Air Quality and Exercise-Related Health Benefits from Reduced Car Travel in the Midwestern United States

Maggie L. Grabow,[1,2] Scott N. Spak,[1,3,4] Tracey Holloway,[1,4] Brian Stone Jr.,[5] Adam C. Mednick,[1,6,7] and Jonathan A. Patz[1,2,8]

[1]Nelson Institute, SAGE (Sustainability and the Global Environment), University of Wisconsin–Madison, Madison, Wisconsin, USA; [2]Department of Population Health Sciences, School of Medicine and Public Health, University of Wisconsin–Madison, Madison, Wisconsin, USA; [3]Now: Public Policy Center, University of Iowa, Iowa City, Iowa, USA; [4]Department of Atmospheric and Oceanic Sciences, University of Wisconsin–Madison, Madison, Wisconsin, USA; [5]School of City and Regional Planning, Georgia Institute of Technology, Atlanta, Georgia, USA; [6]Department of Urban and Regional Planning, University of Wisconsin–Madison, Madison, Wisconsin, USA; [7]Wisconsin Department of Natural Resources, Madison, Wisconsin, USA; [8]Global Health Institute, University of Wisconsin–Madison, Madison, Wisconsin, USA

Background: Automobile exhaust contains precursors to ozone and fine particulate matter (PM ≤ 2.5 µm in aerodynamic diameter; $PM_{2.5}$), posing health risks. Dependency on car commuting also reduces physical fitness opportunities.

Objective: In this study we sought to quantify benefits from reducing automobile usage for short urban and suburban trips.

Methods: We simulated census-tract level changes in hourly pollutant concentrations from the elimination of automobile round trips ≤ 8 km in 11 metropolitan areas in the upper midwestern United States using the Community Multiscale Air Quality (CMAQ) model. Next, we estimated annual changes in health outcomes and monetary costs expected from pollution changes using the U.S. Environmental Protection Agency Benefits Mapping Analysis Program (BenMAP). In addition, we used the World Health Organization Health Economic Assessment Tool (HEAT) to calculate benefits of increased physical activity if 50% of short trips were made by bicycle.

Results: We estimate that, by eliminating these short automobile trips, annual average urban $PM_{2.5}$ would decline by 0.1 µg/m^3 and that summer ozone (O_3) would increase slightly in cities but decline regionally, resulting in net health benefits of $4.94 billion/year [95% confidence interval (CI): $0.2 billion, $13.5 billion), with 25% of $PM_{2.5}$ and most O_3 benefits to populations outside metropolitan areas. Across the study region of approximately 31.3 million people and 37,000 total square miles, mortality would decline by approximately 1,295 deaths/year (95% CI: 912, 1,636) because of improved air quality and increased exercise. Making 50% of short trips by bicycle would yield savings of approximately $3.8 billion/year from avoided mortality and reduced health care costs (95% CI: $2.7 billion, $5.0 billion]. We estimate that the combined benefits of improved air quality and physical fitness would exceed $8 billion/year.

Conclusion: Our findings suggest that significant health and economic benefits are possible if bicycling replaces short car trips. Less dependence on automobiles in urban areas would also improve health in downwind rural settings.

Key words: air pollution, BenMAP, bicycling, built environment, climate change, ozone, particulate matter, physical activity, urban design, vehicle emissions. *Environ Health Perspect* 120:68–76 (2012). http://dx.doi.org/10.1289/ehp.1103440 [Online 2 November 2011]

The current fossil fuel–based transportation system of the United States negatively impacts human health by increasing air pollution and automobile accidents and by decreasing physical activity. Here, we consider how replacing short automobile trips with bicycle transport might yield health benefits through improved air quality and physical fitness, with a focus on the upper midwestern United States as our study region.

