribution with mean equal to the estimate of β and standard deviation equal to the standard error of the estimate. CAPMS takes the n^{th} percentile value of ∃ from this normal distribution — for n = 0.5, 1.5, ..., 99.5 – and produces an estimate of the incidence, given that value of β. The resulting distribution of values for each CAPMS grid cell is then stored, and CAPMS proceeds to the next grid cell, where the process is repeated. A distribution of the region-wide incidence is calculated by summing the n^{th} percentile grid cell-specific estimates, for n = 0.5, 1.5, ..., 99.5.

2.2 CONCENTRATION-RESPONSE FUNCTIONS

2.2.3 Basic Concentration-Response Model

The methods discussed in this sub-section apply to the estimation of ozone's impact on adverse health effects. For expository simplicity, the discussion below refers only to a generic "health endpoint," denoted as y, although several endpoints are associated with ozone. In addition, the discussion refers to the estimation of incidence for a health endpoint at a single location (population cell). Region-wide incidence totals are estimated by summing ozone-related incidence estimates over all population cells in the region.

Corresponding to a change in ozone, ΔO_3 , in a given population cell is an associated change in the health endpoint, Δy, in that population cell. Given a C-R function estimated by a study, and a particular change in ozone, ΔO_3 , the corresponding change in the health endpoint, Δy, corresponding to the particular ΔO_3 can be calculated. This change in the health endpoint, Δy, can be thought of as the health effect incidence that is attributable to ozone exposures in a given area.

When using a particular epidemiological study to develop a C-R function and estimate the effect of ambient ozone levels, it is important to follow the study design as much as possible. Epidemiological studies can differ in a number of ways. Some studies that measure the relationship between ozone and hospital admissions related to respiratory illnesses may include all categories of respiratory health effects, while others may exclude certain symptoms from the study. One study may have measured daily (24-hour) average ozone concentrations while another study may have used two-day averages. Some studies have assumed that the relationship between y and ozone is best described by a linear form (i.e., the relationship between y and ozone is estimated by a linear regression in which y is the dependent variable and ozone is one of several independent variables). Other studies have assumed that the relationship is best described by a log-linear form (i.e., the relationship between the natural logarithm of y and ozone is estimated by a linear regression).[13] Finally, one study may have considered changes in the health endpoint only among members of a particular subgroup of the population (e.g., individuals 65 and older), while other studies may have considered the entire population in the study location.

Estimating the relationship between ozone and a health endpoint, y, consists of (1) choosing a functional form of the relationship and (2) estimating the values of the parameters in the function assumed. The two most common functional forms in the epidemiological literature on ozone and health effects are the log-linear and the linear relationship. The log-linear relationship is of the form:

$$y = Be^{\beta \cdot O_3}$$

13. The log-linear form used in the epidemiological literature on ozone-related health effects is often referred to as "Poisson regression" because the underlying dependent variable is a count (e.g., number of deaths), believed to be Poisson distributed. The model may be estimated by regression techniques but is often estimated by maximum likelihood techniques. The form of the model, however, is still log-linear.

or, equivalently,

$$\ln(y) = \alpha + \beta * O_3$$

where the parameter B is the incidence of y when the concentration of O_3 is zero, the parameter β is the coefficient of O_3, ln(y) is the natural logarithm of y, and $\alpha = \ln(B)$[14]. If the functional form of the C-R relationship is log-linear, the relationship between ΔO_3 and Δy is:

$$\Delta y = y(e^{\beta \Delta O3} - 1)$$

where y is the baseline incidence of the health effect (i.e., the incidence before the change in O_3). For a log- linear C-R function, the relative risk (RR) associated with the change ΔO_3 is:

$$RR_{\Delta O3} = e^{\beta \Delta O3}$$

Epidemiological studies often report a relative risk for a given ΔO_3, rather than the coefficient, β, in the C-R function. The coefficient can be derived from the reported relative risk and ΔO_3, however, by solving for β:

$$\beta = \ln(RR)/\Delta O_3$$

The linear relationship is of the form:

$$y = \alpha + \beta * O_3$$

where α incorporates all the other independent variables in the regression (evaluated at their mean values, for example) times their respective coefficients. When the C-R function is linear, the relationship between a relative risk and the coefficient, β, is not quite as straightforward as it is when the function is log-linear.