Both ozone (O_3) and fine particular matter ≤ 2.5 µm in aerodynamic diameter ($PM_{2.5}$) in the ambient air exacerbate bronchitis and asthma and may contribute to cardiovascular mortality (Brunekreef and Holgate 2002). Asthma affects 8.2% of U.S. citizens, and an estimated 10 million adults have diagnosed chronic obstructive pulmonary disease (COPD) (Centers for Disease Control and Prevention 2009). In addition, recent estimates attribute 63,000–88,000 premature deaths per year due to $PM_{2.5}$ [U.S. Environmental Protection Agency (EPA) 2010c]. In the United States, on-road vehicles are responsible for about 26% of volatile organic compounds (VOCs) and 35% of nitrogen oxide (NO_x) emissions (U.S. EPA 2005c, 2005d). NO_x and VOCs combine to form O_3 and contribute to nitrate and secondary organic aerosols, important components of $PM_{2.5}$. Nearly 240 U.S. counties, with > 118 million total residents, exceeded U.S. EPA O_3 standards in 2011, and > 200 counties (> 88 million total residents) failed to meet $PM_{2.5}$ standards, in part because of pollution from short car trips (U.S. EPA 2011a, 2011b).

Transport-related inactivity, that is, the use of motorized transport rather than walking and bicycling, has been linked to increased mortality and decreases in healthy life years, with the greatest impacts on chronic diseases including heart disease, stroke, colon cancer, diabetes mellitus type 2, obesity, breast cancer, and osteoporosis [World Health Organization (WHO) 2002]. Carlson et al. (2009) estimated that 32.4% of the U.S. population is fully inactive (no moderate-intensity or vigorous-intensity physical activity lasting at least 10 min at a time), while only 33.5% is physically active, defined as 30 min/day with moderate-intensity activity, ≥ 5 days/week. In a recent Dutch study, Johan de Hartog et al. (2010) concluded that shifting from short car trips to bicycle trips would reduce all-cause mortality, with estimated reductions in mortality due to increased physical activity that were nine times greater than estimated increases in mortality due to increased pollution inhalation and traffic-related fatality estimates in the Netherlands.

In the United States, 28% of all car trips are ≤ 1.6 km (1 mi), which is the distance that a typical European would walk (European Commission 2001; Pucher and Dijkstra 2003). Another 41% of all trips are ≤ 3.2 km (2 mi), a distance that many Europeans would be as likely to bicycle as to

Address correspondence to M.L. Grabow, Nelson Institute for Environmental Studies, Center for Sustainability and the Global Environment, University of Wisconsin–Madison, 1710 University Ave., Madison, WI 53726 USA. Telephone: (314) 249-0471. Fax: (608) 265-4113. E-mail: grabow@wisc.edu

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walk (European Commission 2001; Pucher and Dijkstra 2003). If we use European travel behavior as a point of comparison for walking and bicycling activity for the United States, these data suggest that many car trips in the United States could be avoided.

Amplifying the potential benefits of increased bicycle use is the nonlinear relationship of vehicle emissions to travel time. A large fraction of emissions (25% of VOC and 19% of primary $PM_{2.5}$) are emitted in just the first few minutes of automobile operation, often known as "cold start," before pollution-control devices operate [Federal Highway Administration (FHWA) 2006]. Because emissions control systems reach operating temperature only after several miles of travel and typically cool below operating range in < 1 hr (Singer et al. 1999), reducing the number of short trips could disproportionately curtail pollutant emissions from on-road vehicles.

In the present study, we quantified the potential health and monetary benefits of replacing short (≤ 4 km one way) car trips with travel by bicycle (50% of trips) in the 11 largest midwestern metropolitan statistical areas (MSAs): Chicago, Illinois; Cincinnati, Cleveland, Columbus, and Dayton, Ohio; Detroit and Grand Rapids, Michigan; Indianapolis, Indiana; Madison and Milwaukee, Wisconsin; and Minneapolis/St. Paul, Minnesota. This study builds on the Projected Land Use and Transportation (PLUTO) modeling framework developed by Stone et al. (2007). We estimated changes in regional emissions and air quality, as well as resulting health benefits, across the upper midwestern states [see Supplemental Material, Figure 1 (http://dx.doi.org/10.1289/ehp.1103440) for a map of the area]. In addition, we estimated the benefits of increased physical activity using the Health Economic Assessment Tool (HEAT) for cycling developed by the WHO (Rutter et al. 2007).

Methods

We estimated that eliminating short car trips (≤ 8 km round trip) in urban areas of Illinois, Indiana, Michigan, Minnesota, Ohio, and Wisconsin would reduce residential vehicle use by 20%. This estimate is based on a census-tract level travel and mobile emissions inventory by Stone et al. (2007), who combined 1995 Nationwide Personal Transportation Survey (NPTS) responses (FHWA 1997), demographic modeling of household vehicle travel, and the U.S. EPA MOBILE6 emissions factor model (U.S. EPA 2004b). From that contemporary emissions inventory, we estimated current emissions levels if all round trips of ≤ 8 km in urban and suburban census tracts were made using alternate modes of transportation. To inform the potential impact of a range of realistic policies and choices, we used these estimated reductions to quantify the maximum potential response to a change in travel behavior. Although arbitrary, this assumed reduction in short auto trips would be consistent with the use of active (cycling or walking) transportation in European cities similar in density and population to the MSAs considered here. These values represent theoretical upper bounds on short-trip transportation choices under current travel patterns and population density. We assume that no change occurs in rural travel because distances between residential and commercial areas are typically too great for bicycling or walking and because rural populations are too low to support mass transportation.

Specifically, we compared transportation modes used in the study-area cities, with populations ranging from 837 persons/km^2 in Grand Rapids to 4,884 persons/km^2 in Chicago (average 2,051 persons/km^2), to five European cities with similar population densities (range 901–5,971, average 2,910 persons/km^2) [see Supplemental Material, Table 1 (http://dx.doi.org/10.1289/ehp.1103440)]. Although public transportation use was similar, only 39% of trips were made by automobile in the European cities, compared with 80% of trips in the Great Lakes region. Although the configurations and historical growth patterns of the European cities differ from their American counterparts, the fact that half of all trips used active transportation suggests that active transport for 50% of short trips is feasible for similar travel distances in midsized American cities of similar density and that greater active transportation need not be limited to areas of highest density.

We estimated changes in emissions only for on-road light-duty passenger vehicles with internal combustion engines and only for round trips ≤ 8 km. We modeled changes in primary emissions (including NO_x, carbon monoxide, sulfur dioxide, ammonia, VOCs, elemental carbon, organic carbon, and primary fine and coarse particulate matter) from all stages of vehicle operation, as well as emissions from evaporation, brake dust, resuspended road dust, and refueling. Reducing the number of short trips further lessens the frequency of cold starts from 59.9% to 21.9% of trips in urban tracts and from 55.6% to 20.3% in suburban tracts, with corresponding reductions in VOC and NO_x emissions. We mapped emissions from the census-tract level to the 12 × 12 km^2 model grid by area-weighted averaging using the U.S. EPA Sparse Matrix Operator Kernel Emissions (SMOKE) model, version 2.4 (Community Modeling and Analysis System Center 2007). Emissions from sources other than motor vehicles were from the 2001 National Emissions Inventory (U.S. EPA 2005a) and were held constant in both scenarios.

We estimated changes in ambient air $PM_{2.5}$ and O_3 concentrations using hourly regional chemical transport simulations with the Community Multiscale Air Quality Model (CMAQ), version 4.6 (Byun and Schere 2006), driven by meteorology from the weather research and forecasting model for the full year of 2002 (Skamarock and Klemp 2008; Skamarock et al. 2008). Simulations with CMAQ were conducted on a 12 × 12 km^2 grid and included gas phase, aqueous, and heterogeneous chemical reactions and equilibrium aerosol thermodynamics. We followed the model configuration used by Spak and Holloway (2009), with boundary conditions from a 36 × 36 km^2 simulation over continental North America.

We used the Environmental Benefits Mapping and Analysis Program (BenMAP) version 4.0.35 (U.S. EPA 2010a) to estimate health impacts due to CMAQ-simulated changes in ambient air pollution resulting from reduced car travel. Because BenMAP addresses both mobile and stationary sources (U.S. EPA 2004a, 2008), it has been used to support the creation of environmental regulations in several countries.

After air quality data is loaded into BenMAP, the program determines the change in ambient air pollution. BenMAP then uses concentration–response functions (CR) to calculate the relationship between the pollution and certain health effects, applying the relationship to the exposed population (Abt Associates 2010). Finally, BenMAP uses a "damage function" to estimate health costs and benefits from changes in air quality. A damage function quantifies the health benefits and economic value of reduced exposure to pollutants (Davidson et al. 2007).