Studies using linear functions usually report the coefficient directly. If the functional form of the C-R relationship is linear, the relationship between Δ_{O3} and Δy is simply:

$$y = \beta * O_3$$

For a given O_3 change, ΔO_3 (using a measure consistent with the O_3 measure used in the C-R function — e.g., daily average O_3), and a value for the O_3 coefficient, β, the corresponding change in the health endpoint, Δy, is estimated — using equation (1), if the C-R function is log-linear, or equation (2), if the C-R function is linear.

A few epidemiological studies, estimating the relationship between certain morbidity endpoints and O_3 , have used functional forms other than linear or log-linear forms. Of these, logistic regressions are the most common. Abt Associates (1999) provides further details on the derivation of dose-response functions.

14. Other covariates besides pollution clearly affect mortality. The parameter B might be thought of as containing these other covariates, for example, evaluated at their means. That is, B = Boexp{\$1x1 + ... + \$nxn}, where Bo is the incidence of y when all covariates in the model are zero, and x1, ... , xn are the other covariates evaluated at their mean values. The parameter B drops out of the model, however, when changes in y are calculated (see equation (7)) and is therefore not important.

2.2.4 Baseline Incidences

Most of the C-R functions used in this analysis are log-linear, and the estimation of incidence changes based on a log-linear C-R function requires a baseline incidence. The baseline incidence rate is the fraction of a given population that incurs an adverse health effect in a given time period. For example, in 1994, people ages 65 and older had a daily incidence rate for respiratory-related hospital admissions of 1.187 E-4; in other words, out of a population of one million individuals, about 119 would be admitted to the hospital for a respiratory condition on any given day[15]. Unfortunately, incidence rates are not available for each CAPMS grid-cell, so national rates are used, or in some cases the rate found in the (epidemiological) study population was used to develop an incidence rate.

2.2.5 Population

Many epidemiological studies focus on a particular age cohort. The age group chosen is often a matter of convenience (e.g., extensive Medicare data may be available for the elderly population) and not because the effects are necessarily restricted to the specific age group. To avoid overestimating the benefits of reduced pollution levels, this analysis applies the given C-R relationships only to those age groups corresponding to the cohorts studied.

Some people are especially sensitive to ozone, and suffer effects of a different kind or to a different degree than does the rest of the population. Factors that result in a higher risk include: (1) a person's biological response to ozone, (2) pre-existing lung disease, (3) activity patterns, (4) personal exposure, and (5) personal factors, such as age and nutritional status.[14] Asthmatics and others with pre-existing respiratory conditions are examples of potentially sensitive populations. There are a limited number of epidemiological studies, so it is not possible to capture all of the ozone-related effects for all groups of interest. In this analysis, we have C-R functions that cover asthmatics, children, adults, the elderly, and persons of all ages.

There is little doubt that ozone affects children's health, however, the type of the effect is uncertain. Studies by Avol et al (1985), Portney and Mullahy (1986), and Krupnick et al. (1990) found children did not develop the ozone-related respiratory symptoms suffered by adults. However, in a comprehensive review of the health effects literature, EPA (1996b, p. 9-36) concluded that: "Human studies have identified a decrease in pulmonary function responsiveness to ozone with increasing age, although symptom rates remain similar." So it appears that children may indeed be sensitive to ozone, however we have a limited number of studies to estimate this effect.

Finally, some of the C-R functions are derived from epidemiological studies of relatively small populations. Nevertheless, we apply these functions to all areas in the OTAG region. Our results may be inaccurate, to the extent that the OTAG region differs from the study population used as the basis for the C-R function.

2.2.6 Pooling Study Results

When only a single study has estimated the C-R relationship between a pollutant and a given health endpoint, the estimation of a population cell-specific incidence change is straightforward. For some endpoints, however, C-R functions have been estimated by several studies, often in several locations. In this case, a pooled, "central tendency" C-R function can be derived from multiple study-specific C-R functions in several ways.

15. All respiratory hospital admissions are based on first-listed discharge figures for the latest available year, 1994. The rate equals the annual number of first-listed diagnoses for discharges (1.437 million) divided by the 1994 population of individuals 65 years and older (33.162 million), and then divided by 365 days in the year. The discharge figures are from Graves and Gillum (1997, Table 1), and the population data is from U.S. Bureau of the Census (1997, Table 14).