BenMAP 4.0 (i.e., version 4.0.35) incorporates hourly air pollution data and county-level baseline incidence rates for the following health outcomes: overall mortality, asthma exacerbations, chronic bronchitis, hospital admissions, acute myocardial infarctions, acute and chronic respiratory infections, upper and lower respiratory infections, work-loss days, and school-loss days. Spatial specificity in baseline incidence data varies by health outcome and location; where county-level data are not available, BenMAP distributes state estimates to the county level using age-specific rates for each health outcome within each county. For mortality estimates, BenMAP combines national-level census mortality rate projections and county-level age-specific incidence rates from 2006 with projected changes in study area populations to derive county-level mortality rate projections for 2010. For the present study, BenMAP used state-level hospitalization data to estimate county-level incidence for Minneapolis/St. Paul, Chicago, and Indianapolis; county-level incidence data for all cities in Ohio; and city hospital discharge data for Milwaukee, Madison, Detroit, and Grand Rapids. For emergency room (ER) admissions, we used midwest regional incidence data for Detroit, Grand Rapids, Chicago, and Indianapolis; state-level data for Minneapolis/St. Paul; county-level data for Ohio cities; and hospital discharge-level data for Milwaukee and Madison. For all cities, nonfatal acute myocardial infarction incidence rates were based on regional hospitalization data. All other health end point data were based on national figures.

BenMAP assigns monetary values to the reduction of adverse health effects based on national averages that do not reflect intracity or intercity variability in costs. The BenMAP analysis was conducted on the 12 × 12 km^2 grid, using 2010 census projection allocation to the grid by the U.S. EPA. Valuation is in 2010 dollars.

We combined air quality estimates for 2002 from CMAQ with 2002 U.S. EPA monitoring using spatial scaling by Voronoi nearest neighbor averaging (e.g., Chen et al. 2004). This pairing yields air quality inputs to BenMAP including complete spatial and temporal coverage by high-resolution hourly modeling, constrained to match concentrations observed near monitors. We then used the expert-derived $PM_{2.5}$ CR functions, valuation estimates, and pooling methods used for the U.S. EPA 2006 Regulatory Impact Analysis, plus O_3 exposure–response functions for 2008 National Ambient Air Quality Standard (NAAQS) evaluations (U.S. EPA 2004a, 2008; University of North Carolina Institute for the Environment, Community Modeling and Analysis System Center 2008). Because multiple studies exist for each given health incidence, pooling techniques are often used to statistically combine the results. Using BenMAP, we ran each CR function and pooling of incidence and valuation for each health end point in a 5,000-member Monte Carlo ensemble. Sources of CR functions used in this analysis are presented in Tables 1 and 2. As standard practice, the U.S. EPA does not pool mortality studies. Thus, we used the Harvard Six Cities study (Pope et al. 2002) as BenMAP input for $PM_{2.5}$ mortality; that study included the most representative sites. We selected the 2010 population database to use in BenMAP because the sensitivity studies we conducted indicated that choice of year has no substantial impact (1–2% difference) on incidence of health threats.

Table 1. Estimated $PM_{2.5}$ reductions, reductions in health impacts, and valuation of reduced $PM_{2.5}$ exposure.