One potential method of pooled analysis is simply averaging the coefficients from all the studies. This has the advantage of simplicity, but the disadvantage of not taking into account the measured uncertainty of each of the estimates. Estimates with great uncertainty surrounding them are given the same weight as estimates with very little uncertainty.

An alternative approach to pooling studies is to apply subjective weights to the studies. If the analyst is aware of specific strengths and weaknesses of the studies involved, this prior information may be used as input to the calculation of weights which reflect the relative reliability of the estimates from the studies.

A third approach to pooling the estimates from different studies – and the one primarily used in this analysis – is to give more weight to estimates with little reported uncertainty and less weight to estimates with relatively more uncertainty. The exact way in which weights are assigned to estimates of ozone coefficients from different studies in a pooled analysis depends on the underlying assumption about how the different estimates are related to each other. If, for example, there is actually a distribution of true ozone coefficients, or β's, that differ by location (referred to as the "random effects" model), the different coefficients reported by different studies may be estimates of *different* underlying coefficients, rather than just different estimates of the same coefficient. In contrast to what one might call a "fixed effects" model (which assumes that there is only one β everywhere), the random-effects model allows the possibility that different studies are estimating different parameters.

In deciding whether to use a fixed or random effects pooling estimate, we use the fixed effects approach as our null hypothesis, and then determine whether the data suggest that we should reject this null hypothesis. The test statistic is asymptotically chi square distributed; if the test statistic exceeds a value that would occur no more than 5 percent of the time, then a random effects pooling estimate is used. Further details on how studies are pooled are given below; in addition, Abt Associates (1999, p. A-48) discusses general issues involved with pooling in more detail.

In some cases, studies reported several estimates of the C-R coefficient, each corresponding to a particular year or particular study area. For example, Ostro and Rothschild (1989) reported six separate regression coefficients that correspond to regression models run for six separate years. This analysis combined the individual estimates using a meta-analysis on the six years of data.

2.3 ADVERSE HEALTH EFFECTS RELATED TO OZONE

Health effects studies have found that ozone is associated with a variety of adverse health outcomes, ranging from relatively minor symptoms, to hospital admissions, chronic illness, and even premature mortality. Not all studies have found ozone linked to these effects. In particular, there is a significant amount of uncertainty about the relationships between ozone and both mortality and chronic illness. In this section, we discuss the studies we use to estimate ozone-related adverse health effects. In addition, we briefly discuss the ozone-mortality and ozone-chronic illness relationships (although we do not estimate them). Appendix I-a presents the C-R functions that are used in this analysis (see website: www.theoec.org/ovalley/A-I-a).

2.3.1 Hospital Admissions

The hospital admissions estimated in this analysis are respiratory admissions. There is not much evidence linking ozone with other types of hospital admissions; most of the evidence supports the link between ozone and respiratory admissions. In addition to hospital admissions, we consider separately the association between ozone and ER visits. Hospital admissions require the patient to be examined by a physician, and on average may represent more serious incidents than ER visits (Lipfert, 1993, p. 230).

2.3.2 Respiratory Hospital Admissions

To estimate the number of hospital admissions for respiratory illness, we pool the incidence estimates from a variety of U.S. and Canadian studies, using a random-effects weighting procedure. These studies may differ in two important ways: (1) some studies considered people of all ages while others considered only people ages 65 and older; and (2) the types of effects included in studies of respiratory hospital admissions and air pollution may vary substantially.

The epidemiological studies used in this analysis used version 9 of the International Classification of Diseases (ICD-9)[16]. The broadest classification of respiratory admissions includes ICD-9 codes 460-519.

Other studies, however, considered only subsets of the broader classification. For example, Burnett et al. (1997) consider ICD-9 codes 466, 480-486, 490-494, and 496. If the broadest category (460-519) is too broad, including respiratory illnesses that are not linked to ozone, we would expect the estimated pollutant coefficients to be biased downward; however, they would be used in combination with a larger baseline incidence in estimating changes in respiratory hospital admissions associated with changes in pollutant concentrations. If the broadest category is correct (i.e., if all the respiratory illnesses included are actually associated with ozone), then studies using only subsets of ICD-9 codes within that category would presumably understate the change in respiratory hospital admissions. It is likely, however, that all the studies have included the most important respiratory illnesses, and that the impact of differences in the definition of "all respiratory illnesses" may be less than that of other study design characteristics. We therefore treat each study equally, at least initially, in the pooling process, assuming that each study gives a reasonable estimate of the impact of air pollution on respiratory hospital admissions.

There are several steps in our estimation process:

(1) Develop the C-R function for ozone in a model from a given study. (Note that a study may include multiple pollutants in the final model.) We generally prefer this, especially if the model includes a measure of particulate matter, which has been found to be related to a number of adverse effects. For example, Burnett et al. (1997) included $PM_{10\text{-}2.5}$, O_3, NO_2, and SO_2 in their final model for respiratory admissions[17]; we just use the O_3 coefficient from this model to develop a C-R function.

(2) If a study estimated separate models for non-overlapping respiratory illness categories, we sum the estimated incidence changes across these non-overlapping categories. For example, Burnett et al. (1999) estimated separate models for asthma admissions (ICD-9 code 493) and respiratory infection admissions (ICD-9 codes 464, 466, 480-487, 494); we calculated the ozone-related incidences for these two categories, and then summed the two estimates.

(3) We then pooled studies from the same geographic location, using an inverse-variance weighted average. Three studies based on data from Toronto and two studies based on data from Minneapolis were pooled in this fashion, and these two estimates were then combined with the estimates from the studies in other locations (e.g., Buffalo, NY). We used a random effects pooling procedure for this final step.

Exhibit 3-2 (see www.theoec.org/ovalley/A-I-Exhibit3-2) summarizes the studies used in estimating respiratory admissions. Since two of the 13 studies did not include ozone in the final model, this raises the question of whether these studies should be included anyway, with the mean ozone estimate for these studies then set to zero. However, we would also need an estimated standard error, to include these in our pooled estimate, since the pooling weights are based on the estimated standard error. We considered several approaches. One approach is to use a simple average: 11 studies (out of 13) would be used to develop a pooled ozone estimate; the pooled estimate could then be given a weight of 11/13, with the remaining 2/13 weight applied to a zero estimate. Alternatively, we could simply drop those studies that do not include ozone, and base the pooling on 11 studies. We used the latter approach, however the interested reader can easily apply an alternative weighting to derive an overall estimate.

16. The International Classification of Diseases is often used to define health outcomes in epidemiological studies. The Center for Disease Control provides detailed information:
17. They defined respiratory admissions using the following codes from the ninth version of the International Classification of Disease (ICD): 464-466, 480-486, 490-494, 496

2.3.2 Emergency Room Visits

There is a wealth of epidemiological information on the relationship between air pollution and hospital admissions for various respiratory and cardiovascular diseases; in addition, some studies have examined the relationship between air pollution and ER visits. Because most ER visits do not result in an admission to the hospital — the majority of people going to the ER are treated and return home — we treat hospital admissions and ER visits separately, taking account of the fraction of ER visits that do get admitted to the hospital, as discussed below. The only types of ER visit that have been explicitly linked to ozone in U.S. and Canadian epidemiological studies are asthma visits. However, it seems likely that ozone may be linked to other types of respiratory-related ER visits. So we include two separate estimates: (1) an estimate for asthma-related ER visits, and (2) an estimate for total respiratory-related ER visits based on the observed relationships between ER visits and hospital admissions.

2.3.3 Asthma ER Visits

We use three C-R functions for asthma-related ER visits, based on epidemiological studies by Cody et al. (1992), Weisel et al. (1995), and Stieb et al. (1996). The first two studies, Cody et al. and Weisel et al., were conducted in Northern New Jersey. The Cody et al. study examined asthma-related ER visits over a 16-month period between May, 1988 and August, 1989, and found that ozone was linked to asthma-related ER visits. No significant effect was seen for PM_{10} or SO_2 . Using a one-pollutant model, Weisel et al. also found ozone linked to asthma-related ER visits in an all-age 1990 population for eight New Jersey counties. Finally, Stieb et al. examined asthma-related ER visits over an eight-year period from May through September in St. John, New Brunswick, Canada. Ozone was linked to ER visits within the all-ages population, especially when ozone levels exceeded 75 ppb. No significant effect was seen for the other pollutants.