City/data	Mean annual $PM_{2.5}$ reduction[a]	Mortality	Asthma	Chronic bronchitis	Respiratory problems[b]	Cardiovascular problems[c]	Work-loss days[d]	Total savings
Chicago								
Incidence	0.05 (0.02, 0.15)	162 (63, 260)	802 (91, 2,301)	29 (5, 53)	36,690 (30,233, 43,145)	253 (99, 407)	5,923 (5,161, 6,685)	
Savings		1,273 (176, 3,361)	0.04 (0.01, 0.13)	13 (1, 44)	2.47 (1.44, 3.61)	16.7 (5, 40.9)	1 (0.96, 1.2)	1,305 (184, 3,449)
Cincinnati								
Incidence	0.03 (0.01, 0.10)	26 (10, 42)	100 (11, 288)	4 (0.7, 7)	4,751 (3,918, 5,583)	34 (13, 54)	763 (665, 861)	
Savings		212 (29, 549)	0.005 (0.001, 0.02)	1.6 (0.14, 6)	0.32 (0.18, 0.46)	2.27 (0.68, 5.6)	0.13 (0.11, 0.15)	212 (30, 561)
Cleveland								
Incidence	0.05 (0.02, 0.16)	53 (21, 85)	184 (21, 527)	7 (1, 13)	8,804 (7,264, 10,345)	74 (28.5, 119)	1,405 (1,224, 1,586)	
Savings		418 (58, 1,105)	0.01 (0.001, 0.03)	3 (0.3, 11)	0.60 (0.35, 0.88)	4.86 (1.4, 11.9)	0.23 (0.2, 0.27)	427 (60, 1,129)
Columbus								
Incidence	0.04 (0.02, 0.14)	27 (11, 43)	124 (14, 355)	4 (1, 8)	5,854 (4,829, 6,879)	35 (13.5, 55.9)	951 (828, 1,073)	
Savings		212 (29, 561)	0.007 (0.001, 0.02)	1.9 (0.2, 6.7)	0.39 (0.23, 0.57)	2.37 (0.71, 5.8)	0.16 (0.14, 0.18)	217 (31, 574)
Dayton								
Incidence	0.04 (0.03, 0.10)	14 (6, 23)	47 (5, 136)	2 (0.4, 3.4)	2,278 (1,880, 2,676)	18 (7, 29)	365 (318, 412)	
Savings		112 (15, 296)	0.003 (0, 0.008)	0.8 (0.07, 2.8)	0.15 (0.09, 0.22)	1.22 (0.36, 3)	0.06 (0.05, 0.07)	114 (16, 302)
Detroit								
Incidence	0.05 (0.02, 0.16)	106 (41, 171)	462 (52, 1,325)	17 (3, 31)	21,181 (17,462, 24,899)	158 (61.5, 254)	3,395 (2,958, 3,832)	
Savings		836 (115, 2,208)	0.025 (0.003, 0.08)	7.3 (0.6, 26)	1.4 (0.83, 2.09)	10.46 (0.26, 2.2)	0.66 (0.58, 0.75)	856 (120, 2,262)
Grand Rapids								
Incidence	0.03 (0.02, 0.06)	7 (3, 11)	45 (5, 130)	2 (0.3, 2.9)	2,023 (1,667, 2,379)	13 (4.75, 20.3)	327 (285, 369)	
Savings		54 (7, 143)	0.002 (0, 0.007)	0.66 (0.06, 2.3)	0.13 (0.08, 0.19)	0.88 (0.26, 2.2)	0.054 (0.047, 0.061)	56 (8, 148)
Indianapolis								
Incidence	0.03 (0.01, 0.09)	19 (7, 30)	85 (10, 243)	3 (0.5, 5.2)	3,676 (3,024, 4,328)	24 (9.3, 38.8)	592 (516, 669)	
Savings		146 (20, 386)	0.005 (0.001, 0.01)	1.2 (0.11, 4.3)	0.25 (0.14, 0.36)	1.59 (0.48, 3.9)	0.1 (0.09, 0.12)	149 (21, 394)
Madison								
Incidence	0.02 (0.02, 0.04)	1 (0.44, 1.8)	10 (1, 28)	0.42 (0.08, 0.8)	565 (469, 661)	3 (1.2, 5)	93 (81, 105)	
Savings		8.8 (1.2, 23.3)	0.001 (0, 0.002)	0.18 (0.02, 0.6)	0.037 (0.022, 0.053)	0.23 (0.067, 0.55)	0.016 (0.014, 0.018)	9 (1, 24)
Milwaukee								
Incidence	0.04 (0.02, 0.08)	12 (5, 19)	73 (8, 210)	3 (0.5, 5)	3,407 (2,809, 4,005)	21 (7.7, 34.3)	545 (475, 616)	
Savings		93 (13, 246)	0.004 (0, 0.012)	1.18 (0.10, 4.2)	0.22 (0.13, 0.33)	1.72 (0.51, 4.23)	0.095 (0.08, 0.1)	96 (14, 254)
Twin Cities								
Incidence	0.01 (0.00, 0.06)	7 (2.7, 11)	87 (10, 248)	3 (0.65, 6)	4,379 (3,619, 5,139)	27 (10, 44)	709 (618, 800)	
Savings		54 (7, 142)	0.005 (0.001, 0.014)	1.46 (0.13, 5.2)	0.28 (0.17, 0.42)	1.95 (0.58, 4.8)	0.13 (0.11, 0.15)	57 (8, 152)
Total MSAs								
Incidence	0.01 (0.00, 0.16)	433 (169, 698)	2,018 (228, 5,790)	75 (14, 137)	93,607 (77,175, 110,037)	659 (255, 1,062)	15,067 (13,128, 17,006)	
Savings		3,484 (480, 9,199)	0.109 (0.012, 333)	32 (2.8, 112)	6.16 (3.59, 9.01)	43.4 (13, 106.3)	2.6 (2.38, 3.1)	3,570 (500.7, 9,875)
Outside MSAs total								
Incidence		92 (35, 149)	541 (60.9, 1,552)	21.64 (3, 40.2)	579 (278, 878)	200.6 (71.8, 332)	4,280 (3,729, 4,830)	
Savings		726.7 (100, 1,919)	0.0227 (0.0026, 0.069)	7.35 (0.640, 26)	1.36 (0.792, 1.99)	11.1 (3.27, 27.4)	0.489 (0.426, 0.552)	747.02 (105.1, 1,975)
Region total								
Incidence		525 (204, 806)	2,559 (289, 7,342)	97 (17, 177)	94,186 (77,453, 110,915)	860 (327, 1,394)	19,347 (16,857, 21,837)	
Savings		4,143 (571, 10,937)	0.132 (0.015, 0.402)	39.1 (3.41, 138)	7.52 (4.38, 10.99)	54.5 (16.2, 132)	3.24 (2.82, 3.65)	4,247.5 (598, 11,222)

Values for incidence represent estimated incidence per adverse health effect avoided due to a change in air pollution in the given city per year; savings are presented in millions of dollars. Values in parentheses are 95% confidence intervals, and all changes are annualized.
[a]Change in $PM_{2.5}$ (μg/m^3) was calculated as area averaged and reported with a range of minimum and maximum grid cell values; data for $PM_{2.5}$-related health effects estimated in this analysis (and the source of the PM concentration–response functions used to estimate the change in incidence) are from Abbey et al. (1995), Dockery et al. (1996), Ito (2003), Laden et al. (2006), Moolgavkar (2000a, 2003), Norris et al (1999), Ostro (1987), Ostro and Rothschild (1989), Ostro et al. (2001), Peters et al. (2001), Pope et al. (1991, 2002), Schwartz and Neas (2000), Sheppard (2003), and Vedal et al. (1998). [b]Respiratory problems include upper and lower respiratory symptoms, hospital admissions (respiratory), emergency room visits (respiratory), and cases of acute bronchitis. [c]Cardiovascular problems include nonfatal acute myocardial infarctions and cardiovascular hospitalizations. [d]Yearly work-loss-day incidence based on estimates from the 1996 National Health Interview Survey (Adams et al. 1999).

To address the potential health and economic co-benefits that would result if half of all short trips were made by bicycle, we used HEAT. This model uses relative risk data (Anderson et al. 2000) to estimate cost savings from reduced all-cause mortality. Controlling for socioeconomic variables (e.g., age, sex, smoking) and leisure time activity, HEAT calculates risk reduction for days spent cycling based on estimates of total number of days cycled, distance, and average speed (Rutter et al. 2007).

We used HEAT analysis to estimate the monetized health benefits associated with the conversion of one-half of short trips (< 8 km round trip) by car to be made by bicycle. This represents 10% of vehicle miles traveled (VMT) for the region. We used the U.S. EPA value of a statistical life ($7.4 million) (U.S. EPA 2010b) and the annual percentage of all-cause working-age mortality [0.00390; 95% confidence interval (CI): 0.00277, 0.00503] (Wilkinson and Pickett 2008). We assumed an average of 124 days of cycling per year, HEAT's default value (Rutter et al. 2007), which is representative of the climate of the upper midwest, where bicycle commuting is most common from April through October. We also assumed that only 50% of these trips would be undertaken by people who do not currently cycle, thus excluding the small percentage of the population already benefitting from cycling, as well as elderly individuals or those physically unable to bicycle. We used the NPTS average commute distance for each MSA (from 3.34 to 3.98 km) with an average speed for commuter cyclists of 14 kph. Finally, we used the HEAT-recommended default percentage (90%) of cyclists completing a round trip each day.

Results

Simulations yielded unique hourly estimates of surface-level $PM_{2.5}$ throughout the year (Figure 1A) and O_3 during the warm season (1 May 30–September) (Figure 1C) on a 12 × 12 km^2 grid for 2002. The CMAQ simulations described here captured spatial and temporal variability in $PM_{2.5}$ [see Supplemental Material, Table 2 (http://dx.doi.org/10.1289/ehp.1103440)] and O_3 (see Supplemental Material, Table 3) when compared with U.S. EPA monitoring data throughout the region, with performance for $PM_{2.5}$ and O_3 both exceeding community and U.S. EPA expectations for chemical transport modeling in policy and research applications.