An additional study by Schwartz et al. (1993) failed to find a significant relationship between asthma-related ER visits and ozone. In this study of Seattle residents, Schwartz et al. found PM_{10} to be significantly related to asthma-related ER visits. We include the Schwartz et al. study in Exhibit 3-4, however, we do not include this result in our pooled estimate of the link between ozone and asthma-related ER visits.

2.3.4 Total Respiratory ER Visits

There have been far fewer studies that examine the link between ozone and ER visits than those that examine the link between ozone and hospital admissions. This is curious, in light of the strong evidence linking ozone and a variety of respiratory ailments. However, in part, this is understandable. ER data are not easily obtainable, so epidemiologists have fewer opportunities to explore this relationship. This lack of research results creates obvious problems for the present analysis, but there is a reasonable alternative. Since people who are hospitalized for ozone-related respiratory problems likely pass through the ER – i.e., ozone-related hospitalizations are presumably not planned events and need urgent treatment — it seems reasonable to assume that ozone-related ER visits are some multiple of ozone-related hospital admissions. We estimate total respiratory-related ER visits using the observed relationship between ER visits and hospital admissions multiplied by the estimated number of ozone-related hospital admissions. This is the same procedure used by the ALA (1996, p. 15). The ALA estimated respiratory-related ER visits by first estimating respiratory-related hospital admissions, and then multiplying the estimated hospital admissions by an observed (3:1) ratio between ER visits and hospital admissions[14].

2.3.5 Minor Symptoms

Two studies, one by Ostro and Rothschild (1989) and the other by Krupnick et al. (1990), cover a variety of minor symptoms. To avoid double counting, we treat these two studies as alternative measures of the same health effect, and pool the incidence estimates.

Ostro and Rothschild (1989) estimated the impact of ozone and $PM_{2.5}$ on the incidence of minor restricted activity days (MRAD) in a national sample of the adult working population, ages 18 to 65, living in metropolitan areas[15]. Separate coefficients were reported for each year in the analysis (1976-1981). The coefficient used in this analysis is a weighted average of these coefficients using the inverse of the variance as the weight.

Krupnick et al. (1990) estimated the impact of coefficient of haze (COH, a measure of particulate matter concentrations), ozone and SO_2 on the incidence of any of 19 symptoms or conditions[17]. They used a logistic regression model that takes into account whether a respondent was well or not the previous day. A key difference between this and the usual logistic model is that the model they used includes a lagged value of the dependent variable.

2.3.6 Asthma and Shortness of Breath

A number of studies have linked ozone to acute respiratory problems in asthmatics. Dockery et al. (1989) found ozone significantly related to the asthma rates of grade school children. Holguin et al. (1985) found ozone linked to asthma symptoms in a study of 42 Houston asthmatics ranging in age from 7 to 55 years. Following work by Whittemore and Korn (1980) – discussed below – Holguin et al. developed a sophisticated model that controlled for NO_2, pollen, whether a respondent had an asthma attack on the previous day, and other variables; however, they did not control for particulate matter. Using spectral analysis, Lebowitz et al. (1987) found ozone linked to wheezing, coughing, and reduced pulmonary functioning in asthmatics in Tucson. In subsequent work in Tucson, Krzyzanowski et al. (1992) found asthmatics were more sensitive to ozone exposure, experiencing a greater decline in pulmonary function than non-asthmatics. Ostro et al. (1995) found ozone increased symptoms of shortness of breath in a study of African-American asthmatics living in Los Angeles. Finally, Whittemore and Korn (1980) examined 443 asthmatic children and adults in six communities in southern California, for three 34-week periods in 1972-1975. Controlling for TSP levels, they found oxidants (90 percent of which is ozone) were significantly related to asthma attacks[18].

After comprehensively reviewing the literature, EPA (1996b, p. 7-121) concluded that the epidemiological literature has "generally supported a direct association between ambient ozone/oxidant concentrations and acute respiratory morbidity in asthmatics." However, as this suggests, the published studies have not uniformly found a significant relationship between ozone and asthma symptoms. In a careful study of 24 cities in the U.S. and Canada , published concurrently with the EPA review, Dockery et al. (1996) did not report a significant relationship between the annual average of the daily peak ozone level and asthma symptoms reported in the previous year. It is unclear why Dockery et al. did not support previous findings. One possible explanation is that an annual measure is too crude, and that a daily measure of ozone is more appropriate. Alternatively, the annual average of the daily peak may yield different results than if the annual average of all hours was used as in Dockery et al. (1989).