We estimated that substitution of non-emitting modes for short trips would achieve average annual reductions in the 24-hr average $PM_{2.5}$ concentrations considered in U.S. $PM_{2.5}$ regulations (Figure 1B). Regional O_3 would also be reduced throughout the May–September summer season (calculated based on daily maximum 8-hr and 1-hr averages, consistent with U.S. O_3 regulations) but daytime O_3 would increase in the largest cities

Table 2. Estimated O_3 changes, changes in health impacts, and valuation of changes in O_3 exposure.

City/data	Change in O_3 (ppm)[a]	Acute respiratory symptoms	ER visits (respiratory)	HA (respiratory)	Mortality	School-loss days	Worker productivity	Total savings
Chicago								
Incidence	–0.09 (–0.23, 0.39)	–5,780 (–10,131, –1,431)	–3 (–12, 2.4)	–8 (–18. 0.87)	–14 (–22, –6)	–1,762 (–3,174, –351)	–13,564	
Savings		–0.365 (–0.718, 0.047)	–0.001 (–0.004, 0.001)	–0.19 (–0.41, 0.03)	–108 (–305, 38)	–0.17 (–0.3, –0.03)	–0.017	–109 (–306, 37)
Cincinnati								
Incidence	–0.13 (–0.25, 0.21)	483 (–229, 1,194)	0.08 (–1.1, 1.3)	0.53 (–0.72, 1.76)	0.78	151 (–75, 377)	1,279	
Savings		0.03 (–0.02, 0.09)	0 (0, 0)	0.011 (–0.02, 0.04)	6.2 (–20.2, 37.3)	0.014 (–0.007, 0.036)	0.002	6.2 (–20.2, 37.5)
Cleveland								
Incidence	–0.17 (–0.33, 0.70)	353 (–871, 1577)	–0.15 (–2.6, 2.2)	0.35 (–2.2, 2.8)	0.54 (–2.3, 3.4)	106 (–271, 483)	2,490	
Savings		0.022 (–0.07, 0.12)	0 (–0.001, 0.001)	0.008 (–0.05, 0.06)	4.4 (–47.4, 61)	0.01 (–0.03, 0.05)	0.003	4.412 (–47.5, 61.3)
Columbus								
Incidence	–0.2 (–0.27, 0.23)	199 (–555, 953)	–0.07 (–1.3, 1.09)	0.19 (–0.9, 1.3)	0.45 (–0.85, 1.8)	58 (–182, 297)	1,697	
Savings		0.013 (–0.04, 0.07)	0 (0, 0)	0.005 (–0.02, 0.03)	3.64 (–20, 29.7)	0.005 (–0.017, 0.028)	0.002	3.669 (–20, 29.9)
Dayton								
Incidence	–0.21 (–0.26, 0.01)	713 (362, 1,064)	0.48 (0, 1.4)	0.99 (0.26, 1.8)	2 (1.3, 2.9)	209 (100, 318	2,568	
Savings		0.045 (0.02, 0.08)	0 (0, 0.001)	0.02 (0.004, 0.038)	16 (2.5, 40.6)	0.02 (0.01, 0.03)	0.003	16.462 (2.6, 40.7)
Detroit								
Incidence	–0.17 (–0.29, 0.18)	830 (–976, 2,634)	0.46 (–1.7, 3)	0.58 (–2.3, 3.3)	1 (–2.5, 4.7)	257 (–326, 840)	1,389	
Savings		0.052 (–0.07, 0.19)	0 (–0.001, 0.001)	0.011 (–0.05, 0.07)	8.5 (–48, 68)	0.024 (–0.031, 0.08)	0.002	8.631 (–48.5, 68.7)
Grand Rapids								
Incidence	–0.23 (–0.29, 0.07)	1,124 (571, 1,677)	0.65 (0, 1.9)	1 (0.34, 1.8)	2 (1.5, 3.4)	363 (174, 552)	12,235	
Savings		0.071 (0.033, 0.12)	0 (0, 0)	0.02 (0.005, 0.035)	19.3 (3, 48)	0.035 (0.02, 0.05)	0.015	19 (3, 48)
Indianapolis								
Incidence	–0.15 (–0.22, 0.12)	532 (13, 1,051)	0.32 (–0.39, 1.3)	0.72 (–0.38, 1.8)	1 (0.23, 2.1)	169 (–9.2, 347)	2,036	