We estimate ozone-related reductions in asthma symptoms using three studies. We present results based on the Whittemore and Korn study (1980) because it is the only study that controls for a measure of particulate matter, measures a symptom that is easily recognized, and can be easily used to develop a C-R function[19]. In addition, we present results based on the single pollutant models found in Dockery et al. (1989) and Ostro et al. (1995), because they focus on children, who may be particularly susceptible to the effects of ozone.

The three studies that we use have symptom definitions and populations that overlap somewhat with each other, and the estimates based on these studies should not be added together to get an overall effect. Dockery et al. (1989) measured the prevalence of children ages 10-12 who had doctor-diagnosed asthma over the course of a year; Whittemore and Korn (1980) looked at the number of days that juvenile and adult asthmatics suffered one or more asthma attacks; Ostro et al (1995) looked at the number of days with shortness of breath in African-American asthmatics ages 7-12. Clearly, the populations in Dockery et al. and Ostro et al. are a subset of that covered by Whittemore and Korn. However, the effects examined in the study are still somewhat different and give another view of the impact of ozone on children.

17. Krupnick et al. (1990) list 13 specific "symptoms or conditions": head cold, chest cold, sinus trouble, croup, cough with phlegm, sore throat, asthma, hay fever, doctor-diagnosed ear infection, flu, pneumonia, bronchitis, and bronchiolitis. The other six symptoms or conditions are not specified.
18. EPA (1996b, pp. 7-115 to 7-121) nicely summarizes most of the studies presented here.
19. The study by Krzyzanowski et al. (1992) uses technical measures of lung functioning such as peak expiratory flow rate; Lebowitz et al. (1987) is somewhat qualitative and thus difficult to translate into a C-R function

Appendix II

Cumulative Exposure Methodology - Harvard School of Planetary Studies, (1999)

Cumulative exposure to ozone was analyzed by using the USEPAIRS database to identify the hours of ozone for each monitor above a certain cutoff point. The data in the analysis included ozone-monitoring data from April through October 1998, between the hours of 11am to 7pm. The cumulative background levels were calculated for 30, 40, 50, 60, 70, and 80ppb.

The first line of each listing includes the name for the monitoring site, followed by the AIRS id number, the latitude, longitude, the total number of hours (between 11am - 7 pm) for which there were measurements, & the mean ozone concentration during the period. It is also noted if there were months that weren't operational during Apr-Oct. After the header, there is a line corresponding to each cutoff. The first number is the cumulative ozone level (in ppb) based on that cutoff, followed by the total number of hours where the cutoff was exceeded. The final column contains the fraction of the number of hours where the cutoff was exceeded to the total number of hours in the record

BOSTON:

Lynn 250092006442011 42.4744 -70.9725 1862 43.821

30 ppb	28837	1422	0.764
40 ppb	16562	979	0.526
50 ppb	8732	556	0.299
60 ppb	4488	300	0.161
70 ppb	2265	154	0.083
80 ppb	1098	74	0.04

Chelsea 250251003442011 42.4017 -71.0311 1816 37.185

30 ppb	19635	1089	0.6
40 ppb	10481	696	0.383
50 ppb	5215	354	0.195
60 ppb	2442	198	0.109
50 ppb	1010	86	0.047
80 ppb	390	40	0.022

Waltham 250174003442011 42.3836 -71.2139 1801 44.882

30 ppb	31446	1320	0.733
40 ppb	19626	997	0.554
50 ppb	11186	658	0.365
60 ppb	5948	379	0.21
70 ppb	2972	223	0.124
80 ppb	1259	113	0.063

NEW YORK CITY: (Greenwich, CT included also)

nyc67 360850067442011 40.5969 -74.1264 1848 44.756

30 ppb	31972	1419	0.768
40 ppb	19425	1009	0.546
50 ppb	11337	595	0.322
60 ppb	6422	373	0.202
70 ppb	3272	241	0.13
80 ppb	1400	125	0.068

Greenwich 090010017442011 41.0036 -73.5853 1908 47.524

30 ppb	35999	1582	0.829
40 ppb	21839	1192	0.625
50 ppb	12126	707	0.371
60 ppb	6615	402	0.211
70 ppb	3306	250	0.131
80 ppb	1317	136	0.071

OHIO RIVER VALLEY

Marietta 391670004442011 39.4317 -81.46 1913 51.395 (Washington)

30 ppb	43710	1600	0.836
40 ppb	28731	1344	0.703
50 ppb	16913	987	0.516
60 ppb	8766	607	0.317
70 ppb	4149	333	0.174
80 ppb	1542	165	0.086

Hamilton 390610006442011 39.2758 -84.3661 1838 47.171

30 ppb	36774	1396	0.76
40 ppb	23979	1119	0.609
50 ppb	14116	821	0.447
60 ppb	7376	504	0.274
70 ppb	3486	262	0.143
80 ppb	1468	127	0.069

Huntington 540110006442011 38.4311 -82.4175 1898 51.824 (Cabell)

30 ppb	44971	1544	0.813
40 ppb	30627	1306	0.688
50 ppb	19032	960	0.506
60 ppb	10921	633	0.334
70 ppb	5799	372	0.196
80 ppb	2958	191	0.101

Appendix III

Photochemical Grid Modeling of the 1991 OTAG Episode to Evaluate Electric Utility Impacts

Prepared by Gary Moore, Lloyd Schulman, Elfrun Lehmann; Earth Tech Inc, 1999.

(**Note:** for tables and figures, go to www.theoec.org/ovalley/A-III-tf)

The CAMx2 model was used to conduct photochemical grid modeling for the OTAG region for the period July 17-21, 1991. The CAMx2 model directly utilizes UAM-V input files developed for the OTAG modeling. The OTAG fine mesh UAM-V simulations consisted of a 7 layer structure with layer interfaces at 50, 100, 250, 500, 1500, 2500, and 4000 m. The horizontal extent of the domain, which is identical with the OTAG UAM-V fine mesh (illustrated in Figure 1), consists of a 137 by 110 cell domain mapped on a latitude-longitude grid with an approximate spacing of 12 to 16 km. The focus of the present study was on the Ohio River Valley. A sub-region of 54 by 54 cells was defined for analysis and presentation purposes. The sub-region domain is illustrated by in Figure 2.

The input data files used in the present modeling exercise were obtained from an OTAG data center tape, with the meteorological data files downloaded from the US EPA's web pages. All of the files were in UAM-V input format. Table 1 lists the files by form, variable types, and data base source.

The surface layer emission files were for the 1991 base case rather than the 2007 future case since the future emissions projections are uncertain. The only file modified for the present study was the day specific elevated point source file.

The CAMx2 model was configured for speed. The Smolarkewicz numerical advection scheme was selected over the slower, but less diffusive, Bott scheme. The CAMx2 was tested at several levels of compiler optimization and was exercised as the highest level (fastest) level at which the model solution remained consistent with lower levels of optimization. Initially the CAMx2 was operated using the plume-in-grid (PIG) approach used by UAM-V simulations made during the OTAG project. However, this approach was discontinued when it was found that the PIG produced severe air mass displacement mismatches throughout the domain. Due to shortcuts made in implementation, the PIG as presently configured in CAMx2 cannot be used for small sources and extreme levels of control. The difference between removing a source entirely and zeroing out the emissions is pronounced. It was agreed among the project participants that CAMx2 should be exercised without using the PIG option.

Methodology for Generating the Point Source Emission Scenarios

The identification of the point sources in the UAM-V input file was complicated by the fact that plant identifiers, names, and SCC codes are not contained in the file. The only information present are the plant location, stack parameters, and the emission rates. Also, the plant/stack locations were not identical across the various databases. A spread sheet supplied by the Ohio Environmental Council does not have plants in exactly the same location as does the EPA 1990 base year point source file from which the UAM-V point source file was derived.

A search radius technique was used to identify the utility point sources. Initially, a 4-km search radius was used to match plants from both databases, but this was found to be insufficient. The solution employed was to expand the search radius to 10 km and apply a minimum emission threshold of 1000 moles/hour for NO_x to ensure that only utility sources were identified.