Savings		0.034 (–0.002, 0.076)	0 (0, 0)	0.017 (–0.009, 0.04)	9.4 (–8, 33)	0.016 (–0.001, 0.033)	0.003	9.476 (–8, 33)
Madison								
Incidence	–0.12 (–0.16, 0.04)	135 (33, 237)	0.03 (–0.01, 0.11)	0.11 (0.02, 0.21)	0.25 (0.12, 0.39)	41 (12, 70)	436	
Savings		0.009 (0.002, 0.017)	0 (0, 0)	0.002 (0, 0.004)	1.99 (0.28, 5.2)	0.004 (0.001, 0.007)	0.001	2.002 (0.28, 5.2)
Milwaukee								
Incidence	–0.12 (–0.20, 0.14)	134 (–292, 559)	–0.09 (–0.74, 0.45)	0 (–0.61, 0.57)	0.2 (–0.6, 1.1)	21 (–116, 158)	–60	
Savings		0.008 (–0.02, 0.04)	0 (0, 0)	0.001 (–0.01, 0.01)	1.65 (–11.5, 16.9)	0.002 (–0.01, 0.02)	0	1.66 (–11.6, 16.9)
Twin Cities								
Incidence	–0.05 (–0.11, 0.38)	–2,190 (–3,590, –790)	–0.76 (–2.5, 0.24)	–2 (–3.7, –0.3)	–4 (–6.5, –2.3)	–588 (–994, –181)	–7,881	
Savings		–0.138 (–0.25, –0.039)	0 (–0.001, 0)	–0.039 (–0.07, –0.003)	–34.7 (–88.9, 2.57)	–0.056 (–0.095, –0.0017)	–0.01	–34.9 (–89.3, 2.5)
Total MSAs								
Incidence	–0.07 (–0.33, 0.70)	–3,467 (–15,663, 8,723)	–963 (–8,281, 5,764)	–5.97 (–28, 16)	–9 (–32, 14)	–976 (–4,860, 2,909)	2,627	
Savings		–0.22 (–1.15, 0.72)	–0.00096 (–0.0083, 0.0058)	–0.134 (–0.63, 0.36)	–71.7 (–543, 380)	–0.093 (–0.46, 0.28)	0.0033	–72.14 (–545.38, 381.47)
Outside MSAs total								
Incidence		33,628 (17,087, 50,167)	17 (0, 51.5)	47 (10.9, 83)	91 (53.3, 129.6)	9,608 (4,610, 14,606)	282,669	
Savings		212 (0.995, 3.6)	0.0073 (0, 0.019)	1.01 (0.222, 1.81)	763.2 (118, 1,1893)	0.914 (0.439, 1.39)	0.353	767.6 (120, 1,900)
Region total								
Incidence		30,161 (1,423, 58,891)	14.63 (–22.4, 66.8)	41.13 (–17.2, 98.8)	82.6 (21.3, 143.3)	8,632 (–250.2, 17,515)	285,296	
Savings		1.91 (–0.14, 4.33)	0.0062 (–0.0083, 0.025)	0.876 (–0.42, 2.17)	691.5 (–425, 2,273)	0.822 (–0.0239, 1.67)	0.357	695.5 (–425, 2,281)

Values for incidence represent estimated incidence per adverse health effect avoided due to a change in air pollution in the given city per year; costs are expressed as negative and benefits as positive (millions of dollars). Values in parentheses for incidence and savings are 95% confidence intervals (in most cases rounded to nearest decimal), and all changes are annualized.
[a]Change in O_3 season average daily maximum 8-hr are calculated as mean area (range of grid cell values). Data for O_3-related health effects (and the source of the O_3 concentration–response functions used to estimate the change in incidence) estimated in this analysis are from Bell et al. (2004, 2005), Burnett et al. (2001), Chen et al. (2000), Crocker et al. (1981), Gilliland et al. (2001), Huang et al. (2005), Ito et al. (2005), Jaffe et al. (2003), Levy et al. (2005), Moolgavkar et al. (1997), Ostro and Rothschild (1989), Peel et al. (2005), Schwartz (1994a, 1994b,1995, 2005), and Wilson et al. (2005).