Quality assurance was conducted by using a program, which reads in both the base, and the modified point source input file to CAMx2. The program finds all stack entries that are different and writes out the differences into a special ASCII data file for plotting. The emission rate and the locations were used to plot symbols on the concentration isopleth plots. The size of the symbols was scaled to the logarithm of the NO_x emission rate after it has been summed over all of the stacks at the facility. Plotting the symbols allows a visual inspection of the location and magnitude of emissions.

Development of the Plant Shutoff Emission Scenarios (1, 2, and 3)

In the initial work plan, three scenarios were clearly identified, in addition to the base year simulation. The first scenario consisted of turning off all power plants within Ohio. In addition a set of plants just across the border in Kentucky and West Virginia are also turned off. The Kentucky plants were identified as those with latitude greater than 38.20 degrees. The power plants removed are summarized in Table 2. A total of 2857 tons per day (tpd) of NO_x distributed over 79 stacks were removed. A list of the NO_x emissions by stack is presented as entry A-1 in the appendix.

A second scenario consisted of leaving all of the plants identified in scenario 1 on, but turning off all the power plants in Michigan, Illinois, Indiana, Missouri, and the rest of Kentucky. The list of power plants shut off is presented in Table 3. This scenario resulted in a removal of 5144 tpd of NO_x over 274 stacks.

The third scenario was the same as the second scenario with the addition that the Ohio, Kentucky and West Virginia power plants were also turned off. The complete set of power plants removed for this scenario is summarized in Table 4. This scenario results in the removal of 7806 tpd of NO_X over 342 stacks. A listing of the individual stack emissions are summarized in the appendix as entry A-2.

Several specific urban areas were identified for closer study. The power plants upwind of Cincinnati, OH were one group of particular interest. A second group was those upwind of the Huntington, WV area. A third group was upwind of the Parkersburg, WV area and a fourth group was upwind of Pittsburgh, PA. The Ohio Environmental Council identified a set of eight plants. An additional set of six sites was identified as having potential air impacts in the four urban areas identified. A listing of the 14 power plants removed is presented in Table 5. All of the plants had NO_x emission of the order 100 tpd or larger, and so dominated the upwind point source NO_x budget. The total NOx daily tonnage shut off for the 14 plant scenario was 1863 tpd.

Development of the Reduced Plant Emission Scenarios (3b and 3c)

In order to understand how specific types of planned NO_x emission standards for power plants throughout the mid-west will affect the maximum daily 1 hour and 8-hour ozone concentrations, a third set of power plant emission reductions were modeled with CAMx. The power plant sources whose emissions were reduced were the same as those for the scenario 3 simulation (Missouri, Kentucky, Illinois, Indiana, Ohio, Michigan, and portions of West Virginia bounding Ohio). The emissions were reduced to NO_x emissions rate of 0.25 lb/MMBTU and 0.15 lb/MMBTU. The reductions were made using the emission rates reported in the Excel spreadsheet provided and described earlier. The emission were scaled by the factors $0.25/Q_e$ or $0.15/Q_e$, where Q_e is the NO_x emission rate reported in the Excel spreadsheet for each of the power plants under consideration. The reduction to 0.25 lb/MMBTU is referred to as scenario 3b, while the deeper reduction to 0.15 lb/MMBTU is identified as scenario 3c.

The scaling was not done for units in the OTAG elevated point source file which have a combined NO plus NO_2 molar emission rate of less than 1000 moles/hour (1.2 tpd). Once again this threshold was used to exclude non power plant units that might be within several kilometers of such a facility. The emission rates of all emitted species were scaled in exactly the same manner, although the levels of control for other species such as VOC and CO resulting from the NOx control technology will be significantly different. A list of power plants is presented in Table 6 which was compiled using the program which detects and prints out all points in the scenario point source input file to CAMx that is different from the base case input file. The table of contents summarizes both the base NOx emission rate and the emission rate of the units that have been scaled to 0.15 lb/MMBTU. From Table 6 one can note that that emission reductions have been rather deep and actually on the average consists of a 62% and 75% reduction of NOx for the 3b and 3c scenarios respectively. Of the 342 scenario 3 stacks only 27 had initial emission rates of less than 0.25 lb/MMBTU, and only 8 were less than 0.15 lb/MMBTU.

Appendix IV

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